

AN ABSTRACT OF THE THESIS OF

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Title: THE EFFECTS OF RED ALDER LEAF FALL ON THE WATER
COLOR AND OTHER WATER QUALITY CHARACTERISTICS
OF A SMALL WATERSHED IN NORTHWEST OREGON

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Paul W. Adams

This study was initiated to make some preliminary evaluations of the effects of red alder (Alnus rubra, Bong.) leaf fall on stream water quality, particularly water color. Laboratory tests were also conducted to better understand the effects of leaf loading, duration of leaf leaching, and type of leaching on changes in water quality. The study focused on the Seaside Municipal Watershed, which is typical of many municipal systems in the Oregon Coast Range. In these systems, streamside alder stands are common and water treatment is limited to chlorination.

The potential for coloration of stream water by alder leaves is present but in the fall of 1981 there was no chronic (more than one or two days) low flow water color problem observed at the Seaside diversion in the South Fork of the Necanicum River. The color observed during the lowest flows, which occurred almost through September, averaged 20 platinum-cobalt units of true color. While this is slightly higher than the 5-15 units measured during the winter months and slightly higher than the maximum standard of 15

units set by the USPH for domestic supplies, it is a very faint tint and probably would not cause many complaints by domestic users.

Field results showed that the water color can become elevated for short periods at the diversion. There were two occasions when the color was above 20 units (30 units on October 6 and 25 units on November 11). Elevated water color was also observed in a tap water sample from the city's water works building on October 27 (45 units). Each of these color increases occurred during the beginning of a storm and there was a subsequent drop in color after the storms. This increase in color during storms is probably due to the flushing of storage sites containing dissolved organic material by rising flows. The other water quality characteristics measured (dissolved oxygen, conductivity, pH, nitrate-nitrogen and temperature) showed evidence of some relationships with leaf fall, but there were no signs of water quality impairment at the city diversion during the autumn of 1981.

Laboratory leachings in filtered stream water showed that for a given mass of leaves leached in still water, there is a fairly constant release of color (≈ 2 units/hr) for leaching periods between 2 hours and 4 weeks. Running water was more efficient than still water in leaching colored material. There also appears to be a limit to the amount of material that can be removed from a given leaf mass in the first 48 hours of leaching. In still water a linear increase in color occurred with increases in mass loading, and weight loss (as a percentage of original dry weight) was constant. Laboratory tests also showed that ultra violet absorbance may provide a reasonable estimate of dissolved organic carbon concentrations in

systems dominated by red alder input. This, however, has yet to be verified with field samples.

In a small stream with lower flows than the South Fork of the Necanicum River bounded by an alder stand, color could possibly be a chronic problem during autumn low flows. The South Fork generally appears to have too large a flow for this to normally occur. However, a hypothetical calculation of maximum coloration, under conditions of very low flow and peak leaf fall, resulted in a water color of approximately 110 units in this stream. Short-term increases in color at the beginning of the first few storms would be expected to be more common, in a stream of this size, if there is time for organic matter accumulation in storage sites between storms. Such increases in true color would likely be accompanied by increases in apparent color due to sediment, which may even supercede the coloration due the dissolved or colloidal organic material.

THE EFFECTS OF RED ALDER LEAF FALL ON THE
WATER COLOR AND OTHER WATER QUALITY
CHARACTERISTICS OF A SMALL WATERSHED
IN NORTHWEST OREGON

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The Effects of Red Alder Leaf Fall on the
Water Color and Other Water Quality
Characteristics of a Small Watershed
in Northwest Oregon

I. INTRODUCTION

Small streams that run through forests are largely heterotrophic in nature. That is, the organisms they contain receive much of their energy from terrestrial sources (Cummins, 1974; Fisher and Likens, 1972; Hynes, 1975; Triska and Sedell, 1975; Vannote et al., 1980). Much of this allocthonous input, where deciduous vegetation dominates the streamside, is in the form of leaves and occurs during the autumn leaf fall period (Fisher and Likens, 1972; Kaushik and Hynes, 1971). Considerable study has been made of the role and fate of this material in forest streams (Cummins et al., 1972; Cummins, 1974; Cummins et al., 1980; Dahm, 1981; Kaushik and Hynes, 1971; Merritt and Lawson, 1978; Petersen and Cummins, 1974; Triska and Sedell, 1975). This research has shown that these streams are quite capable of assimilating the material they receive. In fact, many organisms that inhabit small streams have developed life cycles that are intertwined with the autumnal input of leaf material.

There are different concerns involved when referring to stream water quality as opposed to domestic water quality. Stream water quality is concerned with physical and chemical characteristics which affect the organisms present (eg. temperature, dissolved oxygen, turbidity nutrients, toxic materials). Domestic water quality, on the other hand, is more concerned with the aesthetic characteristics of taste, odor, and color, as well as chemical and biological conditions related to human health. It is, therefore, important to

distinguish between domestic water quality (human use) and water quality as it relates to the stream environment. Changes that are unacceptable for domestic consumption may be beneficial to the stream environment.

Aquatic systems can, however, become temporarily overloaded with natural inputs of organic matter. When this occurs the quality of the water can be degraded, especially with respect to domestic use. This degradation can range from small odor, taste and color increases to heavy coloration, highly objectionable tastes and odors, and severe oxygen depletion (the latter an environmental quality problem) (Allen, 1960; Slack, 1964; Slack and Feltz, 1968). This natural overloading often occurs in northern temperate regions during the fall months when streamside deciduous vegetation loses its leaves and stream flows are generally lowest (Rothacher, 1971).

Leaves can exert rapid and significant control over water quality (both domestic and environmental) during these low flow conditions (Allen, 1960; Nykvist, 1962; Ponce, 1974; Slack, 1964; Slack and Feltz, 1968). This is normally only a short term condition because as soon as substantial runoff begins to occur, dilution and flushing of materials in the stream quickly improves water quality. However, even a short term reduction in stream water quality can present problems to potential users and aquatic life (Allen, 1960; Hoak, 1960; Ponce, 1974; Slack and Feltz, 1968).

The Problem

The city of Seaside, located on Oregon's northern coast, has a water distribution system fairly typical

of small cities and towns in Oregon's Coast Range. The water is basically diverted from the South Fork of the Necanicum River and pumped to the users. The only treatment is chlorination. The chlorine is injected into the main line at a point 1.5-3 km downstream of the diversion at highway 26 but still 8-10 km from Seaside. There is one other chlorination point in the distribution system about 1.5 km from the city. The chlorination rate is approximately 40 pounds per day into the entire system and the diversion averages approximately $6800-7600 \text{ m}^3/\text{day}$ (1.8-2.0 million gallons per day) on an annual basis (Personal communication, City of Seaside personnel). During the summer/fall low flows the entire river flow is diverted. Since there is no filtration, the diverted water is checked daily for turbidity. If the turbidity is greater than 1 Nephelometric Turbidity Unit (NTU), the city switches to an alternative water supply. This is a small emergency storage reservoir that can be used for short periods to supply water.

In January 1981, a meeting was held in Seaside to discuss the City personnel's concern with domestic water quality problems associated with fall runoff, particularly color and taste. The city had received complaints during the fall period in the past. It was noted that the situation with color and taste only occurred during a few months in the fall and it was thought that the problem could be caused, at least in part, by the accumulation of alder leaves during fall low flows. This type of problem is relatively common to varying degrees in small streams that have riparian zones dominated by deciduous vegetation (Corbett and Heilman, 1975; Slack, 1964; Slack and Feltz, 1968; State Water Resources Board, Oregon, 1973).

Such areas are quite common in the Oregon Coast Range where red alder quickly revegetates moist sites that have been disturbed (e.g., through logging activities).

Individuals from various local, state and federal agencies in attendance at this meeting expressed the general opinion that the problem warranted more attention. Assistance was solicited from the Forest Engineering Department of Oregon State University, and this study was thus initiated. To narrow the focus of the research, attention was directed primarily at the specific problem of water coloration.

Study Objective

To provide initial documentation and evaluation of the effects of red alder leaf fall on the water quality, particularly water color, of a municipal water supply in northwestern Oregon.

Background

Leaf fall

Deciduous leaf fall occurs in a similar pattern throughout the northern temperate climatic region. The leaf fall period is generally from late August or early September through late November or early December, with a peak occurring in October or November.

During this short period, the amount of leaf material falling can be quite high. Zavitkovski and Newton (1971) found that an average of 521.75 g/m^2 of leaves fell annually in 15 stands of red alder in western

Oregon, ranging in age from 3-34 years. They also stated that 2/3 of this amount fell between September and December. Slack and Feltz (1968) found leaf fall in a small watershed in Virginia to be $5.3 \text{ g/m}^2/\text{day}$ during the first six weeks (beginning in September), with a peak of about $12 \text{ g/m}^2/\text{day}$ in late October and early November. Petersen and Cummins (1974) estimated allocthonous inputs based on standing crop measurements to be $5.0 \text{ g/m}^2/\text{day}$ for a Michigan watershed.

Understory riparian communities can also contribute considerable amounts of organic material to streams during the autumn months. Annual shrubs and herbs can contribute a great deal of organic matter to streams when they die back in autumn (Campbell and Franklin, 1979). This study, done in the Oregon Cascade mountains, found that herbacious communities tend to be found adjacent to streams to which they directly contribute organic matter. Herbacious communities were found to have yearly foliar productions of 5 to 277 g/m^2 , depending on the plant size. Shrub dominated communities produced 28 to 88 g/m^2 . This input also occurs for the most part during the autumn months and could be a major source of organic detritus added to streams bordered by heavy shrub and herbacious growth.

Foliar inputs of shrubs and herbs were not considered in this study because the riparian zones in the Seaside watershed are heavily dominated by red alder. The understory vegetation did not appear to have the potential to contribute greatly to allocthonous inputs.

The results of these studies indicate that a large amount of organic material in the form of leaves can be directly added to a stream system during a few months in the fall. Leaves may also reach streams from

the banks. It was estimated that 1.5 times the amount of leaves falling directly into the stream were being blown in from the banks and side slopes of Watershed 10, a fairly steep watershed in the Oregon Cascades (Coniferous Forest Biome, 1974).

Leaf Decomposition

There are two major processes whereby leaves are broken down. These are physical or mechanical decomposition, and biological decomposition. Both processes occur throughout the decomposition cycle. Physical decomposition refers to the fragmentation (ripping, pounding, and grinding) and leaching of leaf material by physical means. This is accomplished by the interaction of wind, water, sediments, animals, and to a great extent macroinvertebrates (shredders) (Cummins, 1974). The physical leaching of soluble leaf material occurs very rapidly and can account for a large loss of material, with from 5-30% of the original leaf weight being lost in the first 24 hours in water depending on the type of leaves (Kaushik and Hynes, 1971; Klotz and Matson, 1978; Nykvist, 1962; Peterson and Cummins, 1974; Short and Ward, 1980).

Biological decomposition is brought about mostly through the actions of microorganisms (fungi and bacteria), especially fungi initially. Shredding by macroinvertebrates could also be considered biological decomposition. The process of microbial colonization of leaves is a very important one. These microbes not only directly attack and decompose the leaf material, but also increase the nutritional value of the leaf material, making it a more desirable food source for the macroinvertebrates. Shredders have been shown to exhibit

a preference for colonized rather than uncolonized leaves (Cummins and Klug, 1979; Petersen and Cummins, 1974). The microbial population can thus increase the action of shredders (physical decomposition) and the shredders can in turn increase the biological break down by increasing the surface area available to microbes.

Both physical and biological decomposition processes occur simultaneously throughout the decomposition cycle. Initially the physical processes dominate and over the long term the biological processes dominate. This shift from physical to biological processes has been shown as a break in the weight loss curve over time after an initial rapid weight loss during the first one or two days under aerobic conditions (Nykvist, 1962; Petersen and Cummins, 1974). These two break down processes can also occur before the leaves ever reach the stream (e.g., on the stream banks or on the tree itself). Decomposition on land is similar to decomposition in the water but much slower (Merritt and Lawson, 1978; Thomas, 1970). This preconditioning on land also affects the rate of decomposition in the stream.

Temperature can affect leaching (Nykvist, 1962), but Petersen and Cummins (1974) found that over the water temperature range of 4-17°C there was no significant difference in leaching rates (percent dry weight loss).

Perhaps as important as the release through leaching and decomposition to the amount of dissolved organic matter present in stream water, is the removal of these materials (e.g., by microbial uptake). It has been reported that in recirculating chambers containing stream water and sediment from a third order valley stream, 97% of the red alder leaf leachate added was removed

in 48 hours (Dahm, 1981). In a subsequent similar study, as yet unpublished (Personal Communication, Clifford Dahm OSU Fish and Wildlife Department), done with water and sediment from a first order mountain stream only about 50% of the added alder leachate was removed: in 48 hours. It appears that the type of stream environment, most likely the microbial population present, is important to the uptake rate. Also important is the initial level of dissolved organic enrichment (Dahm, 1981). Temperature was probably not a factor since both experiments were conducted at between 11-13°C.

Color

Color in water can be derived from the leaching of standing plant tissues (e.g., tree canopy and bark), the solution of break down products from organic detritus (e.g., leaves, twigs, bark, etc.), and leaching of soil organic matter (Christman and Ghassemi, 1966). As stated earlier, this input of organic material is important to the energy balance of small streams.

There are two categories of water color, apparent color and true color. Apparent color is caused by both dissolved and particulate material in the water, whereas true color is only that color which remains after the suspended particles have been removed (e.g., through filtering) (Am. Pub. Health Assoc., 1975). A generally accepted arbitrary cutoff for dissolved material is anything that will pass through a 0.45 μm filter. Some research suggests, however, that some of the substances causing true color in stream water are not really dissolved but colloidal in nature with a particle size of less than 0.01 μm (Black and Christman, 1963). Colloidal

organic carbon may be considered as a component of the total organic pool and it may take only a small amount of colloidal organic material to noticeably color water (Lock et al., 1977).

Questions still remain as to the make-up and size of the molecules involved with water color. It is generally agreed that color in water is caused by fulvic and humic acid-like substances and that these substances are very similar to the fulvic and humic acids extracted from soils and woody material (Christman and Ghassemi, 1966; Ghassemi and Christman, 1968; Larson, 1978; Research Committee on color problems, 1967). Some studies have shown that the majority of the color is due to compounds in the 700-10,000 molecular weight range (fulvic acid) (Ghassemi and Christman, 1968; Larson, 1978; Wilson, 1959; Research Committee on color problems). Others have shown water color to be derived from a significant amount of high molecular weight (greater than 50,000) humic acid material (Gjessing and Lee, 1967). Gjessing and Lee (1967) also state that low colored waters appear to derive most of their color from the low molecular weight fraction of organic matter, while moderate to highly colored waters tend to get their color from the high molecular weight substances. This seems to be consistent with the observation that humic acids are more highly colored than fulvic acids (Ghassemi and Christman, 1968; Larson, 1978). The chemical make-up of these substances is still largely unknown. Christman and Ghassemi (1966) provide strong evidence that the structure of color producing organic molecules is aromatic polyhydroxy, carboxylic acids with phenolic nuclei. Shapiro (1964) believed it to be aliphatic, polyhydroxy, carboxylic acids. Both agreed that a fairly constant small amount of nitrogen was present (1-2%).

Some other properties of color producing organic molecules are that they exhibit acidic properties and that the ambient pH affects coloration (Black and Christman, 1963; Christman and Ghassemi, 1966; Ghassemi and Christman, 1968). Organic molecules that produce color also react with iron increasing its solubility and this reaction is also affected by pH (Ghassemi and Christman, 1968; Shapiro, 1966).

Significance of Water Color

Water color can have both positive and negative implications for water quality depending on if domestic or environmental water quality is being considered. On the positive side, there is evidence that the organic matter that causes color may be beneficial through its role in primary production (Research Committee on color problems, 1967). It was also stated earlier that allochthonous organic material can be very important to the energy balance of streams. Thus, color may indicate an increase in stream quality with respect to aquatic life. If the color were to become too high it might indicate conditions detrimental to the aquatic system.

On the negative side, natural water color can be aesthetically undesirable in terms of domestic use. Even if colored water is harmless, most people are used to using relatively colorless water and consider colored water "dirty" or unfit to use. The United States Public Health Service (USPHS) set 15 platinum-cobalt units as the maximum for drinking water in 1961 (USPHS Drinking Water Standards, 1961). A numerical standard for color was not found for the state of Oregon. Oregon Water Quality Standards state:

- "k. Objectionable discoloration, scum, oily sleek, or floating solids, or coating of aquatic life with oil film shall not be allowed.
- 1. Aesthetic conditions offensive to the human senses of sight, taste, and smell or touch shall not be allowed." (Oregon Water Quality Standards, 1980).

There are other negative aspects of color that have to do with industrial use, interference with colorimetric analyses and interference with water treatment (Research Committee on color problems, 1967).

Another possible negative aspect of colored water with respect to domestic use is that the organic matter associated with color can interfere with chlorination and produce compounds that are possibly harmful to humans. In recent years, interest has been directed at halogenated organics, especially trihalomethanes (THM's), produced through chlorination of water containing dissolved organic matter (Beller et al., 1974; Rook, 1976; Stevens et al., 1976; Symmons et al., 1975).

THM's have been found to be widespread in finished drinking waters throughout the United States (Bellar et al., 1974; Symmons et al., 1975). Humic acids (especially the fulvic fraction) have been identified as precursors to THM formation and it has been shown that this formation is dependent on pH, precursor concentration, chlorine dose, and the point of chlorination (Rook, 1976; Stevens, et al., 1976; Veenstra and Schnoor, 1980). Rook (1976) states that fulvic acids, which color natural waters, form haloform compounds when they react with chlorine. In a subsequent paper Rook (1977) maintains that all colored waters that are chlorinated must contain at least some quantity of halogenated organic

compounds.

The precursors to THM production are likely a complex mixture of humic substances and low molecular weight compounds containing the acetyl group, with the humic substances dominating near neutral pH (Stevens et al., 1976). The above studies (Rook, 1976; Stevens et al., 1976; and Veenstra and Schnoor, 1980) all indicate that only a small amount of humic acid precursor (<1%) actually reacts with chlorine to form THM's. However, even under low chlorine dosages a measurable amount of haloforms can be formed (Rook, 1976).

The dominant halogenated organic compound generally found in treated drinking water is the trihalomethane, chloroform (Bellar et al., 1974; Symmons et al., 1975; Wilkins et al., 1978). Chloroform is a known carcinogen in rats (Reuber, M.D., 1979) and this has generated human health concerns where THM's may exist and be ingested with drinking water. Considerable study has been made of the possible relationship of chlorinating drinking water and cancer incidence (Carlo and Mettlin, 1980; Safe Drinking Water Committee (NAS), 1980; Tuthill and Moore, 1980; Wilkins et al., 1978).

The review done by the National Academy of Science (NAS) examined the known epidemiological work dealing with drinking water chlorination and cancer. Two different groups of studies were investigated, those where the exposure to THM's was estimated indirectly (10 studies) and those where water quality (THM's) was measured (3 studies). Less confidence was put into information derived from the first group since the degree of exposure was crudely estimated.

The investigation revealed a generally positive association between drinking water chlorination and

cancer rates for one or more body sites (bladder, large intestine, rectum, and stomach were the most common). However, the authors point out that while some of the important demographic variables were included, other complicating factors (e.g., cigarette smoking, ethnic background, and past exposure) were left out of some or all of these studies. The authors, therefore, stressed that these associations should be considered tentative and caution should be used in their interpretation, and that further study is needed.

Wilkins et al., (1978) also conducted an extensive review of the epidemiological research, including many of the same studies as the NAS review. They concluded that even though there have been no strong relationships shown between drinking water and cancer and there are many problems with the studies that have been done, there have been some consistencies in these studies that indicate a need for further work. They also indicate that repeated findings of cancer of the large intestine, rectum and bladder could give some direction to this further research.

Other recent studies have concluded that there is no significant statistical association between chlorination by-products in drinking water and cancer rates or incidence (Carlo and Mettlin, 1980; Tuthill and Moore, 1980).

On the whole, these studies point out the difficulties involved in defining relationships between environmental factors and health problems, especially for substances in small quantities (e.g. THM's). There appears to be evidence for and against a causal relationship between chlorination by-products in drinking water and cancer, so the question remains to be clearly resolved.

II. STUDY AREA

Location and Description

Watershed

The Seaside Municipal watershed is located approximately 8-10 km southeast of Seaside, Oregon on the South Fork of the Necanicum River (Lat. $45^{\circ} 53' 35''$, Long. $123^{\circ} 49' 55''$, T. 5N., R. 9W., Clatsop County, Oregon). The city's diversion dam is a few kilometers upstream of the confluence of the South Fork and the main Necanicum at state highway 26. There is a stream gauge located 45-60 m above the diversion (Water Resources Dept., OR., 1981). The watershed has an orientation just west of north on a line from the mouth straight back to the highest point. The approximate basin relief is 4m/30m or 140m/km. This was estimated along the same line as the orientation, with distances measured on an enlargement of a 15 minute USGS quadrangle map.

The total area of the watershed is 20.2 km^2 (Water Resources Dept., OR., 1981), with three major sub-basins designated the west tributary, middle tributary, and east tributary for identification purposes (Figure 1). The areas are 9.8 km^2 , 5.5 km^2 , and 3.0 km^2 respectively, and represent 54%, 30%, and 16% of the total sub-basin area respectively. The drainage network was drawn on the 15 minute quadrangle using the crenulation method (Marston, 1978). This method produced a fifth order watershed with 60 km of stream channel (34.7 km first order, 11.6 km second order, 8.9 km third order, 3.7 km fourth order, and 0.8 km fifth order) and a drainage density of 7.7 km^{-1} (Linsley et al., 1975). These

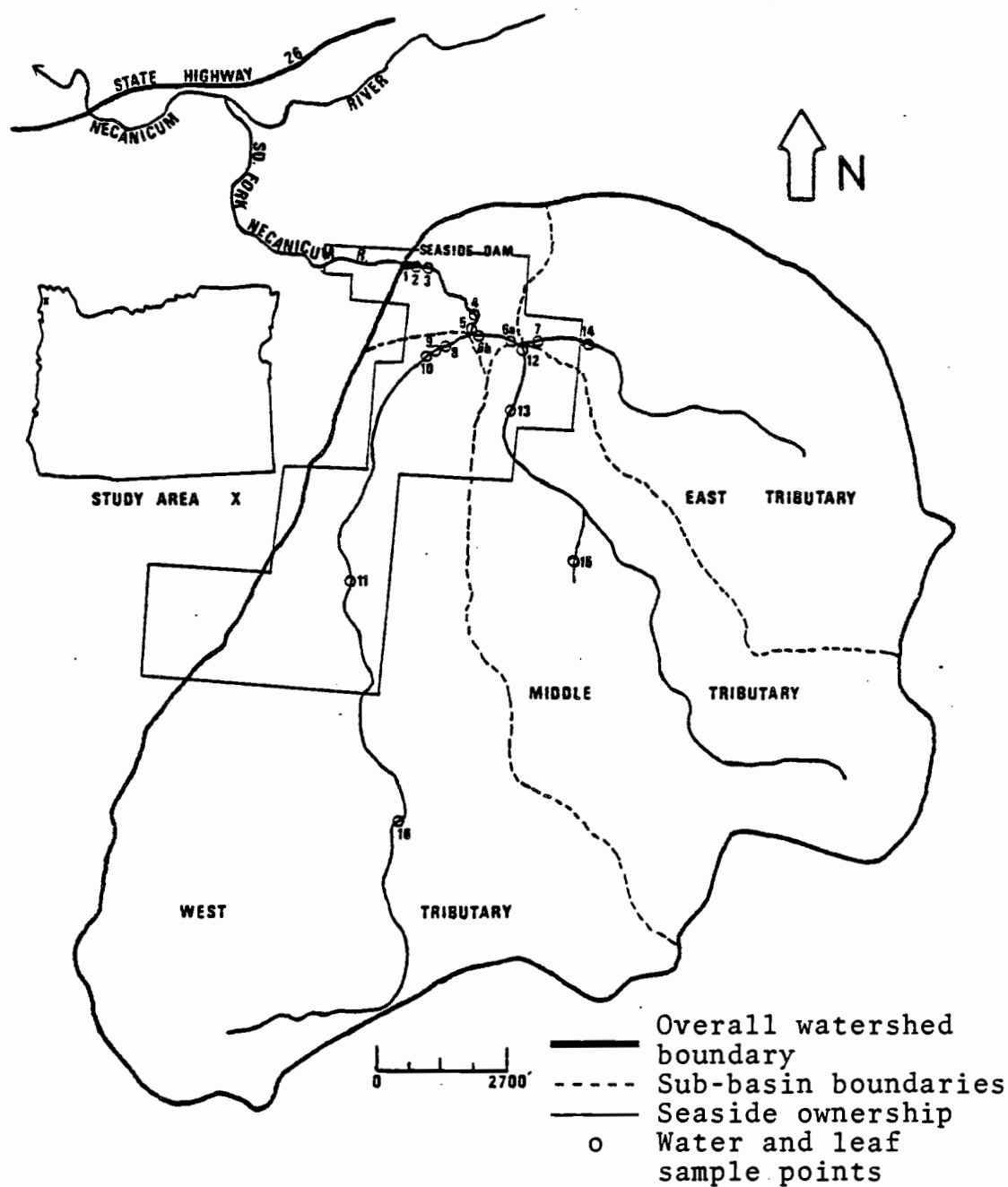


FIGURE 1. Seaside Municipal Watershed, South Fork Necanicum River, with approximate locations of water and leaf sample points and approximate Seaside ownership, Clatsop Co., Oregon

are probably underestimates because of the small map scale used.

The city of Seaside owns approximately 405 ha (one fifth) in the lower portion of the watershed (Figure 1). The rest of the watershed is owned by Crown Zellerbach and Boise Cascade. In 1980 the city of Seaside and Crown Zellerbach began developing a management plan for the watershed. The plan was never completed but a map was developed (primarily from aerial photographs) delineating major vegetation zones, with descriptions of the dominant overstory vegetation and estimated dates of previous logging activity (Appendix 1). This map was redrawn showing only the information on the riparian management zone (Figure 2). The area of this management zone is approximately 85 ha, and is dominated by red alder (Alnus rubra Bong.).

Riparian Zone

A separate ground check was made for this study in July, 1982. Thirteen plots were measured for stem density, basal area, average diameter at breast height (DBH), tree height and age (Dilworth, 1977). A 20 basal area factor (BAF) prism, cloth measuring tape, clinometer and increment borer were used. The thirteen plots were spread throughout the watershed but concentrated in the lower portion (Figure 3). General plot areas were located at points along the stream channel that were previously used for water and leaf sampling, while the remainder were selected using aerial photographs. At each location either an identifiable rock was thrown over the shoulder or 5-10 paces were taken with head down into the stand, to establish the plot center.

