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Fishery biologists and watershed management specialists have recently begun to investigate the pollutional threat posed by organic debris left in or near the forest watercourses. Oxygen content in some streams has fallen below the limits required for fish survival. The biochemical oxygen demand exerted by the debris and the reaeration potential of the stream must be determined to predict the effects of debris accumulation on dissolved oxygen.

The purpose of this study is to specify equations suitable for predicting reaeration in small, turbulent streams and identify the factors affecting reaeration under these circumstances. Reaeration rates in a mountain stream were determined by depleting the oxygen content at one point with sodium sulfite and then measuring the changes in dissolved oxygen as the deaerated water moved downstream. These reaeration rates were then correlated with the hydraulic characteristics of the stream.

Reaeration in Small Mountain Streams

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REAERATION IN SMALL MOUNTAIN STREAMS

CHAPTER I

INTRODUCTION

Water quality is especially significant in Oregon's forests. The majority of Oregon's water resources lie within or originate in forested areas. Most of the state's 300 municipal watersheds are forested. Streams are typically small. Small streams are also the backbone of the region's multi-million dollar salmon and steelhead fishery. These fish seldom spawn in large rivers, but seek small streams instead. Fingerlings remain in these streams for one year before migrating to the sea. Thus, the conditions under which the eggs hatch and the young fish grow are determined by the factors affecting these small streams.

Timber harvest and the related wood processing industries are the largest contributors to the economy of the Pacific Northwest. Logging is commonly practiced on many municipal watersheds and adjacent to fish spawning streams. Because of silvicultural and economic reasons, clearcut logging is the most desirable method of harvest. This method often results in trees being felled or skidded across stream channels. Limbs, bark, and other logging debris accumulate in the stream. Fishery biologists and watershed management

specialists have recently begun to investigate the polluttional threat posed by the organic debris left in or near the forest watercourses. Preliminary results indicate taste, odor, color, and oxygen depletion problems in areas of debris accumulation. In some cases, the oxygen content of the streams has fallen well below the limits required for fish furvival.

Similar to our large rivers, small mountain streams have the ability to assimilate certain quantities of organic pollutants. In the process of natural purification, the dissolved oxygen used in stabilization of the organic matter is replaced from the atmosphere. The rate of reaeration is used as a measure of the ability of a stream to assimilate biodegradable wastes. Oxygen depletions will not occur if the reaeration rate exceeds the rate at which oxygen is used in the decomposition process.

Two factors must be known to predict the effects of debris accumulation on dissolved oxygen. First, the biochemical oxygen demand (BOD) exerted by the debris must be measured or predicted. The analysis of the BOD from logging debris is presently being conducted by other researchers. Second, the reaeration potential of the stream must be determined or predicted.

Previous investigators have suggested various models for predicting stream reaeration. The purpose of these models is to predict reaeration from easily measured hydraulic characteristics of the

stream channel such as slope, depth, velocity, etc. These models can be classified into two general categories. The first category includes those models which were developed theoretically by using the appropriate gas-transfer laws of physical chemistry. Verification of these theoretical models generally came from closely controlled laboratory experiments. The second approach was empirical and predictive equations were developed from reaeration measurements on large river systems. These studies evaluated the reaeration capacity of large rivers subjected to oxygen demands exerted by municipal and industrial wastes. These reaeration equations may also be applicable to small streams. Previous studies have not tested this possibility.

The purpose of this study is to specify equations suitable for predicting reaeration in small, turbulent streams, and identify the factors affecting reaeration under these circumstances. Three specific objectives are stated for accomplishing this purpose. First, the reaeration equations which appear in the literature will be tested for their predictive accuracy when applied to a turbulent mountain stream. Second, a new set of predictive equations will be developed which may predict reaeration more accurately than the existing equations under turbulent conditions. Third, the factors affecting reaeration will be identified and examined with regard to their management potential for improving the reaeration capacity of a

stream and reducing the chances of serious oxygen depletions.

The scope of this study will be limited to investigating the reaeration process as it occurs on a single mountain stream. The stream chosen for this study contained a variety of hydraulic conditions typical of those found in other small streams in the Oregon Coast Range. Reaeration rates in this stream were determined by chemically depleting the oxygen content at one point in the stream and then measuring the changes in dissolved oxygen as the water moved downstream. These reaeration rates were then correlated with the hydraulic characteristics of the stream. The intent is to extend the results of this study only to other small, turbulent streams where the hydraulic conditions fall within the ranges considered in this study.

CHAPTER II

LITERATURE REVIEW

A review of the literature clearly indicates the importance of dissolved oxygen in the aquatic environment. The concentration of dissolved oxygen in a natural stream is viewed by many environmental scientists as being one of the most important parameters controlling water quality. Oxygen content is used not only as a measure of the waste assimilative capacity of a stream, but also as an index to the biochemical quality required for a healthy and normal distribution of aquatic flora and fauna.

Basic to an understanding of dissolved oxygen are the principles of physical chemistry which define the solubility of oxygen in water. A discussion of these factors is presented in order to clarify the following sections. Special attention has been given to the sources and sinks of oxygen since they determine the oxygen content of the stream at any given time. A discussion of the methods used to analyze for dissolved oxygen content is also included since such measurements formed the basis of this research. Because this research is directed towards defining the reaeration capacity of small mountain streams, no discussion would be complete without a review of the previous studies and theories concerned with stream reaeration. These studies and theories have been presented in some detail since they not

only set the stage for the present research, but also offer a convenient basis of comparison for evaluating the results. By examining and defining the process of reaeration, these studies have also shown how certain hydraulic characteristics of the stream can be used to predict its reaeration capacity.

Gas Laws and the Solubility of Oxygen in Water

General concepts from physical chemistry indicate that the solubility of any gas in a liquid depends on the nature of the gas, the temperature of the liquid, and the pressure of the gas over the liquid. Oxygen may be considered a slightly soluble gas in water. Under conditions of standard pressure (760 mm of Hg) the solubility of oxygen gas in water varies from 14.6 mg/l at 0° C to 5.6 mg/l at 50° C (American Public Health Association, 1960). The effect of atmospheric pressure on oxygen solubility is well understood and can be predicted from two basic laws of physical chemistry. Dalton's law of partial pressures states that: "In any mixture of gases, the total pressure is equal to the sum of the partial pressures exerted by the individual gases. The partial pressure of any one gas is proportional to the percentage of its molecules in the total mixture" (Quagliano, 1958). It is generally accepted that the atmosphere of the earth contains 20.9 percent oxygen gas. Thus the partial pressure of oxygen in air is 20.9 percent of 760 mm of Hg or approximately 159

mm of Hg at sea level. Henry's law states that the mass of gas dissolved in a given mass of liquid is directly proportional to the pressure of the gas above the solution (Quagliano, 1958). Given these two laws, it is apparent that the solubility of oxygen in water is directly proportional to the atmospheric pressure. The effect of pressure on oxygen solubility can be calculated directly by applying the following equation:

$$C_s' = C_s \frac{P - p}{760 - p} \quad (2-1)$$

In the above equation, C_s' is the solubility of oxygen in mg/l at some stated pressure, C_s is the solubility in mg/l at a given temperature and at 760 mm of Hg, P is the barometric pressure in mm of Hg, and p is the pressure of saturated water vapor at the temperature of the water. For elevations of less than 3,000 ft. and temperature below 25°C , p can be ignored (American Public Health Association, 1960).

Unfortunately, the relationship between oxygen solubility and temperature is not quite so easily defined. In fact, there is continued disagreement between various investigators as to the true nature of oxygen solubility at a given temperature. Perhaps the most widely used and accepted saturation values of dissolved oxygen are those reported in Standard Methods (American Public Health Association, 1960). The term 'saturation value' as used here is synonymous with 'solubility'. The values found in Standard Methods are those reported

by Whipple and Whipple (1911) which were based on calculations from experimental data presented by Fox (1909). The experiments of Fox were actually performed using pure oxygen instead of air. Using a gasometric technique, he exposed a given volume of pure dry oxygen at 760 mm of pressure to a known volume of deaerated water and measured the reduction in oxygen volume after equilibrium had been established. Tests were performed at different temperatures and from this data, the solubility of atmospheric oxygen was calculated, assuming that air contained 20.90 percent oxygen. Results were reported in cubic centimeters of oxygen absorbed by 1,000 cubic centimeters of water. Using these results, Whipple and Whipple applied corrections for saturation vapor pressure and converted the results into mg/l and ppm as found in Standard Methods. The solubility of dissolved oxygen in distilled water at any pressure may be calculated by the following equation:

$$C_s = \frac{(P - p) 0.678}{35 + T} \quad (2-2)$$

In the above equation, C_s is the saturation value of oxygen in water in mg/l at some temperature, T , P is the barometric pressure in mm of Hg, p is the saturation vapor pressure of air for a given temperature T , and T is the temperature of the water between 0° and 30° C measured in degrees C (American Public Health Association, 1960).

For many years, the work of Fox and later Whipple and Whipple was accepted without further qualification. However, experiments conducted by Truesdale et al. (1955) resulted in saturation values which differed significantly from those previously reported. In general, Truesdale's values were somewhat lower. In the temperature range of 0° to 35° C, they were as much as 0.57 ppm lower or about six percent of the solubility given by Standard Methods. The following technique was used by Truesdale to obtain the published saturation values. Using a 10-liter aspirator bottle, eight liters of distilled water were added and nitrogen gas passed through the bottle for one hour to deoxygenate the water and air. With the oxygen concentration in the water reduced to well below the saturation point, CO₂ free air was pumped into the bottle at a rate of 100 cc/min. The water in the bottle was slowly stirred by a paddle wheel near the bottom to assure complete mixing and uniform oxygen concentration throughout the bottle. Subsamples of water were periodically drawn off and analyzed for dissolved oxygen. The dissolved oxygen content of the samples was determined by standard Winkler titration. The Winkler method is discussed in greater detail in a subsequent section. Sampling and analysis continued until several successive determinations taken over a period of days gave a constant value when corrected for changes in atmospheric pressure. Water temperatures during each run were held constant by immersing the aspirator bottle in a

water bath which was thermostatically controlled to $\pm 0.05^{\circ}$ C.

Through successive runs at different temperatures, Truesdale determined the oxygen solubility in both pure and saline water over a temperature range of 2° to 40° C. The results of the experiment can be expressed by an empirical equation which relates water temperature directly to oxygen solubility. Truesdale's equation is:

$$C_s = 14.161 - 0.3943 T + 0.007714 T^2 - 0.0000646 T^3 \quad (2-3)$$

where C_s is the saturation concentration in parts of oxygen per million parts of water and T is the water temperature in $^{\circ}$ C. In discussing the apparent disagreement between his results and those of previous investigators, Truesdale concludes that either the Winkler method is not accurate, or the previously published gasometric solubilities are incorrect. Truesdale states that the later explanation is more probable since super-saturation may have resulted from the vigorous agitation used to dissolve the oxygen in the experiments of Fox.

Using the results published by Truesdale et al. (1955), Mackereth (1963) proposes a simpler formula for calculating oxygen solubility. The equation is:

$$C_s = \frac{475}{(33.5 + T)} \quad (2-4)$$

where C_s is the solubility of oxygen in mg/l and T is the temperature

in ° C.

Churchill, Buckingham and Elmore (1962) discuss the discrepancy between the results of Fox and those of Truesdale. A careful study of the techniques used by the Truesdale group failed to reveal procedures that could be questioned. Furthermore, the general design of the Truesdale experiments included several advantages that were considered to be superior to the gasometric techniques used by Fox and other early workers such as Winkler. However, since Churchill, Buckingham and Elmore could find no explanation for the differences between the results of Truesdale and Fox, they felt that further investigations were necessary to resolve the conflict. The techniques used by Churchill, Buckingham and Elmore for these investigations were quite similar to those of Truesdale and will not be discussed in detail. The results of these latest studies showed close agreement with the Standard Methods values in the 0° to 10° C range and lower values in the 10° to 30° C range, but not as low as those reported by Truesdale. The best-fit empirical equation developed by Churchill, Buckingham and Elmore is:

$$C_s = 14.652 - 0.41022 T + 0.0079910 T^2 - 0.000077774 T^3 \quad (2-5)$$

where C_s is the saturation concentration of dissolved oxygen and T is the temperature in degrees C. For comparative purposes, Table E-1 has been prepared to show the saturation values predicted by the

three principle investigators, Fox, Truesdale, and Churchill, over the temperature range of 0° to 30° C.

The choice of the saturation value of oxygen for a given water temperature is important from two standpoints. It determines the direction of oxygen transfer and also the rate of transfer. When the oxygen concentration in water reaches the saturation value, the partial pressure of oxygen in the water is the same as the partial pressure of oxygen in the overlying air. Since there is no gradient of partial pressure at saturation, no net reaeration will occur and an oxygen equilibrium exists between dissolved and atmospheric oxygen. When the dissolved oxygen content in water falls below the saturation value, the partial pressure of oxygen in water becomes less than that of the atmosphere. A positive gradient is established such that the net transfer of oxygen is from air to water. The reverse can occur under conditions of high photosynthetic activity when the oxygen concentration in water rises above the saturation value. In this case, a negative pressure gradient exists and oxygen is transferred from water to air. The term generally applied to this transfer is "reaeration" or simply "aeration". These terms are somewhat of a misnomer since they imply that oxygen is being added to the water. They are used not only to describe the process by which oxygen enters the water but also to describe the loss of oxygen from the water to the air. The latter might better be termed "deaeration". However, to avoid

confusion, "reaeration" and "aeration" will be used to describe any gas transfer between water and atmosphere. The saturation value is also related to the rate of oxygen transfer because it defines the partial pressure gradient at the air-water interface. This gradient is directly related to the reaeration rate. The effect of the gradient on the reaeration rate will be discussed more fully in a following section.

Sources and Sinks of Dissolved Oxygen

The concentration of dissolved oxygen which exists in a natural stream at any time is a function of a number of interrelated factors and processes. The influence of any of these factors or processes varies with time and dissolved oxygen content. For discussion purposes, these variables can be classified as either sources or sinks of dissolved oxygen.

Typical sources are oxygen released into the water from photosynthetic activity of algae and other green plants and oxygen from atmospheric reaeration. Sinks of dissolved oxygen result from the respiration of bacteria, algae and green plants, the loss of oxygen to the atmosphere, and the use of oxygen to satisfy biochemical oxygen demands resulting from pollutional loading and bottom deposits. These sources and sinks are shown in a diagrammatic form in Figure 2-1.

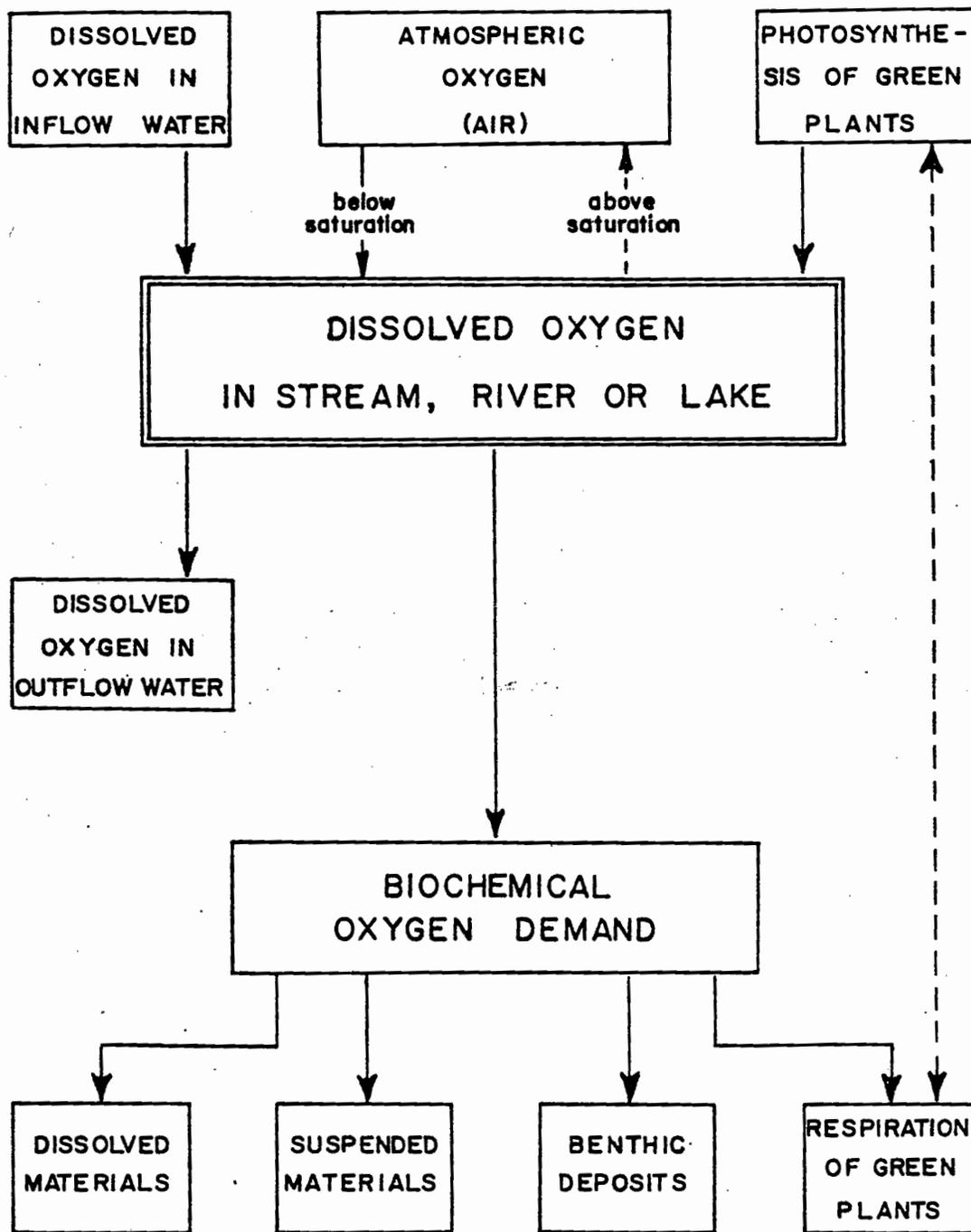


Figure 2-1. Sources and sinks of dissolved oxygen.

O'Conner (1967) uses a mass balance approach to define the oxygen balance in any given stream system. To use this conceptual model, a given volume of water or section of the stream is specified to spatially define the system. All the inputs and outputs of oxygen to the system are then balanced. Because water flowing into and out of the stream section or system represents an input or output of dissolved oxygen, it has been shown in Figure 2-1 as a source or sink of dissolved oxygen.