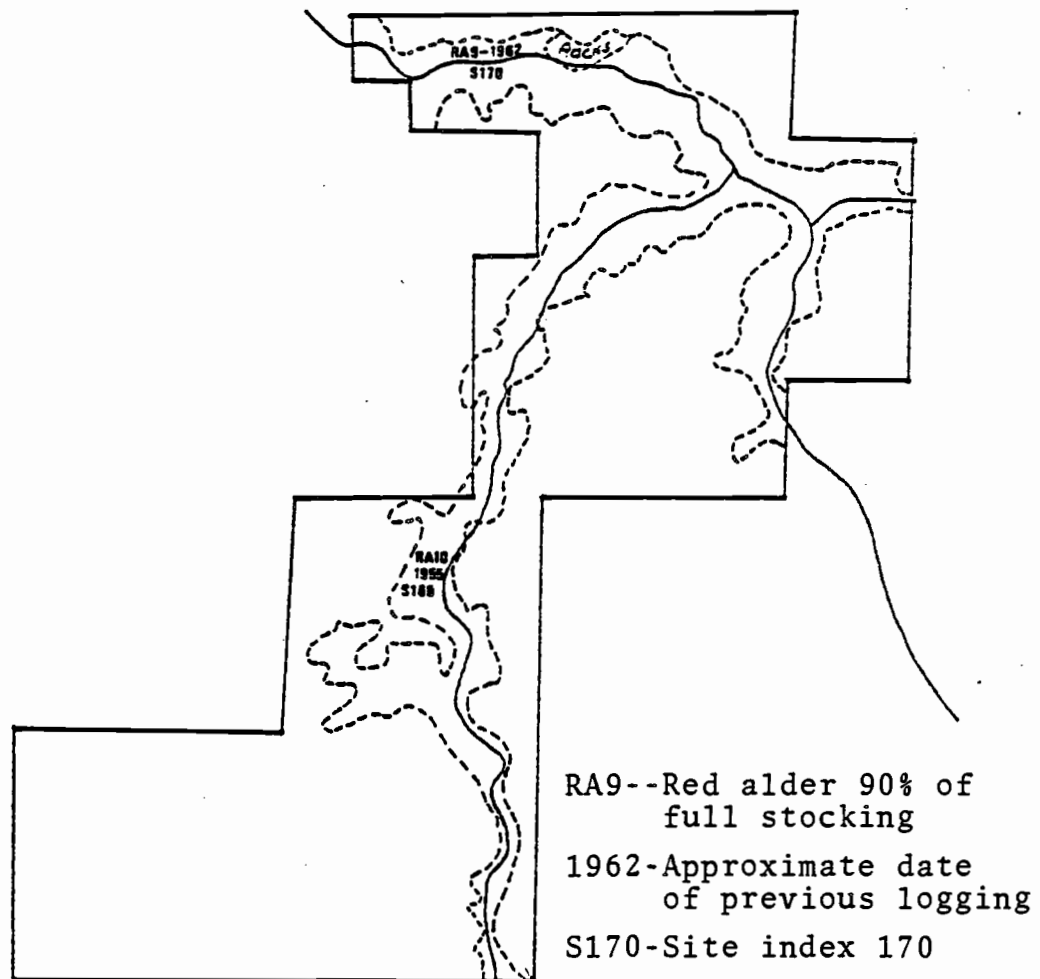


FIGURE 2. Overstory vegetation and dates of previous logging activity in the riparian zone Seaside Municipal Watershed, Clatsop Co., Oregon

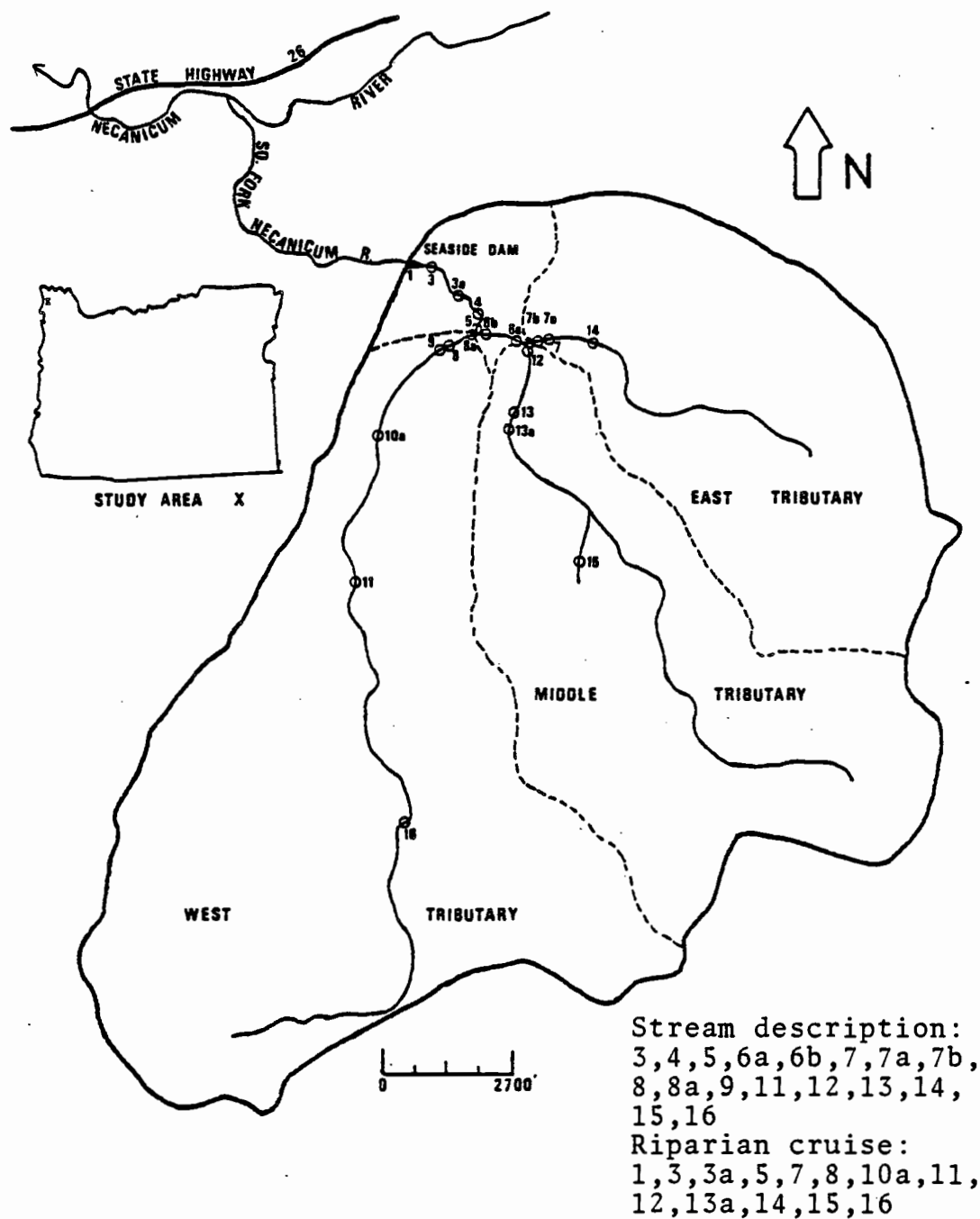


FIGURE 3. Approximate locations of stream description measurement sites and riparian zone spot cruise sample points, Seaside Municipal Watershed, Clatsop Co., Oregon

Within each plot representative "in" trees were measured for DBH and one or two were chosen for height and age determinations. In many instances the tops of the trees could not be distinguished from one another from 15-30 m away (the distance used for measuring height), but when necessary reasonable height estimates were expected because the trees were fairly uniform in height within an area. Along with these standard cruise measurements, an angle canopy densiometer (ACD) was used to get a quantitative measure of canopy cover over the stream. The ACD was used according to a method used for work in the H.J. Andrews Experimental Forest (Rick Summers, personal communication). The results of this spot cruise (Appendix 2) are similar to the results shown on the management map when it is used in conjunction with yield tables. These results are summarized in Table 1 and give a good general description of the watershed riparian zones.

According to the management map the riparian alder is quite uniform, being approximately 20 years old with 90% or normal stocking in the lower watershed and approximately 27 years old with 100% of normal stocking along the upper portion of the west tributary. So, normal stocking was assumed for the whole riparian zone for purposes of using the yield tables. When the estimates from the yield tables (based on the management map) and the field spot cruise are taken together, the average stem density in this zone is about 741 trees/ha, average basal area between 23 and 34 m²/ha, average DBH is approximately 23 cm, and the average tree height is between 18-24 m. The canopy cover is quite uniform and high throughout the riparian zone, ranging from 86% to 97% and averaging 94%. From the management map, the field cruise, the aerial

TABLE 1. Estimates of riparian red alder stand characteristics, July 1982, Seaside Municipal Watershed, Clatsop Co., Oregon

Source	Average Stem Density (trees/ha)	Average DBH (cm)	Average Normal Basal Area (m ² /ha)	Average Tree Height (m)	Average Age (years)
Empirical ¹ Yield Tables	509	23.6	22.3	-----	20-27 ³
Normal ² Yield Tables	647	23.9	28.4	21-24	20-27 ³
Field Spot Cruise	889	21.1	32.7	18-24	23
----- Field Cruise by Sub-basin -----					
South Fork Necanicum	808	21.3	28.7	23	26
West Tributary	956	22.3	37.9	18-24	22
Middle ⁴ Tributary	1400	7.3	34.4	14-18	22
East Tributary	393	27.4	27.5	21	24

¹From: Empirical yield tables for predominantly alder stands in western Washington (Chambers, 1974)

²From: Normal yield tables for red alder (Worthington et al., 1960)

³Ages taken from the Crown Zellerbach-City of Seaside management map (Appendix 1)

⁴Plot 15 not included in the averages due to the much younger age and smaller trees

photographs, and visual observations on the ground the riparian alder stand was considered to be uniform throughout the watershed.

A general survey of the understory vegetation revealed the following major plants at the various plots: salmon berry, sword fern, devils club, Indian plum, Western black current, Alaskan blueberry, colts foot, willow, stinging nettle, lady fern, and fairy bells. There was also some western hemlock and sitka spruce present in the understory. The understory vegetation was generally fairly low lying under the heavy shading of the alder canopy.

Soils

Based on soils maps on file at the Seaside Soil Conservation Service office, the lower one third to one half of the watershed is made up of four major soil types: Klotchie silt loam (3-30% slopes), Klotchie-Necanicum complex (30-60% slopes), Skipanon silt loam (3-30% slopes) and Skipanon silt loam (30-60% slopes). These four soils also dominate the riparian zone in the upper watershed. The Necanicum-Ascar complex (30-60% slopes), Ascar-Rockoutcrop complex (30-60% slopes) and Tropaquepts and Humitropepts (0-20% slopes) are also present but not as prevalent. Tropaquepts and Humitropepts make up the stream zone along the main fork from below the east tributary to the diversion.

The upper watershed is dominated by the Skipanon silt loams (3-30% and 30-60% slopes), Klotchie silt loam (3-30% slopes) and Klotchie-Necanicum complex (30-60% slopes), as well as, the Laderly-Rockoutcrop complex (60-90% slopes). Also present are the

Necanicum-Ascar complex (30-60% slopes), Murnen bouldery complex, Murnen-Cater bouldery complex (30-60% slopes) and in the uppermost reaches, the Humbre silt loams (3-30% and 30-60% slopes), Klickitat-Herrington complex (30-60% slopes) and Rinearson silt loam (30-60% slopes). Small amounts of Murnen silt loam (3-30% slopes), Templeton-Ecola complex (30-60% slopes), and Fluvaquents and Tropofluvents (nearly level) are also found.

Based on Soil Interpretations Records from the Seaside Soil Conservation Service office, the major soils in the watershed (Skipanon silt loams, Klootchie silt loam, and Klootchie-Necanicum complex) are all deep (1-1.3 m), well drained soils formed in colluvium (Klootchie and Necanicum from basalt and Skipanon from mixed materials over marine sediments). The organic matter content of these soils is less than 12%.

Stream Description

Measurements of channel gradient, side slope, channel width and depth, and velocity (flow) were made to describe the stream. Observations were also made on the composition of the streambed and banks. These measurements and observations were made at 17 locations (Figure 3), most of which (12) correspond to leaf fall or stream sampling stations. The rest were chosen as velocity determination sites or to round out the characterization of a stream reach. An approximate 30 m section was paced off at each location for observing the streambed, banks, and channel gradient. The width/depth cross sections were usually taken in the center of this 30 m section.

The width and depth measurements were accomplished

using a 15 m plastic measuring tape, metal pins and a surveying stadia rod. The flow during measurement (May 10, 1982) was approximately $1.0 \text{ m}^3/\text{s}$, which was similar to mid-October 1981 flows. The bank full waterline was located by observing piles of debris, topographic breaks, and vegetation. A metal pin held the end of the tape to the ground at this point and a clinometer was used to find the point on the opposite bank (as much as possible perpendicular with the flow) level with this first point. This was done by sighting directly across to the first at ground level or by sighting to eye level on the stadia rod being held at that point. A second pin was placed at this point and the tape stretched tight and clamped to the pin. Depth measurements were taken every two feet or less where more detailed topographic information was included (e.g., sharp breaks), reading the stadia rod where it intersected the tape. For measurements in the middle of the tape, the tape was lifted slightly while reading to bring it more in line with the pins and reduce sag. The edge of water was recorded on each side so that the low flow dimensions could be calculated.

The channel gradient and side slopes were measured with the clinometer. The channel gradient was observed over the 30 m measurement section on one side of the stream at the waters edge. The side slopes were measured from the approximate center of the stream to the pins on the upper banks. Velocity (flow) measurements were made near the mouths of the three main tributaries using a stop watch and floating sticks. The sticks were tossed into the stream and allowed to float 3-5 m before being timed for 15 m (within the 30 m section). There were four or five

runs made, with different sticks, on each tributary. The velocities obtained were multiplied by 0.83 to account for faster flow at the surface and get an estimate of the average velocity (Linsley et al., 1975).

From these measurements, the weighted mean stream width and depth for the watershed, at a flow of $1.0 \text{ m}^3/2$, were estimated at 2.6 m and 6.1 cm respectively. The weights used were the lengths of stream channels greater than or equal to third order (Linsley et al., 1975) and the lengths of stream channels less than third order. There was only one stream measured that was of first or second order, Station 15, so it was assumed to be typical and used as the "average" first and second order stream. A summary of the dimensions, velocities, and flow estimates for the three main tributaries (near their mouths) is shown in Table 2. It is interesting to note that the percentage of flow that each tributary contributes to the main fork is very similar to the percentage of the total sub-basin area drained by each tributary; 54%, 30% and 16% (page 170).

The channel gradient ranged from 0.25% at the diversion to approximately 2-2.5% for the lower portions of the three tributaries. The gradient in the upper reaches was probably greater than 3% (3.5% and 7.5% for Stations 15 and 16 respectively). The overall gradient, taken from the USGS 15 minute quadrangle, for the stream beginning at the highest elevation (middle tributary) was approximately 7%. The side slopes generally ranged from 10-25% with a few greater than 40%.

The stream itself follows a pool-riffle-pool sequence. For the sections measured, pools and riffles covered about the same area, 35% each. Runs accounted for 20% and the rest was in cascades (upper sites, about

TABLE 2. Stream dimensions, velocity, and flow at selected sample points on the Seaside Municipal Watershed, Clatsop Co, Oregon, May 10, 1972⁴

Sampling Station #	Live Stream Width (m)	Average Depth (cm)	Velocity (m/s)	Flow (m ³ /s)	Flow ¹ (%)
3	14	11.0			
4	9.0	29.0			
5	8.5	25.3			
6a	5.5	44.8			
6b ²	8.5	9.8			
7	3.0	4.9			
7a	6.4	21.0			
7b	3.6	7.0	0.29	0.08	10
8	9.1	10.4			
8a	5.8	13.4	0.53	0.4	52
9 ³	12	7.3			
11	4.9	14.6			
12	6.1	15.2	0.30	0.3	38
13	6.7	28.4			
14	4.3	1.8			
15	1.8	6.4			
16	7.0	11.6			

¹Estimated tributary flow as % of total tributary flow

²Site used before the traps were washed away in October

³There were four sub-channels flowing within the main channel which gave a combined live stream width of 12m

⁴Stream flow at the diversion was approximately 1.0 m³/s

7%) or backwaters (3-4%). The bottom material tended to be mostly gravels (60% or more) near the city's diversion with the rocks fairly rounded, but equally split between gravels and cobbles up to Station 5. Material greater than baseball size was considered cobble and less than baseball size was considered gravel (Personal communication Charles Dewberry OSU Stream Team). Cobbles dominated the rest of the main fork and the three tributaries, except in pools where the distribution evened out or tended toward gravel. There also appeared to be a trend towards angular rocks above Station 5. There was some bedrock showing on the stream bottom but it was a small percentage (about 3%). The stream bottom material was generally coarse throughout the watershed, tending toward the cobble end of the scale.

The upper banks of the stream looked very similar throughout the watershed. They generally appeared quite stable and well vegetated with a lot of roots visible and a fair amount of organic debris incorporated in some areas. There was some undercutting of the banks observed but it was generally light to moderate and, if present, usually occupied less than half of the 30 m section on one bank. The stream channel itself contained very little large organic debris. The large debris that was seen appeared to come primarily from failed log bridges.

An observational inventory was also conducted on off-channel pools and backwaters on October 20, 1981, when flow was $0.9 \text{ m}^3/\text{s}$. This was done by walking the channels of the main fork, the lower portions of the three major tributaries and some of the upper west tributary. Rough visual estimates of size (length/width or diameter and depth) were made and a tally kept

of the number pools encountered.

There were about 110 pools counted, most of them small and shallow. This figure is approximate because some of the pools (a small number) included in the tally were in-channel pools and the distinction between these and the off-channel pools was not always made. In addition, the entire length of stream channels within the study area was not covered. A rough estimate of the volume of an "average" pool is 0.6 m^3 . The total volume of all pools inventoried was 66 m^3 . These are most likely overestimations for the pools observed since many pools were considered rectangular but were actually elliptical and the widths/lengths were normally estimated near the longest and widest points. It was also difficult to judge the average depth visually so it tended to be nearer the maximum depth.

More detailed measurements were made with a meter stick on the pools that were sampled for water quality. A summary of the physical characteristics of these pools is shown in Table 3.

TABLE 3. Dimensions of sampled off-channel and side of channel pools, Seaside Municipal Watershed, Clatsop Co., Oregon

Pool	Length (m)	Width (m)	Diameter (m)	Average Depth (cm)	Volume (m ³)
9Pa	4 ¹	2 ¹	---	30	2.5
9Pb	---	---	1.5	12.8	0.2
13Pa	2.4	1.1	---	35.4	0.9
13Pb	1.8	1.5	---	17.4	0.5
14Pa	3.6	1.5	---	12.8	0.7
14Pb	3.4	0.9	---	7.9	0.2

¹This was a larger pool surrounded by vegetation and debris. Visual estimates of the dimensions were made here.

III. METHODS

Field Study

Leaf Fall

To quantify the leaf material falling directly into the stream, a network of twenty-three litter traps (0.25 m^2) was established at sixteen stations (Figure 1). This trap size was selected because of ease of transport and handling of traps and leaf samples. The traps had wooden sides 10 cm high, with window screening tacked to the bottom and supported by two wood strips 2.5 cm wide.

Twelve traps had short wooden legs that held the traps about 20 cm off the ground, allowing free drainage. These traps were placed on the upper banks, generally within 2-3 m of the live stream at low flow. The other eleven traps had holes in the bottom of the "legs", which were flush with the top and bottom of the traps, so that the traps could be set on one m poles driven into the bed of the live stream.

On September 2, 1981, the litter traps were installed and leaves were collected from the ground at each station using a 0.25 m^2 open wood frame. The frame was thrown randomly to the ground and any leaves lying within it were collected and placed in a labeled plastic bag. Highly decomposed leaves were assumed to be from the previous year and were not included. These samples served as an estimate of the cumulative leaf fall to that date. The traps were cleaned once a week, with a few exceptions, between September 2 and December 2, 1981 and the leaves placed in clean, dry, labeled plastic bags. The objective was to collect leaves

from the beginning of autumn leaf fall until leaf fall was essentially complete. Normally the leaf samples were returned to the laboratory the same day as collection and refrigerated in a cold room at 2-4°C until they could be dried.

There were a few weeks when some of the traps were not emptied due to inaccessibility at high water. Also, no leaves were collected on October 6 because most of the traps had been washed away by high water sometime between September 30 and October 6. The few traps that were found the next week had either been washed clean or tipped over and floated downstream.

On October 12 the missing traps were replaced with 0.25 m² pieces of heavy plastic. These new leaf collectors (traps) were placed on the banks as near as possible to the site of the old traps. At this time the 0.25 m² open wood frame was used, as described earlier, to collect samples representing leaf fall during the two weeks traps were absent. Two instream traps remained intact, 12S and 13S, and were the only instream traps used from October 12 on. The few other wooden traps remaining at various locations were redistributed so that each tributary had at least two of these traps. This situation remained until November 18 when the instream trap at Station 12 was discovered missing due to another high water event, leaving only a plastic trap at that site.

The initial leaf collection, using the open frame and the first three weekly collections, from the traps, were kept in the cold room until the beginning of October. All other samples were generally put into the oven and dried within a few days of being collected. Before being dried, an attempt was made to remove the non-alder leaf material from the samples, sometimes

using a screen to pick over the samples. There was normally little else besides alder leaves present in the samples, but needles, twigs, and other debris were occasionally found, especially in those samples taken from the ground. The leaves were placed into pre-weighed trays and dried at approximately 50°C for approximately 48 hours. Upon removal from the oven leaves were cooled for a few minutes and then weighed on a Mettler p1210 balance to the nearest 0.01 g.

Leaf collection sample sites were located so that leaf fall quantity and distribution could be observed, while the criteria for selecting water sampling sites were somewhat more specific. The objectives of the water sampling were to follow through time the quality of the water: 1) just above the city of Seaside's diversion, 2) in areas upstream with potential influence upon the water quality at the diversion (possible source areas), and 3) in in-channel and off-channel pools where leaf collection and slow-moving or still water occurred together.

The general area of each station was primarily chosen according to the water sampling needs described above. Within a general area, the location of an in-stream leaf trap was influenced by the nature of the streambed, because a relatively soft substrate (e.g., a gravelly deposit) was necessary to firmly set the metal legs. Most instream traps were set near the center of the stream. Bank leaf traps were set near off-channel pools or close to instream traps when pools were not present. In the absence of both, bank traps were placed anywhere close to the stream in the area chosen. Sample point subscripts were used to distinguish between instream (S) and bank or land traps (L) and between live stream (S) and pool (P) water sampling.

When a trap location was being chosen, an effort was made to look only at the ground to avoid bias from the canopy appearance.

Station 1 was situated just above the diversion dam (Figure 1). This was the lowest station and represented the water being diverted by the city of Seaside. Stations 2, 5, 9, 13, and 14 included instream and off-channel pools. Station 2 was just upstream from Station 1 at an instream pool. Station 5 was located at the confluence of the main fork and the west tributary to observe the influence of the latter on the main fork water quality. An instream pool was also located at Station 5. Station 9 was on the west tributary in a wide depositional area where a mid-channel bar separated the channel into two sub-channels at low flow. A third channel ran through the middle of the bar and created an isolated pool (9Pa) at low flows. This and a second pool (9Pb) in the right channel (looking downstream), off to the side, were sampled. At Station 13, on the middle tributary, a high water channel to the right of the main channel formed a series of pools at low flows. Two pools were sampled at this site, one in the high water channel (13Pb) and one next to the right bank in the main channel (13Pa). A large mid-channel bar (more of an island), well stocked with red alder, split the east tributary into two sub-channels at Station 14. The right channel dried up at low flow in 1981 and left a pool at the lower end (14Pb). To the left of the main channel a high water channel left a series of pools at low flows, one of which was sampled (14Pa). Stream samples were taken in the middle of the stream.

The rest of the station locations were chosen for reasons other than the presence of pools. Station

3 was located about 50 m above Station 2 to note any changes in water color occurring in the lower main fork. Station 4 was strictly an in-channel leaf collection station. Stations 7, 8, and 12 were the lowest stations on the east, west and middle tributaries respectively and represented the water entering the main fork. Station 7 was located a few hundred meters above the mouth of the east tributary. Stations 8 and 10 were located just below and above Station 9 to follow any changes in water quality due to that pooled section. Station 8 was also a few hundred meters above the mouth of the west tributary. Station 12 was located within 50 m of the mouth of the middle tributary. Station 11 was chosen as an intermediate site to follow the water quality along the west tributary. Stations 15 and 16 were chosen to represent the upper reaches of the middle and west tributaries respectively. It was discovered later that Station 15 was actually on a first order tributary to the middle tributary, but was still considered a good representative station for the upper watershed.

Water Sampling and Analysis

Water quality sampling normally took place on the same weekly schedule as the leaf sampling. Two grab samples were taken at each sampling station, a general water quality sample and a dissolved oxygen (DO) sample. No replicates were taken at the individual stations because of the number of samples involved and the time required for sampling and analysis. To be able to rely on a single grab sample, an assumption of complete mixing was made.

Water quality samples were taken with 500 ml

plastic, wide mouth bottles. The bottles were scrubbed with a mild soap and rinsed well with tap water. They were rinsed again with stream water just before sampling. The samples were taken at least 15 cm below the surface or about mid-depth in shallow water. The bottles were inverted and pushed, mouth first, by hand to the desired depth then rotated (mouth upstream) until upright and full. When shallow pools were sampled, care was taken to avoid stirring the bottom sediments. Pools at Stations 2 and 5 were sampled using a BOD bottle (described below) clamped to the end of a pole with a stopper connected to a string. The bottle was lowered as close to the bottom as possible, without touching, and the stopper pulled out. An initial two samples were transferred to a plastic bottle and a third was taken for dissolved oxygen analysis. All plastic bottles containing samples were immediately capped and placed in an ice chest for transport.

Dissolved oxygen samples were taken with 300 ml glass biochemical oxygen demand (BOD) bottles. The same technique was employed to fill these bottles as for the water quality samples. This method of filling the BOD bottles undoubtedly caused some turbulence within the bottles and some incorporation of the oxygen in the bottle into the sample may have occurred. However, the possible addition of a small amount of oxygen should not have made much difference because of the generally high (near saturation) DO levels. In small pools, where the DO was sometimes low, the bottles were filled more slowly to avoid stirring up the bottom and probably created less turbulence.

The BOD bottles were capped after being filled and were taken back to the vehicle where the samples

were fixed according to the Azide modification of the Winkler method, using Hach dry chemical powder pillows (Hach Co., 1978). There were a few sampling dates when the samples were fixed before being capped, but it was determined that there was no great difference between the two methods. Once the DO in a sample is fixed in this manner it can be stored up to 8 hours before being titrated, if kept out of direct sunlight. The samples were therefore covered and titrated as soon as they were back in the laboratory, normally within 8 hours.

The samples were titrated with 0.25 normal phenylarsine oxide (PAO) titrant instead of the conventional sodium thiosulfate, which is less stable. This modification, as well as the use of dry chemical powder pillows, has been accepted by the U.S. Environmental Protection Agency for DO determinations, as cited in the Hach Chemical Company's, "News and Notes for the analyst", Kroner, R.C. et al., 1964).

Titration were carried out with 100 ml of sample so that two titrations could be done on a sample if necessary. The titrations were made using a buret (25 or 50 ml) fixed to a ring stand, which was expected to be more accurate than the dropper system supplied with the Hach DR-EL/2 kit. At first titrations were done from a straw color to colorless endpoint, as suggested in the Hach manual. However, by adding a few drops of starch indicator, to get a deep blue color, the colorless endpoint was easier to spot. Starch was, therefore, used for the rest of the titrations. The volume of titrant used to react to the endpoint was measured to the nearest 0.05 ml.

The water quality samples were stored on ice in the field and transferred to the cold room upon

returning to the laboratory. These samples were then analyzed for color, turbidity, conductivity, and nitrate-nitrogen, normally within 48 hours of collection. The Hach DR-EL/2 field laboratory was used for the analysis of color and nitrate-nitrogen. Electrical conductivity was measured with a Hach portable conductivity meter, model 16300. These instruments were used because they were readily available and simple to operate. In addition, comparisons of results provided by the Hach DR-EL/2 kit with those of standard laboratory methods indicate that the Hach kit provides adequate measures for general water quality surveys (Boyd, 1977). Both parameters that were measured with the Hach kit were evaluated in a general way for evidence of gross differences that might occur during the sampling period.