Algae and green plants act as both sources and sinks for dissolved oxygen. During the daylight hours, the oxygen produced by plants through photosynthesis usually exceeds their use of oxygen for respiration, and hence they are sources of dissolved oxygen. At night, photosynthesis ceases but respiration continues and plants become sinks for dissolved oxygen. The general effects of large plant and algae populations are large diurnal variations in dissolved oxygen. The net effect of plants and algae over a 24-hour period depends upon the specifics of the environment such as sunlight, temperature, mass of algae, and nutrients (O'Conner, 1967). In studies on the Fox River, Wisconsin, Wisniewski (1958) found daily net photosynthetic/respiration ratios of 1.4. Thus on a net daily basis, algae under these conditions were definitely a source of oxygen. In discussing Dobbins's (1964) article on BOD and oxygen relationships in streams, Lynch (1965) concludes that respiration and photosynthesis of plankton

and fixed plants may be a major if not dominating factor in certain stream situations. O'Connell and Thomas (1965) clearly demonstrate the truth of Lynch's statement. Their studies of benthic algae in the Truckee River near Reno, Nevada show the large diurnal fluctuations resulting from high algae populations and the dominating influence of algae on the stream's dissolved oxygen content. They conclude that photosynthetic oxygen production will be a liability rather than an asset if it results from benthic algae or rooted aquatic plants rather than phytoplankton. Nighttime respiration requirements can cause seriously low daily minimum dissolved oxygen concentrations. To support this statement, they cite studies by Warren et al. (1964) which indicate that the growth rates of fish subjected to fluctuating dissolved oxygen concentrations are dependent on the minimum concentrations occurring at night or early morning rather than on daily mean concentrations. Besides producing serious nighttime depletions, algae cause abnormally high mid-day oxygen concentrations. Gameson and Barrett (1958) discuss cases where dissolved oxygen concentrations as high as 150 percent of the air-saturation values have occurred as a result of photosynthetic activity.

The exchange of oxygen between the atmosphere and the water is still considered to be the predominant mode of transport of oxygen into the water. However photosynthesis and respiration may exert a considerable influence on dissolved oxygen under certain circumstances.

The atmosphere can act as either a source or sink for dissolved oxygen. The direction of transfer will depend on the relationship between the saturation value and the actual oxygen concentration in the water.

In polluted streams, biochemical oxygen demand is a major sink for dissolved oxygen. Biochemical oxygen demand (BOD) is defined as the amount of oxygen required by bacteria while stabilizing decomposable organic matter under aerobic conditions (Sawyer, 1960). While it is true that certain chemical wastes may exert an oxygen demand exclusive of biological activity, they will not be considered here since such wastes are not normally found in natural streams draining undeveloped watersheds. In the past, most of the work involving BOD test was directed toward predicting the effects of municipal and industrial wastes on the dissolved oxygen content of the receiving waters. In fact, waters containing significant amounts of BOD were only associated with the more heavily populated areas. However, recent literature suggests that there are also some natural sources of BOD which can affect the oxygen concentrations in natural streams flowing from undeveloped areas. Chase and Ferullo (1957) have investigated the oxygen demand of leaves stored underwater. The purpose of their research was to identify the possible oxygen demand created by autumn leaf fall in deciduous forests. These leaves, when water-logged, sink to the bottom of the stream and

in the slow process of decay exert a sustained demand on the dissolved oxygen of the overlying water. Their tests on maple leaves showed about 750 mg of oxygen consumed per gram of leaf material over a period of 396 days. This means that maple leaves consumed a weight of oxygen equivalent to 75 percent of their initial dry weight. About 500 mg of oxygen per gram of leaf material were consumed by either oak leaves or pine needles during a similar period.

Slack and Feltz (1968) studied the effects of autumn leaf fall on the water quality characteristics of a small Virginia stream. They found that organic loading by tree litter can exert a significant control on water composition, especially during low flow periods. Dissolved oxygen and pH decreased; and water color, specific conductance, iron, manganese, and bicarbonate values increased as the rate of leaf fall increased. They note that depletion of dissolved oxygen by decomposing litter often results in fish kills, decreases in bottom fauna, and in nuisance odors from anaerobic decomposition. During their study, a few dead fish were observed on October 16, 1963. On October 25, more dead fish were present, and many fish showed distress by gulping air at the surface. This period coincided with the peak rate of leaf fall. During August and September, the rate of leaf fall was minimal and dissolved oxygen values averaged 6-8 mg/l. By the end of October, the leaf fall had reached its peak and dissolved oxygen concentrations in the stream had fallen to less than 1 mg/l.

The flow during this period remained fairly constant at about 0.3 liters/second (0.01 cfs) showing that the low dissolved oxygen did not result from simply a drop in the volume of flow.

Hall and Lantz (1969) reported on the effects of logging on the habitat of coho salmon and cutthroat trout in coastal Oregon streams. They found significant reductions in stream dissolved oxygen during and immediately following logging operations. Three experimental watersheds were used in the study. One was clearcut, one patch-cut, and one left undisturbed to serve as a control. Felling began on the clear-cut watershed in March, 1966 and was complete by mid July, although yarding of felled trees continued through late summer. As a result of felling and yarding, considerable amounts of fine debris composed largely of Douglas-fir needles, twigs, bark chips, organic duff and soil were left in the stream channel, forming small dams that ponded and slowed the flow of water. Periodic sampling of dissolved oxygen concentrations in the surface and intragravel waters showed oxygen levels too low to support salmon and trout in late June and through most of July, 1966. Juvenile coho salmon placed in live boxes in the stream survived less than 40 minutes, and oxygen concentrations as low as 0.6 mg/l were observed. There was substantial improvement in surface dissolved oxygen levels downstream and after debris was cleared from the stream in mid September. There was further improvement after fall freshets moved much of the fine debris

downstream. The oxygen content of the surface water returned to approximately pre-logging levels by autumn. The low dissolved oxygen concentrations found in surface and intragravel waters of the clearcut watershed during and immediately following logging probably resulted from a combined effect of three factors. First, it seems reasonable to assume that the finely divided organic debris left in the stream channel created a biochemical oxygen demand in the process of decomposing. Second, Brown (1969) has shown that the removal of riparian vegetation exposes the stream to direct solar radiation, causing rises in surface water temperatures which in turn decreases the solubility of oxygen in water. The higher water temperatures would also tend to increase the solubility of organic leachates from the debris. Lastly, the ponding and damming effect of the debris probably reduced the reaeration capacity of the stream. The effects of ponds and dams on reaeration will be discussed in detail in later sections.

Justification for assuming that logging debris causes a biochemical oxygen demand can be found in the literature. In studies of water quality in log storage ponds, McHugh (1964) shows that floating logs add soluble organic compounds as well as color and odor producing substances to the water. In turn, some of these soluble organic compounds would be broken down by bacterial activity, producing a biochemical oxygen demand in the process.

Research to determine the possible pollutional effects of floating

log storage is presently being conducted by the Department of Civil Engineering at Oregon State University. Laboratory tests show that 60-80 percent of leachates from selected species of wood stored in water are organic in nature (Graham and Schaumburg, 1969). Cross-cut sections of Douglas fir and Ponderosa pine submerged in water for 40 days exerted a chemical oxygen demand (COD) of 6,000 to 7,000 mg per square foot of exposed surface area. Recent work shows that the BOD from submerged logs is between 20 to 50 percent of the COD value. The reason for the lower BOD values is that not all of the soluble organic leachates can be readily broken down by bacterial activity. The BOD values varied depending on type of wood and amount of attached bark. The cross-cut ends were sealed with paraffin in some samples and the bark or uncut surface in others. Graham and Schaumburg (1969) found the leachates were passed to the water in greater quantities through the cross-cut ends. The finely divided organic debris resulting from logging operations would probably exert a much higher BOD on a unit volume basis as compared with cut log sections. This would result from the greatly increased surface area and the large number of exposed end-cuts. Large quantities of debris are left on a watershed following logging. Debris ranges in size from sawdust and small wood chips to stumps and trees which have been felled but not yarded because of their unmerchantable quality. Williamson and Schaumburg (1969) indicate that

18 percent of the bark from Douglas fir is dislodged during felling and yarding operations. In conclusion, it seems evident from the foregoing discussion that the potential sources for biochemical oxygen demand would certainly exist on a recently logged watershed.

Reaeration and the Oxygen Balance

The reaeration capacity of a stream, the ability of water to entrain and absorb atmospheric oxygen, has been a subject of increasing interest and importance. Its importance lies in the fact that the reaeration capacity is often used as a measure of the total assimilative capacity of the stream. Biodegradable organic compounds present in the stream use dissolved oxygen in the decomposition process. Once the initial oxygen content of a stream has been depleted, atmospheric reaeration becomes the major vehicle of transport supplying oxygen to the water. Extreme oxygen depletions may occur in situations of high polluttional loading and low reaeration capacity. Such depletions pose a threat to the natural flora and fauna of the stream. If oxygen values remain low, anaerobic decomposition may replace the aerobic process. Odor problems from hydrogen sulfide gas generally accompany anaerobic decomposition. To avoid these problems, efforts have been made to predict the effect of any given waste on the oxygen content of the stream. Theoretical and empirical equations have been developed to predict the reaeration capacity of various streams.

Streeter and Phelps (1925) were the first to intensively study the phenomenon of oxidation and reaeration in natural streams. Their classic work on the pollution and natural purification of the Ohio River lead to the formulation of the oxygen sag equation, a conceptual model still widely applied by sanitary engineers. The mathematical expression for the oxygen sag model is composed of two differential equations, one to account for the rate of oxygen use by the pollutant and a second to specify the rate of oxygen replacement from the atmosphere. The first equation states that in the absence of reaeration the rate of biochemical oxidation is proportional to the concentration of unoxidized matter (BOD). The assumption made is that the rate of biochemical oxidization can be expressed by a first order decay equation in which the exerted oxygen demand decreases exponentially with time. Streeter's first equation takes the following form:

$$\frac{dL}{dt} = -K_1 L \quad (2-6)$$

where L is the BOD concentration in mg/l, K_1 is termed the BOD rate constant (base e) in units of 1/days, and t is the time in days.

The second equation is based on the generally accepted proposition that the rate of gas absorption by a liquid is a first order rate process. Therefore in the absence of BOD the rate of oxygen absorption will be proportional to the oxygen deficit. The oxygen deficit (D) is defined as the difference between the saturation value and the

actual concentration of oxygen in water of a given temperature. The form of the second equation is:

$$\frac{dD}{dt} = -K_2 D \quad (2-7)$$

where D is the oxygen deficit in mg/l, K_2 is termed the reaeration coefficient (base e) in units of 1/days, and t is time in days. Equations 2-6 and 2-7 must then be combined to express the effects of both oxidation and reaeration. To do this, equation 2-6 must be expressed in terms of the oxygen deficit (D). In equation 2-6, L is the BOD concentration in mg/l. BOD is measured in terms of the amount of oxygen used in the decomposition process over a given period of time. For each mg/l of oxygen used, the amount of BOD remaining is reduced by one mg/l. This can be expressed mathematically as:

$$\frac{dL}{dt} = \frac{dC}{dt} \quad (2-8)$$

where dL/dt is the rate of oxidation and dC/dt is the rate of oxygen consumption. It is convenient here to express the rate of oxygen use as the rate change of the oxygen deficit. To use the previous example, for each mg/l reduction in L , the oxygen deficit (D) will increase by one mg/l. Mathematically expressed:

$$-\frac{dL}{dt} = \frac{dD}{dt} \quad (2-9)$$

Equation 2-9 states that the rate decrease in L is proportional to the rate increase of the oxygen deficit (D). Equation 2-6 can now be expressed in terms of D by:

$$-\frac{dL}{dt} = \frac{dD}{dt} = K_1 L \quad (2-10)$$

and equations 2-6 and 2-7 combined to yield:

$$\frac{dD}{dt} = K_1 L - K_2 D \quad (2-11)$$

Solving equation 2-11 for D results in equation 2-12. The derivation of equation 2-12 is presented in Appendix B.

$$D = \frac{K_1 La (e^{-K_1 t} - e^{-K_2 t})}{K_2 - K_1} + Da e^{-K_2 t} \quad (2-12)$$

In equation 2-12, La is the initial BOD in mg/l at t equal to 0, Da is the initial dissolved oxygen deficit in mg/l at t equal to 0, e is the base of Napierian or natural logarithms (2.71828) and the remaining terms are those previously defined.

In engineering, the first order decay coefficients (K_1 and K_2) are often expressed to the base 10 rather than the base e and noted as such by the use of the lower case letters k_1 and k_2 . These coefficients may be converted from one base to the other by the following relationship:

$$k \text{ (base 10)} = 0.434 K \text{ (base } e) \quad (2-13)$$

Using the conceptual framework of the Streeter-Phelps equation (2-12), other authors have extended the mathematical model to include additional factors which may influence the oxygen deficit in the stream. For instance, O'Connell and Thomas (1965) proposed the following modification to account for the influence of benthic algae on oxygen deficit:

$$D = \frac{K_1 La}{K_2 - K_1} (e^{-K_1 t} - e^{-K_2 t}) - \frac{(P_o - R_s)}{K_2} (1 - e^{-K_2 t}) + Da e^{-K_2 t} \quad (2-14)$$

where $P_o - R_s$ is the net effect of photosynthesis minus respiration in terms of mg/l. Dobbins (1964) proposes several additional modifications to the basic Streeter-Phelps equation to account for multiple additions of BOD along a given river reach, removal of BOD by settling of organic material, and several other cases too numerous to be included in this discussion.

Streeter's original equation (2-7) defines the rate change in the oxygen deficit resulting from reaeration. It is possible to solve for the reaeration coefficient K_2 by the following integration and manipulations:

$$\frac{dD}{dt} = -K_2 D \quad (2-7)$$

The solution to equation 2-7 in terms of D is similar to the one

presented in Appendix B and will not be repeated here. Solving equation 2-7 for D yields:

$$D = D_a e^{-K_2 t} \quad (2-15)$$

Equation 2-15 may be resolved for K_2 as follows:

$$\ln D = \ln D_a - K_2 t \quad (2-16)$$

$$-K_2 t = \ln D - \ln D_a \quad (2-17)$$

$$K_2 = \frac{\ln D_a - \ln D}{t} \quad (2-18)$$

$$\text{or } k_2 = \frac{\log D_a - \log D}{t} \quad (2-19)$$

where D_a is the oxygen deficit at t equals 0, D is the oxygen deficit at time t , t is the time in days, K_2 is the reaeration coefficient (base e) in units of 1/days, and k_2 is the reaeration coefficient (base 10) in units of 1/days. The practical significance of these equations is that the reaeration coefficient may be calculated directly from empirical data. Provided there is no biochemical oxygen demand exerted within the stream, the reaeration coefficient K_2 or k_2 can be calculated directly from equation 2-18 or 2-19. D_a is taken as the observed oxygen deficit at some upstream point, D as the deficit of some downstream point, and t as the time of travel between these points. In cases where a biochemical oxygen demand does exist within the reach,

the BOD rate constant K_1 must be determined and equation 2-12 used to calculate the reaeration coefficient K_2 .

Development of Reaeration Models

The purpose of the foregoing discussion has been to show the effect of reaeration on the oxygen concentration in a stream under conditions of oxygen deficit. The remaining remarks will be directed toward identifying the factors which influence reaeration and specifying how they can be used to predict reaeration in a natural system. This is generally accomplished in one of two ways. First, reaeration coefficients can be deduced from theoretical models using the appropriate gas transfer laws and other pertinent principles of physical chemistry. Second, reaeration coefficients can be predicted from empirical models which relate reaeration to the hydraulic characteristics of the stream channel. The theoretical development relating reaeration to gas transfer laws will be examined first.

Theoretical Models

About the same time that Streeter and Phelps published their work, Lewis and Whitman (1924) were examining the principles of gas absorption as related to reaeration. They proposed a two-film theory to explain the process of gas transfer at the liquid-gas interface. Their theory postulates the existence of two very thin and stagnant

layers at the interface, one gas and another liquid. Gas must pass through these layers by a process of molecular diffusion, and the resistance to transfer encountered in the thin layers determines the rate of diffusion. Their equation specifying this transfer rate is:

$$\frac{dW_t}{A dt} = \frac{V_o}{A} \frac{dC}{dt} = K_g (P_g - P_i) = K_L (C_i - C) \quad (2-20)$$

$$\text{or} \quad \frac{dC}{dt} = \frac{A K_L (C_i - C)}{V_o} \quad (2-21)$$

where W_t is the weight of the solute gas, K_g is the mass transfer coefficient through the gas film, K_L is the mass transfer coefficient through the liquid film, P_g is the concentration of the gas in the main body of the gas and P_i is the concentration in the gas film at the interface, C is the concentration of the gas in the main body of the liquid and C_i is the concentration in the liquid film at the interface, A is the area of the gas-liquid interface and V_o is the volume of liquid undergoing aeration.

For a slightly soluble gas such as oxygen, it is assumed that the resistance of the gas film is negligible, and that the concentration of the solute gas at the interface is determined by Henry's law, or C_i equals C_s , where C_s is the saturation value at a given temperature. Equation 2-21 may then be rewritten as:

$$\frac{dC}{dt} = \frac{A K_L (C_s - C)}{V_o} = \frac{A K_L D}{V_o} \quad (2-22)$$

where $D = C_s - C_i =$ oxygen deficit. Since V_0/A equals the depth (H), a simpler and more familiar form of equation 2-22 is:

$$\frac{dC}{dt} = \frac{K_L}{H} D \quad (2-23)$$

The relationship between the reaeration coefficient K_2 and the liquid film or mass transfer coefficient K_L can be specified as follows:

$$- \frac{dD}{dt} = \frac{dC}{dt} = K_2 D = \frac{K_L}{H} D \quad (2-24)$$

or

$$K_2 = \frac{K_L}{H} \quad (2-25)$$

Lewis and Whitman maintained that with a stagnant film at the liquid surface in which steady-state molecular transfer controls the rate of mass transfer, the mass transfer coefficient (K_L) is equal to the molecular diffusivity (D_m) divided by the film thickness (L_f) or:

$$K_L = \frac{D_m}{L_f} \quad (2-26)$$

However, their theory predicting K_L from D_m and L_f has been criticized from two standpoints. First, the existence of a laminar film in a stable form in the presence of a turbulent flow regime normally present in most streams is difficult to reconcile physically. Second, the two-film theory assumes complete and uniform gas

concentration in the main body of the liquid below the film layer. This implies a very high level of turbulent mixing and is not consistent with the hypothesis of a stagnant surface layer. Furthermore, any hydrodynamic effect on the process must be accounted for in the thickness of the hypothetical film.

Objecting to the steady-state model of Lewis and Whitman, Higbie (1935) proposed a model called the penetration theory. Higbie felt that the role of turbulence was to bring elements of bulk fluid to the surface, where absorption takes place for a prescribed time T_e , after which the element returns to the main body of the fluid and is replaced by another. His theory states:

$$K_L = 2 \sqrt{\frac{Dm}{\pi T_e}} \quad (2-27)$$

where Dm is the molecular diffusivity or diffusion coefficient.