Color measurements were made according to the Hach DR-EL/2 manual procedures (Hach Co., 1978) which were adapted from the platinum-cobalt technique (Am. Pub. Health Assoc., 1975). Initial color analyses were of apparent color, however it was decided, near the end of the fall sampling period, that true color should be measured to evaluate the color caused by "dissolved" materials. Therefore, the samples taken throughout the fall were reanalyzed for true color by filtering with a millipore filter apparatus and Whatman GF/C glass microfibre filters (effective pore size of $1.2\ \mu\text{m}$). The turbidity of some samples was measured after filtering and it was found that the GF/C filters brought the turbidity below 1.0 NTU with some variability between 0-1 NTU. It was therefore decided, that only samples that had been measured with turbidities greater than 1.0 NTU would be filtered for true color analysis. Measurements were

assumed to be of true color for samples that had a turbidity of less than 1.0 NTU, despite the absence of filtering. This functional definition of true color coincides with the fact that the city of Seaside only uses water from the river if the turbidity is less than 1.0 NTU.

There were some samples with turbidities less than 1.0 NTU that were also rechecked for true color because there was some question as to the reliability of the initial color measurements. During the measurements on the first six weekly groups of samples, the readings on the Hach kit spectrophotometer were somewhat erratic due to needle drift. Readings were taken two or three seconds after inserting the sample as a means of consistently reading the instrument. After the first six weeks, however, the machine appeared to settle down and the readings became quite steady. It is possible that the initial drift problem was caused by insufficient warm-up time for the machine. Storage of these samples for almost two months should have had little effect on the color, although in the presence of oxygen color may not be quite as stable as in a nitrogen atmosphere (Black and Christman, 1963).

Turbidity measurements were made with a Hach model 2100A laboratory turbidimeter. The samples were inverted a few times and 25 ml were carefully poured into a glass cuvette to minimize the formation of air bubbles. The cuvette was wiped to remove any water from the exterior and placed into the turbidimeter. The precision of the measurements depended on the turbidity. If the turbidity was in the range of 0-1 NTU, it was read to the nearest 0.01 NTU. If the turbidity was in the range of 1-10 NTU, it was read to the nearest 0.1 NTU.

The electrical conductivity was measured in the laboratory with a Hach model 16300 conductivity meter. Sample aliquots of 40-60 ml were warmed following removal from the cold storage, so that all conductivity measurements were made with the sample between 20°C and 26°C. Most measurements were made at 22°C-24°C, only a few fell outside this range.

Nitrate-nitrogen was measured on the Hach DR-EL/2 spectrophotometer. The cadmium reduction method using NitraVer V powder pillow reagent was followed as outlined in the DR-EL/2 methods manual (Hach Co., 1978). Nitrate-nitrogen was measured to the nearest 0.25 mg/l.

A calibrated Corning 130 pH meter was used to measure the pH of a 40-60 ml aliquot of the water samples, warmed to 24°C (room temperature). The meter was read to the nearest 0.01 pH units after the reading had stabilized. This value was then rounded to the nearest 0.1 pH unit.

Stream temperatures were taken at the time water samples were taken. A field thermometer with a Fahrenheit scale was used. The thermometer bulb was held approximately 15 cm underwater for about 30 seconds, read, then placed underwater for a short time and read again. If there was no change between the two readings, the temperature was recorded. If there was a difference, the thermometer was submerged again until there was no change.

Following the weekly sampling during the autumn leaf fall period, water sampling and analyses were continued on a monthly basis, at six stations, from January 1982 through March 1982. The six stations sampled were; 1S, 5S, 6S, 7S, 8S, and 12S. The March sample was actually taken at the beginning of April.

Laboratory Studies

To compliment the field work, a set of leaching experiments was conducted to investigate the potential for alder leaves to change water color. These experiments were carried out for the most part to observe the physical leaching process and, therefore, were on a shorter time frame (hours and days) than the field sampling (weekly). This discrepancy is not as great as it would seem since the color observed on a given day in the field would be a combination of the color derived from the initial physical leaching of newly fallen leaves and the continued decomposition of leaves already present in the stream, as well as from other sources. There were a couple of longer leachings (2 and 4 weeks) conducted also. All leachings were carried out with leaves picked from trees and filtered stream water at a constant temperature.

Leaves for these experiments were obtained from trees bordering Oak Creek, at the edge of MacDonald Forest near the Oak Creek Experiment Station, just northwest of Corvallis, Oregon. The leaves were picked by hand and with the use of a pruning pole on November 10, 15, and 16, 1981 and placed into polyethylene bags. They were picked from the trees, rather than off the ground, to simulate relatively unweathered leaves falling directly into a stream. This also helped to keep the leaf quality more uniform. Leaves still on the tree can vary in quality but this variation would be expected to be smaller than that for leaves that have been on the ground for an unknown period of time. Leaves were collected from three or four different trees. The leaves were dried at approximately 50°C for approximately 48 hours and then placed into

new polyethylene bags for storage in the cold room. Drying the leaves at this temperature did not seem likely to interfere with the integrity of organic compounds within the leaf but it is possible some changes may have occurred. Drying leaves at this temperature did probably killed the microbial population present but would probably not have affected spores on and in the leaf material. This then produced a bias as to the type of microbial population present in the long term leachings.

The water used for the laboratory leachings was taken from the South Fork of the Necanum River. Approximately 150 liters were collected, from just below the diversion in early March 1982 with 20 liter hard plastic cans, for use in the still water leaching experiments. Another 50-60 liters were collected from just above the diversion in early August 1982 for use in the running water leaching experiment. The collection cans used were rinsed with tap water and rinsed again with stream water before being filled. The cans were stored in the cold room. Water was removed as needed and filtered using a millipore filter apparatus, vacuum pump, and Whatman GF/F grade glass microfibre filters (effective pore size of 0.7 μm). The Whatman GF/F filters were recommended by members of the O.S.U. Fish and Wildlife Department Stream Team (personal communication) who use them in studies of dissolved organic carbon. Filtering was done to remove any apparent color from the water and to remove as much of the microbial population as possible since physical leaching was the main interest. The leaves were weighed and placed in jars using metal tweezers and were handled as little as possible by hand to help minimize microbial contamination. The leachings were not

poisoned because it was assumed that the microbial populations present would be low enough that they would not contribute greatly to decomposition during the first two days of leaching. This seemed to hold true as is stated on page 45. However, the water used for the long term (2 and 4 week) leachings should have probably been left unfiltered since biological decomposition was of interest as well in these leachings. Filtering effectively removed the aquatic microorganisms but terrestrial microorganisms from the air, from handling the leaves and from any spores already present on the leaves would have remained. This means that another bias was created in the type of microbial population present.

All leachings were conducted in the dark at a constant water temperature of 10°C, which was in the middle of the temperature range found (6.7-13.8°C or 44-57°F) in the South Fork of the Necanicum River during the fall months, 1981. In this temperature range, the leaching of soluble substances from leaves has been shown to be independent of temperature (Petersen and Cummins, 1974; Barnes, Ovink and Cummins, 1978). Nykvist (1962) did show temperature to be important to leaching, but this work was done at much higher temperatures (0-80°C) and the observed changes in leaching were small. In the range of 0-10°C the change in leaching loss, as a percentage of dry weight, was between 0.1 and 0.2% per degree. Therefore, leaching was assumed to be independent of temperature in the range encountered and used.

When the leachings were to be started, leaves were taken from the plastic storage bags and weighed on a Mettler H16 balance to the nearest milligram. They were then placed into labeled jars and stored at 10°C until they could be filled with filtered stream water.

The leaves were handled with metal tweezers, as much as possible, to reduce microbial contamination. However, sometimes large leaves had to be bent by hand and curled slightly to get them into the jars. For leaching requiring 450 ml of water, 500 ml wide mouth glass jars were used, with 2 l jars used when a one liter volume was needed. The jars were filled using volumetric flasks, 200 and 250 ml flasks for the 450 ml volumes and one liter flasks for the one liter volumes. The beginning of each leaching period was recorded as being near the middle of the time it took to fill all the replicates of a given leaching situation.

A brief preliminary leaching experiment showed some variability in color generation between individual leaves. Five replicates were, therefore, run for each leaching time and mass loading to account for this variability. A smaller number could have probably been used if the leaves were crumbled and mixed, but the intention was to evaluate the leaves whole as they would be after naturally falling off the tree. There were times, however, that pieces of leaves had to be used to get the correct leaf mass. A blank of filtered stream water without leaves was also included for most leaching situations.

When the jars were filled the leaves had a tendency to float, with some leaves partially above the surface. To keep leaching conditions more uniform an attempt was made to keep all leaves completely submerged using glass stirring rods to pin some portion of the leaves to the bottom of the jar. This worked well, although occasionally petioles or a small portion of leaf material would rise above the surface and have to be forced back under. The procedure did not work quite as well in the large jars (one liter) since

the rods did not reach from the top to bottom. Usually the leaves stayed submerged without assistance by the end of the first day.

The first 2 laboratory experiments involved leaching alder leaves in still water. This type of leaching would be analogous to off-channel pools, impoundments, and to some extent backwater areas.

Time Series Leaching (Still Water)

One experiment was conducted to evaluate the changes in water color for different leaching time lengths, with a constant leaf mass (1.0 g) and water volume (450 ml). There were 9 leaching times: 2, 6, 12, 24, 48, 72, 96, 336, and 648 hours. The last 2 being 2 and almost 4 weeks respectively. The leaves for this leaching experiment were taken from those collected on November 10, 1981 near Corvallis.

The first 7 leaching times were expected to show color generation primarily due to the physical leaching process. These jars were immediately covered to help minimize microbial contamination from the air. The caps were removed periodically to push the leaves under, and the contents of each jar were gently swirled at least once a day usually with the caps in place. This swirling was done to mix up any dissolved material in the water and insure that the whole water column was included in the leaching process. After swirling, the leaves were checked and resubmerged if necessary.

The last 2 leaching times represented the longest times leaves might be expected to remain in the stream, in substantial numbers, before being flushed out by a storm. These long term leachings were conducted to observe any additional changes in color and water

quality due to microbial decay. For this reason they were left uncovered and stirred vigorously (creating many bubbles), normally twice a day, once in the morning and once in the evening, to keep them aerobic. The stirring rod was rinsed once with distilled water before the jars were stirred and the leaves were re-submerged if necessary after each stirring. The stirring caused some breakage of the leaves which could be considered analogous to mechanical breakage in a stream. Approximately once a week 20-25 ml of filtered stream water were added to each jar to replace water lost by evaporation and any small amount lost during stirring. The same amount of water was added to each jar.

Mass Loading Series Leaching (Still Water)

A second still water leaching experiment focused on changes in water color with differences in leaf loading. The leaves used for this series were taken from a composite of the leaves collected. The leaves were mixed because the November 10 leaves were expected to run out and mixing insured that the leaves for the whole series would be taken from the same batch. Leaching time was kept constant, at 48 hours, while the leaf mass to water volume ratio was varied.

Five leaf loadings were chosen to represent a range that would encompass loadings likely to be found in nature. The loadings used were: 0.25 g/l, 0.50 g/l, 1.00 g/l, 1.00 g/450 ml, and 2.50 g/450 ml. The last 2 leachings were equivalent to 2.22 g/l and 5.60 g/l respectively. Each leaf loading was leached in capped 500 ml or 2 l jars, and again the jars were swirled at least once a day.

The 48 hour leaching time was chosen because

physical leaching was of primary interest and the 48 hour time series leaching of 1.00 g/450 ml exhibited strong color, which indicated that the smallest mass loading (0.25 g/l) would probably show measurable color in that time. The time series leachings also showed that 48 hours was probably short enough to minimize microbial action. After the 96 hour leaching, the water was quite cloudy, but the 72 hour leaching remained clear. Therefore, it was assumed that little microbial growth had taken place in 72 hours and even less in 48 hours.

Analysis of the Leachates

At the end of the designated leaching time, the leaves and pieces that could be removed easily with tweezers, or by hand without touching the water, were taken out and placed in preweighed, labeled trays for drying. The leachates were then filtered through preweighed Whatman GF/F filters using the equipment described earlier. The filtering apparatus was rinsed with distilled water before a new sample was filtered. The filters and residue along with any pieces recovered were placed into the appropriate tray. Most samples only took one or two GF/F filters for complete filtering. Exceptions to this procedure occurred with the long term (2 and 4 week) leachings, where suspended particulate material present at the end of the leaching made filtering very difficult. With these samples the filtering was done in steps.

Where multiple filtering was necessary the leachate was first poured through the fine mesh screen of the millipore filter apparatus to remove as many large pieces as possible. These were placed in the appropriate

drying trays. Next, the filtrates were passed through Whatman #41 quantitative filters (effective pore size of 20-25 μm) until it only took one or two filters to filter the entire sample. The same filtrate was then passed through Whatman GF/C filters (effective pore size of 1.2 μm) until only one or two were needed to filter the whole sample. Finally, the Whatman GF/F filters were used. It usually only took one or two of these to filter the sample at this point. The same jar used for leaching was used to hold the filtered leachate between filterings so the filter apparatus could be rinsed.

Since it took a large number of filters to complete the filtering of these samples, the filters were not weighed or included in the drying trays following filtering. Because of this, some fine leaf material that was not removed by the mesh screening before filtration was possibly lost. This means that the mass of leaves measured after leaching is likely to be slightly less than the actual mass left.

When the filtering was complete the jars with filtered leachate were returned to the incubator and the trays with leaf material were dried for approximately 48 hours at about 50°C. Upon removal from the oven the trays were allowed to cool for 5-10 minutes then weighed to the nearest milligram on the Mettler H16 balance. The leaves were weighed again after being left to equilibrate with the moisture in the air, which would be more representative of their condition when weighed before leaching. This equilibration time varied between 12 and 72 hours and resulted in an increase of 4 to 5% of the weight taken immediately after drying. These weights were used to determine leaf weight loss due to leaching as a percentage of

original dry weight.

Leachates from the various treatments were analyzed for color, pH, conductivity, and turbidity normally the same day as filtering. If not analyzed the same day, the leachates were stored at 10°C until analyzed. The analyses were made with the instruments and procedures used for the field samples. Color was measured as described previously, except that samples from the filtered stream water controls were used for the blanks instead of distilled water. Turbidity was measured primarily as a check on the adequacy of the filtering procedure.

Spot checks for total iron were also made using the DR-EL/2 kit, and the 1,10 Phenanthroline method (Hach Co., 1978). Iron was checked because organic molecules that cause color can increase the solubility of iron and when it combines with these organic molecules it can cause a darkening of color (Shapiro, 1966).

Time Series Leaching (Running Water)

A third leaching experiment was done in mid-August 1982 to compare leaching in still water (jars) with leaching in running water (recirculating chambers). The water was circulated at <0.2 m/s using submersible pumps.

The leaching water was, again, filtered stream water (GF/F filters). The leaves used were from the composite sample collected in November 1981. The same leaf mass to water volume ratio as used in the time series still water leachings was chosen (1.00 g/450 ml), and the leaves were leached for 48 hours. The actual

leaf mass and water volume used was 22.222 g in 10 l in 3 replicate chambers. The leachings were carried out in the cold room with the pumps submerged in ice baths to keep the water temperature fairly constant and close to 10°C.

The chambers were filled the night before leaching was to begin and placed in the cold room. Prior to the addition of the leaves the pumps were turned on for 15-20 minutes and water samples were taken. These samples represented the blanks or controls. After the addition of the leaves, samples were taken at 2, 6, 12, 24, and 48 hours. All samples were approximately 50 ml and were taken with a plastic syringe. Each time 50 ml were removed, 50 ml of filtered stream water were added.

Each sample was filtered with a Whatman GF/F filter and the same apparatus previously described. The samples were then analyzed for color, turbidity, conductivity and pH using the same analytical techniques employed for the other leaching experiments. Blanks for the color determinations were distilled water and the color of the controls was subtracted from the readings of the samples. The water temperature in the chambers was checked periodically to insure that it stayed fairly constant.

DOC vs. Absorbance

Another experiment was carried out to explore the relationship between dissolved organic carbon (DOC) in alder leaf leachates and light absorbance in the visible (color) and ultra violet (UV) wavelength ranges. A single leaching was conducted to produce a highly colored, concentrated leachate. Determinations

were made of DOC concentration of this leachate and the absorbance (UV and visible) of a series of distilled water dilutions. With the absorbance and the DOC (by calculation) of each dilution, the relationship between these two variables could then be statistically evaluated.

The concentrated leachate was obtained by placing 9.989 g of leaves, from the composite sample, into a clean 2 l glass jar with 1500 ml of filtered stream water (collected in March 1982). The weighing and leaching were carried out as before. The jar used for leaching and the flask used in filtering were washed with a soapless glass cleaner and rinsed several times with tap water and distilled water. After filtering the initial leaching water, any glassware used for filtering, storage, or subsequent dilutions was covered with foil and heated to between 400 and 450°C for 6-8 hours to burn off any carbon present.

The leaves were leached for 47 hours at approximately 10°C. After leaching, the leaves were removed as described earlier and the leachate filtered with Whatman GF/F filters. The leaves were discarded and the filtered leachate prepared for analysis.

The determination of DOC concentration in the leachate concentrate was done by the Cooperative Chemical Analytical Laboratory (CCAL) located at the Forest Sciences Laboratory, 3200 Jefferson Way, Corvallis, Oregon. A wet oxidation method using sealed ampules was employed with an Oceanographic International carbon analyzer (Oceanographic International Corp., 1978). Dilutions of 1:100, 1:200, and 1:500 were made and fixed for DOC analysis by the CCAL staff. The remaining leachate was capped and stored at 10°C for absorbance determinations.

The same day the leaves were removed and the leachate filtered, 8 more dilutions were made for the absorbance determinations. These dilutions and how they were made are as follows:

<u>volume of leachate conc. to total vol. with dist. water</u>	<u>dilution ratio</u>
concentrate	----
50 ml to 75 ml	0.67
50 ml to 100 ml	0.50
25 ml to 75 ml	0.33
25 ml to 100 ml	0.25
10 ml to 50 ml	0.20
15 ml to 100 ml	0.15
5 ml to 50 ml	0.10
5 ml to 100 ml	0.05
2 ml to 50 ml	0.04
1 ml to 50 ml	0.02

The last two dilutions were made the next day when the absorbance readings were conducted so as to include a few more dilute samples. These 10 dilutions plus the 3 made for the DOC determination produced a total of 13 dilutions for the absorbance analysis.

A spectral analysis and the absorbance readings of the leachate dilutions were made with a Hitachi model 100-60 double beam spectrophotometer. The objective of the spectral analysis of the leachate was to determine where the peak absorbance occurred so that this wavelength could be used to make the absorbance measurements. This analysis was conducted on the leachate concentrate over the wavelength range of 290 nm to 880 nm. Only one peak was found (305) and it was too large for the readings to be accurate on the machine. Therefore, spectra were run on the 0.33 and 0.20 dilutions. Again only one peak was found in each case,

near 300 nm. Quartz cuvettes were then used to insure that the peaks being observed were not being caused by the glass cuvettes. No peak was observed with either dilution when quartz cuvettes were used, instead there were increasing absorbances (beyond the accurate range of the machine) with decreasing wavelength. The same observation was made with a very dilute sample (a 1:100 dilution of the 0.20 dilution). This spectrum was run from 300 nm to 200 nm (the limit of the machine) and no peak was observed, just increasing absorbances. These results are similar to the results of Wilson (1959).

Since there was no apparent peak observed over the wavelength range of the machine, 254 nm was chosen for the UV absorbance readings. There were two other reasons, besides the lack of a peak, why 254 nm was chosen: 1) the maximum bulb intensity of the spectrophotometer light source was approximately 250 nm, and 2) many high pressure liquid chromatography and other UV detection systems routinely use 254 nm for a wide range of substances (Personal Communication, Clifford Dahm, OSU Stream Team). Absorbance readings for each dilution of 0.25 or less were taken at 254 nm. The more concentrated samples had absorbances beyond the accurate range of the spectrophotometer and were thus omitted from further evaluation.

Absorbance readings of the 13 dilutions were also done at 455 nm on the Hitachi spectrophotometer to compare with the color readings taken with the Hach DR-EL/2 spectrophotometer at 455 nm. The color readings on these dilutions were taken with the Hach kit as described for the other color measurements.

IV. RESULTS

Field Study

Leaf Fall

Weekly leaf fall in the riparian zone of the Seaside watershed is shown in Figure 4 along with stream flow and water color at Station 1S. Leaf fall data points represent the mean of all stations collected for each week. The data expressed as cumulative leaf fall with time are shown in Figure 5. As stated earlier, the only material generally collected was alder leaves and an attempt was made to remove other debris from the samples before weighing. It was noted in the background section that shrubs and herbs did not seem likely to contribute much material to the stream. This observation appears to be supported by the fact that little else other than alder leaves was found even on the plastic ground traps used after October 12. There was probably some unknown contribution of organic material to the stream by herbaceous plants and shrubs, but it appears that red alder leaves was the dominant organic input from September to December.

Leaf fall was low at the beginning and end of the sampling period with a marked increase and peak during the month of October. The pattern and quantities shown are typical of deciduous forests and similar to those observed in a similar study in Virginia (Slack and Feltz, 1968). The measured leaf fall on the Seaside watershed started out at about $2 \text{ g/m}^2/\text{day}$, increased to about $4 \text{ g/m}^2/\text{day}$ by October and peaked in late October at $11.7 \text{ g/m}^2/\text{day}$. In the Virginia study peak leaf fall was just over $12 \text{ g/m}^2/\text{day}$.

FIGURE 4. Daily stream flow and weekly water color for the South Fork Necanicum River near Seaside diversion (Station 1S), and weekly riparian zone leaf fall during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon
(Note: no collection made on Oct. 6. Amount collected Oct. 12 was evenly divided between Oct. 6 and Oct. 12 (see p. 30))

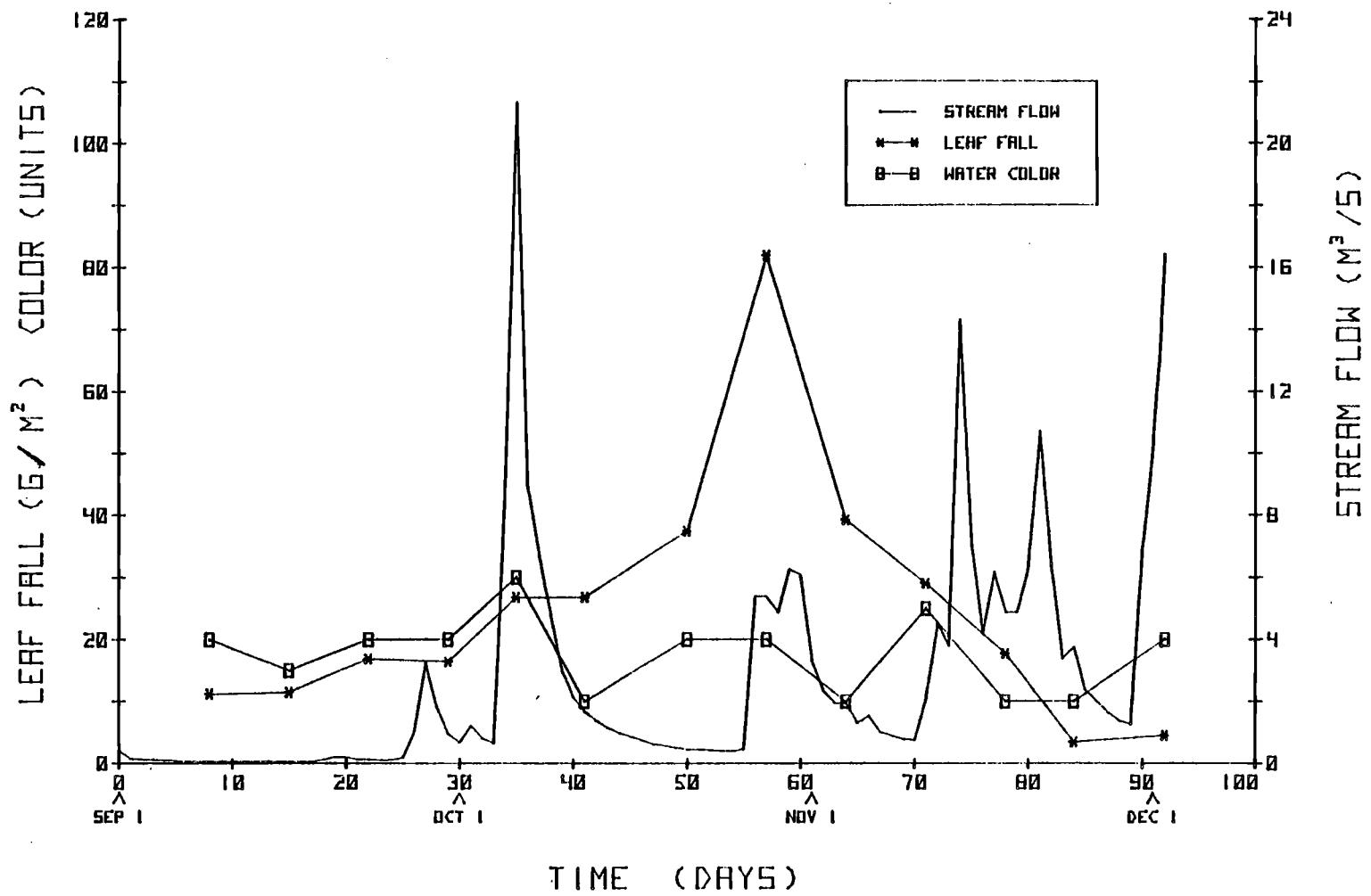
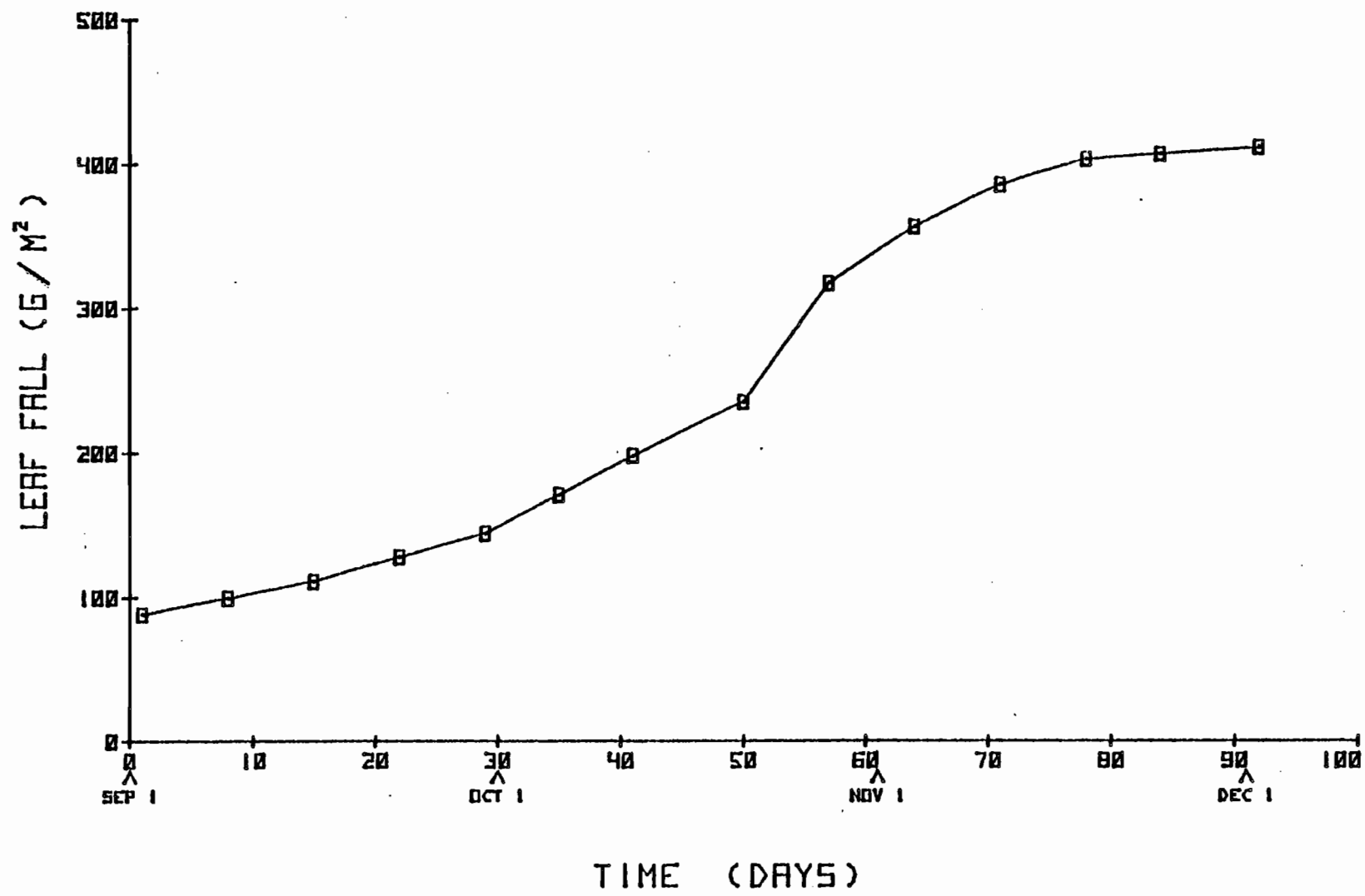


FIGURE 5. Cumulative mean riparian zone leaf fall, autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon (Note: no collection made on Oct. 6. Amount collected Oct. 12 was evenly divided between Oct. 6 and Oct. 12 (see page 30))



The total cumulative leaf fall between September and December of 322 g/m^2 was also in close agreement with the average annual leaf fall found by Zavitkovski and Newton (1971) for 15 red alder stands aged 3-34 years, in western Oregon in 1967 and 1968. This assumed that $2/3$ of the annual leaf fall occurs between September and December. Zavitkovski and Newton also found that for those two years the peak leaf fall occurred in October and early November.

Leaf fall appears to have been fairly uniform throughout the watershed, except the east tributary (Table 4). Measured total leaf fall for this tributary was slightly higher than for the rest of the watershed riparian zone. The largest differences occurred in the first cumulative collection and during the weeks of heaviest leaf fall. However, some bias may have occurred because only 2-3 leaf collectors were used on this tributary and they were not as widely distributed as on the other tributaries, especially the upper reaches.

Although some variation in leaf fall measurement was likely attributable to location, it was assumed that leaf fall was fairly constant throughout the watershed. This is based on the fact that for the various tributaries leaf fall data was generally fairly consistent and the riparian alder stand appeared to be fairly uniform throughout the watershed, especially the canopy cover (see site description).

Water Color

Mean daily stream flow and weekly water color at the city diversion (Station 1S) for autumn 1981 are also shown in Figure 4. Water color refers to true color

TABLE 4. Estimated average weekly and total autumn 1891 leaf fall in the riparian zone of the South Fork Necanicum River and its three main tributaries, Seaside Municipal Watershed, Clatsop Co., Oregon

Sample Date	Leaf Fall g/m ² ¹			
	Main Fork n=5-6	West Tributary n=5-8	Middle Tributary n=3-5	Tributary n=2-4
Sep. 2 ²	75.1 ±36	84.8 ±50	95.0 ±14	126.7 ±49
Sep. 9	11.7 ±6	12.5 ±6	11.2 ±6	7.2 ±2
Sep. 16	11.0 ±4	12.8 ±6	11.0 ±4	10.1 ±3
Sep. 23	14.4 ±9	19.5 ±6	16.1 ±8	16.1 ±2
Sep. 30	12.7 ±4	17.1 ±3	17.6 ±5	19.3 ±11
Oct. 6 ³	24.7 ±4	22.2 ±4	30.7 ±4	38.8 ±15
Oct. 12 ³	24.7 ±4	22.2 ±4	30.7 ±4	38.8 ±15
Oct. 21	36.8 ±23	33.2 ±8	29.6 ±13	56.0 ±15
Oct. 28	93.3 ±34	62.2 ±35	63.3 ±4	116.1 ±28
Nov. 4	46.0 ±15	48.4 ±32	26.3 ±9	27.6 ±3
Nov. 11	37.3 ±12	31.9 ±16	17.1 ±9	23.5 ±7
Nov. 18	18.8 ±10	15.7 ±9	20.6 ±12	15.6 ±7
Nov. 24	4.2 ±2	3.2 ±2	2.2 ±2	3.8 ±5
Dec. 2	4.4 ±5	5.1 ±3	1.2 ±8	2.7 ±3
TOTAL	415.2	390.8	372.7	502.3

¹Mean and standard deviation

²Cumulative 1981 leaf fall to date

³No collection Oct. 6. Amount collected Oct. 12 evenly divided between Oct. 6 and Oct. 12

as stated in the methods section (p. 37).

Water color remained fairly low throughout the sampling period. This is based, however, on weekly observations and it is possible that the water color could have been higher between sampling dates. The relatively high water color at the tap in Seaside on October 27 supports this possibility (Figure 6). Figure 4 does show that there were no color increases longer than a week.

The data in Figure 4 indicate that the background or baseline water color at the diversion during the 1981 low flow period was about 20 platinum-cobalt units (hereafter referred to simply as units). This is slightly higher than the water color observed during the winter 1982 (Table 5). At the beginning of storms in autumn 1981, water color tended to increase with a subsequent decrease after the storm peak. There were only two sampling dates that the water color at the diversion rose above 20 units (Oct. 6 and Nov. 11) and stream flow was rising or peaking on both occasions. Although stream water was not sampled, rising flows also occurred the day (Oct. 27) that water color at the tap in Seaside increased to 45 units (Figure 6).

Stream water sampling revealed no specific source area for the observed color. Water color near the city diversion (Station 1S) and the lowest sampling stations on the three main tributaries (7S, 8S, 12S) is shown in Figure 7. The east tributary (7S) tended to be the most colored, especially during low flows. It is also the smallest of the three tributaries, has the lowest flows and probably would not have had a controlling influence over color at the diversion since its color was generally not that much higher than that of the other tributaries (Figure 7). This is also

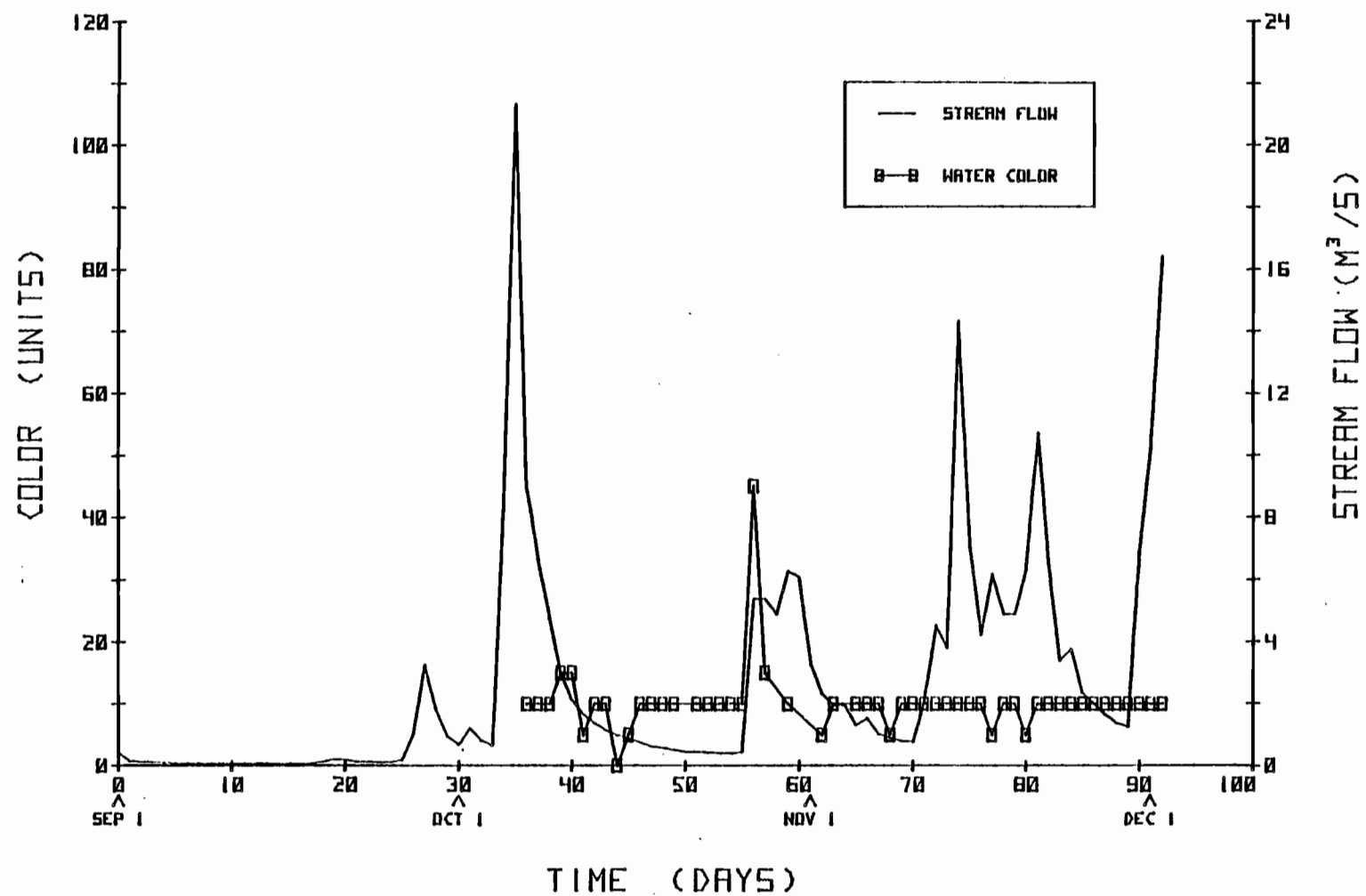


Figure 6. Daily water color at tap in city of Seaside and daily stream flow at city diversion during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon

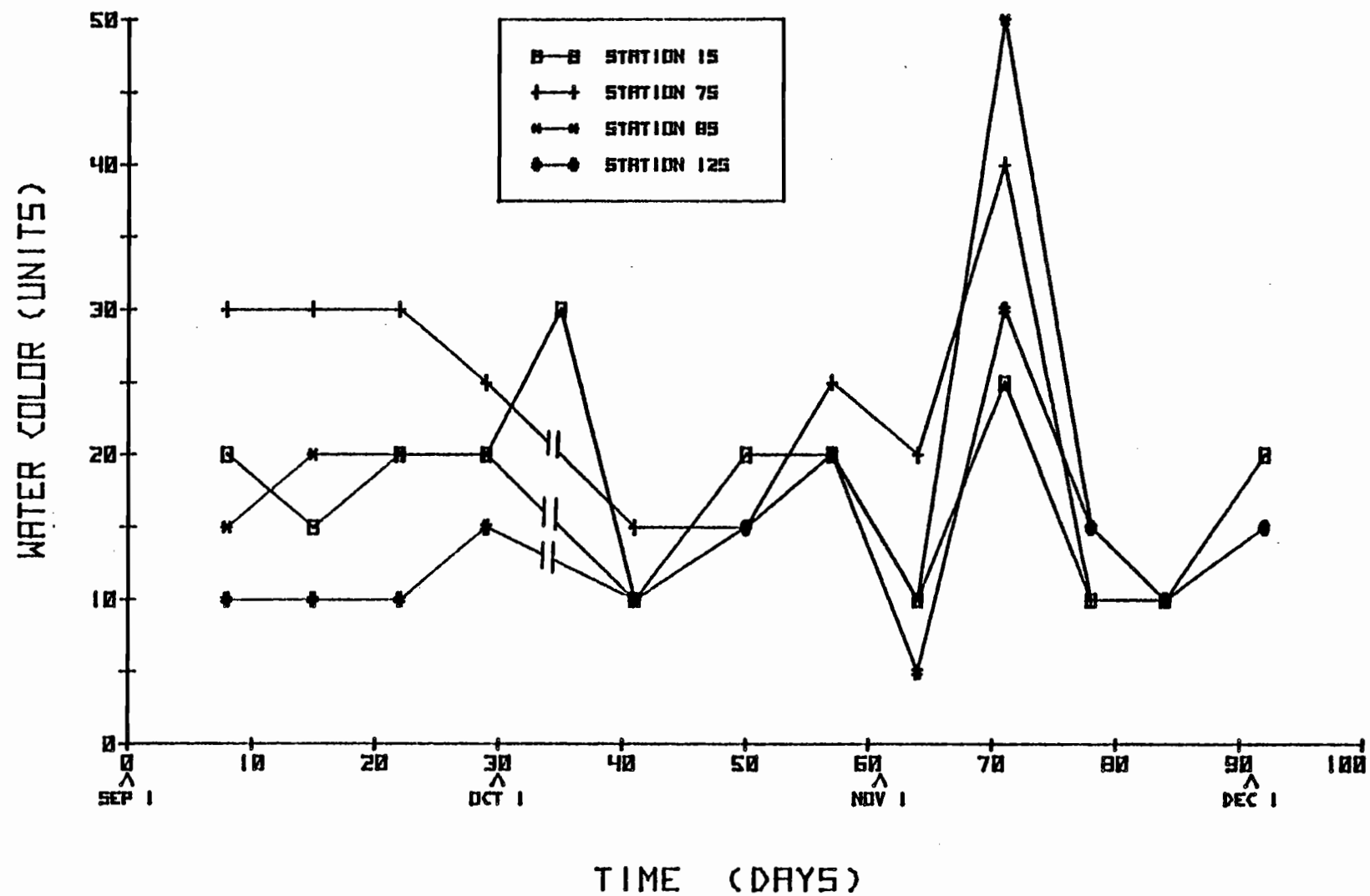
TABLE 5. Stream water quality during winter 1982, Seaside Municipal Watershed, Clatsop Co., Oregon

Date	Station #	Apparent ¹ Color (units)	True ² Color (units)	Turbidity ¹ (NTU)	Conductivity (μ mhos/cm)	pH	Dissolved Oxygen (mg/l)	Temperature (°C)
Jan 26	1s	25	5	2.3	35	6.7	11.9	8.0
	5s	20	5	2.2	34	6.7	12.0	8.0
	6s	20	5	2.1	34	6.8	11.8	8.0
	7s	20	5	1.6	31	6.6	11.8	8.0
	8s	20	5	2.1	34	6.8	12.0	7.8
	12s	20	5	2.3	37	6.9	12.0	8.0
Feb 16	1s	30	15	3.5	30	6.6	12.1	7.8
	5s	30	15	3.2	30	6.6	12.3	7.8
	6s	25	10	2.7	29	6.6	12.1	8.0
	7s	25	15	2.0	27	6.6	12.0	8.3
	8s	25	10	3.2	28	6.6	12.3	7.8
	12s	25	10	3.3	31	6.7	12.2	7.8
Apr 4	1s	10	--	0.56	42	6.8	12.6	6.9
	5s	10	--	0.49	41	6.9	12.6	6.9
	6s	10	--	0.57	40	6.9	12.4	7.2
	7s	10	--	0.71	34	6.8	12.4	7.5
	8s	10	--	0.41	39	6.8	12.6	6.7
	12s	10	--	0.41	43	7.0	12.5	7.2

¹Unfiltered samples

²Filtered samples

FIGURE 7. Water color near the city diversion (Station 1S) and at the lowest stations on the three main tributaries (7S, 8S, 12S) during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon

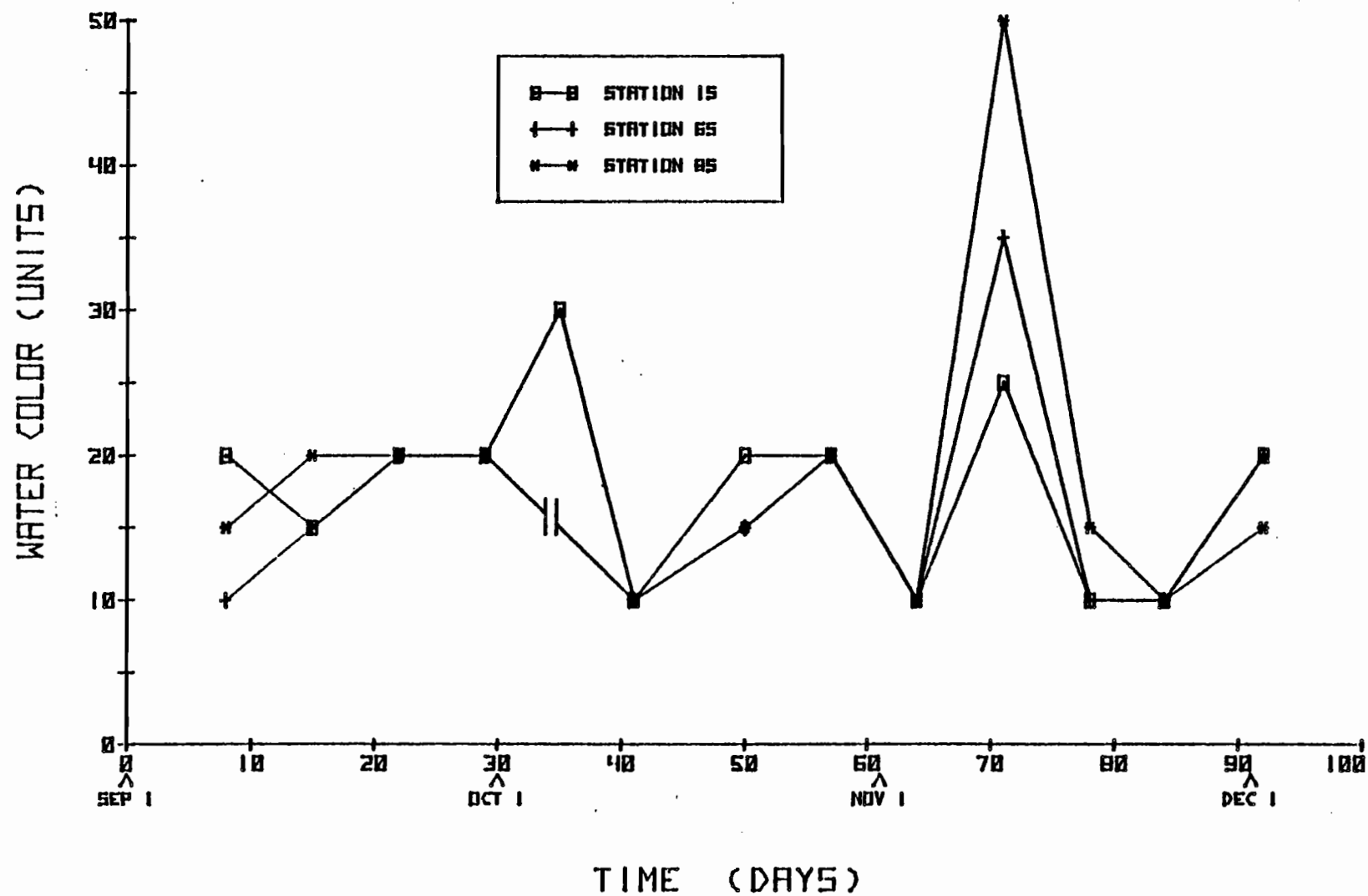


likely because the middle tributary (12S) tended to be the least colored and carried about three times the flow of the east tributary (Table 2).

The color at the diversion coincided fairly well with that of the west tributary (8S) and the combined east and middle tributaries (Station 6S, Figure 8), but there were dates when it was different than both. There were three dates that the color at the diversion was higher than both the west and combined east-middle tributaries. On these three dates (September 9, October 21, and November 11) the diversion had a color only 5 units higher than the most colored tributary. This slight color increase could be due to organic inputs below Stations 6S and 8S especially during the lower flows (Sep. 9 and Oct. 21). There was one date that the color at the diversion was lower than the color of both these main tributaries. This is difficult to understand considering that the trend throughout the rest of the sampling period was for the diversion to be the same or slightly higher in color. It may be possible that the time of sampling was important on this date. The order of sampling was Station 1S first, Station 6S second, and Station 8S third. There was probably two hours or more between each of these samples. On that day the flow was beginning to rise due to a storm (see Figure 4, day 71). It may be that as the storm progressed more storage sites were flushed and the color increased so that the higher colors observed on the tributaries had not yet reached the diversion when it was sampled. Similarly Station 6S was sampled before Station 8S and was less colored.

It appears that the color seen at the diversion is a composite of all the inputs in the watershed, including subsurface flow entering the stream below the

FIGURE 8. Water color near the city diversion (Station 1S), the lowest station on the west tributary (8S), and below the confluence of the east and middle tributaries (6S) during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon



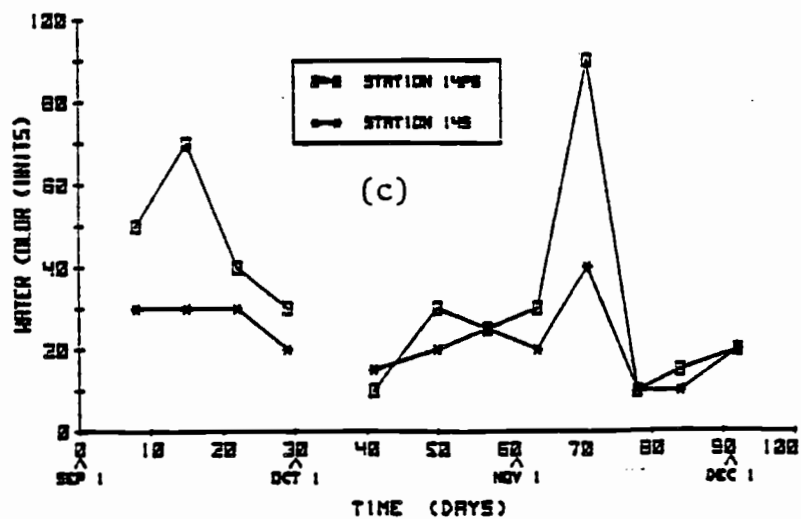
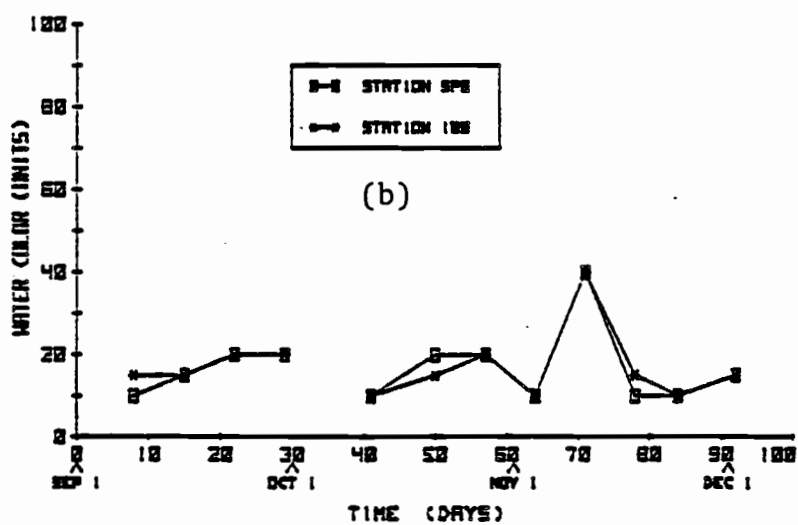
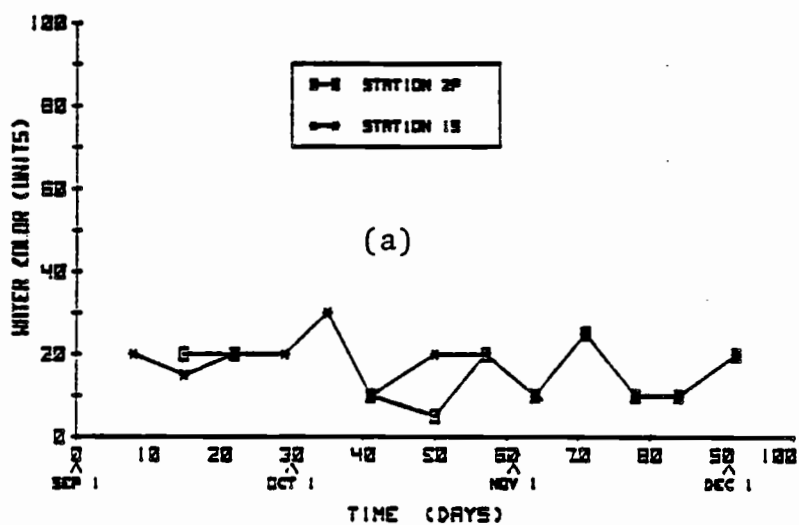
confluence with the west tributary, with no one specific controlling source.

Pools within the high water stream channel were carefully examined as one possible source of color. Isolated pools and pools connected indirectly to the flow have been observed to become highly and moderately colored, respectively (Slack and Feltz, 1968). This same pattern was observed in this study. Weekly water color levels for an isolated pool (14Pb), a pool to the side of the channel out of the main flow (9Pb), and a pool in the main flow (2P) are shown in Figure 9. Stream water color at or near each of these locations is also shown in this figure.

The isolated pool (14Pb) was generally more highly colored and more persistently colored than the other two pools (Figure 9c). The pool in partial contact with the flow (9Pb) had only one date when the color was above 20 units (40 units on Nov. 11) and closely followed the color in the flowing stream at that location (Figure 9b). The pool directly subjected to the main flow (2P) did not show any major color fluctuations and was either at or below the color level of the stream (Figure 9a). Pools in contact with the flow tended to be less colored and reflected the water color of the flowing stream at that location. As soon as the water level increased enough to flow through an isolated pool, the pool was flushed out and the color quickly dropped (compare stream flow, Figure 6, and Figure 9c) to a level similar to the flowing stream.

The amount of colored material present in water at a given time is going to depend not only on the amount and rate of input but also on the removal rate. In most natural stream systems removal of dissolved organic

FIGURE 9. Comparison of water color in three different pools and the nearest stream station to each pool, Seaside Municipal Watershed, Clatsop Co., Oregon: 9a) a pool in direct contact with the flow (2P) and Station 1S, 9b) a pool in indirect contact with the flow (9Pb) and Station 10S. 9c) an isolated pool (14Pb) and Station 14S



carbon (colored organics) is probably due primarily to microorganisms (Dahm 1981). So, anything affecting the microbial population could affect color intensity. Dissolved oxygen, temperature, and pH could all affect color intensity through their actions on the microbial population. The pH can also affect color abiotically. Increasing the pH causes an increase in color intensity but this relationship is not linear (Black and Christman, 1963; Christman and Ghassemi, 1966). The highest color intensities were found at pH 10 and 11. An increase in pH also causes an increase in the number of particles and a decrease in particle size (Black and Christman, 1963). On the other hand, optimum removal of colored material by coagulation (alum) occurs at low pH (<5) due to the characteristics of the aluminum ion (Black and Christman, 1963). Turbidity is also related to color. Both are measurements of optical properties of the water. Turbidity is a measure of light scattering due to particles suspended in the water. Color is a measure of light absorption at a specific wavelength (455 nm in this study). Turbidity can have an impact on the appearance (apparent color) of water at the tap in a water distribution system with no filtration, settling or other means of removing suspended particles.