Danckwerts (1951) modified Higbie's model to allow for a random distribution of exposure times dependent on the rate of replacement of surface elements. Called the surface renewal theory, it is expressed as:

$$K_L = \sqrt{Dm r_n} \quad (2-28)$$

where r_n is the rate of surface renewal and the other terms are the same as previously defined. The advantage of the penetration and surface renewal theories is that they account for the hydrodynamic

effects of turbulence and eddies. Both specify the mass transfer coefficient (K_L) as being proportional to the square root of the molecular diffusivity (D_m). However, as Forthescue and Pearson (1967) and Metzger and Dobbins (1967) point out, both theories involve arbitrary parameters, T_e or r_n respectively, which must either be determined experimentally or deduced from physical arguments. Although a turbulent flow field can yield characteristic times, there were few attempts to relate these directly to T_e or r_n . Thus, the usefulness of these theories was somewhat limited by the lack of a verified theory for the determination of these arbitrary parameters.

Dobbins (1964) and Metzger and Dobbins (1967) feel that the surface renewal theory of Danckwerts provides the best physical model for reaeration. They discuss and experimentally verify various equations which predict the r_n value used in equation 2-28. However, these equations also include several arbitrary parameters which must be determined from graphs based on empirical observations of previous investigators. The result is better curve fits and correlations between reported and calculated values of K_L , but with considerably more complex calculations.

O'Conner and Dobbins (1958) propose a simpler method of estimating the terms of the Danckwerts surface renewal equation. Their approach has the advantage that the gas transfer principles of the Danckwerts model are incorporated into the predictive equations, but

in a way that allows the reaeration coefficient (k_2) to be calculated directly from easily measured hydraulic parameters. They initially proposed two reaeration equations, one for nonisotropic flow characterized by a marked velocity gradient, and another for isotropic flow in comparatively deep channels where the velocity gradient approaches zero. For flow depths less than five feet, turbulence was assumed to be nonisotropic and the reaeration coefficient predicted by the following equation:

$$k_2 = 480 Dm^{0.5} S^{0.25} H^{-1.25} \quad (2-29)$$

For depths greater than five feet, isotropic turbulence was assumed, and the reaeration coefficient calculated by:

$$k_2 = 127 Dm^{0.5} V^{0.5} H^{-1.5} \quad (2-30)$$

where k_2 is the reaeration coefficient (base 10) in terms of 1/days, Dm is the liquid-film diffusion coefficient in ft^2/day , S is the channel slope in ft/ft , H is the mean depth in feet, and V is the mean velocity in ft/sec . In a subsequent article, O'Conner (1958) abandons the non-isotropic flow equation 2-29 by showing that equation 2-30 works equally well for conditions of either isotropic or nonisotropic turbulence. The numerical coefficient (127) in equation 2-30 is used to convert the units of measurement into a dimensionally correct form. O'Conner and Dobbins (1958) show that the r_n term in Danckwerts'

equation 2-28 can be approximated by V/H . With this substitution, equation 2-30 can be derived directly from equation 2-28.

Empirical Models

Streeter and Phelps (1925) pioneered the research to define the relationship between reaeration rates and the physical characteristics of the stream. They state that "the reaeration rate should be closely related to and governed by those conditions which influence turbulence of flow, and hence values of K_2 , as actually determined in a given river reach, should bear a close relation to the several measurable factors of physical condition which cause varying degrees of turbulence." They also point out the practical application of correlating values of K_2 with certain measurable factors of physical stream condition. These factors may be used to estimate the reaeration capacities of streams in which no direct measurements have been made of the reaeration coefficient. From their work on the Ohio River, Streeter and Phelps propose the following general relationship:

$$K_2 = \frac{c V^{n_b}}{H^2} \quad (2-31)$$

where K_2 is the reaeration coefficient (base e) in 1/days, V is the flow velocity in feet per second, H is the mean depth in feet, and c and n_b are empirical coefficients defined as follows:

$$c = 0.39 (10^{1.16} S) + 17 \quad \begin{array}{l} \text{(for smooth channels,} \\ \text{irregularity factor} \\ \text{equal 2.0 to 3.0)} \end{array} \quad (2-32)$$

$$c = 1.1 S^{2.3} \quad \begin{array}{l} \text{(for irregular channels,} \\ \text{irregularity factor} \\ \text{equals 3.1 to 5.0)} \end{array} \quad (2-33)$$

$$n_b = \frac{y - 0.17}{y - 1.17} \quad (2-34)$$

where S is the channel slope in feet per mile, the irregularity factor is the number of changes in slope (exceeding one foot) per mile of channel, and y is the mean increase in velocity of flow per five foot increase in gage height. Equations 2-31 through 2-34 were derived using the empirical data from the Ohio River study.

Churchill, Buckingham and Elmore (1962) point out that there are certain inherent sources of error associated with the method used by Streeter and Phelps to determine the value of K_2 . In their study, Streeter and Phelps substituted measured K_1 values into equation 2-12 and solved the equation to obtain the K_2 values. These K_2 values were used to derive the empirical relationships shown in equations 2-31 through 2-34. The source of error arises from underestimating the BOD in a particular stream reach. Bottom deposits can exert a BOD which is not measured by the standard BOD analysis. The total or ultimate BOD as measured by standard analysis will be lower than the actual BOD exerted in the natural environment. Errors of this type will result in calculated reaeration coefficients which are

too low.

Subsequent reaeration research has been directed toward methods which circumvent the problems caused by biochemical oxygen demand. There are two methods which eliminate these problems. First, reaeration coefficients can be determined in systems where no BOD is present. The oxygen deficits required for reaeration studies can be artificially created under natural or laboratory conditions by the addition of chemical reducing agents. Second, reaeration coefficients can be calculated directly from equation 2-18 or 2-19 if dissolved oxygen measurements are made in streams where natural oxygen deficits exist but minimal BOD concentrations are encountered. Situations such as this are not normally found in the natural environment since BOD is generally the cause of oxygen deficits and thus both are usually present. However, Churchill, Buckingham and Elmore (1962) were able to take advantage of an unusual situation where waters released from the deep storage impoundments of the Tennessee Valley Authority dams were low in dissolved oxygen but free from organic pollutants. This natural phenomenon resulted from prolonged storage of water under conditions of thermal stratification. As the water flowed downstream from the impoundments and absorbed air, they measured the changes in dissolved oxygen and were able to calculate the reaeration rates in successive river reaches. By accurately measuring the hydraulic characteristics of each river

reach, they were able to statistically relate these characteristics to reaeration. Through dimensional analysis and multiple regression techniques, they arrived at a set of four predictive equations relating the reaeration coefficient (k_2) to various combinations of hydraulic variables. These equations and the resulting coefficients of multiple correlation (R^2) are:

$$k_2 = 5.026 V^{0.969} H^{-1.673} \quad (R^2 = 0.822) \quad (2-35)$$

$$k_2 = 44.55 V^{-1.297} S^{0.230} \quad (R^2 = 0.805) \quad (2-36)$$

$$k_2 = 1.477 V^{1.049} H^{-2.262} f^{-0.823} \quad (R^2 = 0.841) \quad (2-37)$$

$$k_2 = 0.041 V^{2.361} H^{-2.753} S^{-0.669} \quad (R^2 = 0.835) \quad (2-38)$$

where k_2 is the reaeration coefficient (base 10) in 1/days, V is the mean velocity in ft/sec, H is the mean depth in feet, S is the energy slope in ft/ft, and f is the resistance coefficient defined as follows:

$$f = \frac{8 g H S}{V^2} \quad (2-39)$$

The correlation coefficient for equation 2-35 is slightly less than some of the other predictive equations. However Churchill, Buckingham and Elmore recommend this equation as being the simplest and easiest to apply. Without doubt, the work of Churchill, Buckingham and Elmore is the most detailed and exacting study of

reaeration ever performed on a natural system.

Churchill, Buckingham and Elmore compared their actual k_2 values to those predicted by the equations of O'Conner and Dobbins (1958). They found that the values predicted by the O'Conner and Dobbins isotropic flow equation (2-30) agreed fairly well with the actual values. However, the nonisotropic flow equation (2-29) proposed by O'Conner and Dobbins underestimated the actual values. In developing their nonisotropic flow equation, O'Conner and Dobbins assumed that the turbulence near the surface of the stream was directly related to the velocity gradient near the surface. Churchill, Buckingham and Elmore criticize this assumption and show that different states of turbulence can exist with identical velocity distributions.

Krenkel and Orlob (1962) studied turbulent diffusion and reaeration in a laboratory flume where oxygen deficits were created by additions of sodium sulfite reducing agent. They concluded that reaeration is directly related to the degree of turbulence, and that the energy dissipated per unit mass of water as it flowed through the flume was a good index of the degree of turbulence. The source of energy for the flow process is the rate loss of potential energy as the water moves downslope and can be calculated by the following equation:

$$E = S \quad V \quad g \quad (2-40)$$

where E is the rate of energy dissipated per unit mass of water, S is the slope of the energy gradient (taken as the slope of the water surface profile) in ft/ft, V is the mean flow velocity in ft/sec, and g is the acceleration due to gravity in ft/sec². Relating energy dissipation (E) and flow depth (H) to the reaeration coefficient (k_2 - base 10), Krenkel and Orlob obtained a correlation coefficient of 0.86 with the following predictive equation:

$$k_2 = (1.141 \times 10^{-4}) E^{0.408} H^{-0.660} \quad (2-41)$$

Thackston (1966) used laboratory and field data to develop an equation relating reaeration to the physical properties of the stream. Laboratory studies of reaeration were conducted with a recirculating flume. Reaeration rates were determined under various conditions of slope, velocity, depth and bottom roughness. Wire fence and then bricks were used to simulate various types of bottom roughness. Field measurement of reaeration were made on the same river reaches used by Churchill, Buckingham and Elmore (1962). The resulting reaeration equation is:

$$k_2 = 0.000125 \left\{ 1 + \left[\frac{V}{\sqrt{gH}} \right]^{\frac{1}{2}} \right\} \frac{\sqrt{HSg}}{H} \quad (2-42)$$

In equation 2-42, k_2 is the reaeration coefficient in units of 1/seconds and the remaining terms and units of measurement are the same as

those used in previous equations. Equation 2-42 can be rewritten as:

$$k_2 = 0.000125 (1 + Fr^{1/2}) \frac{V^*}{H} = (a+b Fr^c) \frac{V^*}{H} \quad (2-43)$$

where Fr is the Froude number and V^* is the friction or shear velocity. Thackston determined the values of a, b and c in equation 2-43 by what he called "a semi-trial-and-error" method of curve fitting. In equation 2-43, the term V^*/H was used to estimate k_2 . The Froude number is used to correct for the difference between the actual and projected surface area.

Using the reaeration data published by O'Conner and Dobbins (1958), Churchill, Buckingham and Elmore (1962), Krenkel and Orlob (1962), and Streeter, Wright and Kehr (1936), Langbein and Durum (1967) present an additional equation to predict reaeration capacity. It is based on a best fit regression line relating k_2 to mean depth and velocity and uses the observations of the mentioned investigators as data points. The regression equation resulting from the combined data is:

$$k_2 = 3.3 V H^{-1.33} \quad (2-44)$$

Langbein and Durum (1967) also show some interesting relationships between the reaeration coefficient (k_2) and mean discharge (Q). Using hydraulic data from the Kansas, Missouri, and Mississippi rivers, they plot mean discharge versus reaeration coefficients calculated

from equation 2-44. The result is a logarithmic decrease in the reaeration coefficient with increasing discharge. This relationship can be explained by the fact that as discharge increases, mean depth increases at a faster rate than mean velocity. However, this pattern did not hold true for smaller streams. Under these conditions, the reaeration coefficient (k_2) decreased with discharge in riffle sections, while in pool sections it increased with discharge. This would seem to indicate that with increased discharge in shallow sections, depth increases faster than velocity and in deep sections, velocity increases more rapidly.

The third method of studying reaeration in natural streams is the one used in this study and is based on the artificial depletion of oxygen with chemical reducing agents. This method is generally limited to smaller streams since the amount of reducing agent required for large streams would be prohibitively large. The advantage of this method is that streams without measurable BOD concentrations can be studied and reaeration coefficients computed directly from equation 2-18 or 2-19. Owens, Edwards and Gibbs (1964) used this approach in their studies of reaeration in small English streams. The dissolved oxygen in certain reaches was depleted by additions of sodium sulfite reducing agent which chemically combined the free oxygen gas. By measuring the changes in dissolved oxygen content as the depleted water moved downstream, they were able to calculate

the reaeration rates and related these to the hydraulic characteristics of the stream. A multiple regression analysis of their data resulted in a correlation coefficient of 0.94 for the following predictive equation:

$$k_2 = 10.09 V^{0.73} H^{-1.75} \quad (2-45)$$

By combining their data with the results of previous investigators, a second predictive equation was derived with a calculated correlation coefficient of 0.95. The equation is:

$$k_2 = 9.41 V^{0.67} H^{-1.85} \quad (2-46)$$

In these two equations, k_2 is the reaeration coefficient (base 10) in 1/days, V is the mean velocity in ft/sec, and H is the mean depth in feet.

Effect of Temperature on Reaeration

All of the previous predictive equations have been corrected for temperature effects. The calculated reaeration coefficients represent the k_2 or K_L values in water at 20° C. A standard temperature of 20° C was selected because it represented a mean temperature of the water during most of the reaeration experiments. In general, the reaeration coefficient increases with increased temperature. As Churchill, Buckingham, and Elmore (1962) point out, a rise in water

temperature results in a decrease in viscosity, density, and surface tension, and conversely, the velocity of molecular vibrations is increased. It is difficult to distinguish the exact role played by each of these factors but it appears probable that the increased velocity of molecular vibration is the most important. When the temperature of the water is raised, the rate of movement across the interface increases. At the same time molecular diffusion allows oxygen penetration to a greater depth as measured in molecular diameters.

Most investigators agree on the general form of the equation which relates k_2 at a given temperature to k_2 at a standard temperature of 20° C. This equation is:

$$k_{2T} = \theta k_{220}^{(T-20)} \quad (2-47)$$

$$\text{or} \quad k_{220} = \theta k_{2t}^{(20-T)} \quad (2-48)$$

where k_{2T} is the reaeration coefficient at some temperature T, k_{220} is the reaeration coefficient at a standard temperature of 20° C, T is the temperature in degrees C, and θ is an empirically derived coefficient. The difference in opinion among various researchers comes in specifying the value of θ . Streeter and Phelps (1925) used a mean value of 1.047 for θ in their work on the Ohio River and based their calculation of θ on their experimental observations and those of previous investigators. Using the results of laboratory flume studies

of reaeration, Streeter, Wright and Kehr (1936) revised the previous estimation of θ and redefined its value as 1.016. This figure was accepted and used by O'Conner and Dobbins (1958) and later by Krenkel and Orlob (1962). In a detailed laboratory analysis evaluating the effects of temperature on reaeration, Churchill, Buckingham and Elmore (1962) conclude that a value of 1.0241 for θ gives the best agreement with their data. This value was later used by Owens, Edwards and Gibbs (1964) in their work on English streams. In recent papers by Metzger and Dobbins (1967) and later by Metzger (1968), the value of θ is determined by a graphical solution based on their observations and the data presented by the previously mentioned authors. Their values of θ are very close to Streeter's value of 1.016 in the 10 to 20 degree C temperature range.

Methods of Measuring Dissolved Oxygen

Technological advances in analytical chemistry and instrumentation during the past half-century have provided a wide variety of methods for measuring dissolved oxygen. Each of these methods utilizes a basic physical or chemical property of molecular oxygen. In titrimetric and colorimetric methods, the oxidizing property of oxygen is utilized. Voltammetric methods depend on the rate of diffusion of molecular oxygen. Other methods of analysis are based on the determination of the thermal conductivity or the paramagnetic

susceptibility of molecular oxygen (Mancy and Jaffe, 1966). Each of the many methods has an application to which it is best suited. Mancy and Jaffe (1966) have presented the most definitive and complete discussion of the various methods to be found in the literature. Standard Methods (American Public Health Association, 1960) provides the best supplement to the work of Mancy and Jaffe, and contains exact instructions for the preparation of reagents used in the various titrimetric methods for measuring dissolved oxygen.

In the field of sanitary engineering and water quality chemistry, the Winkler titrimetric technique and its various modifications are probably the most widely used and accepted methods for determining dissolved oxygen in water and wastewater. The Azide modification of the Winkler technique is the method most commonly applied to natural streams and it eliminates the interferences caused by nitrite nitrogen in the water. The Winkler procedure provides an accurate although time consuming method of determining dissolved oxygen.

Dissolved oxygen concentrations can also be measured with membrane electrode systems. The membrane electrode technique has been developed within the last few years and recent improvements make the accuracy of the results comparable with those of the Winkler method. This method has several advantages over the Winkler titration. Almost instantaneous measurements of dissolved oxygen can be made, which greatly reduces the time required to perform the test.

Oxygen concentrations can be continuously and automatically monitored over extended periods of time.

Membrane electrode systems can be described as finite-diffusion electrochemical cells. The rate of diffusion of oxygen through a gas-permeable membrane is measured by two electrodes inside the cell. The rate of diffusion through the membrane is proportional to the partial pressure gradient across the membrane. The partial pressure gradient in turn is proportional to the oxygen concentration in the water.

There are two main types of finite-diffusion electrochemical cells: the voltammetric or polarographic membrane electrodes, and the galvanic membrane electrodes. The basic difference between voltammetric and galvanic systems is that the former rely on an external source of applied voltage to polarize the indicator electrode, similar to the conventional polarographic analysis. Galvanic electrodes on the other hand make oxygen a reactant at one of the electrodes, thereby generating an electrical current without the aid of an external electromotive force. Thus, the galvanic system becomes a primary battery or fuel cell in the general sense (Mancy and Jaffe, 1966).

The polarographic or voltammetric membrane electrode system was used in this study and its principle of operation will be discussed in greater detail. The oxygen sensor is a complete polarographic

system in itself. A thin membrane of Teflon stretched over the end of the sensor isolates the sensor elements from the environment. The membrane is permeable to all gases and allows them to enter the interior of the sensor. When a suitable polarizing voltage is applied across the cell, oxygen will react at the cathode, causing a current to flow through the cell. The oxygen - current relationship at the cathode is stoichiometric. The membrane passes oxygen at a rate proportional to the partial pressure gradient across it. Since oxygen is rapidly consumed at the cathode, it can be assumed that the oxygen pressure inside the membrane is zero. Hence, it can be seen that the force causing oxygen to diffuse through the membrane is proportional to the absolute pressure of oxygen outside the membrane. If the oxygen pressure increases, more oxygen diffuses through the membrane and more current flows through the cell. A lower pressure results in less current (Yellow Springs Instrument Company, 1967). Since the partial pressure of oxygen in the water varies directly with the oxygen concentration (mg/l), the voltammetric output from the sensor can be displayed on a meter dial which is calibrated directly in units of oxygen concentration (mg/l) rather than in units of partial pressure. The meter and sensing probe can be easily and quickly calibrated by exposing the probe directly to atmospheric air. This is possible because of the fact that the sensing element reacts to differences in partial pressure. Since atmospheric oxygen exerts the same partial

pressure as oxygen saturated water, the meter will read the same saturation value of oxygen regardless of whether it is in air or in oxygen saturated water. The meter is calibrated in air by measuring the air temperature and adjusting the meter to read the saturation value of oxygen in water at that temperature.