Dissolved Oxygen and Temperature

Stream dissolved oxygen (DO) and temperature through time for Stations 1S, 7S, 8S, and 12S are shown in Figures 10-13. Stations 7S, 8S, and 12S were considered representative of the three main tributaries since the temperature and DO were very similar at the various stations along each tributary.

The stream DO levels throughout the watershed

FIGURE 10. Stream dissolved oxygen and temperature near city diversion (Station 1S) during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon (Note: weekly sampling was done except on October 6 during high water)

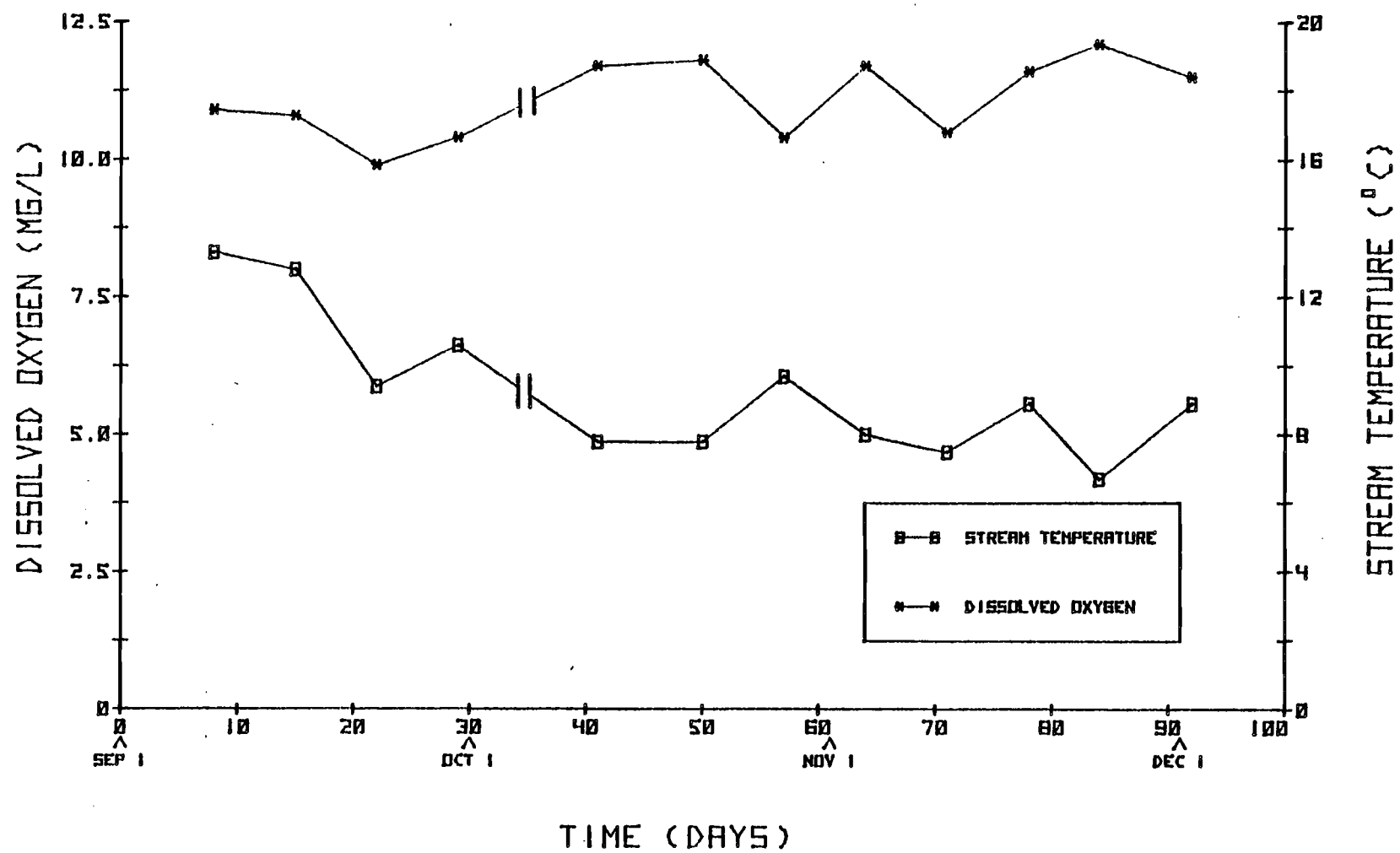


FIGURE 11. Stream dissolved oxygen and temperature in east tributary (Station 7S) during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon (Note: weekly sampling was done except on October 6 during high water)

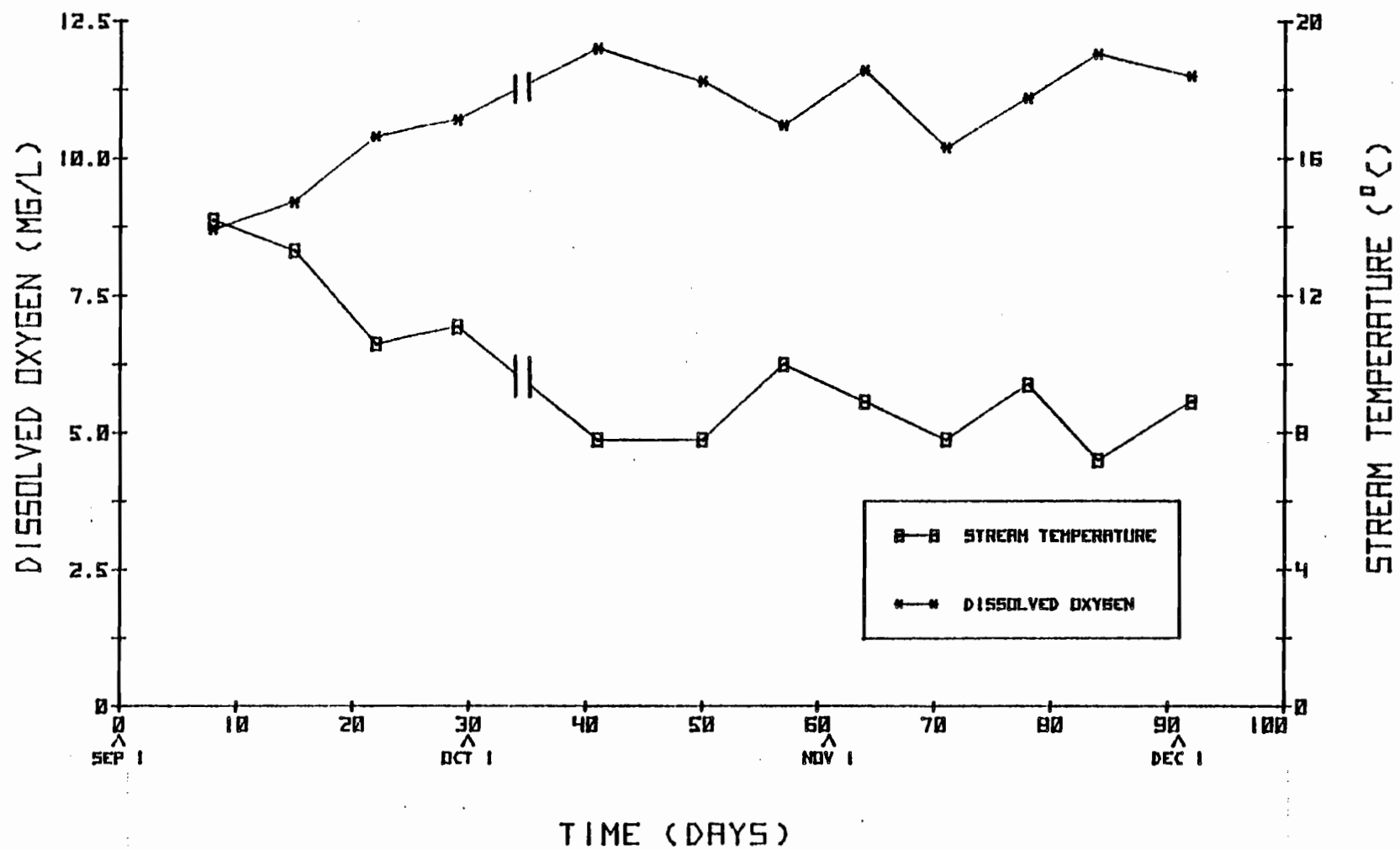


FIGURE 12. Stream dissolved oxygen and temperature in west tributary (Station 8S) during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon (Note: weekly sampling was done except on October 6 during high water)

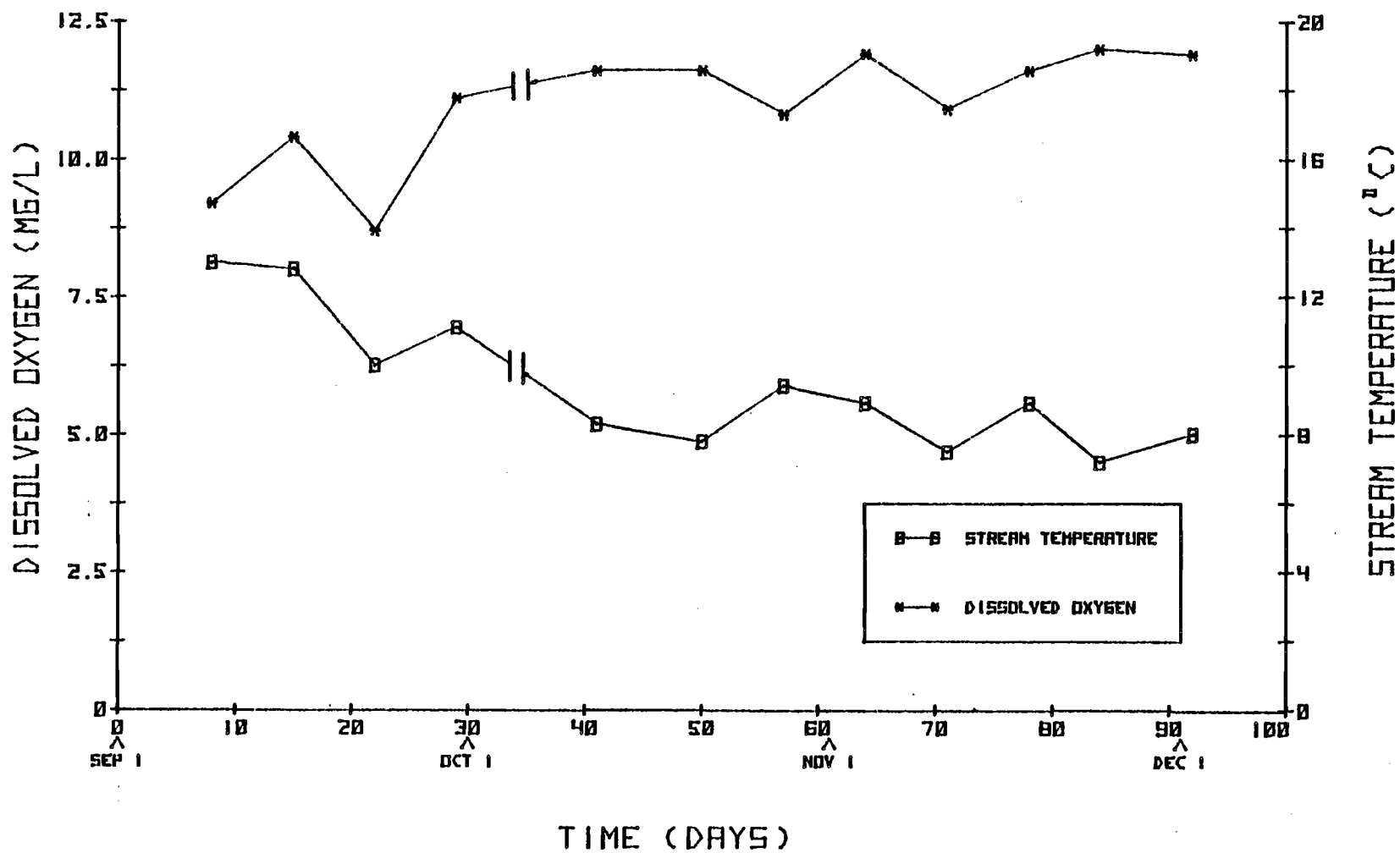
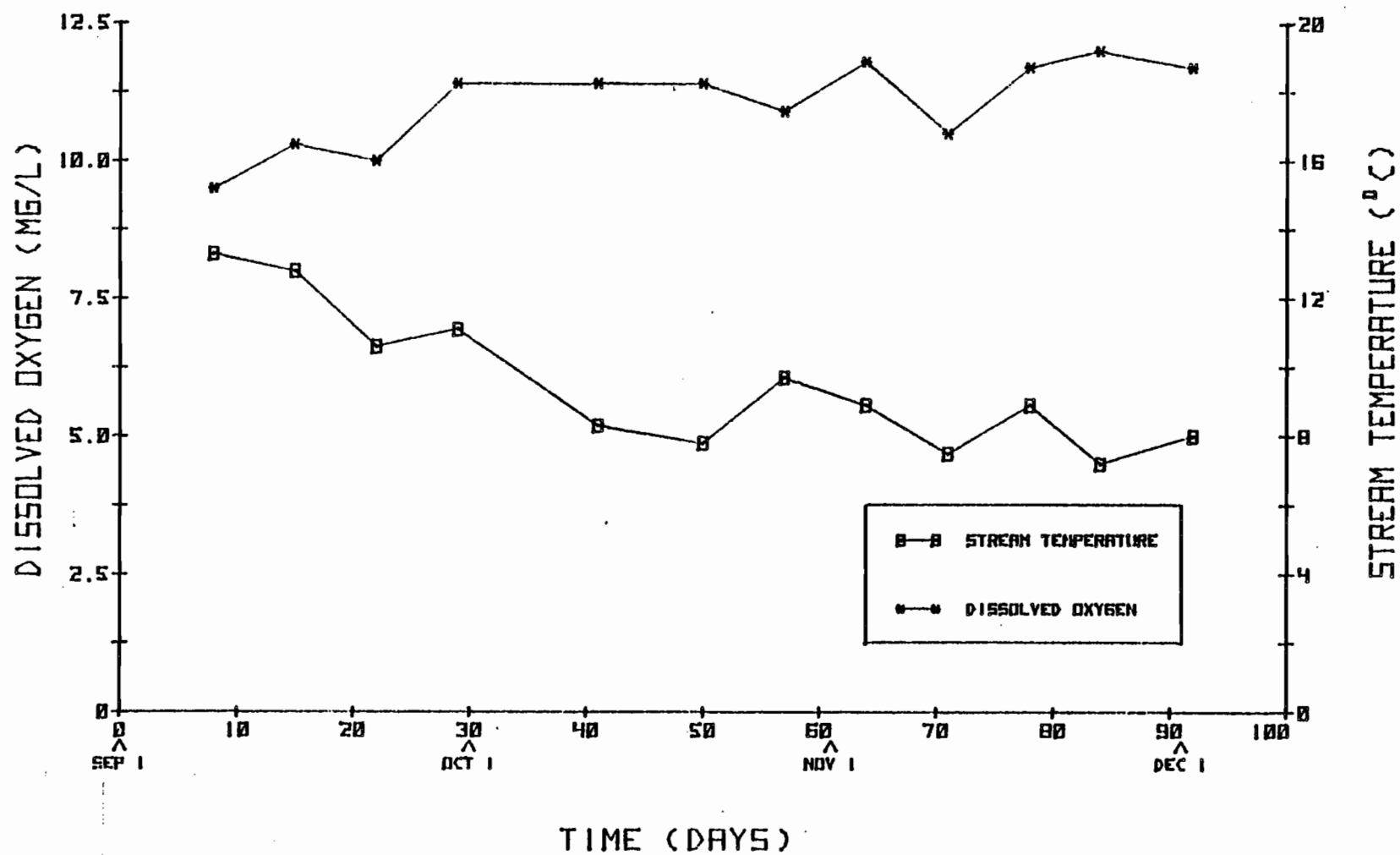


FIGURE 13. Stream dissolved oxygen and temperature in middle tributary (Station 12S) during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon (Note: weekly sampling was done except on October 6 during high water)



remained near saturation for the observed temperature most of the time. Table 6 lists water temperature and the corresponding saturation DO concentrations. At Station 1S, near the diversion, DO was never measured below 10 mg/l (Figure 10). The lowest stream DO concentration measured was 8.7 mg/l at Station 7S on the east tributary (Figure 11). Even this concentration was close to the saturation level of 10.2 mg/l for the stream temperature of 14.2°C. The highest temperatures and lowest DO concentrations occurred during September when flows were lowest and air temperatures highest. After September 30 (through April 4, 1982) measured DO concentrations in flowing water remained at or above 10 mg/l.

There did not appear to be any problem with temperature or DO during the fall 1981 in this stream. Stream DO concentrations appeared to remain high enough for even the most intolerant organisms. The temperature range of 6.7-14.2°C would have also been expected to pose no serious stream quality problems. Since these were only weekly measurements, there may have been times that the DO was lower or water temperature higher, but in general it appears that there were no DO or temperature problems from either a stream or domestic quality perspective.

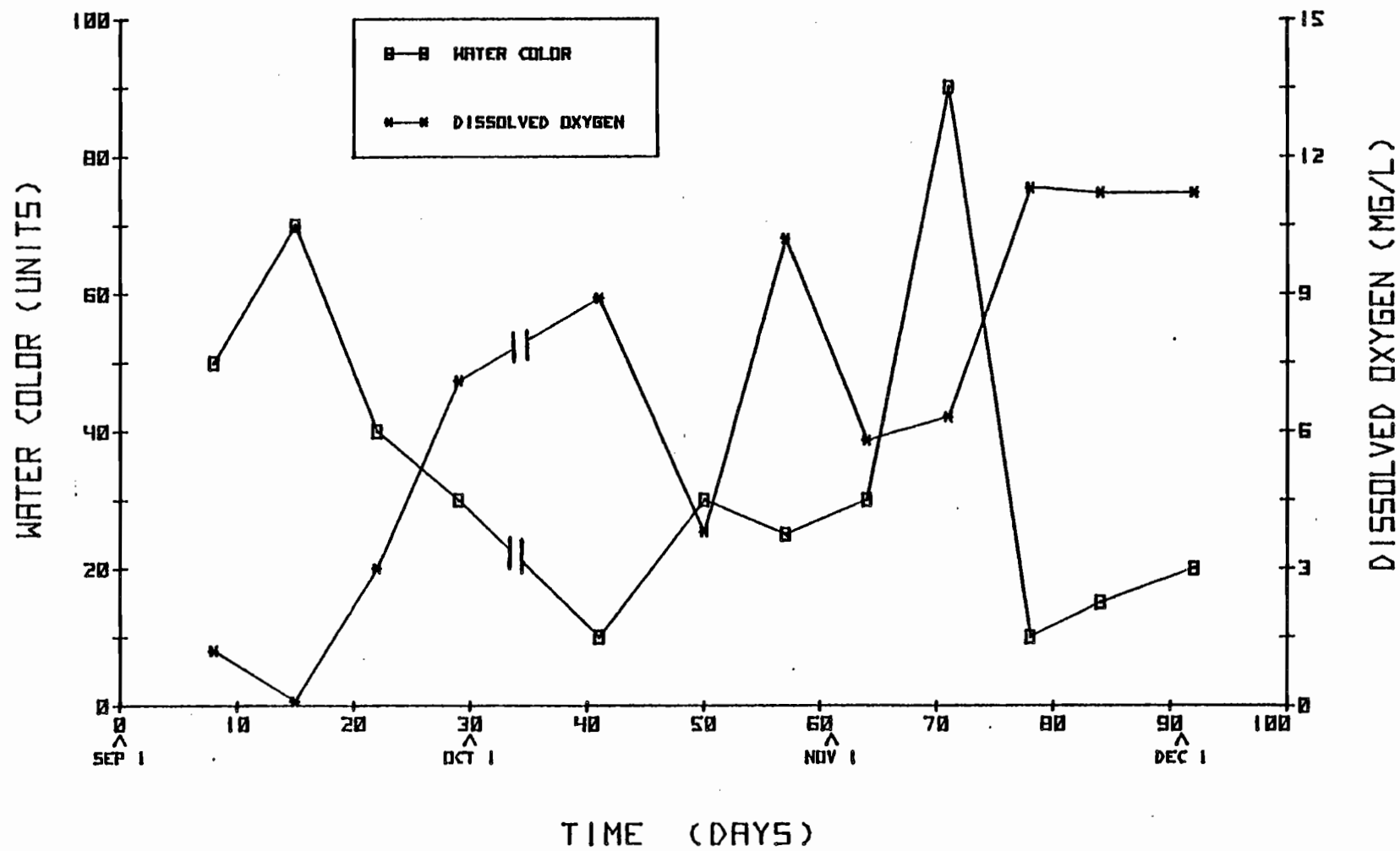
Isolated pools were a different matter. DO and color during autumn 1981 for an isolated pool (14Pb) are shown in Figure 14. Water quality data for all pools sampled during autumn 1981 are given in Appendix 3. DO varied over a much wider range in this pool than in the stream. The DO also tended to be inversely related to the water color, as has been previously observed (Slack and Feltz, 1968). The lowest DO level measured was 0.1 mg/l in pool 14Pb on September 16.

TABLE 6. Maximum dissolved oxygen concentration (saturation concentration) at various water temperatures (after Brown, 1980)

Water Temperature (°C)	Dissolved O ² at ¹ Saturation (mg/l)
7.2	12.1
7.8	11.9
8.3	11.7
8.9	11.6
9.4	11.4
10.0	11.3
10.6	11.1
11.1	11.0
11.7	10.8
12.2	10.7
12.8	10.6

¹Saturation dissolved oxygen concentration =
 $14.652 - 0.41022^{\circ}\text{C} + 0.0079910^{\circ}\text{C}^2 - 0.000077774^{\circ}\text{C}^3$

FIGURE 14. Water color and dissolved oxygen in an isolated pool (Station 14Pb) on the east tributary during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon (Note: weekly sampling was done except on October 6 during high water)



Whenever the flow increased enough to flush through the pool, however, the DO concentration increased until it was similar to the DO level of the flowing stream.

As with water color, pools in direct contact with the flow tended to have similar DO concentrations to the stream. During low flows these pools sometimes had depressed DO concentrations but these concentrations were not as low as in isolated pool 14Pb. In-channel pools with indirect contact to the flow generally had intermediate DO concentrations when the flows were low (Appendix 3). At higher flows these pools were directly affected by the flow and closely followed the stream DO.

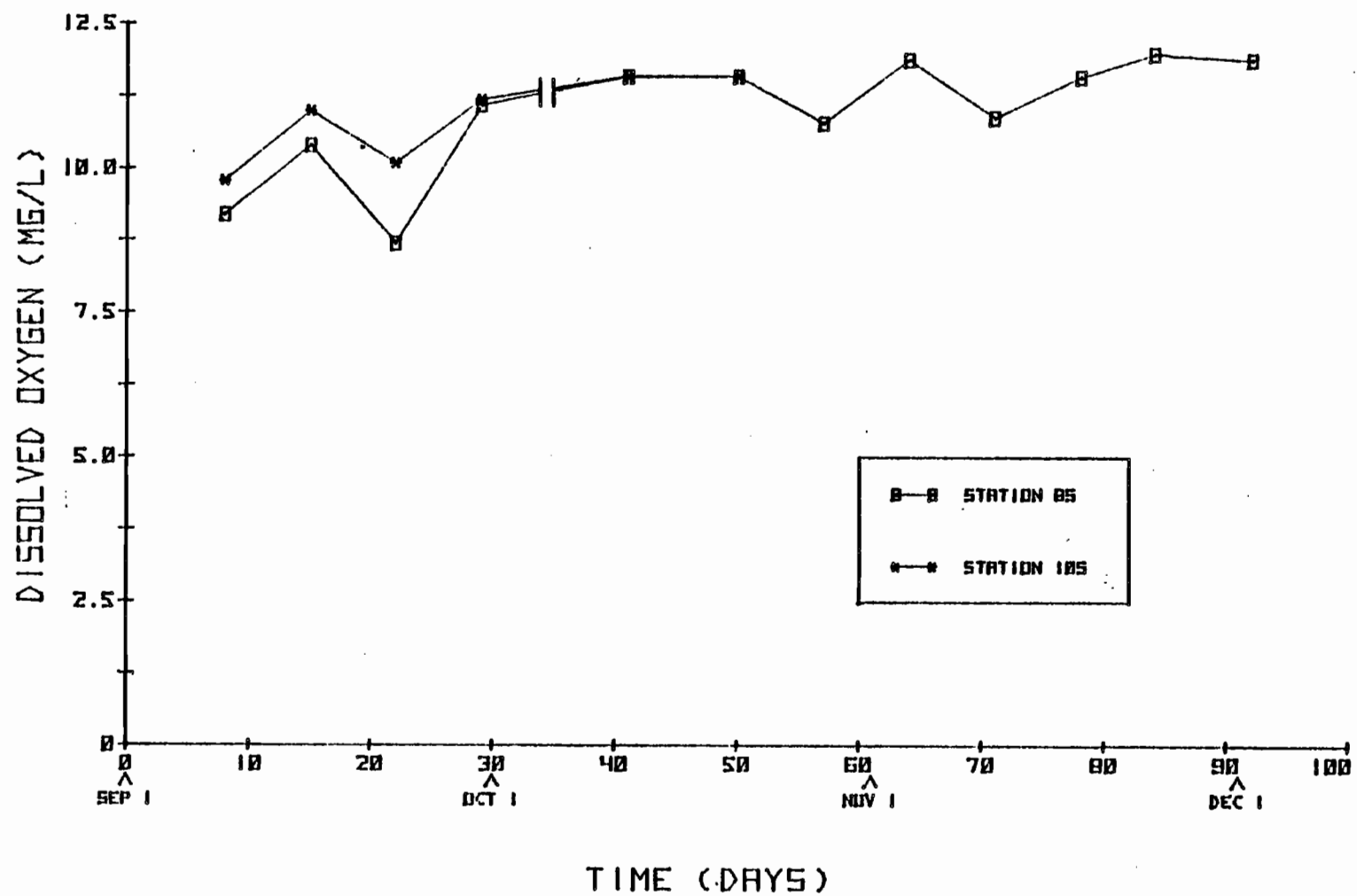
It is interesting to note that while the slower moving section at Station 9 did not seem to have much of an effect on water color, at low flows, it did seem to show some control over the DO concentration downstream. During the first few weeks of sampling, the DO concentration at Station 8S was lower than that at Station 10S (Figure 15). As soon as flow increased, the DO at Stations 8S and 10S remained the same. Because of this and the fact that these stations were in close proximity, only Station 8S was sampled after October 21.

There was not very much difference in temperature between pools and the flowing stream (Appendix 3). The temperature remained fairly constant throughout the watershed on a given sampling date.

pH and Conductivity

Conductivity and pH measurements were made to help characterize the general chemical make up of the stream water. The electrical conductivity provides information

FIGURE 15. Stream dissolved oxygen in the west tributary (Stations 8S and 10S) during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon (Note: weekly sampling was done except on October 6 during high water)



on the amount of electrolytic (charged) salts present in the water. pH was measured to observe the acidic or basic nature of the water and how it changes during the autumn months with the input of organic material.