Mancy and Jaffe (1966) view the membrane electrode system as perhaps the most dramatic development in the analysis for dissolved oxygen since the Winkler test was developed. As the use of such systems becomes more widespread, they will probably equal and surpass the Winkler titration as a method for measuring dissolved oxygen. Ideally suited to applications in environmental science and engineering, they are easily portable for field use and their ruggedness along with their ability to analyze dissolved oxygen in situ represent considerable advantages as compared with other methods.

CHAPTER III

HYDRAULIC CHARACTERISTICS RELATED TO REAERATION

Two different approaches have been used in earlier reaeration studies. The theoretical approach seeks to define reaeration in terms of the physical laws of gas transfer (Lewis and Whitman, 1924; Higbie, 1935; Denckwerts, 1951). Verification for these theoretical models came from carefully controlled laboratory experiments. Using these models for the prediction of in situ reaeration rates requires certain assumptions. First, one must assume that the reaeration process occurring under laboratory conditions adequately describes the process occurring in the natural stream. No clear proof exists to support this assumption. Second, one must assume that all of the parameters in the theoretical model can be calculated from measurements of the natural system. As previously discussed, there is a lack of verified theory for determining some of the arbitrary parameters in these theoretical models.

The second approach is basically empirical but with some theoretical justification. Using some statistical technique, the measured hydraulic properties of the stream are correlated with the observed reaeration rates. The result is a best-fit predictive equation based on the measured variables. Recent investigators (Churchill, 1962; Owens and Gibbs, 1964) have used this technique to develop

equations which predict reaeration rates quite accurately. This approach has been criticized by those who feel that stronger theoretical framework is required. Questions have been raised regarding the contribution of such empirically derived equations to the understanding of the reaeration process. Such criticism is not entirely valid. It can be shown that empirical equations may reflect theoretical considerations. Furthermore, these equations may help explain the reaeration process in that they indicate how various hydraulic properties are related to reaeration rates.

Rationale for Selecting Predictive Variables

In any empirical equation, there should be a theoretical or practical justification for including a variable in the predictive model. Various hydraulic characteristics will now be examined with regard to their hypothetical effect on reaeration.

The importance of turbulence in the reaeration process should be stressed again. Streeter and Phelps (1925) recognized the direct relationship between turbulence and reaeration. The theoretical models of Higbie (1935) and later Danckwerts (1951) supported the earlier conclusions drawn by Streeter and Phelps. Their models emphasized the role of turbulence and included arbitrary parameters designed to estimate the degree of turbulence.

The effect of a given hydraulic property on reaeration can often

be described more adequately by showing its effect on turbulence rather than relating it directly to reaeration. Stream depth has a significant inverse relationship to turbulence and hence should have an inverse relation to reaeration. As depth increases, the turbulence created by bottom roughness is dissipated through a larger vertical section. This results in a proportional decrease in vertical mixing and less turbulent transfer between the saturated surface film and the main bulk of the fluid. In a natural stream, depth is also related to reaeration because of the inverse relationship which generally exists between depth, velocity, and stream width. As depth increases, velocity and stream width generally decrease. A decrease in either velocity or stream width should result in lower reaeration rates for reasons discussed below.

Turbulence in a stream channel increases as the velocity increases. At low velocities, laminar flow is approached in the channel and the mixing of the fluid is by molecular activity only. As velocity increases the inertial forces become greater than the forces of viscosity. When this occurs turbulent flow begins and the laminar flow pattern breaks into a series of random eddies. Mixing is now accomplished by the random eddy motion of finite masses of molecules.

The intensity of turbulence is also directly related to velocity. The term "intensity of turbulence" refers to the speed of the angular

rotation of the eddies. Increases in velocity result in corresponding increases in the speed of eddy rotation and thus more intense mixing. As velocity continues to increase, the laminar sublayer covering the bottom roughness is destroyed. This results in an even higher level of turbulent mixing. Reaeration should be directly related to velocity because of the direct relationship between velocity and turbulence.

Stream width is related to reaeration because of its effect on the surface area of the stream. As stream width increases so does the surface area. As the surface area increases, the total transfer of oxygen across the interface increases even though the rate of oxygen diffusion per unit area remains the same. However this does not mean that surface area can be correlated with reaeration. The surface area of a given stream reach can be calculated from the mean width and length of the reach. Obviously, the longer the reach, the greater the surface area. But a longer reach may simply mean a larger volume of water undergoing reaeration. Thus if surface area is to be related to reaeration, it must be specified as surface area per unit length. If the unit of length is taken as one foot, then the surface area is simply one foot times the width. For this reason, width rather than surface area should be used in a predictive equation.

Channel slope is related to reaeration through its effect on velocity. This effect is shown by the Chezy-Manning equation which states:

$$V = \frac{1.49}{n} R^{0.66} S^{0.5} \quad (3-1)$$

where V is the velocity in ft/sec, n is the Manning roughness coefficient, R is the hydraulic radius in ft., and S is the channel slope in ft/ft. The channel slope provides the energy necessary for the flow process by converting potential energy to kinetic energy through the force of gravity. The hydraulic radius and Manning roughness coefficient account for the energy loss due to the friction drag of the channel banks and bottom. Krenkel and Orlob (1962) maintain that the loss of potential energy as the water flows downslope is dissipated through turbulence. The energy dissipation and thus the slope of the stream channel should then be directly related to reaeration. Slope would also be an important predictive parameter from a second standpoint. There were two waterfalls in the experimental stream section. As expected, the highest measured reaeration rates occurred in the waterfall reaches. The reaeration rates in these two reaches differed markedly from those of any other reaches, as did the channel slopes. Considering all of the possible hydraulic characteristics, channel slope seems to best define the waterfall phenomena in a predictive equation.

Discharge was considered as a variable to be included in a predictive model. However, the effect of discharge on reaeration is largely dependent on the character of individual reaches. As

pressure as oxygen saturated water, the meter will read the same saturation value of oxygen regardless of whether it is in air or in oxygen saturated water. The meter is calibrated in air by measuring the air temperature and adjusting the meter to read the saturation value of oxygen in water at that temperature.

Mancy and Jaffe (1966) view the membrane electrode system as perhaps the most dramatic development in the analysis for dissolved oxygen since the Winkler test was developed. As the use of such systems becomes more widespread, they will probably equal and surpass the Winkler titration as a method for measuring dissolved oxygen. Ideally suited to applications in environmental science and engineering, they are easily portable for field use and their ruggedness along with their ability to analyze dissolved oxygen in situ represent considerable advantages as compared with other methods.

Langbein and Durum (1967) point out, stream discharge can be directly or inversely related to reaeration. He shows that at higher discharges reaeration decreases in riffle sections but increases in pool sections. This apparent inconsistency can be explained by the different rates of increase of velocity and depth as discharge increases. In a given section, changes in discharge could be related to changes in reaeration. However, discharge could not be used to account for reaeration differences between one section and another. An obvious example of this would be several successive reaches with different reaeration rates but the same discharge. For this reason, discharge was eliminated as a possible variable for predicting reaeration.

All of the hydraulic variables examined thus far have been parameters which can be obtained from discrete and independent measurements of the stream system. Statisticians sometimes apply the term "peer variables" to such parameters to indicate that they are measured directly rather than computed from other variables. Besides these peer variables, there are several computed variables which may prove useful for predicting reaeration rates. These computed variables will be considered next.

One of these is the Manning roughness coefficient. The Manning roughness coefficient is defined by Albertson, Barton and Simons (1960) as:

$$n = \frac{1.49}{Q} A R^{0.66} S^{0.5} \quad (3-2)$$

where n is the Manning roughness coefficient, Q is the discharge in cfs, A is the cross-section area of the stream in sq. ft, R is the hydraulic radius in feet and S is the channel slope in ft/ft. Churchill, Elmore and Buckingham (1962) point out that the hydraulic radius R may be assumed equivalent to the mean depth in streams where the width greatly exceeds the depth. Such was the case with the experimental stream section used for this study. The Manning roughness coefficient should be directly related to reaeration. As channel roughness increases, so does the scale of turbulence. The scale of turbulence is the mean size of the eddies. Thus, increased bottom roughness promotes larger eddy size and greater vertical mixing. The effect of channel roughness on turbulence should be most significant at higher velocities after the laminar sublayer covering the bottom roughness has been destroyed.

The Darcy-Weisbach resistance coefficient (f) was developed as a measure of resistance to flow encountered in closed conduits. However, it can be applied equally as well to open channel flow (Albertson, Barton, and Simons, 1960). The resistance coefficient is similar to the Manning roughness coefficient in that it provides an index to channel roughness. Churchill, Elmore and Buckingham (1962) state that in channels of equal depth and slope, the rougher channels (those with larger f values) will generate larger eddies than smoother channels. Larger eddies indicate a greater scale of turbulence and higher

reaeration rates. Thus the resistance coefficient should be useful in predicting reaeration rates. The resistance coefficient can be computed from the following formula:

$$f = \frac{8 g H S}{V^2} \quad (3-3)$$

where f is the resistance coefficient, g is the acceleration due to gravity in ft/sec^2 , H is the stream depth in feet, S is the channel slope in ft/ft , and V is the velocity in ft/sec .

The point was made in discussing the relationship between velocity and reaeration that the mixing characteristics of the stream were quite different for turbulent flow as compared with laminar flow. This being the case, the Reynolds number should be considered for inclusion in a predictive equation since it is an index to the type of flow which prevails. The Reynolds number is the ratio of the inertial force per unit area to the viscous force per unit area. A small Reynolds number indicates laminar flow while a large one characterizes turbulent flow. Fortescue and Pearson (1967) have shown that as the Reynolds number increases the coefficient of mass transfer (K_L) also increases. Since the coefficient of mass transfer (K_L) and the reaeration coefficient (K_2) are directly related, the Reynolds number should be directly related to the reaeration rate. Albertson, Barton and Simons (1960) show that the Reynolds number can be computed by

the following equation:

$$Re = \frac{4 V H}{\nu} \quad (3-4)$$

where Re is the Reynolds number, V is the velocity in ft/sec, H is the depth in feet and ν is the kinematic viscosity in sq. ft/sec. The kinematic viscosity is temperature dependent and the value at any given temperature can be obtained by consulting standard tables such as those found in Albertson, Barton and Simons (1960).

The reaeration equation proposed by Thackston (1966) uses the friction velocity (V^*), the flow depth (H) and the Froude number (Fr) in the predictive model. Thackston defines the friction or shear velocity (V^*) as:

$$V^* = (H S g)^{0.5} \quad (3-5)$$

where H is the flow in feet, S is the channel slope in ft/ft., and g is the acceleration due to gravity in ft/sec². The results of Thackston's experiments show a direct relationship between the reaeration coefficient (k_2) and V^*/H . For this reason, the friction velocity will be considered as a variable for predicting reaeration in small streams.

The Froude number is used to account for waves and other surface irregularities at the air-water interface. Albertson, Barton and Simons (1960) state that in a study of flow phenomena involving

irregularities at an interface, the Froude number must be taken into consideration. They specify the Froude number (Fr) as:

$$Fr = \frac{V}{(gH)^{0.5}} \quad (3-6)$$

where V is the velocity in ft/sec, g is the acceleration due to gravity in ft/sec², and H is the depth. Since reaeration is dependent on the nature of the surface or interface, the Froude number should be treated as a variable for predicting reaeration. Dobbins (1964) suggests the use of the Froude number to account for the difference between the actual and projected surface area undergoing reaeration. Thackston (1966) uses the Froude number for this purpose in his reaeration research.

Both Dobbins (1964) and Krenkel and Orlob (1962) feel that the energy dissipation in a given channel reach should be directly related to turbulence and thus to reaeration. They compute the energy dissipation (E) by the following equation:

$$E = V S g \quad (3-7)$$

where S is the slope of the stream channel in ft/ft, V is the mean velocity in ft/sec and g is the acceleration due to gravity in ft/sec². As Dobbins (1964) points out "the energy is withdrawn from the main flow to create the kinetic energy of turbulence that finally is dissipated

by viscous action into heat. For the flow as a whole, the rates of withdrawal, creation of turbulent energy, and dissipation are equal." If the rate of energy dissipation is equal to the rate at which turbulent energy is created, then it follows that energy dissipation should be closely related to reaeration rates.

Two additional variables must be considered for inclusion in the predictive model. They are the temperature of the stream water (T) and the molecular diffusivity (Dm). These variables are not generally considered as hydraulic properties. However they must be included to account for the effect of temperature on reaeration. The molecular diffusivity (Dm) is a function of temperature and specifies the rate of gas transfer per unit area across the air-water interface. Dobbins (1964) defines the molecular diffusivity (Dm):

$$\begin{aligned} D_m &= 2.037 (1.037)^{T-20} \quad (\text{units are } 10^{-5} \text{ cm}^2/\text{sec}) \\ &= 2.191 (1.037)^{T-20} \quad (\text{units are } 10^{-8} \text{ ft}^2/\text{sec}) \end{aligned} \quad (3-8)$$

where T is the temperature in ° C.

General Predictive Model

The hydraulic characteristics which have been discussed in the previous section can be summarized and stated in the form of a general predictive model.

$$k_2 = f(H, V, W, S, n, f, Re, V^*, Fr, E, T, Dm) \quad (3-9)$$

The symbols and units of measurement in the above equation are defined below in Table 3-1.

Table 3-1. Symbols and units of measurement used in equation 3-9.

Symbol	Variable	Units of measurement
k_2	Reaeration coefficient (base 10)	1/days
H	Mean depth	feet
V	Velocity	ft/sec
W	Width	feet
S	Channel slope	ft/ft
n	Manning roughness coefficient	none
f	Frictional resistance coefficient	none
Re	Reynolds number	none
V^*	Friction velocity	ft/sec
Fr	Froude number	none
E	Energy dissipation	ft ² /sec ³
T	Temperature	°C
Dm	Molecular diffusivity	ft ² /sec

CHAPTER IV

FIELD PROCEDURES

Selection and Description of Stream and Reaches

The South Fork of Berry Creek, a small, natural woodland stream in the Willamette River Basin of Oregon, was chosen as the site for the reaeration studies. This site was selected primarily because Berry Creek contained a variety of hydraulic conditions typical of those found in other small mountain streams in the Oregon Coast Range. Secondary considerations which made the Berry Creek area attractive were the ease of access, the upstream weir for determining streamflow and the variety of hydrologic characteristics in a relatively short section of stream channel.

The Berry Creek drainage basin is ten miles north of Corvallis, Oregon on the Adair Tract of MacDonald forest owned by Oregon State University. Above the study section, the South Fork of Berry Creek drains an area of 1.83 square miles or approximately 1,170 acres. Just below the research area, the South Fork joins the Middle and North Forks of Berry Creek and flows into Soap Creek, which enters the Luckiamute River, a tributary of the Willamette River.

The aspect of the South Fork watershed is north-facing with fairly steep topography ranging from 1,890 feet above MSL at Forest

Peak to approximately 320 feet above MSL at the study site. Typical winter and spring flows through the upstream weir range from 10 to 20 cfs (Warren et al., 1964). Flows as low as 0.1 cfs frequently occur during late summer and early fall.

The section of the stream used for the reaeration studies began about 100 feet below a concrete weir installed by the Department of Fisheries and Wildlife and extended downstream for a distance of 342.6 feet. The weir did not control the hydraulic characteristics of the downstream channel. This section was divided into 16 subsections or reaches with each reach marked by wooden survey stakes driven into the stream bottom. Each reach was selected such that slope, depth, and velocity remained fairly uniform throughout the reach. In this way, pools, riffles, and small waterfalls were divided into separate and distinct reaches. The length of the reaches varied between 7.3 feet and 29.0 feet. The pools were narrow and shallow and separated by riffle sections. The two waterfalls were small. One had a drop of about one foot and the other a drop of about three feet. Figure 4-1 shows the experimental section of Berry Creek and the individual reaches as mapped during the 1969 summer field season.

Measurement of Hydraulic Characteristics

The six hydraulic characteristics used to define each reach of the study section were length, width, slope, flow, velocity, and

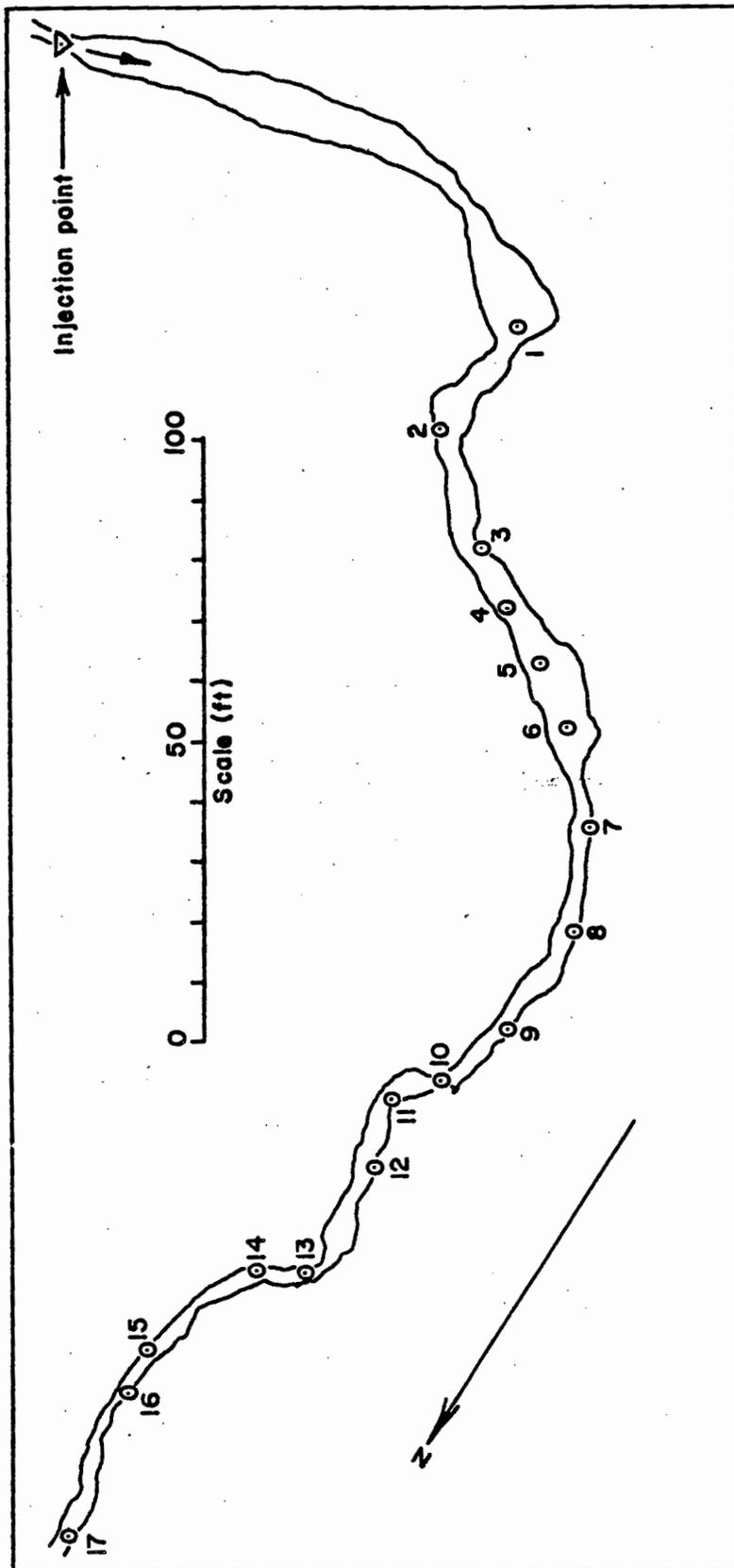


Figure 4-1. Experimental section of Berry Creek showing stations and reaches.

depth. The length, width and survey stake elevations were measured in September, 1968 and again in June, 1969. Some of the survey stakes used to define the reaches were washed away during the high fall and winter flows of 1968-1969. They were replaced early the next summer in approximately the same positions. This accounts for the small changes in the reaches between one summer and the next. The length of each reach was measured with a steel tape. Widths were measured perpendicular to the stake line at two-foot intervals down the reach.