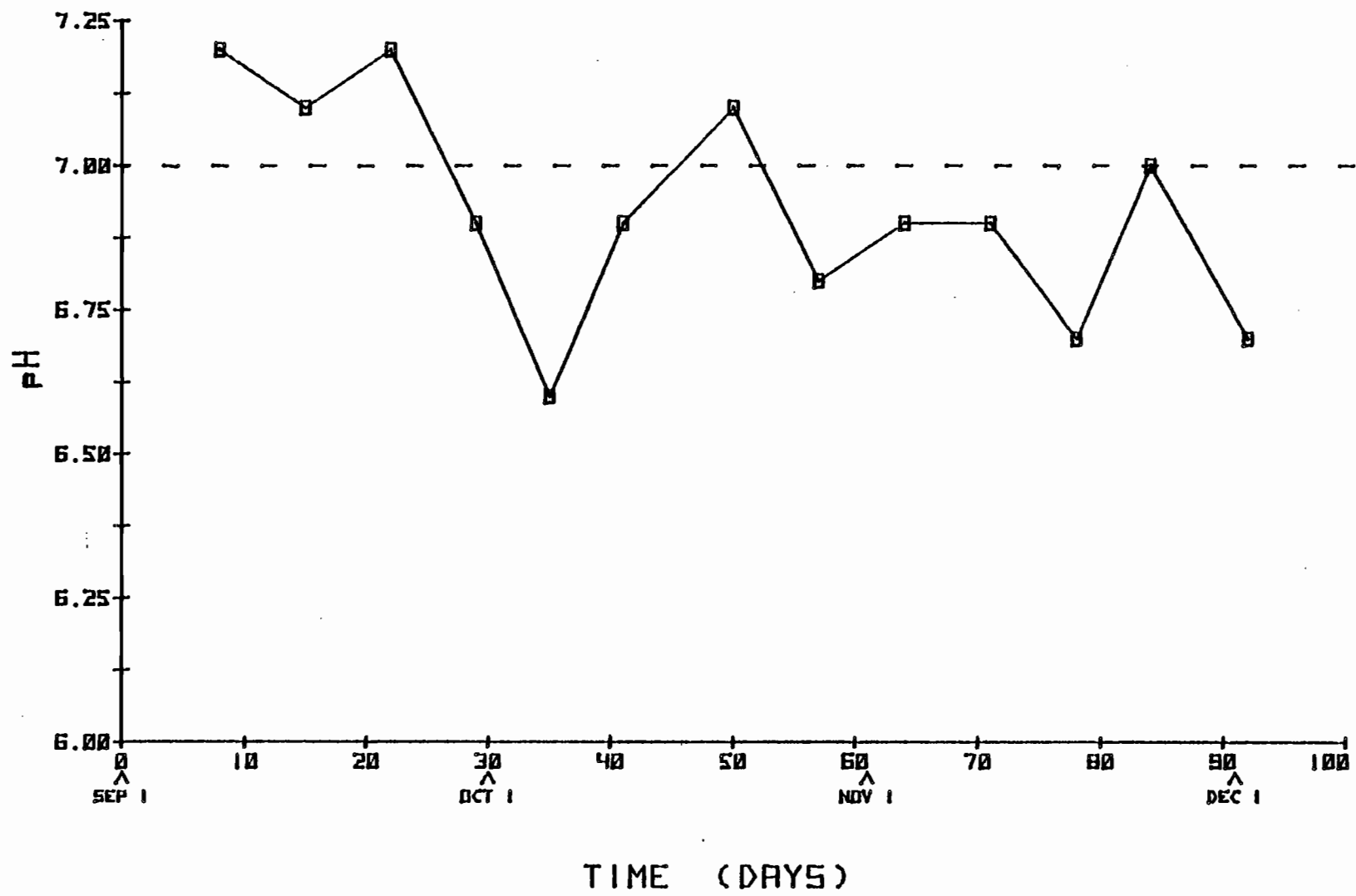
The pH of the stream at the diversion remained near neutral (pH 7) during the autumn 1981 sampling period (Figure 16). During the first three weeks (low flows) the pH was just above 7.0 and as the flows increased it tended to be slightly acidic. The pH was never measured more than 0.4 units above or below neutral in the running stream (Appendix 4) the pH in pool 14Pb (the most highly colored pool) became more acidic than that of the live stream but the lowest it fell was to 6.1 units (Appendix 3). Pools in contact with the stream flow generally had pH values similar to that of the stream, being closest where direct contact with the flow was greatest (Appendix 3).

Conductivity generally remained below 50 $\mu\text{mhos/cm}$ except for the first weeks of sampling and a few samples after that time (Appendix 4). Only once was the conductivity measured above 60 $\mu\text{mhos/cm}$ and this was in an isolated pool, 14Pa (Appendix 3).

Nitrate-Nitrogen and Iron

Stream nitrate-nitrogen was measured to determine if the influx of alder leaves raised the nitrate levels one mg/l or more. Increases in nitrate levels can increase stream biological activity since nitrogen is often a limiting nutrient. From a domestic use standpoint, nitrate levels greater than 10 parts per million can be a health hazard to infants. Increases of this magnitude were not anticipated but some increase seemed possible,

FIGURE 16. Stream pH near the city diversion (Station 1S) during autumn 1981, Seaside Municipal Watershed, Clatsop Co., Oregon



since alder is a nitrogen-fixing species and is known to have relatively nitrogen rich tissue (Turner et al., 1977). It should be remembered that the Hach kit has limited sensitivity, being capable of accurately measuring nitrate-nitrogen only at concentrations greater than 0.25 mg/l.

In early autumn when the flows were lowest, stream nitrate-nitrogen was generally 0.25-0.50 mg/l, with some samples up to 1.00 mg/l. However, the nitrate levels dropped and remained consistently low as soon as flows increased. After September 30, most of the samples taken showed nitrate-nitrogen concentrations less than or equal to 0.25 mg/l. Only selected stations were sampled for nitrates from October 12 through November 11 and the analyses were discontinued after that for the rest of the autumn due to low level of nitrates present. Nitrate-nitrogen was determined for samples taken January 26, 1982 and was found to be less than 0.25 mg/l. Therefore, it was assumed from the end of autumn into the winter nitrate-nitrogen levels remained below 0.25 mg/l.

From the observed conditions, autumnal organic inputs of alder leaves do not appear to significantly increase nitrate-nitrogen concentrations with respect to domestic water quality. Under extremely low flow conditions with high alder leaf inputs the increase might be more dramatic, since nitrate levels did rise slightly during the first few weeks of this study. It is possible, however, that even with increased nitrate inputs much of this nitrate could be removed or transformed by native aquatic microorganisms.

Iron can contribute to water color by combining with colored organics and causing a darkening of color (Shapiro, 1966). Combination of iron with colored

organics may also improve the possibility of color removal by coagulation (Shapiro, 1964).

Stream iron concentrations were not measured in autumn 1981, but a spot check made in April, 1982 at six stations (1S, 5S, 6S, 7S, 8S, 12S) showed total iron concentrations less than 0.1 mg/l. Limited water quality data available from the city of Seaside also indicated that stream iron concentrations were less than 0.1 mg/l in autumn. In addition a check made on the highest leaf mass to water volume laboratory leaching (5.56 g/l) showed that the filtered stream water originally contained 0.06 mg/l total iron and after 48 hours of leaching this increased to 0.3 mg/l. This was the most highly colored of the laboratory leachings after 48 hours and was far above the range of color found in the live stream (Figure 4 and Appendix 4). It thus appears that iron concentrations high enough to affect stream water color are unlikely.

Laboratory Experiments

The results of the laboratory experiments showed that alder leaves have the potential to quickly and significantly change water color and other water quality characteristics. There were, however, some differences found between leaching in still water and running water.

Time Series Leaching (Still Water)

Results of this experiment showed that for up to four weeks of leaching alder leaves, water color increased with time of leaching (Table 7). Rates of color change in the leaching water (determined by dividing the color change over a given time interval by the time

TABLE 7. Water color, conductivity, pH, and leaf weight changes with time during laboratory leaching of red alder leaves in still water¹

Time (hrs)	Color (units) mean	Conductivity (μ mhos/cm) mean	pH ² mean	Leaf Weight % mean
0	0	41	6.8	0
2	15	53	6.6	-5.2
6	22	66	6.4	-10.3
12	34	72	6.3	-13.9
24	58	78	6.2	-16.6
48	90	82	6.0	-22.0
72	133	83	5.8	-22.5
96	198	89	5.9	-23.7
336	655	100	6.5	-26.4
648	1310	97	6.5	-30.7

¹Actual leaf mass and water volume used were 1 g leaves and 450 ml water, a leaf mass to water volume ratio of 2.2 g/l

²Means obtained by converting pH to $[H^+]$ (-antilog pH), averaging and converting back to pH

interval) are shown in Table 8.

Water color increased rapidly in the first two hours. This rapid increase in water color may have been from material on the leaf surface. The data (Table 8) indicate that after 2 hours there is a generally constant rate of color increase with increasing leaching time (approximately 2 units per hour). There were two time intervals that didn't follow this trend (24-48 hours and 72-96 hours). A simple linear regression of color (y) against time (x) also supports the idea that color generation was fairly constant (Figure 17). For this relationship ($y = 2.0x + 2.6$) the r^2 was 0.94 and the rate of color change over time (slope) was 2 units per hour.

It appears that color generation from still water leaching of red alder leaves is fairly constant, at least after the first 2 hours, up to 4 weeks. This consistency indicates that the mechanism for removal of colored material from leaves remains the same, at least during the first four weeks, and is likely to be largely a physical leaching process. At some point, microbial decay will begin to dominate the break down of leaf material but the actual removal of break down products probably continues to be by physical leaching. In this experiment this may have occurred between 72 and 96 hours when the cloudiness of the samples noticeably increased.

Leaf weight changes, conductivity, and pH also showed some distinct patterns over the course of the 4 week experiment (Table 7). Table 8 shows the rate of weight change in percent (of original dry weight) per hour for different time intervals. There was an initial rapid weight loss during the first few hours, followed by a steady decline in weight loss rate up to

TABLE 8. Rate of change in water color, conductivity, and leaf weight at various intervals during laboratory leaching of red alder leaves in still water¹

Time Interval (hrs)	Color (units/hr)	Conductivity (μ mhos/cm/hr)	Leaf Weight (%/hr)
0-2	7.5	6.0	-2.6
2-6	1.8	3.2	-1.2
6-12	2.0	1.0	-0.59
12-24	2.0	0.3	-0.22
24-48	1.3	0.2	-0.22
48-72	1.8	<0.1	-0.02
72-96	2.7	0.2	-0.05
96-336	1.9	<0.1	-0.01
336-648	2.1	0.0	-0.01

¹Actual leaf mass and water volume used were 1 g leaves and 450 ml of water, a leaf mass to water volume ratio of 2.2 g/l.

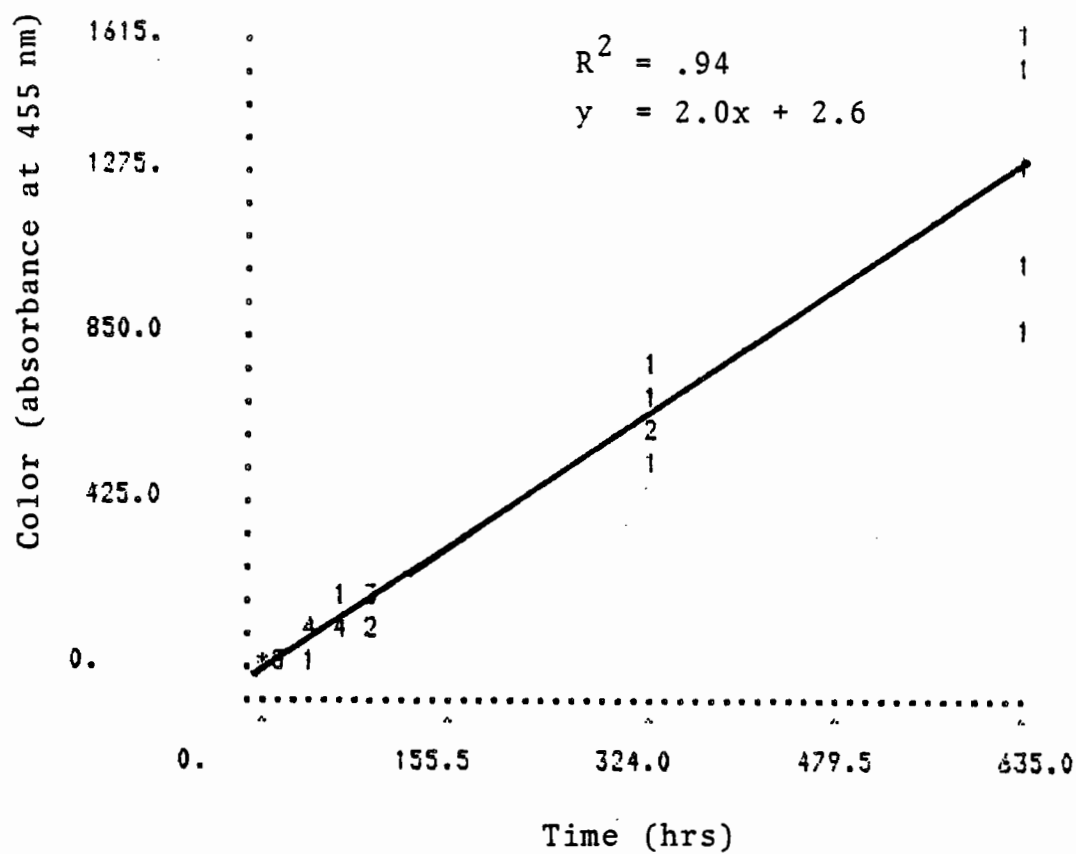


FIGURE 17. Relationship between water color and leaching time of red alder leaves in still water (laboratory). The leaf mass to water volume ratio was 2.2 g/l

48 hours of leaching. After 48 hours the rate of weight loss was relatively small (Table 8). Most of the total weight loss observed after 648 hours occurred within the first 48 hours with the most rapid weight loss occurring in the first 6 hours.

Electrical conductivity increased at each sampling time during the first 24 hours of leaching (Table 7). The most rapid increases again occurred during the first 6 hours, after which increases in conductivity dropped off until they were extremely small after 24 hours (Table 8). From 96 to 336 hours the rate of increase was less than 0.1 $\mu\text{mhos/cm/hour}$. This indicates that most of the electrolytic (charged) salts are removed quickly from the leaves and that subsequent breakdown of the leaf material does not release many more charged species, at least for up to 4 weeks of leaching.

Values for pH clearly decreased during the first few days of leaching, from an initial pH of 6.8 to a pH of 5.8 after 72 hours (Table 7). This is not surprising since the colored leachate from organic material is known to be acidic in nature (Christman and Ghassemi, 1966). Over the long term under aerobic conditions, however, the pH increased almost to its initial level. This increase in pH was probably due to the microbial population engaged in decomposing the leaf and leachate material. Nykvist (1962) cites studies in which leaching time and the type of microbial population present determined changes in pH and the direction in which they occur.

Time Series Leaching (Running Water)

Results of the running water leaching experiment showed trends that were similar to the results of the

still water experiment. The duration of the experiment was only 48 hours, so it was focused on the initial physical leaching process.

Color increased continuously for the first 48 hours, but in a different manner than the still water leaching (Table 9). For example, color generation appeared to be less uniform over time than in still water leaching. Leaching leaves in running water produced larger changes in water color for a given length of leaching time, except for 2 hours of leaching when running water and still water appear to leach the same (Table 10). This again indicates that the color removed during the first 2 hours is readily taken from the surface of the leaves. The amount of color at any given time after 2 hours was always greater for running water (Figure 18). After 48 hours the color in the running water was about one and one half times the color in the still water. Running water appears to be more efficient in removing color from alder leaves than still water; it removed more color in a shorter period of time.

Weight loss from the leaves was not measured for the running water leachings but other studies have measured leaf weight loss in running water systems that include natural streams, artificial streams, and recirculating chambers. This work has shown that 5 to 30% of the original leaf weight is lost in the first 24 hours (Kaushik and Hynes, 1971; Nykvist, 1962; Petersen and Cummins, 1974; and Short and Ward, 1980). The weight loss in this experiment was near the upper end of this range since the still water leaching exhibited an approximate 17% weight loss in the first 24 hours.

Changes in conductivity and pH in running water occurred in a manner almost identical to the still water

TABLE 9. Water color, conductivity, and pH changes with time during laboratory leaching of red alder leaves in running water¹

Time (hrs)	Color (units) mean	Conductivity (μ mhos/cm) mean	pH ² mean
0	0	57	7.2
2	20	71	6.8
6	48	84	6.8
12	68	93	6.7
24	102	102	6.6
48	140	102	6.5

¹Actual leaf mass and water volume used were 2.22 g leaves and 10 l of water, a leaf mass to water volume ratio of 2.2 g/l.

²Means obtained by converting pH to $[H^+]$ (-antilog pH), averaging and converting back to pH.

TABLE 10. Differences in color, conductivity, and pH between still and running water laboratory leaching of red alder leaves¹

Time (hrs)	Difference \pm 95% C.I. ²		
	Color (units)	Conductivity (μ mhos/cm)	pH
0	n.s. ⁴	n.s.	n.s.
2	n.s.	n.s.	0.2 ³
6	26 \pm 9	n.s.	n.s.
12	34 \pm 18	n.s.	n.s.
24	44 \pm 23	8 \pm 5	n.s.
48	50 \pm 26	n.s.	n.s.

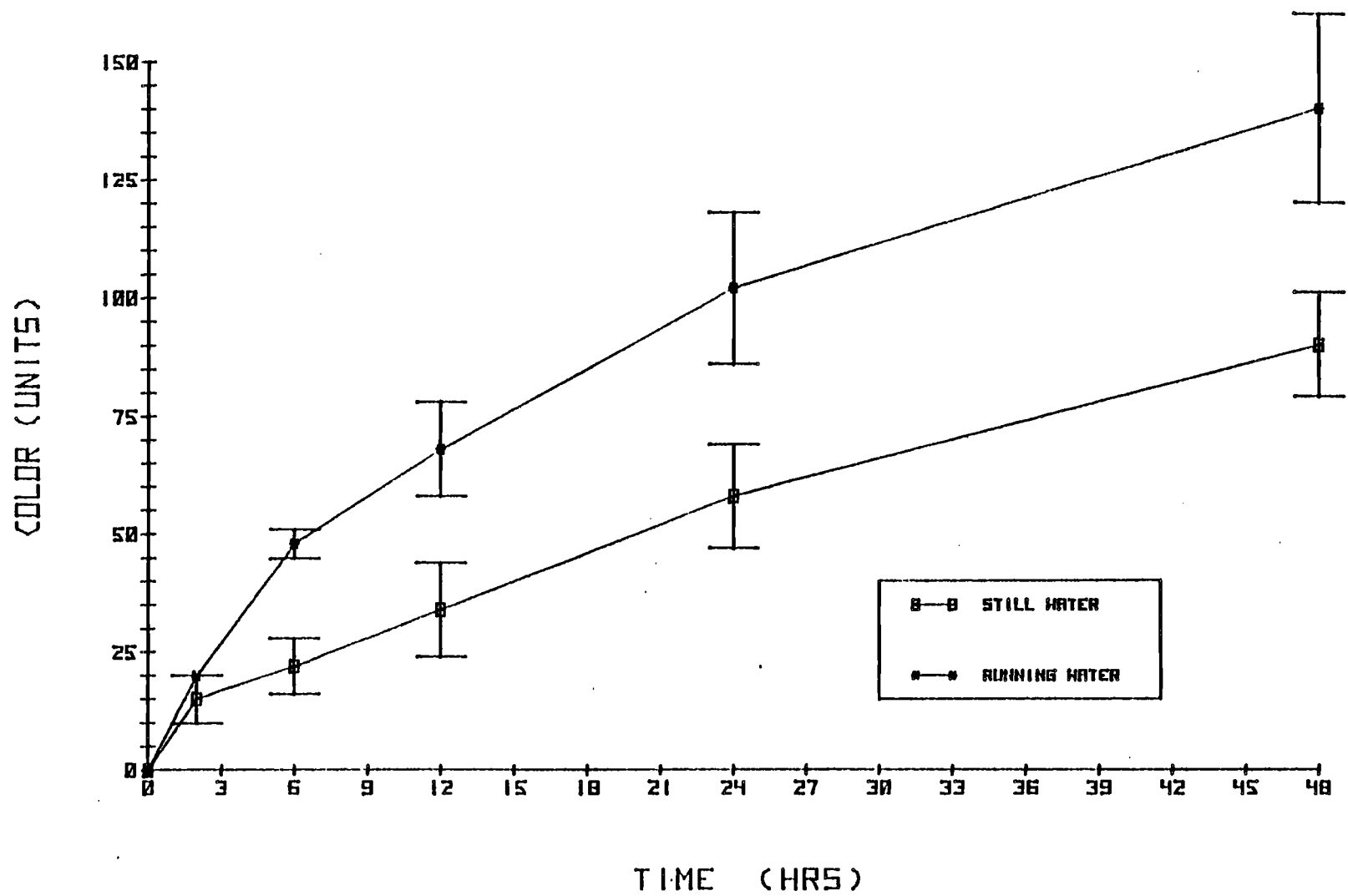
¹The leaf mass to water volume ratio in both situations was 2.2 g/l.

²Differences in means between still and running water treatments with 95% confidence interval at given times.

³Because of the log relationship between pH and $[H^+]$ it was not possible to convert differences with 95% confidence intervals from $[H^+]$ back to pH units. Therefore, confidence intervals not shown.

⁴n.s. indicates no significant difference between the two treatments (two tailed t-test, $\alpha=0.5$).

FIGURE 18. Comparison of color generation from alder leaves (2.2 g/l) leached in still and running water. Bars represent \pm one standard deviation



leaching. A comparison of the results from the two experiments is shown in Table 10. There was only one time for each of these measurements when the values were significantly different between still and running water treatments (24 hours for conductivity and 2 hours for pH) after the difference between blanks was taken into account. Unlike color, changes in conductivity and pH are apparently not greatly affected by whether or not the leaching water is moving. This is probably because these inorganic components are labile and are removed quickly in running or still water.

Within the range observed, water temperature variations were not expected to produce differences between running and still water treatments. Although there were slight temperature increases in the recirculating chambers due the electric pumps, the temperature never rose more than 2°C and always remained under 15°C. The highest temperature reached in any of the chambers was 13°C and the temperature of the still water leachings remained near 10°C. Such temperature differences should not have affected the leaching process (Petersen and Cummins, 1974).

Mass Loading Series Leaching (Still Water)

Results of the 48 hour mass loading series leaching show that increases in leaf mass loading cause increased changes in water quality (Table 11). Color and conductivity increase with increases in leaf loading, while pH decreased. Simple linear regressions of these variables against mass loading suggest that the relationships are linear (Figures 19, 20, 21).

The percent decrease in leaf weight after 48 hours

TABLE 11. Water color, conductivity, pH, and leaf weight changes with changes in mass loading of red alder leaves in still water (laboratory)¹

Mass/ Volume Ratio ²	Color (units) mean	Conductivity (μ mhos/cm) mean	pH mean ³	Leaf Weight (%) mean
0.00	0	41	6.7	0
0.25	19	49	6.6	-24.2
0.50	29	54	6.5	-23.0
1.00	55	65	6.3	-24.8
2.22	94	88	5.9	-24.0
5.56	225	170	5.3	-25.3

¹Leachings were carried out for 48 hours.

²Refers to leaf mass/water volume ratio (g/l).

³Means obtained by converting pH to $[H^+]$ (-antilog pH), averaging and converting back to pH.

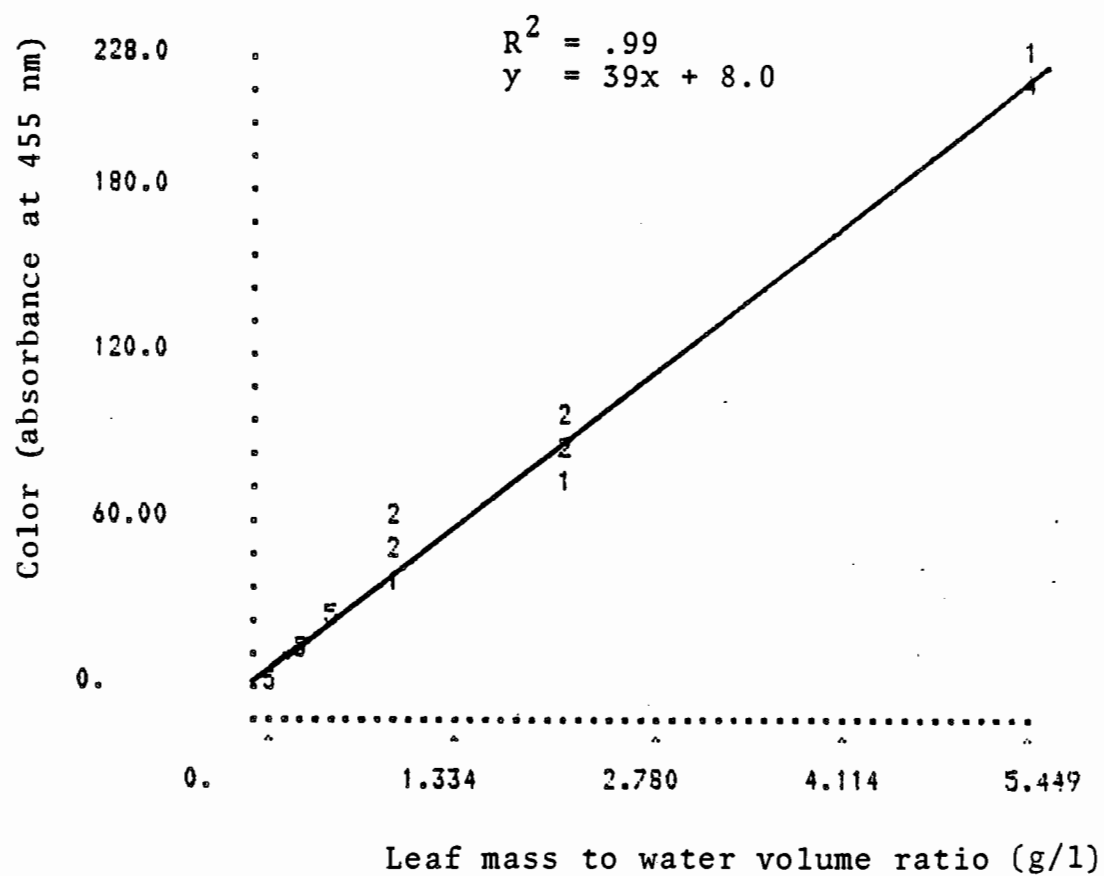


FIGURE 19. Relationship between water color and mass loading of red alder leaves leached in still water (laboratory) for 48 hours