A Zeiss level and survey rod were used to determine the elevation at the top of each survey stake. These elevations were referenced to a lag bolt set into a large tree trunk. The lag bolt served as a datum reference point and its elevation was arbitrarily chosen as 320.00 feet above MSL. On the days when reaeration measurements were conducted, the distance from the top of each station stake to the water surface below was measured with a steel tape. By knowing the water surface elevation at the head and tail of each reach and the length of the reach, it was possible to compute the slope of the water surface profile within the reach.

Streamflow through the study section was determined by discharge measurements made with a pygmy current meter in a uniform section of channel between Stations 14 and 15. These values were compared with the discharges indicated by the stage-discharge rating

table for the upstream weir, and found to be within ten percent of the rated discharge. The weir discharges tended to be higher than those obtained by current meter measurements. Part of this difference may be explained by the observed buildup of algal growth on the weir crest, and the subsequent rise in stage in the stilling pond.

The pygmy current meter proved inadequate for measuring the average velocity in many of the reaches. The very shallow flows in the riffle sections and the low velocities in the reaches containing pools made it impossible to obtain reliable velocity measurements with a current meter. Timing floats and cork granules as they moved through a reach also proved unreliable because reproducible results could not be obtained.

Dye cloud measurements were found to be the most successful and reproducible method for measuring velocities within the reaches. Using an eyedropper, one milliliter of Rhodamine B dye was injected at the upstream edge of the reach. A thin red film of dye immediately spread out on the water surface, but was ignored. The majority of the dye remained concentrated in a dense orange cloud and was carried through the reach. A stopwatch was started just as the dye was injected, and stopped when the most concentrated part of the cloud passed the downstream edge of the reach. It can be argued that velocities obtained in this manner were not really average velocities but something between the average and maximum velocity in the reach.

While this may be true, there did not seem to be any better method available for determining characteristic velocities within the reaches.

Mean depths were computed from cross-section measurements made within each reach. Depth calculations for the 1968 data were based on two cross-sections per reach. In 1969, the number of cross-sections per reach was increased to four or five in order to obtain more representative mean depths. It should be noted that even with five cross-sections per reach, it would be possible in extremely shallow sections to have a 20 percent error in the mean depth. The tape used in measuring individual depths could be read to an accuracy of ± 0.01 feet. A computed mean depth of 0.05 ± 0.01 feet represents a possible error of 20 percent. However, it is unlikely that errors of this magnitude occurred. Random errors tend to cancel each other out, especially with the large number of samples used to compute the mean depths. Care was taken to avoid any deterministic errors which would consistently overestimate or underestimate individual depth measurements.

Artificial Depletion of Dissolved Oxygen Using Sodium Sulfite

An oxygen deficit had to be created in order to measure and study the process of reaeration in the study stream. Various methods have been used by previous investigators to artificially create an

oxygen deficit. Probably the most convenient and widely used method of removing free dissolved oxygen from water is by reducing the oxygen with a sodium sulfite reducing agent. The chemical balance for this reaction is found in Appendix C.

Any precalculated deficit of oxygen can be achieved by injecting sodium sulfite into the stream at a constant rate. Subsequent reaeration rates may be calculated by measuring the changes in the dissolved oxygen content of the water as it moves downstream from the point of injection. The advantage of artificial depletion is that oxygen deficits can be created in the natural environment without accompanying biochemical oxygen demands. For reasons discussed in Chapter II, the accuracy and ease of calculating reaeration rates is considerably improved if biochemical oxygen demands are either not present or small enough to be negligible.

A Mariotte vessel was used to inject the sodium sulfite reducing agent into the stream. A schematic diagram of the Mariotte vessel is shown in Figure 4-2. The vessel was constructed using a 20-liter carboy. The advantage of using this type of injection system is that the flow rate out of the vessel remains constant, regardless of the level of liquid in the container. The constant flow-rate principle of the Mariotte vessel is discussed in Appendix D. The siphon is established by either blowing into the air inlet tube or by suction at the orifice. A vacuum is created in the air space at the top of the

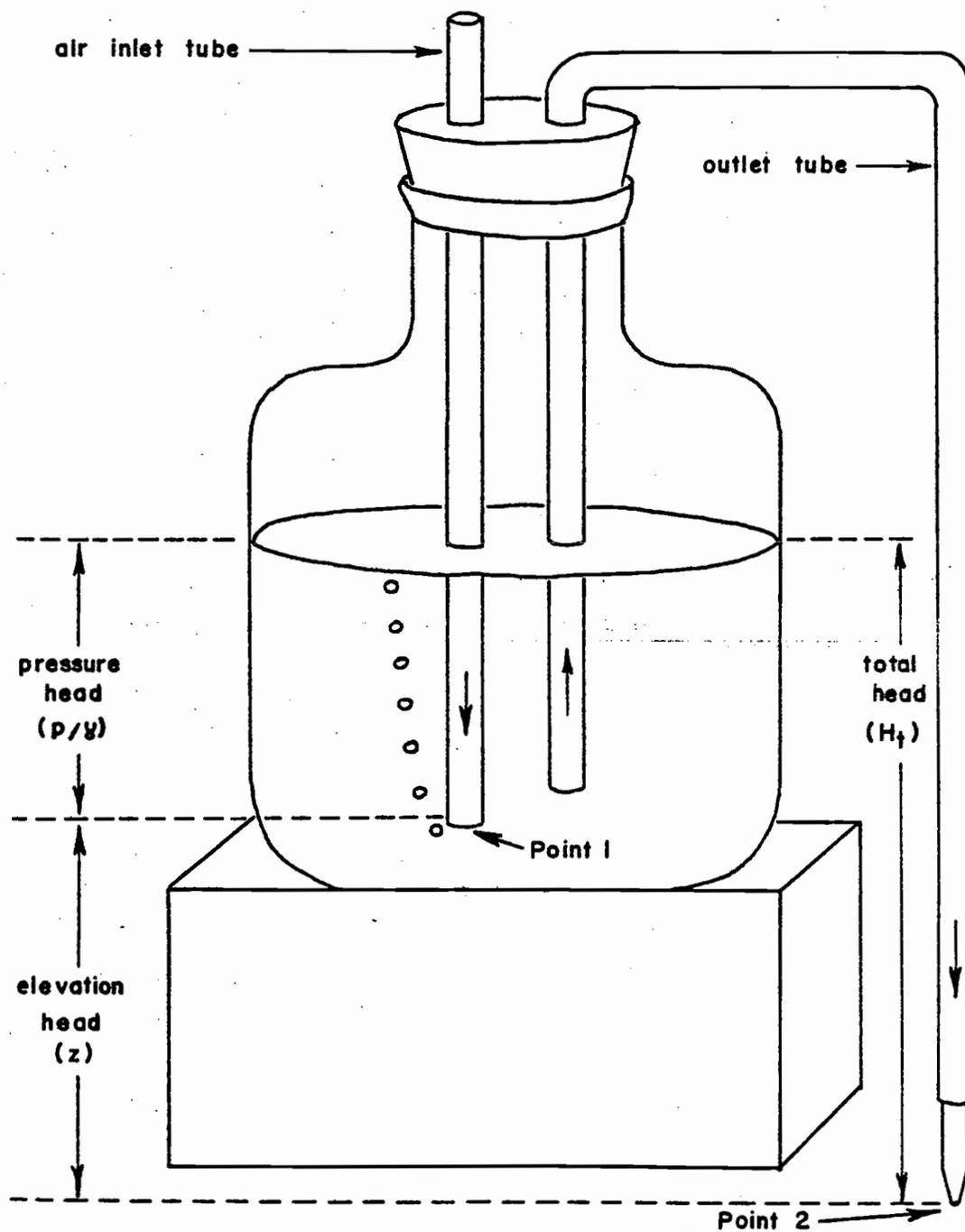


Figure 4-2. Mariotte injection vessel.

container, equal but opposite to the pressure head. Atmospheric pressure operates at Point 1. As the level of the liquid falls, the pressure head becomes smaller as does the vacuum at the top of the container. These opposing forces remain in balance until the level of the liquid falls below the outlet tube, at which time the siphon is broken and injection ceases.

The dissolved oxygen in the stream can be artificially depleted to any given level by using the proper concentration and injection rate of sodium sulfite reducing agent. To insure a uniform and reproducible injection rate, several orifices were made to fit into the 3/16" Tygon tubing used for the outlet tube shown in Figure 4-2. The orifices were constructed by heating 3/16" hollow glass rods and stretching them to the desired thickness. The ends were then cut and fire-polished. Table 4-1 shows the orifice injection rates using 1.3 feet of effective head, as measured from the bottom of the air inlet tube to the orifice opening of the Mariotte vessel. Under field conditions, 1.3 feet of effective head was obtained by placing the Mariotte vessel on a metal can near the stream and supporting the orifice by a ring stand and clamp such that the orifice was level with the bottom of the can and just above the water surface.

Table 4.1. Injection rates from the Mariotte vessel using different orifices at 1.3 feet of effective head.

Orifice number	Injection rate (mls/min)
0 ¹	673
1	454
2	215
3	143

¹ 3/16" outlet tube used without any inserted orifice

The required concentration of sodium sulfite was calculated using equation 3-7. This equation is derived in Appendix C.

Na_2SO_3 concentration (gm/liter) =

$$\frac{1.335 \times 10^4 \times D \text{ (mg/liter)} \times \text{flow (cfs)}}{\text{injection rate (mls/min)}} \quad (3-7)$$

Where D = oxygen deficit below saturation value
 = saturation value (mg/liter) - desired minimum
 oxygen concentration (mg/liter)

The sodium sulfite reducing agent was prepared in the 20-liter glass carboy used for the Mariotte vessel injection system. The carboy was filled with 15 liters of distilled water and placed on a magnetic stirrer. When all of the reagent had dissolved, 2 mls of Rhodamine B dye were added and the carboy filled to the 20-liter mark with distilled water. The dye, a brilliant red tracer, proved very useful

in following the oxygen depleted water as it moved downstream. It also gave some visual assurance of complete lateral and vertical mixing of the reducing agent with the stream water.

Cobalt chloride is recommended as a catalyst for the sulfite-oxygen reaction and should be added at a concentration of 5 mg/liter of reducing agent (Eckenfelder, 1966). To reduce any premature sulfite-oxygen reaction, the catalyst was not added until just before actual injection into the stream. At this time, a solution containing 100 mg of CoCl_2 and 5 mls of distilled water was mixed with the reducing agent in the carboy.

Dissolved Oxygen and Temperature Measurements

Measurements of dissolved oxygen concentration in Berry Creek were made with a Yellow Springs Instrument Company Model 54-RC oxygen meter. This particular instrument employs a Clark type membrane-covered polarographic probe as the sensing element. The operating principle of this type of sensor has been discussed in Chapter II.

This method of measuring dissolved oxygen is relatively new. A discussion of this technique has not appeared in the literature. The measurement techniques and problems encountered in using the oxygen meter are discussed here for the benefit of future researchers who may wish to use this method in their studies.

The instrument is designed to read directly in parts per million (dissolved oxygen) and is compensated for temperature effects on both probe membrane permeability and oxygen solubility in water. The Model 54-RC is well adapted to field operation since it is easily portable and has a self-contained power supply. The power is supplied by four 1.25 volt Ni-Cd rechargeable batteries.

Field observations indicate that some form of agitation is required when the flow velocity past the probe drops below about 0.5 fps. At these low velocities, oxygen is transmitted through the membrane faster than it can be supplied by the surrounding water. The result is a localized depletion of oxygen in the region immediately surrounding the outer surface of the probe's membrane. The effects of this depletion can be observed by the reaction of the meter needle. Instead of coming to rest on some value after a short equilibrium period, the needle continues to drift downward to lower and lower readings. In all cases, this needle drift occurred while measuring the dissolved oxygen in deeper ponded sections of the stream where velocities were quite low. In situations such as this, it was necessary to provide some mechanical agitation of the probe to offset the low flow velocities. Experience proved that by slowly swinging the probe through the water or by raising and lowering the probe in deeper areas, a sufficient velocity past the probe could be created and a reliable and reproducible reading obtained. The only probe

maintenance required was recharging the saturated KCl electrolyte solution in the probe. This was normally done after eight hours of continuous operation. Drifting of the meter needle during measurements or long lag times (time required for the needle to steady on some value) indicated that the probe needed to be recharged with KCl. Whenever the KCl solution was changed, the Teflon membrane was also replaced. After recharging the probe, it was necessary to recalibrate the meter against some standard reference.

Laboratory and field tests showed that a warm-up period of about 30 minutes was required before standardizing the meter calibration. If attempts were made to calibrate the meter as soon as it was turned on, the needle tended to drift away from the calibration point and would not remain in calibration until the meter had been operating for a sufficient length of time.

Two methods listed in the instruction manual (Yellow Springs Instrument Company, 1967) were used for calibrating the oxygen meter. The first method calls for the probe to be placed in a water sample of known oxygen concentration and the meter calibrated to this concentration. This method was used initially as the primary standard for calibration. The oxygen concentration of the water was determined by a standard Winkler titration using the azide modification as reported in Standard Methods (American Public Health Association, 1960). The second method of calibration was to place the probe in air

of known temperature. The meter was then adjusted to the saturation value of oxygen in water at that temperature. This method was possible because of the fact that oxygen pressure in air-saturated water is the same as the pressure in the air above the water. Further explanation of this principle can be found in Chapter II. This method of calibration was far more convenient since it requires only the measurement of air temperature rather than the more complicated and time consuming Winkler titration. Field and laboratory checks showed good agreement between these two methods of calibration. When the probe was air calibrated and placed in a water sample, the meter reading of dissolved oxygen agreed with the oxygen concentration as measured by the Winkler titration. Conversely, when the meter was calibrated in water and set to the oxygen concentration indicated by the Winkler titration, the meter showed the correct saturation value when placed in air. The air calibration method was used throughout most of the experiment, but periodic checks were made using the Winkler titration. These checks showed the continued agreement between the methods.

One possible criticism of the air calibration method is that the saturation value of oxygen depends not only on ambient temperature but also on ambient pressure. No pressure corrections were made when calibrating the meter and a standard pressure of 760 mm of mercury was assumed. In order to check the possible errors

introduced by not correcting for ambient pressure, barometric pressure records were obtained for the days on which reaeration studies were made. These pressure readings were those recorded at the nearest U. S. Weather Bureau station at Salem, Oregon, and appear in Table E-2 of Appendix E. Calculations indicate that the largest possible error that could have resulted from not correcting for pressure was less than 0.6 percent of the true air saturation value. Errors this small would not significantly affect the results.

The YSI Oxygen Meter is also equipped for direct readout of temperature. The temperature-sensing thermistor used to compensate for temperature effects on the membrane permeability and oxygen solubility may be switched out of the compensating circuit to show ambient air or water temperatures directly on the meter dial. The meter dial is calibrated in both parts per million dissolved oxygen and in degrees Centigrade. Temperature measurements made with the oxygen meter were checked periodically with a mercury-filled glass thermometer and found to be in close agreement.

Dissolved oxygen concentrations and stream temperatures were measured at each of the 17 station stakes. This was done before any sodium sulfite was injected into the stream. After the initial survey had been completed, the Mariotte vessel was placed in position upstream and a siphon established to start the injection of the sodium sulfite reducing agent. Using the YSI oxygen meter, the dissolved

oxygen in the stream was monitored at a point downstream from the place of injection. As the leading edge of the deaerated water reached the probe,, the meter reading began to drop and continued to do so for several minutes. Figure 4-3 shows actual depletion curves for three different concentrations of reducing agent, each of which were recorded during a separate injection cycle. A total of one liter of reducing agent was injected during each cycle. For most of the re-aeration studies, the amount of reducing agent injected was increased to 20 liters. The effect was similar to the depletion curves shown in Figure 3-3 with the exception that the plateau of minimum dissolved oxygen values was extended for considerably longer periods.

It was necessary to wait until the rate change of dissolved oxygen at any one station became quite small to accurately measure dissolved oxygen changes within each of the reaches. The probe was then moved to the next downstream station stake and a dissolved oxygen reading taken. As soon as the downstream reading had been recorded, another measurement was taken at the upstream stake. The two upstream readings were averaged and the change in dissolved oxygen within the reach was taken to be the difference between the downstream reading and the averaged upstream readings. This process was repeated at each successive downstream station. Because of the travel time between injection point and measuring site, it was often possible to continue measurements for several minutes after all of

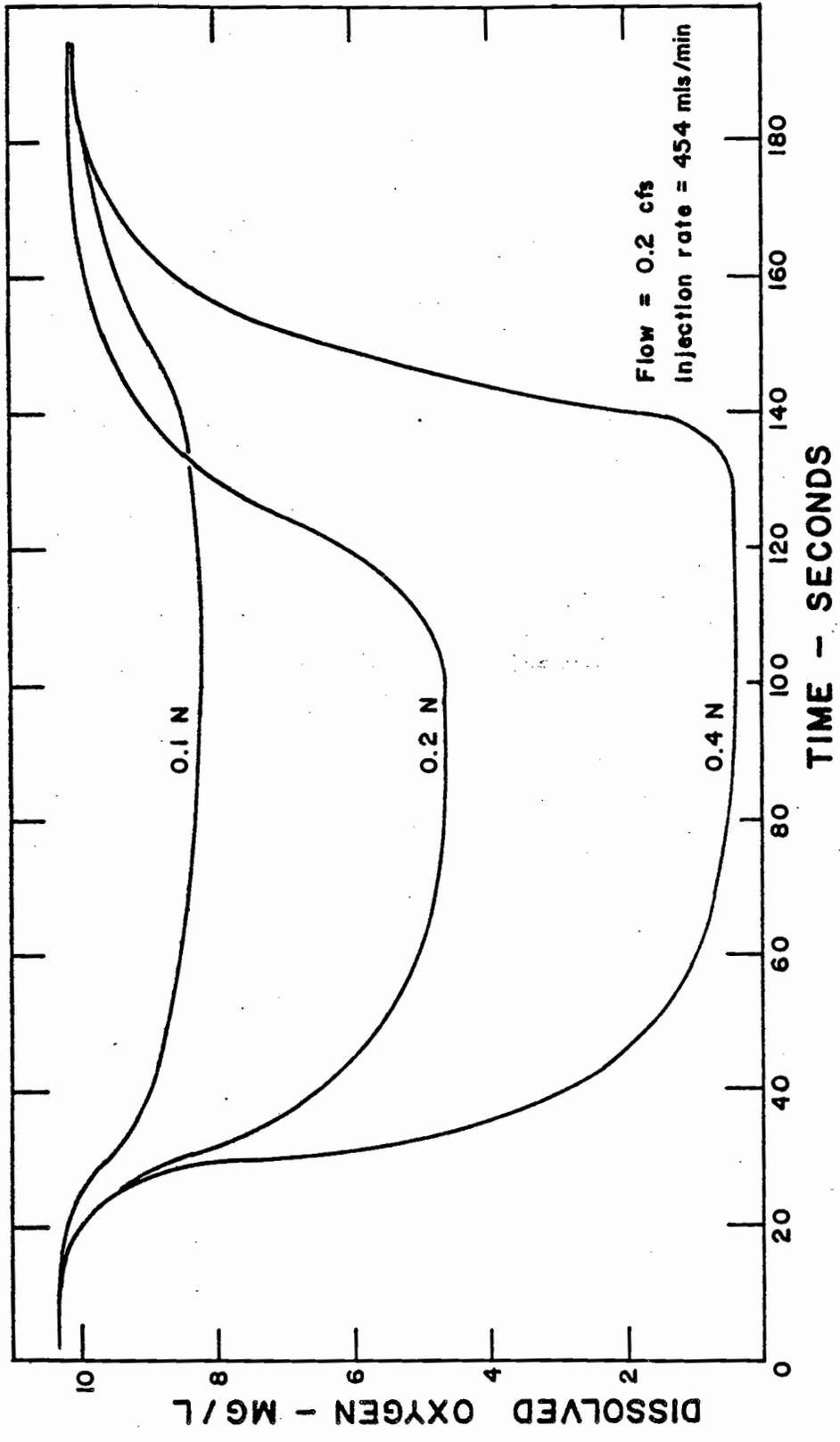


Figure 4-3. Actual oxygen depletion curves at different sodium sulfite concentrations.

the reducing agent had been injected. Measurements were suspended after the trailing edge of the deaerated water passed the measuring site. The trailing edge could be recognized by a sudden and rapid increase in dissolved oxygen at a downstream measuring site.

Meter lag-times initially presented timing difficulties. Lag-times of about 50-70 seconds were observed. The lag-time is the time required by the meter to reach a steady state after being placed in the stream. With three measurements per reach, this meant about three minutes per reach, or about one hour to measure all of the 16 reaches. The larger the oxygen depletion, the longer the meter lag time. Since many of the runs were made using injection rates of 500 mls/min or greater, the 20-liter Mariotte vessel injected for only 30-40 minutes. This was not enough time to measure dissolved oxygen changes throughout the entire study section. Tests with the YSI oxygen meter showed that the lag time was very short when the probe was moved from a region of low oxygen concentration to one of high oxygen concentration, such as removing the probe from deaerated water and exposing it to the air. However, lag times of about one minute were observed when the probe was placed in deaerated water after having equilibrated to the air saturation value. Since the probe was exposed to the air each time it was moved from one station to another, a long lag time was predictable for each reading. The problem of lag time was overcome by transporting the meter probe

in a beaker of deaerated water collected at the last measuring site. In this way, the probe was never exposed directly to the air and it was possible to reduce the lag times to 10-20 seconds.

CHAPTER V

DATA ANALYSIS

Statistical Techniques

The purpose of the statistical analysis was twofold. First, simple linear regression was used to compare the reaeration rates calculated from the study stream with those predicted from the equations of investigators. Second, stepwise multiple linear regression analysis was applied to the variables listed in Table 3-1. This resulted in 12 best-fit predictive models based on the data from the study stream. These 12 models were then tested for predictive accuracy and precision by simple linear regression analysis. The results of this analysis were compared with the regression results for the reaeration equations cited from the literature. This comparison showed which equations were most suitable for predicting reaeration in small, turbulent mountain streams. The CDC 3300 Computer System at Oregon State University was used to perform these analyses.

The first part of the statistical analysis provided a measure of the predictive accuracy and precision of the reaeration equations cited in the literature. The reaeration coefficients (k_2) predicted by these equations were calculated by substituting the appropriate field

measurements into the equations and solving for k_2 . These predicted values will be referred to as k_2 (predicted). The actual reaeration coefficients determined from the measurements of dissolved oxygen in the study stream will be referred to as k_2 (actual).

Simple linear regression was used to compare k_2 (actual) with k_2 (predicted). The slope of the regression line was used as a measure of the predictive accuracy of the equation. A slope of 1.0 indicates a 45 degree slope or a 1:1 relationship between k_2 (actual) and k_2 (predicted). The accuracy of the predictive equation increases as the slope approaches 1.0. Slopes less than 1.0 indicate an over-estimation of k_2 (actual). Slopes greater than 1.0 indicate an under-estimation of k_2 (actual). The regression line was forced through the origin in order to give a uniform basis for comparing the slopes of different equations.

The simple correlation coefficient (r) was used as a measure of the predictive precision of the equation. The simple correlation coefficient is a measure of the variance or scatter of points about the regression line. The precision of the predictive equation increases as the simple correlation coefficient approaches its maximum value of 1.0. Figure 5-1 illustrates the conclusions which can be drawn from this type of analysis. The results of this analysis appear in the following chapter.

Stepwise multiple linear regression analysis was used to develop

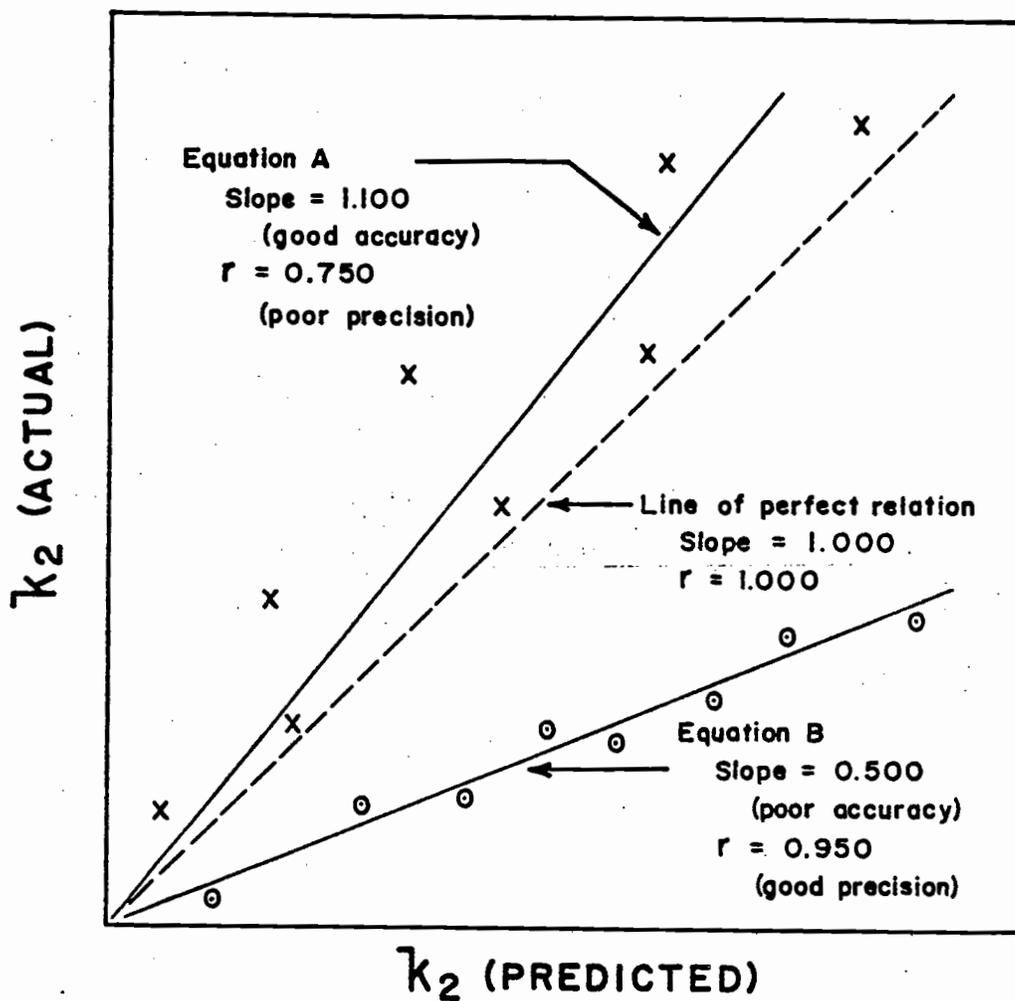


Figure 5-1. Statistical inferences from simple linear regression analysis.

the reoperation equations proposed in this study. The computer program for this type of analysis is stored as a library subroutine under the code name *STEP on the CDC 3600 computer at Oregon State University (Yates, 1969). As applied to this study, the *STEP program treated the first variable (k_2) in Table 3-1 as the dependent variable, and the next ten variables (H through E) as the independent or predicting variables. The last two variables in Table 3-1 (T and Dm) were not included in this analysis. The reason for this will be discussed later. The *STEP program assigns a best-fit linear relationship to the following generalized equation:

$$y = a + bx_1 + cx_2 + dx_3 + \dots + mx_n \quad (5-1)$$

where y is the dependent variable, x_1 through x_n are the independent variables, and a through m are computer-assigned constants. Equation 5-1 is an additive model. The same program can be used for a multiplicative model if the appropriate transformations are observed.

The format for a multiplicative model would be:

$$y = a x_1^b x_2^c x_3^d \dots x_n^m \quad (5-2)$$

The multiplicative model 5-2 can be stated in the terms of an additive model through logarithmic transformations. Making the appropriate transformations, equation 5-2 can be restated as:

$$\log y = \log a + b (\log x_1) + c(\log x_2) + d(\log x_3) \dots + m(\log x_n) \quad (5-3)$$

Equation 5-3 is now in a suitable form for analysis by the *STEP program. The option of transforming the input variables before analyzing them is available through a subroutine of the *STEP program. The computer output from the program specifies the values of log a, b, c, d m. The terms b through m in equation 5-3 become the exponents in equation 5-2. The antilog of 'log a' from equation 5-3 becomes the 'a' value in equation 5-2. Both a multiplicative and an additive model were used in developing the proposed reaeration equations. In these analyses the regression line was not forced through the origin. This is shown by the fact that intercept 'a' has a finite, non-zero value.

Step 1 of the stepwise multiple linear regression analysis enters the independent variable (x_1) which explains the greatest amount of variance in the dependent variable (y). Step 2 adds a second independent variable (x_2) to the regression model. This process is repeated until all of the statistically significant independent variables (x_1 through x_n) have been added to the regression equation. The test for statistical significance is based on an F-level test. Variables which were not significant at the 95 percent level were omitted.

Regression equations were also developed by replicating the form of selected literature equations and recomputing the coefficients.

The purpose of this type of analysis was to see how well the factors used in the literature equations would predict reaeration when fit to the actual reaeration data from the study stream.

The temperature (T) and the molecular diffusivity (Dm) were not included in the regression analysis. Instead, equation 5-4 was used to correct for temperature in the proposed reaeration equations. Equation 3-8 shows that molecular diffusivity is a function of temperature. The effect of molecular diffusivity on reaeration is accounted for in the temperature correction. The small range in temperatures and the large range in reaeration coefficients made an independent analysis of temperature effects impractical. The temperature correction used in the proposed equations is:

$$k_{2T} = 1.016 k_{220}^{(T-20)} \quad (5-4)$$

where k_{2T} is the reaeration coefficient at some temperature 'T', k_{220} is the reaeration coefficient at a standard temperature of 20° C, and T is the temperature of the stream in degrees C. This equation was first recommended by Streeter, Wright and Kehr (1936) and later verified by Metzger and Dobbins (1967) and Metzger (1968).

The proposed reaeration equations are listed in the following chapter. They were tested for their predictive accuracy and precision by the same techniques applied to the literature equations.

This provided a standard basis of comparison on which to judge the relative merits of each reaeration equation.

Calculation of the Actual Reaeration Coefficient

The actual reaeration coefficient (k_2 - actual) determined from dissolved oxygen measurements was calculated for each reach of the stream using equation 2-19.

$$k_2 = \frac{\log D_1 - \log D_2}{t} \quad (2-19)$$

The oxygen deficits, D_1 and D_2 , were computed in the following manner. The temperature and dissolved oxygen concentration were measured at the beginning (subscript 1) and the end (subscript 2) of each reach. The saturation values of oxygen, Cs_1 and Cs_2 , were computed by applying the Churchill formula (equation 2-5).

$$Cs = 14.652 - 0.41022T + 0.0079910T^2 - 0.000077774T^3 \quad (2-5)$$

The Churchill equation was selected after a careful review of the various formulas for predicting the saturation value of oxygen. The values obtained by Churchill are in good agreement with those obtained by several other investigators. Furthermore, the use of these values has been recently endorsed by the Committee on Sanitary Engineering Research (Isaacs and Gaudy, 1968). Using these values,

the oxygen deficits were calculated as follows:

$$D_1 = Cs_1 - C_1 \quad (5-5)$$

$$D_2 = Cs_2 - C_2 \quad (5-6)$$

where C_1 and C_2 are the measured oxygen concentrations at the beginning and the end of the reach. The travel time through the reach was determined by dye cloud measurements as discussed in Chapter III. The travel time in seconds was converted to days and substituted for "t" in equation 2-19. By converting travel time to days, the unit for the reaeration coefficient becomes 1/days rather than 1/seconds. This is consistent with the existing literature.

Data Modification

Some modification of the field data and the calculated reaeration coefficients was required. In six of the original 122 reaeration calculations, the reaeration coefficient was negative. This meant that oxygen was removed rather than added to the stream in these six reaches. The only significant sink for dissolved oxygen in the stream was the sodium sulfite reducing agent. The negative reaeration rates can be explained in two ways. In all six cases, the negative reaeration rates occurred within the first three reaches immediately below the injection point.

Initially, it was assumed that the reaction between the sodium sulfite and the dissolved oxygen was instantaneous. However, some small but finite reaction time may have been required to carry the reaction to completion. If this was the case, the water entering the reach could be higher in dissolved oxygen than the water leaving the reach. This would result in and account for the negative reaeration rates.

A second explanation was also considered. The oxygen concentration was assumed to be uniform throughout the cross-section of the stream. It is possible that complete mixing had not occurred in the reaches just below the injection point. Thus, the measured oxygen concentration would depend on the position of the sensing probe in the cross-section. Calculated reaeration rates could be negative if the measured oxygen values were not representative of the mean concentration in the cross-section.

The data for these six reaches were not used in the statistical analysis. There seemed to be reasonable justification for disregarding these data. Justification for eliminating these data could be based on either incomplete mixing or insufficient reaction time.

One further modification of the data was made. In several instances, the calculated reaeration coefficient was zero. This meant that there was no change in the dissolved oxygen content within the reach. This situation was generally found to occur in the deeper

reaches where the velocity was quite low. It is probable that some reaeration did occur, but that it was too small to be measured. To perform the necessary statistical analyses, a logarithmic transformation of the variables was required. Since the logarithm of zero has no mathematical definition, a calculated reaeration coefficient of zero could not be transformed. To avoid this problem, the zero reaeration coefficients were assigned a value of 0.0001 so that the transformations could be made. This assigned value was several orders of magnitude less than the next highest reaeration coefficient. It would have no significant effect on the outcome of the analysis.

CHAPTER VI

RESULTS

The results of the statistical comparison between k_2 (actual) and k_2 (predicted) are shown in Tables 6-1 and 6-2. In both tables, the proposed reaeration equations have been ranked in the order of their predictive accuracy. The predictive accuracy and precision in this discussion refer to the comparison between the predicted reaeration rates and the actual reaeration rates calculated from measurements on the study stream. As previously discussed, the slope of the regression line was used as a measure of predictive accuracy. Those equations with a slope nearest to 1.0 have been ranked first. The predictive precision of the various equations is shown by the simple correlation coefficient (r). The higher the simple correlation coefficient, the higher the predictive precision.

The reaeration equation proposed by Owens, Edwards and Gibbs (1964) comes closer than any of the other literature equations to predicting the actual reaeration rates measured in the stream. A plot of k_2 (actual) versus k_2 (predicted) by this equation appears in Figure 6-1. This figure shows that the equation tends to overpredict reaeration at the lower values and slightly underpredict it at the higher values. All but one of the equations shown in Table 6-1 tend to underpredict the reaeration rates found in the mountain stream.

Table 6-1. Equations for predicting the reaeration coefficient (k_2) proposed by previous investigators.