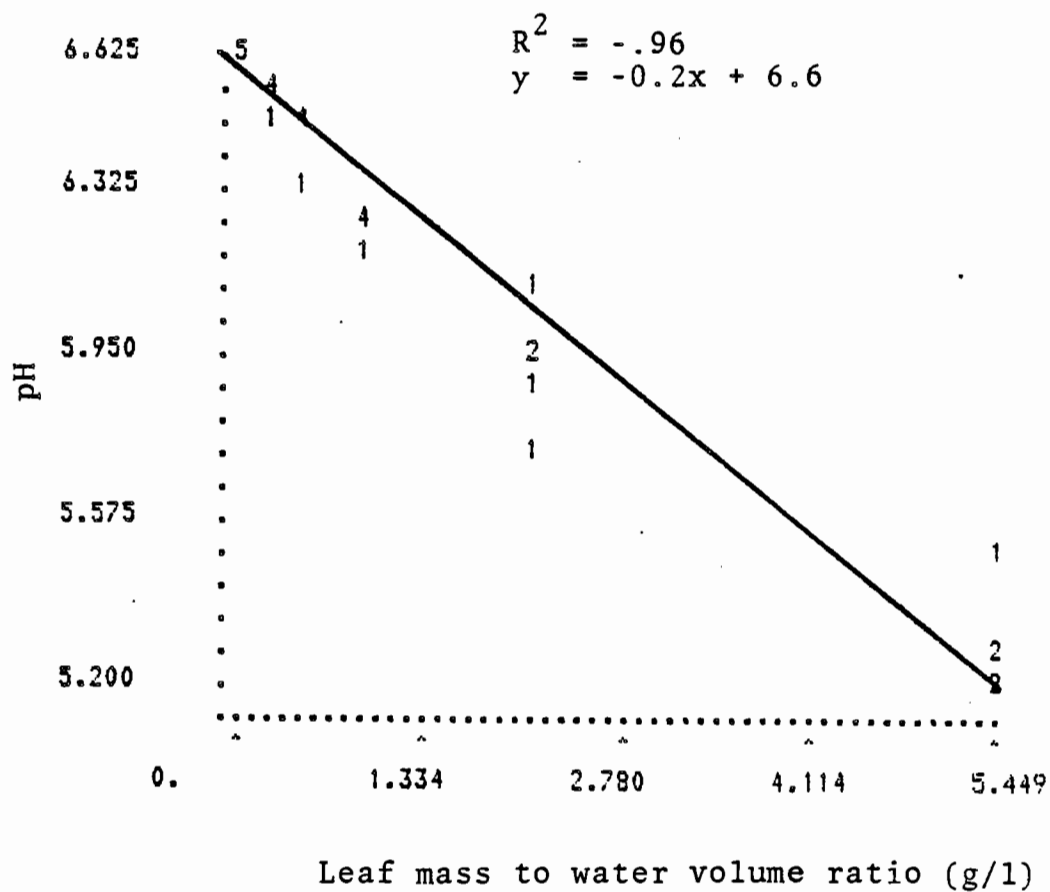


FIGURE 20. Relationship between pH and mass loading of red alder leaves leached in still water (laboratory) for 48 hours

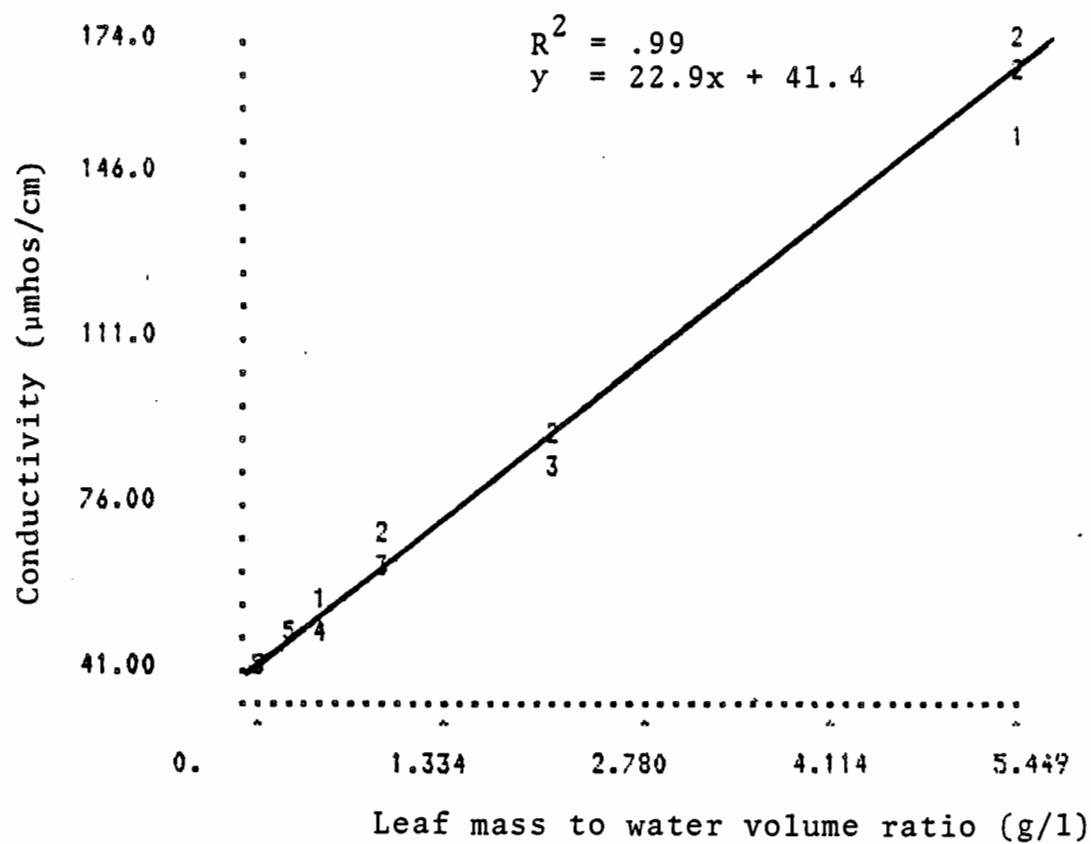


FIGURE 21. Relationship between water conductivity and mass loading of red alder leaves leached in still water (laboratory) for 48 hours

was essentially the same regardless of the quantity of leaves leached (Table 11). This indicates that there is a given, relatively uniform, amount of material available to be readily leached per unit mass of leaf material within 48 hours (primarily physical leaching).

DOC vs Absorbance (UV and Visible)

Red alder leaf leachate showed a strong linear relationship between dissolved organic carbon (DOC) and visible light absorbance (color) at 455 nm and also between DOC and ultra violet (UV) absorbance (Figures 22 and 23). These results indicate that a relatively simple and inexpensive measurement such as absorbance might provide a reasonable estimate for the amount of DOC present in stream water. These results, however, are only for leachates made under laboratory conditions with only one source of organic matter (alder leaves). Whether or not these relationships occur in natural systems would have to be determined from samples collected in the field.

Past studies indicate that color is not a very good estimator of total organic carbon (TOC) in streams because of the variety of sources of organic matter (Christman and Ghassemi, 1966; Larson, 1978). These conclusions were based on varying color/carbon ratios found among streams, in the same stream, and for leachates from tree bark produced in the laboratory. Another study (Wilson 1959) used UV absorbance spectra to analyze natural waters for fulvic acids and concluded that the results depended on the calibration curve. It was also noted that the calibration curve should be prepared with substances as similar as possible to those being studied.

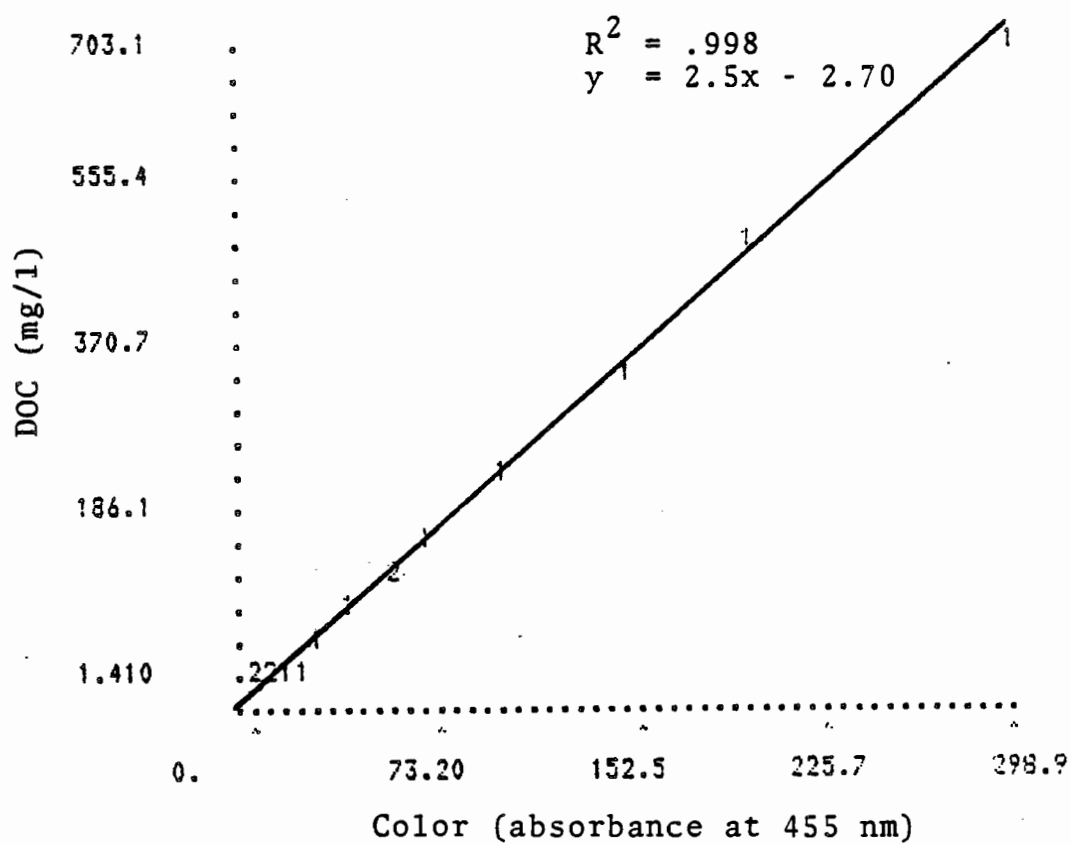


FIGURE 22. Relationship between dissolved organic carbon (DOC) in mg/l and water color (absorbance at 455 nm) in red alder leaf leachates (laboratory)

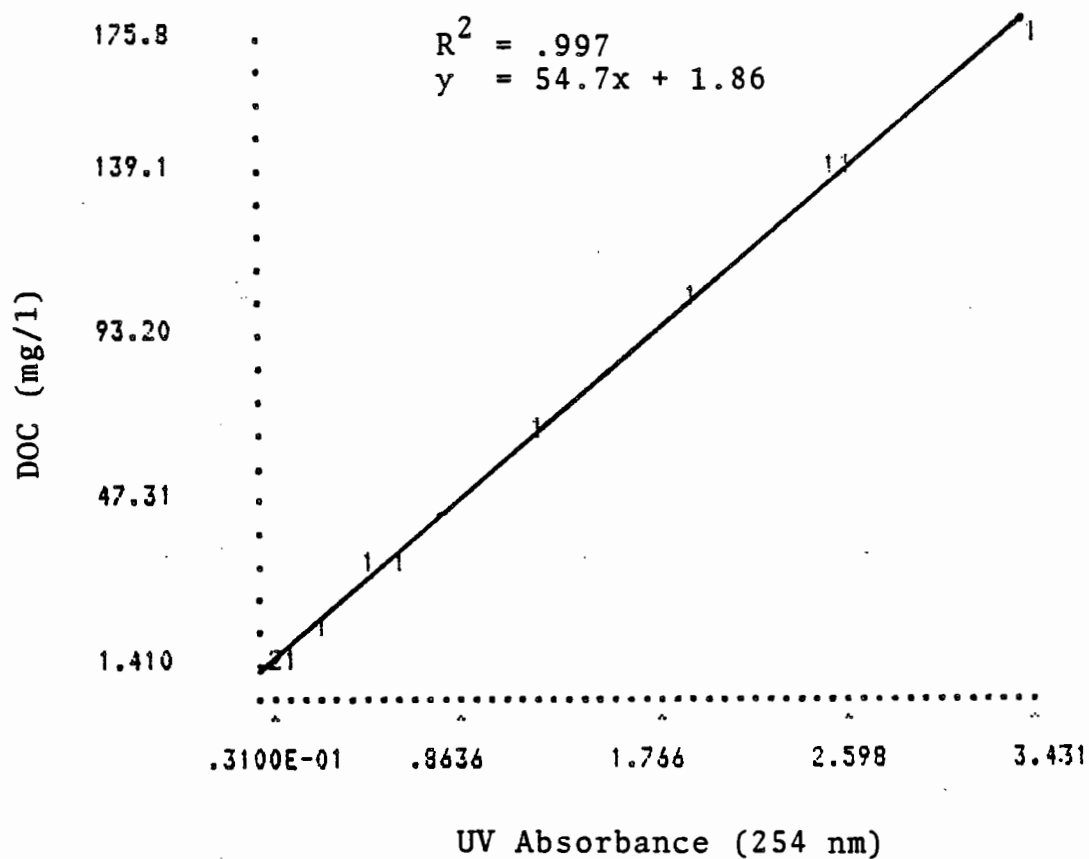


FIGURE 23. Relationship between dissolved organic carbon (DOC) in mg/l and ultra violet absorbance (254 nm) in red alder leaf leachates (laboratory)

The color/carbon ratios for the samples in this experiment were fairly uniform for colors of 30 units and above, but below 30 units there was variability (Table 12). Therefore, even though the relationship appears to be linear (Figure 22), in the lower range of color, characteristic of most streams, it may not be a very good predictor of DOC. On the other hand, the ratios of UV absorbance to DOC were relatively constant throughout the range measured (Table 12) and the relationship is apparently linear (Figure 23). Even though these tests were performed with laboratory produced leachate, UV absorbance may provide a useful estimator of DOC concentrations in aquatic systems dominated by red alder leaves.

TABLE 12. Color/carbon and UV absorbance/carbon ratios in red alder leaf leachates (laboratory) of varying concentrations

DOC (mg/l)	Color ¹ (units)	Absorbance (@ 254 nm)	Color/Carbon Ratio	Abs/Carbon Ratio
740	305	-----	0.41	-----
493	200	-----	0.40	-----
370	150	-----	0.40	-----
247	100	-----	0.40	-----
185	70	3.50	0.38	0.019
148	60	2.63	0.40	0.018
111	40	1.91	0.36	0.017
74.0	30	1.271	0.40	0.017
37.0	20	0.638	0.54	0.017
29.6	15	0.513	0.51	0.017
14.8	10	0.254	0.67	0.017
7.4	10	0.118	1.35	0.016
3.7	5	0.062	1.35	0.017
1.41	0	0.031	0	0.020

¹These values were taken at 455 nm using the platinum-cobalt scale.

V. DISCUSSION AND CONCLUSIONS

The field study showed that stream water color at the city of Seaside diversion (Station 1S) was slightly higher during the September 1981 low flows than the color (about 10 units) measured during the winter 1982 (Figure 4, Appendix 4, Table 5). This early autumn coloration (20 units) was also slightly higher than the maximum standard of 15 units set by the USPHS for drinking water (USPHS Drinking Water Standards, 1961). However, such very faint color would probably not cause many complaints about the appearance of the water.

There was one instance during the autumn 1981 when coloration (true color) of the tap water in Seaside reached 45 units (October 27, Figure 6). This, however, did not occur during low flow conditions but at the beginning of a storm. Other increases in color were also observed in the stream at the beginning or near the peak of storm flows (Figure 4, October 6 and November 11). This suggests that as flows rise, storage sites of dissolved organics and leaf material (e.g., stream-side zones, pools, roadside ditches, and soil) are flushed and enough material is quickly added to increase water color. This condition would only occur a few times each fall during the first storms when there is a chance for accumulation of organic matter between storms.

The area directly adjacent to the live stream may provide a large amount of leaf material and dissolved material as the stream level rises. A great deal of leaf litter can accumulate in this area if the stream flow remains low for a week or two, especially during

a period of heavy leaf fall. These leaves would already be partially leached and decomposed and some of the products from these processes may be present on the leaves or in the soil, available to some extent to the stream along with the leaf material as the stream rises.

Roadside ditches may be similar to the streamside area. These ditches trap and hold leaves and are generally quite moist. The leaves would leach and decompose and much of the material would probably remain in the ditch until washed out by a storm. Usually there is a fairly direct route to the stream from these ditches, through culverts and small feeder drainages, and this material could get into the system rapidly with the progression of a storm.

Pools not in direct contact with the stream flow under low flow conditions are another important source of colored material. Given a week or two for leaf accumulation and leaching, these pools could become highly colored and contribute to the stream water color when flushed by higher flows.

Although not directly examined in this study, soil organic matter can contribute to stream water coloration. Christman and Ghassemi (1966) showed the importance of contact time (water and soil) and organic matter content to the generation of color in water leached through soil. Their results showed that a soil with 4% organic matter only caused water color to increase 7 units in 2 days and 10 units in 30 days. A soil with 14% organic matter caused a rise of about 18 units after 2 days and 22 units after 30 days. In contrast, a soil with 70% organic matter caused a rise of about 60 units after 2 days and 130 units after 30 days. Since the major soils in the Seaside watershed have organic matter

contents of less than 12% (possibly higher along stream-sides), it does not seem likely that the soil would be adding a great deal of color. Also, once the soil is recharged by fall rains the bulk of the soil water contributing to stream flow is likely to have a relatively brief residence time in the soil.

During the low flow period, however, soil organic matter may be very important to the background color of 20 units that was measured. Soil may also act as a storage site for colored material derived from the litter at the surface. Leached material from leaves at the surface could move down into the soil and with a storm be flushed out. There could also be an increase in sediment load during storms, so that the apparent color seen at the tap could include the influence of this suspended sediment.

The other water quality measurements add support to the suggestion that autumn leaf fall did not have a substantial chronic impact on the water quality of the South Fork of the Necanicum River at low flow. For example, DO concentrations remained near saturation even at the lowest flows when a slight DO depression was observed. Conductivity remained low and did not show much difference between fall and winter values (Appendix 4 and Table 5). The pH increased somewhat during the lowest flows but it was never more than 0.2 units above neutral pH (the pH of distilled water) at the diversion (Figure 16).

The laboratory results demonstrate the potential for alder leaves to cause rapid and substantial color changes in stream water. Appreciable increases in water color occurred after just a few hours contact time with leaves in still, filtered stream water and color increases continued even with leaves that were

leached for 4 weeks. It appears that the rate of color generation was fairly constant (approximately 2 units/hour) in still water over the entire 4 week period except possibly during the first few hours. This constant rate indicates that the physical leaching mechanism for color removal probably occurred throughout the experiment.

Running water was more efficient than still water at removing color from alder leaves. There appeared to be a somewhat diminishing rate of color generation, but except for the first few hours the running water leachings were more highly colored. In contrast, the removal of colored material was slower in still water and thus the removal rate was more constant. In moving water physical leaching occurred more rapidly most likely because it constantly carried material away from the leaf surface.

After 48 hours of leaching, color increased linearly as the mass of leaf material per unit volume of water was increased. However, the percent weight loss of the original dry weight of leaves was constant regardless of the mass loading level. This indicates that there is a limit to the amount of material which can be physically leached in a short period of time from a given leaf mass. Therefore, where leaves are the dominant source of organic input, a primary factor that determines water color is the ratio of leaf mass to water volume. Where leaf input is fairly constant spatially the controlling factors are contact time and flow. Watershed area, relief, and stream morphology would thus be important by their effects on velocity and flow and, therefore, dilution and contact time of the water with leaves in the stream.

Two calculations of expected water color were made in Appendix 5. The first was made using a stream flow of $0.9 \text{ m}^3/\text{s}$, measured during the week of peak leaf fall in 1981 and typical of the flows at this time (Figure 4), and the peak leaf fall rate ($11.7 \text{ g/m}^2/\text{day}$). The stream measurements made in May 1982 (stream flow approximately $1 \text{ m}^3/\text{s}$) (Table 2) were used to estimate average stream width and velocity. Stream length was measured on a USGS 15 minute quadrangle map. These values were used to estimate leaf-water contact time (water travel time), stream surface area, leaf loading, and leaf mass to water volume ratio. Then, using the information in Figures 18 and 19 water color was estimated at 15 units. This is quite close to the water color (20 units) actually measured in the Seaside watershed during the peak leaf fall period in 1981.

The second calculation was made to estimate a maximum water color level for this stream. This calculation was done with the same values as used in the first calculation except the stream flow was taken as $0.1 \text{ m}^3/\text{s}$ (4 cfs), the lowest average monthly flow found in the records for the South Fork Necanicum River (August 1979). The stream velocity and surface area, and thus the estimates of contact time and leaf loading, were somewhat off for this calculation since the velocity and stream width used were measured at a flow of $1 \text{ m}^3/\text{s}$. The color estimate under these conditions of low flow and high leaf fall was approximately 110 units (Appendix 5).

There are, however, other variables affecting water color such as the removal of color by uptake by microorganisms, and other sources of color besides direct leaf fall. There were also a number of assumptions and simplifications made. Therefore, the calculation

is not meant to be a prediction of color but rather to demonstrate the potential for alder leaves to cause visible coloration of stream water under natural conditions. There is certainly a strong likelihood that this could occur in a small stream where the flows are much lower with more pooling and the same leaf input.

Based on this study and the existing literature, a stream the size of the South Fork of the Necanicum River appears to normally be too large to have a chronic color problem due to autumn leaf inputs. The water color observed on October 27, 1981 at the water works building tap in Seaside, however, was easily visible and could be considered objectionable by people used to relatively colorless water. Also, during the November 11 storm flow water color on the 3 tributaries ranged from 30-50 units although color at the diversion was only 25 units. The highest color probably occurs at the beginning of storms when the flow is high enough to flush out storage sites but not too high to cause dilution. Water color in this stream can become great enough to be easily noticeable but it appears that this condition would normally occur only a few times a year and would be short lived. Also, at least in Seaside, if the suspended sediment load leads to excessive turbidity, an alternate water supply is used. This does not mean that a chronic color problem cannot occur in other Northwest watersheds in the fall. Given the right conditions, chronic (more than a few days) water coloration due to alder leaf inputs in streams is a distinct possibility.

VI. MANAGEMENT CONSIDERATIONS

It does not appear that water coloration is a major problem during normal autumn low flows in the Seaside watershed. Chronic color increases in the fall may be a problem in other watersheds dominated by red alder if conditions of stream flow and velocity are considerably lower than those observed at Seaside, or if other local conditions increase the contact time or leaf mass to water volume ratio. Taste and odor problems associated with increased quantities of dissolved organics may also occur at Seaside during the autumn (personal communication, Seaside Water Works Department).

Although color and related problems do not appear serious for the Seaside watershed, harvesting of the riparian alder stand has been suggested as a means of eliminating them. In fact, some streamside alders just above the city diversion were harvested in 1982. There are some important things to consider, however, in taking such an approach.

Color, taste, and odor are primarily of concern to domestic water quality (human consumption). Cutting trees may alleviate these problems but could cause changes in the stream environment that could result in domestic water quality problems as bad or worse than the original problems. In addition, such changes can be of considerable concern to stream ecologists or fisheries biologists.

Removal of streamside trees would open up the stream to sunlight in the summer and fall. Since the canopy cover over the stream in the Seaside watershed averaged 94%, this is likely to produce a large change in the amount of sunlight reaching the stream. If the trees were completely removed along the banks of the main

channel between the diversion and the confluence with the west tributary (approximately 800 m or 2640 ft), for example, the increase in stream temperature could be as much as 5.6°C (10°F) (Appendix 6). A stream temperature of 14.2°C (57.5°F, the highest stream temperature measured in the watershed, September 9, 1981) could be increased to about 20°C (68°F). The increase in temperature and light could cause an increase in the growth of algae and mosses. This was evident when the site was revisited in the late summer of 1982 after some of the alders near the diversion had been removed. Filamentous growth covered the rocks in this area that had not been evident the year before when trees were present. The growth was absent just upstream where the riparian stand was still intact.

The growth of algae, while not likely to cause color problems because of the probable rapid uptake of dissolved organic matter released from algae (Personal communication, Clifford Dahm Department of Fisheries and Wildlife OSU), could cause taste and odor problems and could affect the clarity or turbidity of the water. Such problems would likely occur as the algae bloom and die back.

In terms of environmental water quality, a temperature increase and increased algal growth could cause a decrease in stream DO. A temperature change similar to the one described would cause a drop on the saturation DO concentration of over 1.0 mg/l (Table 6). An increase in primary production due to the higher temperature and light, while enhancing the DO through photosynthesis during the day, would increase the demand for oxygen at night when only respiration is occurring. In addition, when the algae and mosses died their tissue could exert a substantial oxygen demand. Elevated

temperatures and decreased DO concentrations or both could stress the stream organisms present, especially those with narrow tolerance ranges for temperature and dissolved oxygen (from invertebrates to anadromous fish).

Other factors to consider with respect to removal of streamside alder trees are the regrowth of the trees and the release of understory shrubs and herbaceous vegetation. If the alder are not inhibited in some way (e.g., chemical treatment) following cutting, they will probably resprout and leaf inputs could return to pre-cutting levels within 3 to 5 years (Zavitskovski and Newton, 1971). If, however, the alder were effectively removed, the understory vegetation could be released and dominate stream organic inputs, possibly supplying enough material to cause increased water color. This scenario would depend on timing of the inputs, rate of release of dissolved organic matter, and rate of uptake of dissolved organic matter.

Perhaps a more attractive management alternative to clearing the streamside alder would be to underplant shade tolerant conifers (e.g., spruce and hemlock) which could be carefully released through selective cuttings of the alder. The shade (temperature and light) problem could be minimized with selective cutting. The alder stumps would still require some type of treatment, however, since they would likely resprout and suppress conifers that are planted. This may be a more attractive alternative not only because deciduous trees are being replaced with conifers but also because monetary gains are likely to be greater if the conifers are eventually harvested commercially.

The riparian zone is a diverse and dynamic system, where the terrestrial and aquatic environments overlap.

Any changes in the terrestrial environment will probably be reflected in the aquatic environment so great care must be taken when attempting to manage riparian zones, especially those affecting domestic water supplies.

VII. SUGGESTIONS FOR FURTHER RESEARCH

- 1) A survey of small coast range water supplies in Oregon and perhaps Washington could supply information on the extent and severity of the problem. Some important questions to be answered would be: 1) are there complaints about water color during the late summer and autumn months?, 2) how many complaints are received?, 3) what is the appearance of the water when complaints are made? and 4) what are stream flow conditions when complaints are received?
- 2) More intensive field study on a smaller stream (less flow) or a small reach of a smaller stream on a daily or perhaps diurnal basis. Focus on daily or diurnal changes in water color, fate of colored materials, or sources of color other than alder leaves (eg. soil and foliage from herbs and shrubs).
- 3) Field studies investigating the effects of elevated water color (increased organic loading) on disinfection (chlorine demand) and the resulting concentration of the microbial population in a public drinking water supply.
- 4) Field studies on the validity of using ultra violet absorbance to predict DOC concentration in systems dominated by red alder leaves.