Predictive equation	Slope (S)	Simple correlation coefficient (r)
<u>Owens, Edwards and Gibbs, 1964 (Eq. 6-1):</u> $k_{2T} = 1.0241^{(T-20)} 10.09 V^{0.73} H^{-1.75}$	1.006	0.9016
<u>Churchill, Buckingham and Elmore, 1962 (Eq. 6-2):</u> $k_{2T} = 1.0241^{(T-20)} 44.55 S^{0.230} H^{-1.297}$	1.168	0.8002
<u>Owens, Edwards and Gibbs, 1964 (Eq. 6-3):</u> $k_{2T} = 1.0241^{(T-20)} 9.41 V^{0.67} H^{-1.85}$	0.811	0.8959
<u>Churchill, Buckingham and Elmore, 1962 (Eq. 6-4):</u> $k_2 = 5.026 V^{0.969} H^{-1.673}$	2.128	0.9303
<u>Churchill, Buckingham and Elmore, 1962 (Eq. 6-5):</u> $k_{2T} = 1.0241^{(T-20)} 1.447 V^{1.049} H^{-2.262} f^{-0.823}$	2.158	0.9265
<u>Churchill, Buckingham and Elmore, 1962 (Eq. 6-6):</u> $k_{2T} = 1.0241^{(T-20)} 5.026 V^{0.969} H^{-1.673}$	2.379	0.9323
<u>O'Conner and Dobbins, 1958 (Eq. 6-7):</u> $k_{2T} = 1.016^{(T-20)} 480 D_m^{0.5} S^{0.5} H^{-1.25}$	2.810	0.8046
<u>Churchill, Buckingham and Elmore, 1962 (Eq. 6-8):</u> $k_{2T} = 1.0241^{(T-20)} 0.041 V^{2.361} H^{-2.753} S^{-0.669}$	2.976	0.9320
<u>O'Conner and Dobbins, 1958 (Eq. 6-9):</u> $k_{2T} = 1.016^{(T-20)} 37,497 D_m^{0.5} V^{0.5} H^{-0.66}$	3.330	0.8343
<u>Thackston, 1966 (Eq. 6-10):</u> $k_2 = 10.08 \left\{ 1 + [V(gH)^{-0.5}]^{0.5} \right\} (SgH)^{0.5}$	6.148	0.8916
<u>Langbein, 1967 (Eq. 6-11):</u> $k_2 = 3.3 V H^{-1.33}$	8.357	0.9196
<u>Krenkel and Orlob, 1962 (Eq. 6-12):</u> $k_{2T} = 1.016^{(T-20)} 0.9858 E^{0.408} H^{-0.660}$	10.037	0.9013

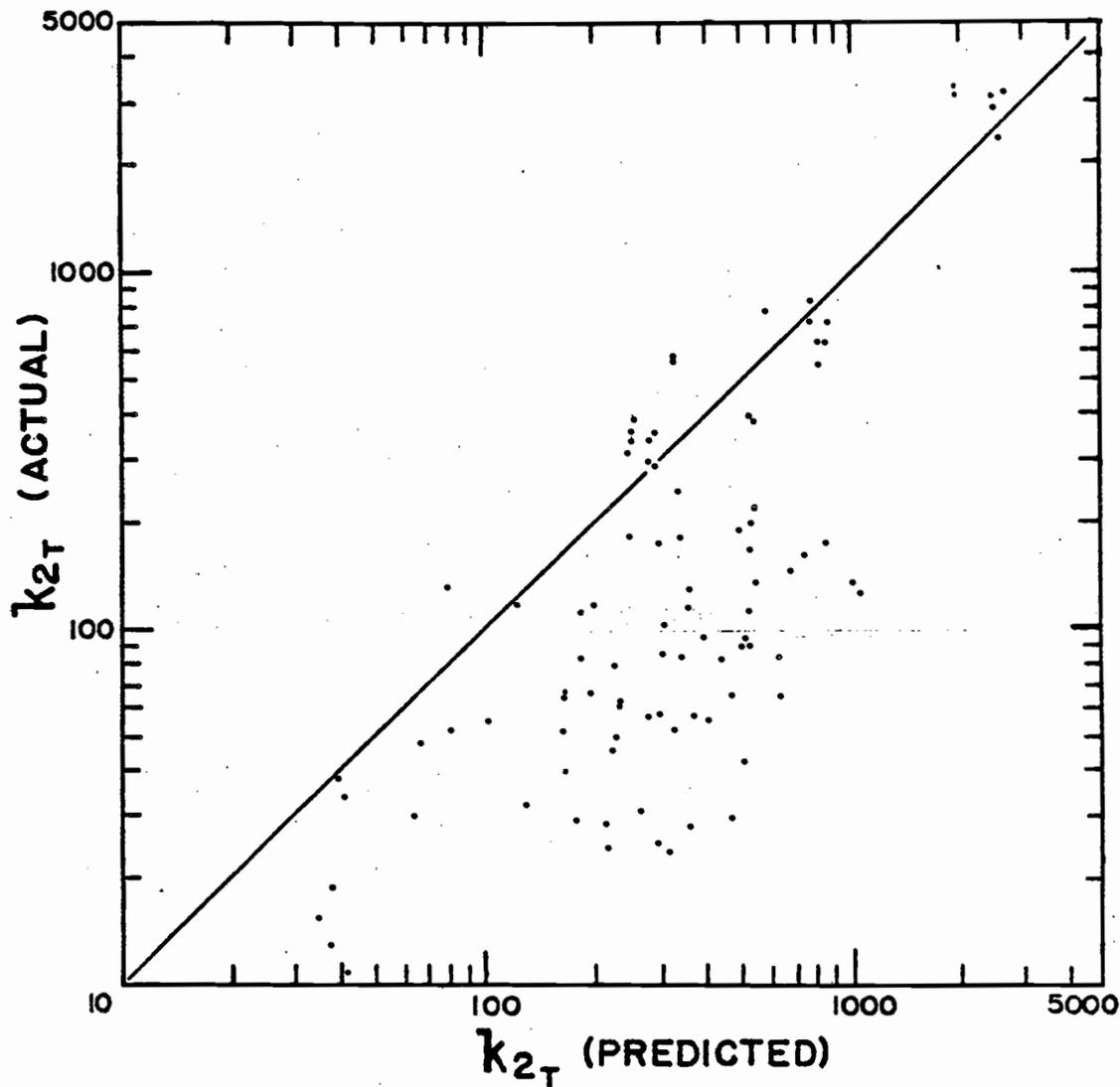


Figure 6-1. Actual versus predicted values of k_{2T} from equation 6-1.

$$k_{2T} \text{ (predicted)} = 1.0241^{(T-20)} 10.09 V^{0.73} H^{-1.75}$$

Underprediction is indicated by slopes which exceed 1.0.

Table 6-2 presents the reaeration equations proposed for small, turbulent streams which were developed by stepwise multiple linear regression analysis. As discussed in the previous chapter, both an additive and a multiplicative model were used to develop these equations. Equations 6-13 through 6-19 in Table 6-2 would be defined mathematically as additive models. However the energy dissipation term which appears in the first four equations is a product of slope, velocity and gravity. Velocity in turn is a function of slope and gravity. Therefore equations 6-13 through 6-16 reflect a multiplicative process but are stated in an additive form.

Equations 6-13 through 6-18 predicted reaeration rates more accurately than any of those cited in the literature. The slopes of these six equations were very close to 1.0 because the regression equation was fitted to the data. Since the slopes did not differ significantly, the simple correlation coefficients were used in choosing the best predictive equation. The precision of the first four equations was very good. All had simple correlation coefficients above 0.99. Equations 6-15 and 6-16 demonstrated the best predictive accuracy and precision. Statistically, there is little difference between these two equations. Selection of the "best" equation can therefore be made on such non-statistical grounds as ease of application.

Each of the first four equations contains the energy dissipation

Table 6-2. Equations for predicting the reaeration coefficient (k_2) developed from multiple linear regression analysis.

Predictive equation	Slope (S)	Simple correlation coefficient (r)
<u>Equation 6-13:</u> $k_{2T} = 1.016^{(T-20)} (152.1 E - 20.16)$	1.000	0.9903
<u>Equation 6-14:</u> $k_{2T} = 1.016^{(T-20)} (152.6 E + 48.90 H - 30.26)$	1.000	0.9903
<u>Equation 6-15:</u> $k_{2T} = 1.016^{(T-20)} (181.6 E - 1657 S + 20.87)$	1.000	0.9920
<u>Equation 6-16:</u> $k_{2T} = 1.016^{(T-20)} (184.6 E - 1789 S + 14.25 W - 9.085)$	0.999	0.9921
<u>Equation 6-17:</u> $k_{2T} = 1.016^{(T-20)} (854.5 H + 4573 S + 1043 V - 804.1)$	1.001	0.9514
<u>Equation 6-18:</u> $k_{2T} = 1.016^{(T-20)} (7552 S - 168.7)$	1.001	0.9332
<u>Equation 6-19:</u> $k_{2T} = 1.016^{(T-20)} (5616 S + 468.5 V - 364.7)$	1.047	0.9375
<u>Equation 6-20:</u> $k_{2T} = 1.016^{(T-20)} 17,260 S^{1.852}$	1.164	0.9838
<u>Equation 6-21:</u> $k_{2T} = 1.016^{(T-20)} 6.928 S^{1.152} H^{-2.549}$	0.794	0.9555
<u>Equation 6-22:</u> $k_{2T} = 1.016^{(T-20)} 9.623 S^{1.056} V^{0.589} H^{-2.369}$	0.640	0.9571
<u>Equation 6-23:</u> $k_{2T} = 1.016^{(T-20)} 56.84 E^{1.483}$	0.634	0.9841
<u>Equation 6-24:</u> $k_{2T} = 1.016^{(T-20)} 0.3451 E^{0.971} H^{-2.295}$	0.564	0.9560

term (E). Energy dissipation has been defined by equation 2-40 and it states that: $E = S V g$. Gravitational acceleration (g) is constant and does not require field measurement. However, channel slope (S) and flow velocity (V) must be determined in order to calculate energy dissipation (E). Therefore, all of the first four equations require at least three variables in order to predict reaeration. The third variable is stream temperature (T). Equations 6-14 and 6-16 require a fourth measured variable for predicting reaeration. From a practical standpoint, the most desirable equation contains the fewest variables, if this can be done without sacrificing the statistical accuracy and precision of the prediction. A case in point is equations 6-15 and 6-16. Both have the same accuracy and precision, yet equation 6-15 requires one less variable or field measurement. On this basis, equation 6-15 was selected as the best predictive equation.

Figure 6-2 shows a plot of k_2 (actual) versus k_2 (predicted) using equation 6-15. The scatter of points about the line is much more uniform than those in Figure 6-1. There does not appear to be any tendency to overpredict or underpredict reaeration in any given region. A graphical solution to equation 6-15 is presented in Figure 6-3.

Several hydraulic characteristics generally associated with turbulence and reaeration do not appear in the equations proposed for small turbulent streams. Table 6-3 lists all of the hydraulic

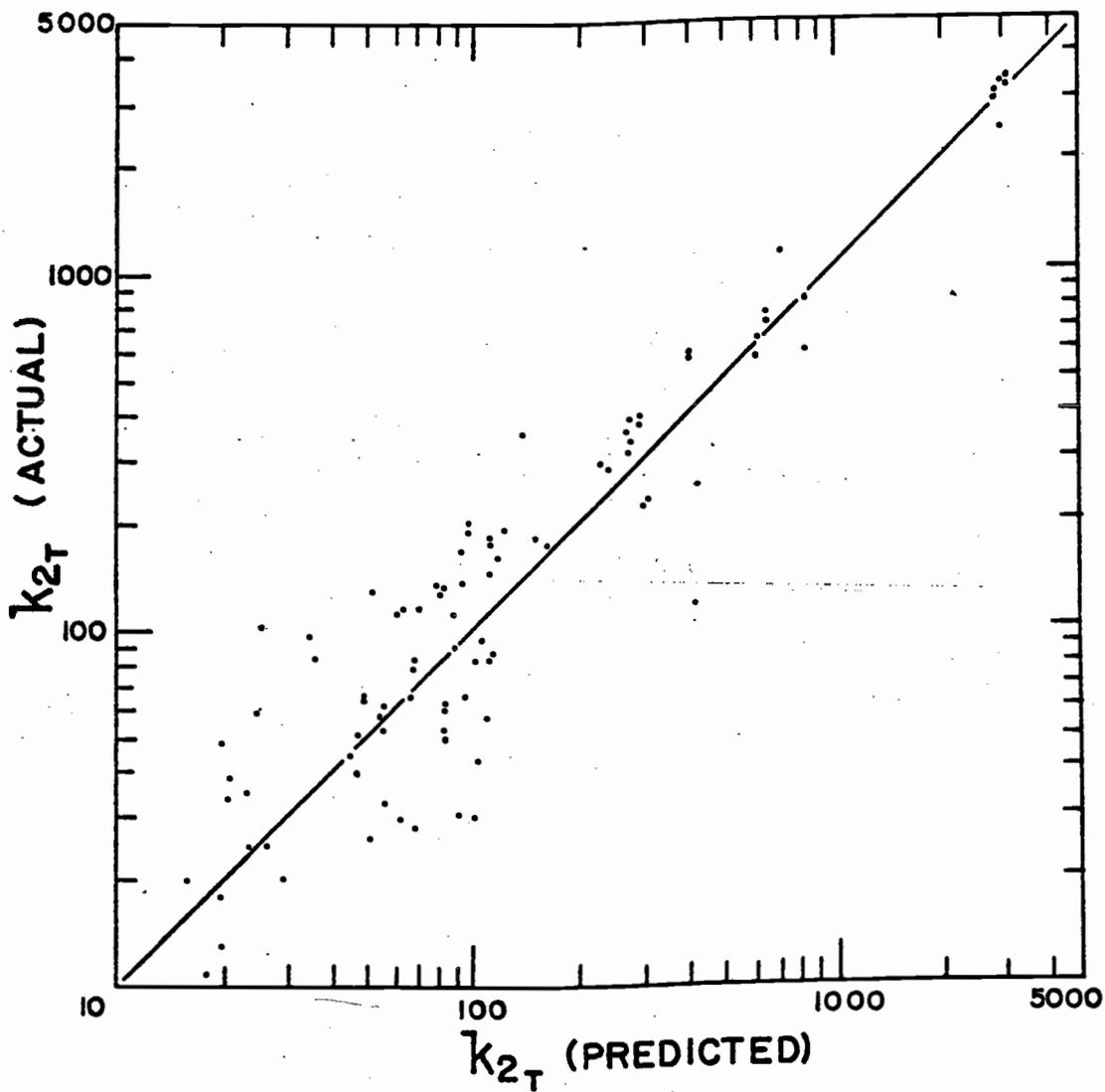


Figure 6-2. Actual versus predicted values of k_{2T} from equation 6-15.

$$k_{2T} \text{ (predicted)} = 1.016^{(T-20)} (181.6 E - 1657 S + 20.87).$$

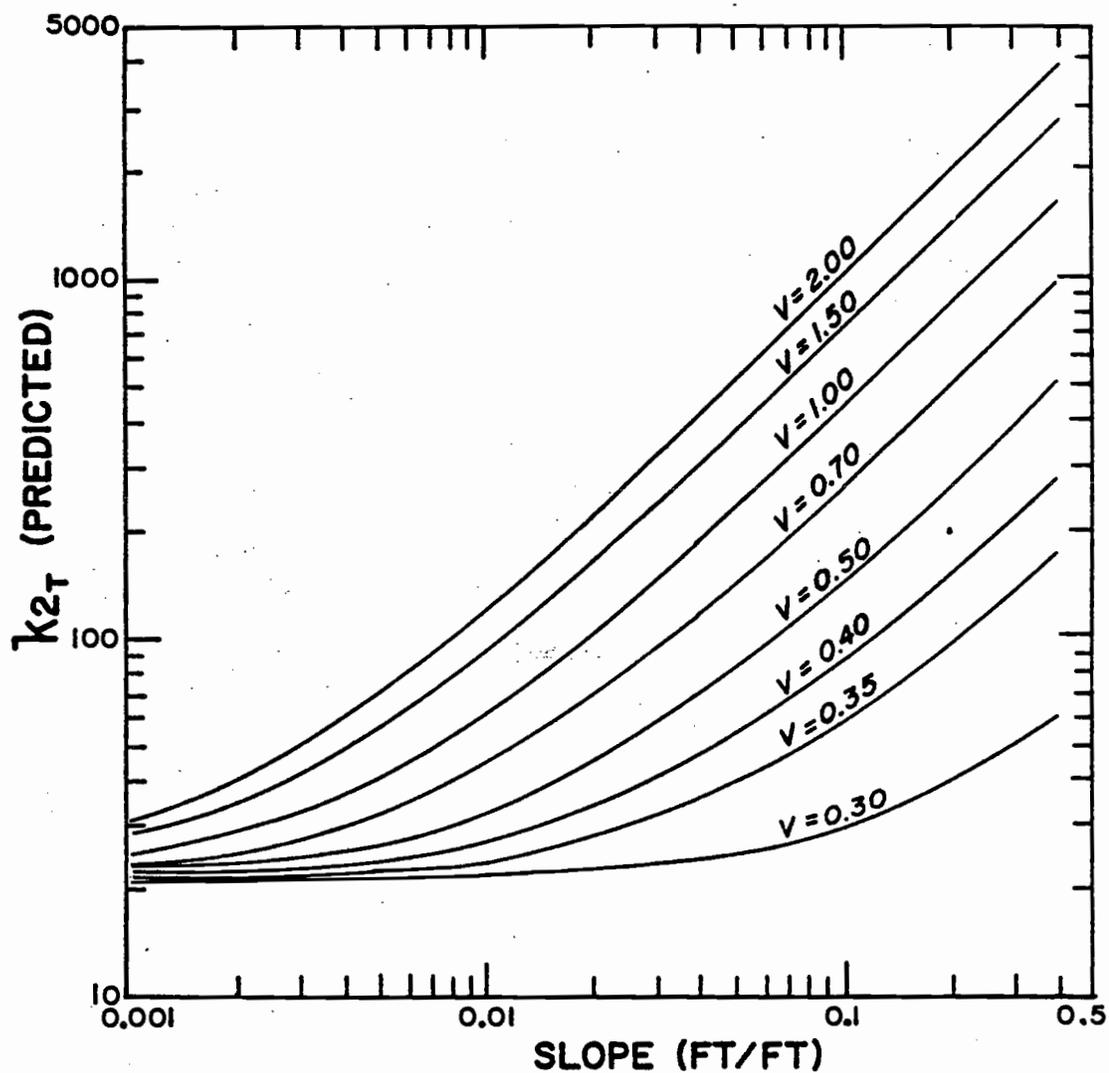


Figure 6-3. Graphical solution to equation 6-15 at T equal to 20° C.
 k_{2T} (predicted) = $1.016^{(T-20)} (181.6 E - 1657 S + 20.87)$.

characteristics used in the stepwise multiple linear regression analysis. The theoretical relationships between these characteristics and reaeration has been discussed in Chapter III. The factors in Table 6-3 have been listed in the order in which they entered the stepwise regression analysis. The factor which explained the greatest amount of variance in the dependent variable (k_2) was entered first. The next factor entered was the one which explained the greatest amount of the remaining variance.

Table 6-3. Factors used in the stepwise multiple linear regression analysis, showing order entered and simple correlation coefficients (r).

Order entered	Symbol	Factor	Simple correlation coefficient (r)
1	E	Energy dissipation	0.9885
2	S	Slope	0.9200
3	W	Width	-0.2218
4	Re	Reynolds number	0.1207
5	V*	Friction velocity	0.6373
6	Fr	Froude number	0.9186
7	V	Velocity	0.8686
8	f	Frictional resistance coefficient	-0.0567
9	n	Manning roughness coefficient	-0.1196
10	H	Depth	-0.2391

Only the first three factors entered were statistically significant. This means that the addition of any of the other factors would not significantly improve the predictive equation. The amount of unexplained variance was so small after entering the energy dissipation term (E) that most of the remaining terms dropped out of the equation. For this reason, most of the other factors from Table 6-3 do not appear in the predictive equations listed in Table 6-2 unless they were forced into an equation.

The best combination of factors for predicting reaeration is not necessarily a combination of factors having the highest simple correlation coefficients. Table 6-3 shows that the Froude number (Fr) and the velocity (V) had the third and fourth highest simple correlation coefficients. However, they were the sixth and seventh factors to enter the equation. This apparent anomaly can be explained by the close inter-relationships among some of the factors. The velocity (V) is used in computing the energy dissipation term (E). Therefore its effect on reaeration is reflected in the energy dissipation term. A similar relationship exists between velocity and the Froude number. The high correlation between reaeration and the Froude number is mainly a result of using velocity to calculate the Froude number. The statistical significance of any factor containing the velocity term is greatly reduced after accounting for the influence of velocity through the energy dissipation term.

The sign of the simple correlation coefficient (r) indicates a direct or inverse relationship between a variable and reaeration. Theoretically, all of the terms in Table 6-3 except stream depth (H) should be directly related to reaeration. However the width (W), the frictional resistance coefficient (f) and the Manning roughness coefficient (n) showed inverse relationships. The correlation coefficients for the latter two terms were too small to be significant. In this case, the correlation coefficients should be interpreted as showing a lack of relationship rather than an inverse relationship.