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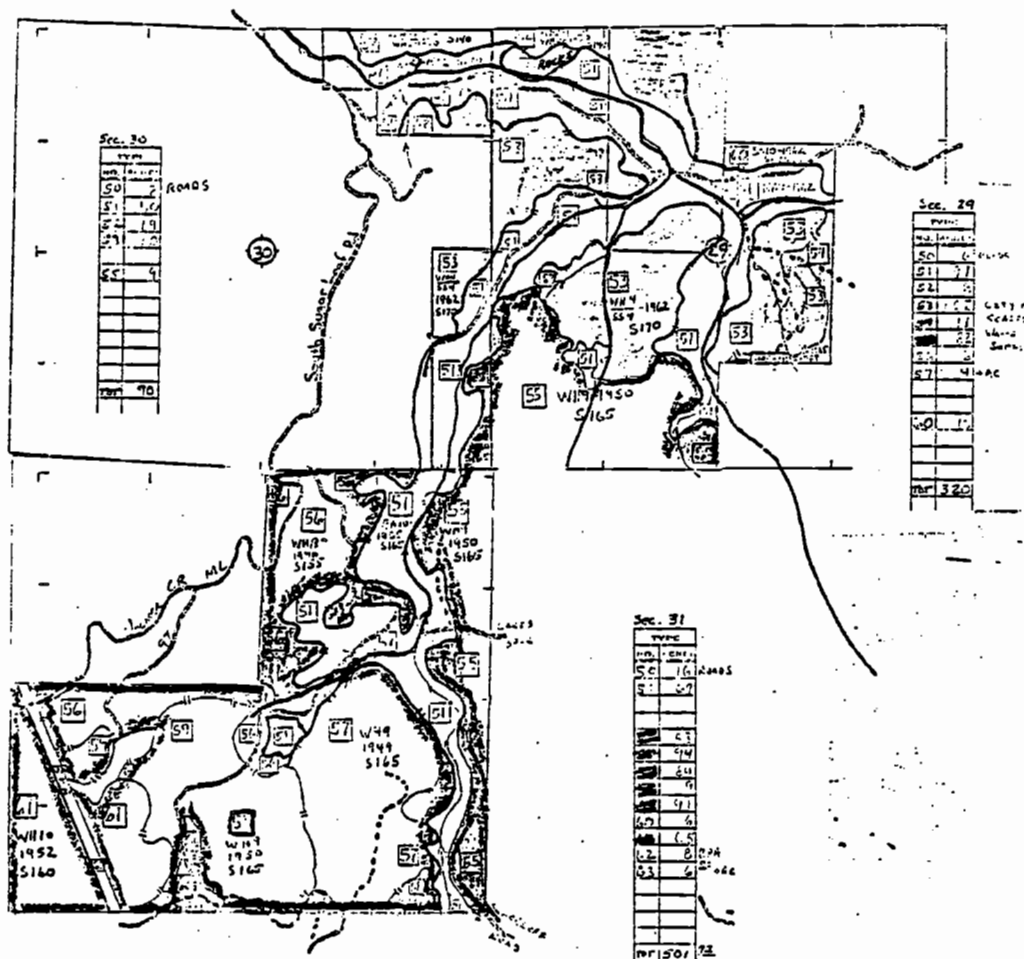
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APPENDICES

APPENDIX 1

Overstory type map from Crown Zellerbach and City
of Seaside Municipal Watershed Management Plan
T.5N., R.9W., Sec. 29,30,31



RA9 - Timber type (red alder 90% normal stocking)
S170 - Site index
1962 - Approximate date of previous logging.

APPENDIX 2

Spot check field cruise of riparian zone,
July 1982, Seaside Municipal Watershed,
Clatsop Co., Oregon

PLOT #	GENERAL PLOT LOCATION	# OF "N" TREES (20 CAF)	CIRCUMFERENCE OF SELECTED "N" TREES (IN)	DBH OF SELECTED "N" TREES (IN)	HEIGHT (FT)	AGE RANGE (YEARS)	ACD (% COVER)
1	ACROSS FROM STATION 15 ON RIGHT BANK	1 1/2 111 1/2 1/2 1/2 (7)	2.67, (2.71) 2.37 2.84, 3.28	10.2, 10.4, 9.0, 10.8, 12.5	80	27	—
3	ACROSS FROM STATION 3 RIGHT BANK	TH 1 1/2 (6 1/2)	2.63, (2.53) 1.57 (3.61), 2.70	10.0, 5.8, 6.0 13.8, 10.3	81	26, 28	96.4
3a	ABOUT 1/2 WAY DOWN	111 1/2 111 (6 1/2)	2.13, 1.49, 2.83 (2.46)	8.1, 5.7, 10.8, 9.4	72 (ESTIMATE)	26	96.9
5	ACROSS FROM STATION 5 RIGHT BANK	1/2 111 1/2 1 (5)	1.23, 0.74, 1.70 1.77	4.7, 2.7 6.5, 6.8	69 (ESTIMATE)	21	97.2
7	ACROSS FROM LAND TRAP STA. 7 RIGHT BANK (14M)	1/2 1/2 1111 (14M) TH	2.71, (2.33) 2.77 3.48, 2.98, 2.77	10.4, 13.3 11.4, 12.5	68-72 (ESTIMATE)	24	93.0
14	ACROSS FROM STA 14 ABOUT 30 FT. FROM STREAM RIGHT BANK	(14M) TH	3.05, 2.51, 2.80 2.67	11.6, 9.6 6.9, 10.2	68	25	95.1
12	ACROSS FROM STA 12 20 FT. FROM STREAM LEFT BANK	TH 11 (7)	1.60, 1.39, 1.90 1.86, 4.37	6.1, 5.3, 7.2 7.1, 16.7	65-80 (ESTIMATE)	22	95.4
13a	ABOVE STA. 13 100-200 FT. ON LEFT BANK	11 1/2 1 1/2 11 11 1/2 (9 1/2)	0.94, 1.21, 1.91 1.53, 1.58, 2.24	3.6, 4.6, 7.3 5.8, 6.0, 8.46	60-80 (ESTIMATE)	22	95.8
8	ACROSS FROM STATION 8 LEFT BANK ON HILL	111 1/2 111 (6 1/2)	1.70, 2.48, 1.73 2.41, 3.08	6.5, 9.4, 6.6 9.2, 11.8	72	26	86.4
10	ABOVE STA. 10 ABOUT 1/2 MI. ON LEFT BANK	TH 1/2 11 1/2 111 (11)	(3.35) 2.90, 2.48 1.52, 2.56, 2.46, 3.15	15.1, 11.1, 9.4 5.8, 9.8, 9.4, 11.0	60-80 (ESTIMATE)	22	95.7
11	ACROSS FROM STATION 11 RIGHT BANK	TH 111 1/2 (8 1/2)	1.13, 2.33, 2.53 2.43, 2.10	4.3, 8.9, 9.8 9.3, 8.0	60-80 (ESTIMATE)	21	96.7
16	STATION 16 AT LAND TRAP LOCATION	1/2 TH 1/2 1 (7)	2.25, 0.92, 1.32 (2.62) 3.63	8.6, 3.5, 5.0 10.0, 13.9	68	19	88.1
15	STATION 15 AT LAND TRAP LOCATION	TH 1 (6)	0.36, 0.75 0.79, (1.31)	1.4, 2.9, 3.0 5.0	30-40 (ESTIMATE)	12	93.6
							89.9 ²

¹ Circled observations indicate trees used for height measurements. Tree height was sometimes estimated from average canopy height because it was difficult to distinguish tops of individual trees. Problems were not anticipated from this because trees within a given area were very similar in height.

² Taken at water/leaf sampling Station 13, no plot done here.

APPENDIX 3. Water quality data for pools in direct contact with stream flow (2P, 5P, 9Pa) sampled autumn 1981, Seaside Municipal Watershed, Clatsop Co., OR

Date	Color (units)			Conductivity (μ mhos/cm)			pH			Dissolved O ₂ (mg/l)		
	Sampling points											
	2P	5P	9Pa	2P	5P	9Pa	2P	5P	9Pa	2P	5P	9Pa
Sep 9	--	--	20	--	--	51	---	---	7.0	---	---	7.2
Sep 16	20	--	20	51	--	52	7.0	---	7.2	9.9	---	9.5
Sep 23	20	--	30	51	--	50	7.2	---	7.2	11.4	---	8.9
Sep 30	--	20	20	--	40	40	---	---	7.0	10.6	10.9	11.0
Oct 6	--	--	--	--	--	--	---	---	---	---	---	---
Oct 12	10	--	10	42	--	42	7.0	---	7.0	11.6	---	11.5
Oct 21	5	15	15	46	46	47	7.1	7.1	7.1	11.6	11.5	11.3
Oct 28	20	20	20	35	35	35	6.8	6.8	6.8	10.6	11.0	11.2
Nov 4	10	10	10	44	42	45	7.1	7.1	7.1	11.9	12.0	11.8
Nov 11	25	25	40	44	44	39	7.0	7.0	6.9	10.6	10.8	10.6
Nov 18	10	15	15	39	39	40	6.8	6.9	7.0	11.6	11.6	11.1
Nov 24	10	15	10	41	41	41	7.0	7.1 ¹	7.1	12.1	11.8	12.1
Dec 2	20	--	15	32	--	34	6.7	---	6.7	11.7	---	11.7

¹ pH data for this date actually from sample point 5S, which was expected to be very similar to 5P.

APPENDIX 3. Water quality data for pools with indirect contact with stream flow
(9Pb, 13Pa, 16p) sampled autumn 1981, Seaside Municipal Watershed
Clatsop Co., OR

Date	Color (units)			Conductivity (μ mhos/cm)			pH			Dissolved O ² (mg/l)		
	9Pb	13Pa	16P	9Pb	13Pa	16P	9Pb	13Pa	16P	9Pb	13Pa	16p
Sep 9	10	10	10	51	43	44	7.0	6.5	7.0	7.4	3.4	8.5
Sep 16	15	20	10	49	55	45	7.1	6.8	7.1	8.1	1.4	8.9
Sep 23	20	20	20	51	56	45	7.2	7.0	7.0	8.4	7.2	7.9
Sep 30	20	15	15	41	44	35	7.0	7.1	6.9	10.8	10.7	11.1
Oct 6	--	--	--	--	--	--	---	---	---	---	---	---
Oct 12	10	10	--	42	47	--	7.0	7.2	---	11.5	11.4	---
Oct 21	20	5	15	47	48	41	6.9	7.0	7.0	10.8	11.3	11.3
Oct 28	20	25	20	36	38	30	6.8	6.9	6.7	11.1	10.8	10.9
Nov 4	10	10	10	44	46	40	7.1	7.1	7.1	11.6	11.7	11.3
Nov 11	40	30	50	39	43	36	7.0	7.0	6.7	10.5	10.8	10.6
Nov 18	10	15	10	40	41	35	7.0	7.1	6.9	11.5	---	11.5
Nov 24	10	10	10	41	44	36	7.1	7.2	7.0 ¹	11.9	12.0	11.8
Dec 2	15	10	5	34	36	--	6.7	6.9	6.6 ¹	11.7	11.7	11.4

¹ pH data for these dates actually from 16S, which were expected to be very similar to 16P.

APPENDIX 3. Water quality data for isolated pools (13Pb, 14Pa, 14Pb) sampled during autumn 1981, Seaside Municipal Watershed, Clatsop Co., OR

Date	Color (units)			Conductivity (μ mhos/cm)			pH			Dissolved O ² (mg/l)		
	Sampling Points											
	13Pb	14Pa	14Pb	13Pb	14Pa	14Pb	13Pb	14Pa	14Pb	13Pb	14Pa	14Pb
Sep 9	15	20	50	53	57	48	6.8	6.6	6.4	3.4	3.2	1.2
Sep 16	10	20	70	41	50	55	7.0	7.2	7.1	3.5	2.8	0.1
Sep 23	15	25	40	47	64	50	6.8	6.7	6.4	6.0	2.8	3.0
Sep 30	15	25	30	47	45	45	6.9	6.7	6.5	9.8	9.4	7.1
Oct 6	--	--	--	--	--	--	---	---	---	----	---	----
Oct 12	5	15	10	48	44	40	7.0	6.8	6.7	10.1	8.8	8.9
Oct 21	15	25	30	48	57	48	7.1	7.0	6.3	9.8	6.1	3.8
Oct 28	20	25	25	39	40	34	6.9	6.6	6.5	8.7	8.9	10.2
Nov 4	10	20	30	45	42	44	6.8	6.4	6.3	10.6	8.8	5.8
Nov 11	35	100	90	44	54	40	6.7	6.4	6.1	9.7	8.1	6.3
Nov 18	10	15	10	42	38	36	6.9	6.2	6.6	11.6	9.8	11.3
Nov 24	10	10	15	45	40	38	6.7	6.2	6.3	11.0	9.2	11.2
Dec 2	10	15	20	37	31	31	6.9	6.5	6.6	11.7	11.2	11.2

APPENDIX 4. Water color for stream stations sampled during autumn 1981, Seaside Municipal Watershed, Clatsop Co., OR

Date	Sampling Points												
	1S	3S ¹	5S	6S	7S	8S	10S	11S	12S	13S	14S	15S	16S
Sep 9	20	15	15	10	30	15	15	15	10	10	30	5	15
Sep 16	15	15	15	15	30	20	15	10	10	10	30	5	10
Sep 23	20	20	25	20	30	20	20	20	10	10	30	10	20
Sep 30	20	20	20	20	25	20	20	15	15	15	20	10	15
Oct 6	30	--	--	--	--	--	30	20	--	--	--	25	--
Oct 12	10	10	10	10	15	10	10	10	10	10	15	5	10
Oct 21	20	15	15	15	15	15	15	15	15	10	20	5	15
Oct 28	20	--	20	20	25	20	20	20	20	20	25	20	20
Nov 4	10	--	10	10	20	10	10	10	5	5	20	0	5
Nov 11	25	--	30	35	40	50	40	40	30	30	40	35	50
Nov 18	10	--	15	10	10	15	15	10	15	10	10	10	10
Nov 24	10	--	15	10	10	10	10	5	10	10	10	10	10
Dec 2	20	--	15	20	15	15	15	15	15	10	20	10	20

¹From October 28 on Station 3S was not sampled because it was so similar to Station 1S in water quality.

APPENDIX 4. Dissolved oxygen (mg/l) of sampled stream water during autumn 1981, Seaside Municipal Watershed, Clatsop Co., OR

Date	Sampling Points												
	1S	3S	5S	6S	7S	8S	10S	11S	12S	13S	14S	15S	16S
Sep 9	10.9	9.8	10.3	9.5	8.7	9.2	9.8	9.7	9.5	--	8.5	9.9	9.7
Sep 16	10.8	11.2	9.6	11.4	9.2	10.4	11.0	11.3	10.3	9.7	10.3	10.2	10.5
Sep 23	9.9	10.8	10.3	9.6	10.4	8.7	10.1	10.8	10.0	9.6	10.6	8.4	9.8
Sep 30	10.4	11.0	10.8	10.8	10.7	11.1	11.2	11.0	11.4	11.2	10.7	10.9	10.6
Oct 6	----	----	----	----	----	----	----	----	----	----	----	----	----
Oct 12	11.7	11.9	11.6	11.6	12.0	11.6	11.6	11.6	11.4	11.5	12.5	11.3	11.5
Oct 21	11.8	11.5	11.5	11.6	11.4	11.6	11.6	11.5	11.4	11.5	11.3	11.2	11.3
Oct 28 ¹	10.4	----	10.8	10.6	----	10.8	----	----	10.9	----	10.6	11.0	11.0
Nov 4	11.7	----	12.0	11.6	11.6	11.9	----	----	11.8	----	----	11.7	11.6
Nov 11	10.5	----	10.8	10.5	10.2	10.9	----	----	10.5	----	----	10.9	10.7
Nov 18	11.6	----	11.6	11.4	11.1	11.6	----	----	11.7	----	----	11.7	11.6
Nov 24	12.1	----	12.2	12.0	11.9	12.0	----	----	12.0	----	----	11.8	11.8
Dec 2	11.5	----	11.6	11.6	11.5	11.9	----	----	11.7	----	----	11.7	11.8

¹From October 28 on only certain stations were sampled for dissolved oxygen because the dissolved oxygen seemed to remain fairly constant among stations in the same stream reach. Therefore, Station 3S was assumed to be the same as 1S, Stations 10S and 11S the same as 8S, Station 13S the same as 12S, and Station 14S the same as 7S, from October 28 on.

APPENDIX 4. pH of stream water sampled during autumn 1981, Seaside Municipal Watershed, Clatsop Co., OR.¹

Date	Sampling Points												
	1S	3S	5S	6S	7S	8S	10S	11S	12S	13S	14S	15S	16S
Sep 9	7.2	7.1	7.2	---	7.1	7.1	---	---	7.2	---	---	7.2	7.1
Sep 16	7.1	---	7.2	---	7.0	7.2	---	---	7.2	---	---	7.3	7.2
Sep 23	7.2	---	7.2	---	6.9	7.2	---	---	7.3	---	---	7.2	7.1
Sep 30	6.9	---	6.9	---	6.8	7.0	---	---	7.1	---	---	7.1	7.0
Oct 6	6.6	---	---	---	---	---	6.6	6.7	---	---	---	6.8	---
Oct 12	6.9	---	7.1	---	6.9	7.0	---	---	7.2	---	---	7.2	6.9
Oct 21	7.1	7.0	7.1	6.9	6.8	7.1	7.1	7.0	7.2	7.2	6.8	7.1	7.1
Oct 28	6.8	---	6.9	6.8	6.6	6.8	6.9	6.8	6.9	6.8	6.7	7.0	6.8
Nov 4	6.9	---	7.1	7.1	6.9	7.1	7.1	7.0	7.2	7.2	6.9	7.1	7.1
Nov 11	6.9	---	7.0	6.9	6.8	6.9	6.9	6.8	7.0	7.0	6.8	7.0	6.7
Nov 18	6.7	---	6.9	6.8	6.7	7.0	7.0	6.9	7.0	7.0	6.7	7.1	6.9
Nov 24	7.0	---	7.1	---	6.9	7.1	---	---	7.2	---	---	7.1	7.0
Dec 2	6.7	---	6.8	---	6.6	6.7	---	---	6.9	---	---	6.8	6.6

¹Because of the similarity in pH between stations in the same stream reach, not all samples were analyzed for pH. It was assumed that the following stations had the same pH: 1S and 3S; 5S and 6S; 7S and 14S; 8S, 10S and 11S; 12S and 13S. pH measurements were not begun until October and it was after a few weeks of analyzing for pH that it was decided to only analyze certain samples.

APPENDIX 4. Electrical conductivity ($\mu\text{mhos/cm}$) of stream water sampled during autumn 1981, Seaside Municipal Watershed, Clatsop Co., OR

Date	Sampling Points												
	1S	3S ¹	5S	6S	7S	8S	10S	11S	12S	13S	14S	15S	16S
Sep 9	54	52	51	52	49	51	51	45	52	53	50	52	44
Sep 16	48	51	50	47	50	49	51	45	51	51	50	52	44
Sep 23	52	52	51	52	51	50	50	46	52	51	51	55	45
Sep 30	42	41	41	43	41	40	41	38	44	43	42	47	34
Oct 6	30	--	--	--	--	--	28	28	--	--	--	30	--
Oct 12	42	42	42	43	39	42	42	39	46	46	37	45	37
Oct 21	46	46	46	46	42	47	47	42	48	48	41	49	41
Oct 28	34	--	35	35	36	35	35	33	38	38	34	39	30
Nov 4	44	--	43	45	37	44	42	40	50	45	37	45	40
Nov 11	46	--	43	41	38	40	39	35	44	43	37	41	37
Nov 18	38	--	39	39	36	40	39	36	42	42	36	42	35
Nov 24	43	--	41	41	36	41	41	38	44	44	36	43	36
Dec 2	32	--	33	33	30	33	34	31	36	36	30	35	28

¹From October 28 on Station 3S was not sampled because it was so similar to Station 1S in water quality.

APPENDIX 5

Estimated water color at City of Seaside diversion
under conditions of moderate and low flow and high
leaf

- 1) Maximum travel time of in-channel water (contact
time of water with leaves)

The longest stream distance in the watershed
is from the diversion to the uppermost reach
of the west tributary. This distance was es-
timated from a USGS 15' quadrangle at 6,915 m
(22,680 ft).

The water velocity was taken as the average of
the velocities measured at the mouths of the
three main tributaries in May 1982, 0.4 m/s.
This is a weighted mean using the percent of
the total flow ($1.0 \text{ m}^3/\text{s}$) found in each tribu-
tary as the weights:

$$\begin{aligned} \text{Ave.} \\ \text{Velocity (V)} &= \\ &= \frac{(0.29 \text{ m/s})(0.1) + (0.53 \text{ m/s})(0.52) + (0.30 \text{ m/s})(0.38)}{0.1 + 0.52 + 0.38} \end{aligned}$$

$$V = 0.4 \text{ m/s}$$

Therefore,

$$\begin{aligned} \text{max. travel} \\ \text{time (t)} &= (6915 \text{ m}) \times (1 \text{ s} / 0.4 \text{ m}) \times (1 \text{ hr} / 3600 \text{ s}) \end{aligned}$$

$$t = \underline{\underline{5 \text{ hr}}}$$

- 2) Leaf loading to stream

- a) Stream surface area

Measurements of stream width and depth were
made at the same time as the velocity meas-
urements (May 1982, flow = $1.0 \text{ m}^3/\text{s}$).

- mean channel width, streams \geq 3rd order
= 6.8 m

- mean channel width, streams < 3rd order
= 1.8 m

- channel length, streams \geq 3rd order¹
= 9,453 m
- channel length, streams < 3rd order¹
= 50,122 m

$$\text{Weighted mean stream width (w)} = \frac{(6.8\text{m})(9453\text{m}) + (1.8\text{m})(50,122\text{m})}{(9453\text{m}) + (50,122\text{m})}$$

$$w = 2.6 \text{ m}$$

$$\text{Total stream channel length}^1 = 59,575 \text{ m}$$

$$\text{Total stream surface area (a)} = (2.6\text{m})(59,575\text{m})$$

$$a = \underline{1.55 \times 10^5 \text{ m}^2}$$

b) Leaf input

Highest leaf input rate observed during autumn 1981 was assumed:

$$\underline{11.7 \text{ g/m}^2 \text{ day}}$$

(Oct 21-28, 1981). Also assumed that low flows prevail for a week and leaves thus accumulate in channel for 7 days.

$$\text{leaf input} = (11.7 \text{ g/m}^2/\text{day})(7 \text{ days}) = \underline{81.9 \text{ g/m}^2}$$

$$\text{Total Stream Leaf Loading (m)} = (81.9\text{g/m}^2)(1.55 \times 10^5 \text{ m}^2)$$

$$m = \underline{1.27 \times 10^7 \text{ g}}$$

3) Water volume during moderate flow

The volume of water receiving the above leaf mass was assumed to equal the volume of water flowing at the diversion over a 5 hour period. The flow during the above stream measurements was $1.0 \text{ m}^3/\text{s}$, which is similar to the flow during peak leaf fall in 1981.

$$\begin{aligned} \text{Water volume (}\ell\text{)} &= (1.0\text{m}^3/\text{s})(1000\ell/\text{m}^3)(3600\text{s/hr}) \\ &= \underline{1.8 \times 10^7 \ell} \quad (5 \text{ hr}) \end{aligned}$$

¹From USGS 15' quadrangle

4) Leaf mass to water volume ratio

If the leaf mass (m) is evenly distributed along the channel so that the water encounters a constant mass as it travels:

$$\frac{\text{leaf mass(m)}}{\text{water vol(l)}} = \frac{1.27 \times 10^7 \text{ g}}{1.8 \times 10^8} = \underline{0.7 \text{ g/l}}$$

5) Estimated water color at moderate flow

From the relationship in Figure 19 ($y = 39x + 8$), the color generated in still water after 48 hours is:

$$\text{for } 0.7 \text{ g/l, } y = 39(0.7) + 8 = \underline{35 \text{ units}}$$

$$\text{for } 2.2 \text{ g/l, } y = 39(2.2) + 8 = \underline{94 \text{ units}}$$

From the relationship in Figure 17 ($y = 2x + 2.6$), the color generated from 2.2 g/l loading after 5 hours is 12.6 units ($y = 2(5) + 2.6$).

However, since actual and predicted color values show a more significant disparity with shorter leaching times, Figure 18 was instead used to obtain an estimated color level of 20 units for 2.2 g/l loading after 5 hours in still water.

Assuming a uniform relationship to color between the 0.7 g/l and 2.2 g/l loading levels, the estimated color generated from 0.7 g/l after 5 hours in still water is therefore:

$$(20 \text{ units})(0.37^*) = \underline{7.4 \text{ units}}$$

$$*0.37 = \frac{35 \text{ units (0.7 g/l color @ 48 hour)}}{94 \text{ units (2.2 g/l color @ 48 hour)}}$$

Finally, since Figure 18 also shows that for 5 hours of leaching, running water produces about twice the color level of still water, the estimated color for the assumed high leaf input and moderate streamflow is:

$$2 \times 7.4 \text{ units} = \underline{15 \text{ units}}$$

This estimated color level is consistent with the stream color measured during October 1981, when the flows and leaf inputs used in the calculation actually occurred.

6) Estimated water color at low flow

In this calculation the previously-used leaf input level was assumed, as were the stream dimension and velocity values, since low flow measurements were not made.

Stream flow was assumed to be $0.1 \text{ m}^3/\text{s}$ (4 cfs), which represents the lowest average monthly flow observed during the five years of record (Aug. 1979). This represents the following volume of water flowing past the city diversion over five hours:

$$\begin{aligned} \ell &= (0.1 \text{ m}^3/\text{s})(1000 \text{ } \ell/\text{m}^3)(3600 \text{ s/hr})(5 \text{ hr}) \\ \ell &= \underline{2.04 \times 10^6 \ell} \end{aligned}$$

The leaf mass to water volume ratio is therefore:

$$\frac{\text{leaf mass (m)}}{\text{water vol } (\ell)} = \frac{1.27 \times 10^7 \text{ g}}{2.04 \times 10^6 \ell} = \underline{6.2 \text{ g}/\ell}$$

Using the above procedure to estimate color under these conditions:

$$\begin{aligned} 6.2 \text{ g}/\ell \text{ still water color} &= 39(6.2)+8 = 250 \\ @ 48 \text{ hr (Fig. 19)} & \quad \quad \quad \underline{\text{units}} \end{aligned}$$

$$\begin{aligned} 6.2 \text{ g}/\ell \text{ running water} &= (20^*)(2.7^{**})(2^{***}) = \\ \text{color @ 5 hour} & \quad \quad \quad \underline{108} \\ & \quad \quad \quad \underline{\text{units}} \end{aligned}$$

$$*20 = 2.2 \text{ g}/\ell \text{ color @ 5 hours (Figure 18)}$$

$$**2.7 = \frac{250 \text{ units (6.2 g}/\ell \text{ color @ 48 hr)}}{94 \text{ units (2.2 g}/\ell \text{ color @ 48 hr)}}$$

$$***2 = \text{running water color is 2x still water color @ 5 hr}$$

NOTE: Since many assumptions and simplifications were made, this calculation is unlikely to be a precise prediction of color level. It is instead intended to demonstrate the potential for alder leaves to cause significant water color on the Seaside watershed.

APPENDIX 6

Estimation of stream temperature change if streamside alders were removed for the half mile above the city of seaside diversion

The prediction equation that was used was from Brown (1980):

$$T(^{\circ}\text{F}) = \frac{\Sigma H \times A}{Q} \times 0.000267$$

Where,

ΣH = net solar radiation
 A = stream surface area receiving radiation
 Q = stream flow

1) Stream surface area (A)

Length of cut = 0.5 mi. = 2640 ft.
 Average stream width = 34 ft..
 Average stream depth = 0.7 ft.

However, since these were not measured at the lowest flow, smaller values were assumed:

Estimated average stream width = 20 ft.
 Estimated average stream depth = 0.5 ft.

Resulting surface area = 2640ft x 20 ft = 52,800 ft.

2) Stream flow (Q)

The lowest average stream flow for August for water years 1977, '78, '79 (available water years) was 4.28 Cfs

3) Net solar radiation (ΣH)

Using the flow and an estimate of the average cross section of flow;

$$\begin{aligned}\text{Stream velocity} &= 4.28 \text{ ft}^3/\text{s} \div (20 \text{ ft} \times 0.5 \text{ ft}) \\ &= 0.428 \text{ ft/s}\end{aligned}$$

Travel time of water through this section would be:

$$\begin{aligned}\text{Travel time} &= 2640 \text{ ft} \times 1 \text{ sec}/0.428 \text{ ft} \\ &= 6168 \text{ sec} = \underline{1.7 \text{ hours}}\end{aligned}$$

- 3) Approximate midday solar angle = 60 degrees
Brown (1980)

Therefore, from Figure III-5 (Brown, 1980)
the net radiation (Σ) would be approximately

$$\Sigma = 4.1 \text{ BTU/ft}^2\text{-min}$$

- 4) Estimated temperature change

The maximum temperature change that would
be expected would be:

$$T(^{\circ}\text{F}) = \frac{(4.1)(52800)}{4.28} \times 0.000267 = 10^{\circ}\text{F}$$