The inverse relationship between width and reaeration was probably a result of the strong influence of slope and velocity on reaeration. Velocity increases in response to increases in slope. The discharge remained constant between one reach and the next. According to the continuity equation ($Q = A V$) the cross-sectional area of the stream must decrease if the velocity increases. The cross-sectional area is generally reduced by proportional decreases in width and depth. Thus, the reaches with high velocities and high reaeration rates were characteristically narrower. This accounts for the inverse relationship between width and reaeration.

The positive and negative signs in equations 6-13 through 6-19 resulted from a statistical fit of the regression line to the data. These signs should not be interpreted as indicating direct or inverse relationships between predictive variables and reaeration.

For comparative purposes, the actual and predicted changes of dissolved oxygen in the study stream were plotted on the same graphs. Reaeration data from three different days are shown. These sets of data were chosen because dissolved oxygen measurements were available for the entire length of the study channel. The actual dissolved oxygen concentrations are shown by the solid lines in Figures 6-4, 6-5 and 6-6. Using the measured hydraulic characteristics, the reaeration rates were predicted in each reach by equation 6-15. These reaeration rates were then substituted into equation 2-15 to determine the predicted oxygen deficit at the end of each reach. The predicted dissolved oxygen concentrations at the end of each reach were obtained by subtracting the calculated oxygen deficits from the saturation values. The dashed lines in Figures 6-4, 6-5 and 6-6 represent the predicted dissolved oxygen concentrations at various points along the stream.

Figures 6-4 through 6-6 show that in most cases the predicted changes in dissolved oxygen are quite close to the changes actually measured. This is demonstrated by the fact that the line joining the predicted values is nearly parallel to the line joining the actual values.

Figure 6-5 shows the greatest discrepancy between predicted and actual values. This discrepancy is largely due to the under-prediction of reaeration in the three reaches between Stations 4 and 7. The predicted values from this point on were less than the actual

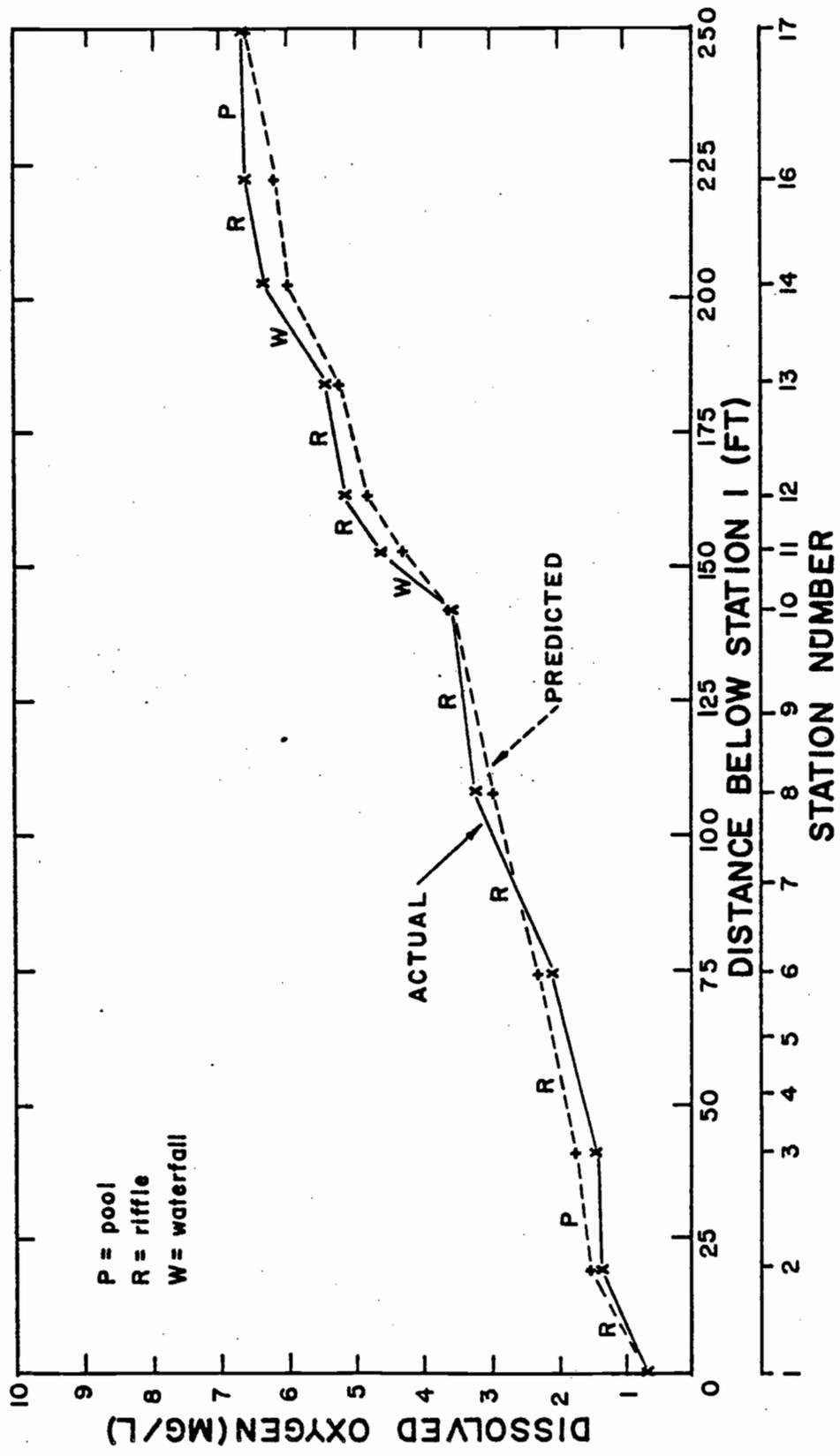


Figure 6-4. Actual and predicted values of dissolved oxygen using equation 6-15 and data points 211 through 221 (see Appendix E-3 and E-4).

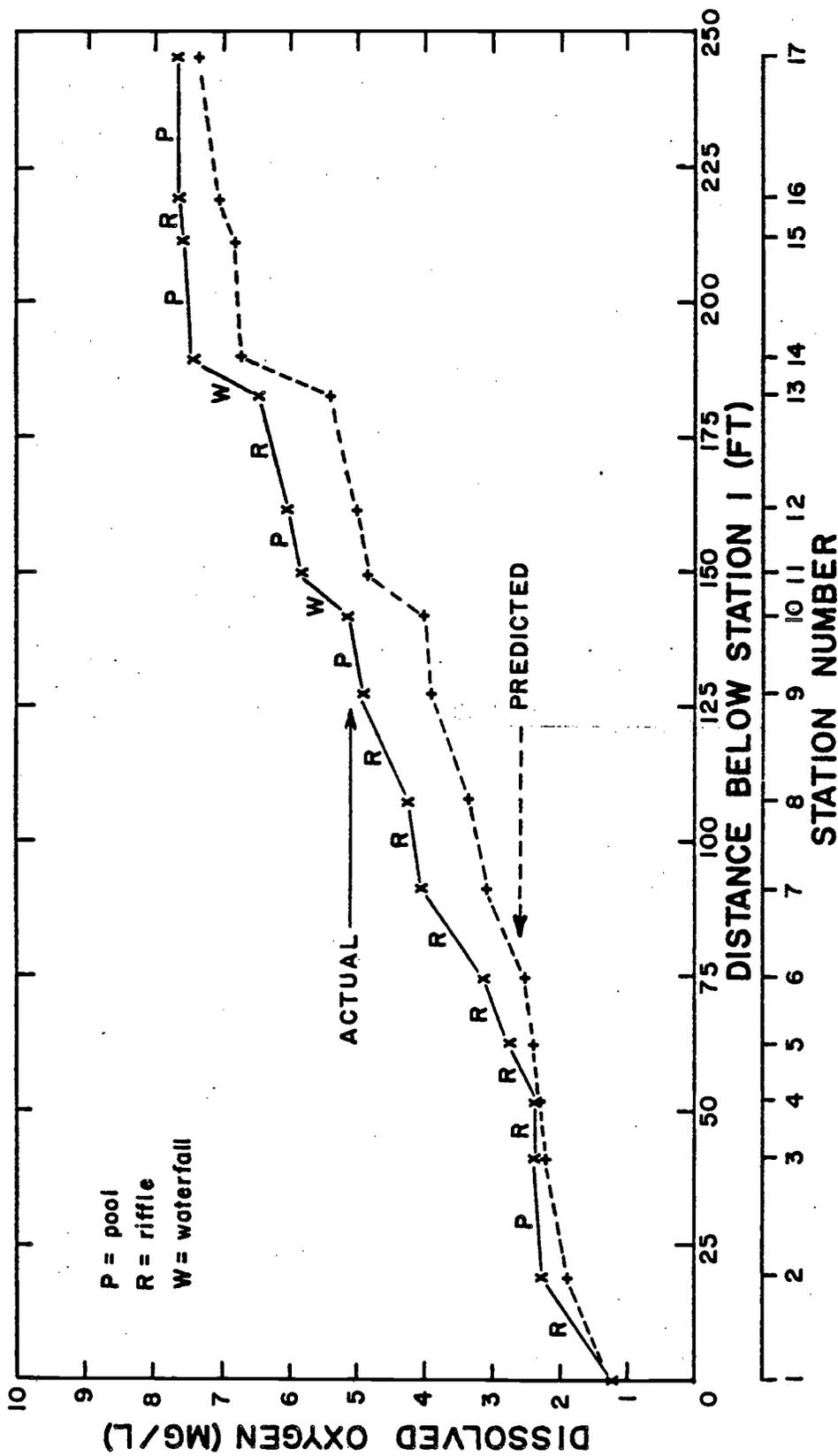


Figure 6-5. Actual and predicted value of dissolved oxygen using equation 6-15 and data points 501 through 516 (see Appendix E-3 and E-4).

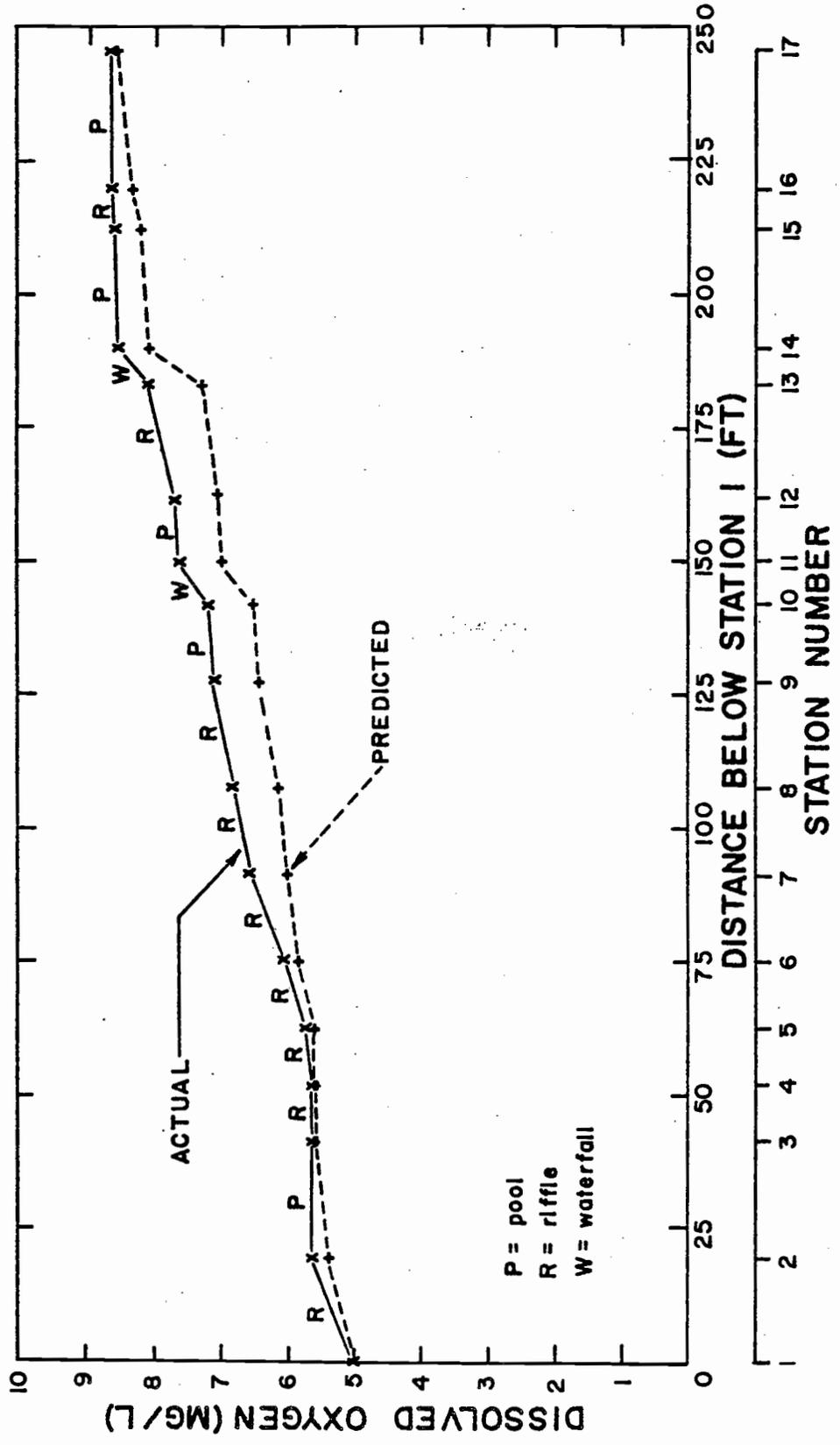


Figure 6-6. Actual and predicted values of dissolved oxygen using equation 6-15 and data points 601 through 616 (see Appendix E-3 and E-4).

values. Figure 6-5 gives the false impression that reaeration was underpredicted in all of the remaining reaches. This is not the case. The predicted and actual dissolved oxygen changes were quite similar as shown by the nearly parallel slopes in the downstream reaches. The divergence of the lines caused by the earlier prediction errors decreases in the last two reaches where the reaeration rates were slightly overpredicted.

Actual and predicted dissolved oxygen values were quite close at the last measured point in the stream. This was a result of coincidence and should not be interpreted as a demonstration of predictive accuracy over an extended section of stream channel. However, one would expect that some of these predictive errors would tend to be compensative. Figure 6-2 shows that the scatter of points is evenly distributed about the regression line. This indicates that there is an equal chance of underpredicting or overpredicting in any single reach. Figure 6-2 also shows that the scatter of points increases at lower values. These points represent the measurements made in the pool reaches where reaeration rates are characteristically lower. This scatter can be explained by the measurement inaccuracies at low velocities and shallow slopes. A measurement error in velocity of ± 0.01 ft/sec represents a ten percent error when the measured velocity is 0.10 ft/sec but only a one percent error when the measured velocity is 1.00 ft/sec. Analogous reasoning can be applied to the

measurement of slope.

It can be concluded from these results that equation 6-15 adequately predicts the reaeration rates in this stream. Extrapolation of these results to other streams with similar hydraulic characteristics will be discussed in the next chapter.

CHAPTER VII

DISCUSSION AND CONCLUSIONS

Application of Proposed Equations

Reaeration studies have been performed under many types of laboratory and natural conditions. The majority of studies conducted under natural conditions were in rivers and streams which were much larger than the one used for this study. One would intuitively expect reaeration rates in these deeper, slow moving rivers to be lower than those in a shallow, swift-moving mountain stream. This is indeed the case. The average reaeration rates found in the mountain stream were about 100 times greater than those reported in the literature for larger streams and rivers.

All of the equations proposed in the literature (Table 6-1) except those of Owens, Edwards and Gibbs (1964) tend to underpredict the actual reaeration rates found in the mountain stream. It seems significant that the type of stream used by Owens, Edwards and Gibbs most closely approximated the type of stream used in this study. This implies that empirical reaeration equations should not be extrapolated or extended much beyond the range of the experimental data. With this in mind, equation 6-15 should not be applied to channels where the slope exceeds 0.4 ft/ft, the velocity exceeds 2.0 ft/sec, or

the depth exceeds 1.0 feet.

Equation 6-15 is recommended as the best equation for predicting reaeration in small, turbulent mountain streams. From a statistical standpoint, the predictive accuracy and precision of this equation is quite good. From the standpoint of practical application, reaeration can be predicted from three easily measured stream characteristics. Equation 6-15 states:

$$\begin{aligned} k_{2T} &= 1.016^{(T-20)} (181.6E - 1657S + 20.87) \\ &= 1.016^{(T-20)} [181.6 (S V g) - 1657S + 20.87] \end{aligned} \quad (6-15)$$

where k_{2T} is the reaeration coefficient (base 10) in units of 1/days, T is the stream temperature in degrees C, E is the energy dissipation in ft^2/sec^3 , S is the channel slope in ft/ft, V is the flow velocity in ft/sec, and g is the gravitational acceleration constant in ft/sec^2 .

The stream temperature (T) can be easily measured with a thermometer. Temperature is the least sensitive variable in equation 6-15.

The slope of the stream channel (S) can be determined by appropriate surveying methods. The slope of the channel can be measured quite accurately in short sections with a transit or engineering level. The slope should be measured over a fairly long section of channel if determined by devices such as a hand level. In this case,

an average value of slope would be used in equation 6-15.

The flow velocity (V) should be determined by dye cloud measurements. As discussed in Chapter III, these characteristic velocities fall between the mean velocity and the maximum velocity of the reach. Measuring velocity with dye has several advantages. First, dye cloud measurements require less equipment and time than conventional current meter measurements. Second, the extremely shallow and turbulent conditions found in these mountain streams often makes current meter measurements impractical. Third, small variations in velocity within a reach are taken into account by the dye cloud method. Variations of this sort would not be reflected by current meter measurements unless measurements were made at numerous points throughout the reach. The velocity can be measured most accurately by dividing a long stream channel into a number of shorter reaches. The dye-cloud method of measuring velocities is most accurate in short reaches where the amount of dispersion in the dye cloud is minimized. The mean velocity would be determined by averaging the individual velocities measured in the shorter sections.

The assumption has been made that the mountain stream used in this study typifies the type of stream flowing through many of the forested areas of the Oregon Coast Range. This assumption seems reasonable. The major criterion used in selecting the study stream was that it contained a variety of hydrologic conditions typical of those

observed in other mountain streams. A second assumption must be made when applying the proposed equation. One must assume that the reaeration process observed in this stream adequately describes the process occurring in other streams of similar size. Complete proof of this assumption would require detailed reaeration data from other streams. However, the number of measurements and variety of hydraulic conditions considered by this study make extrapolation of these results to other streams less questionable.

One final point should be made regarding the use of equation 6-15. One of the objectives of this study was to provide an accurate and convenient method of predicting reaeration in small mountain streams. When applied to a specific stream, the objective will probably be to estimate mean reaeration over a relatively long section of stream channel. This can best be accomplished by subdividing the stream into a number of hydraulically uniform reaches and predicting reaeration for each individual reach.

Process Implications

In Chapter III the point was made that empirical studies often help explain the reaeration process in that they indicate how various hydraulic properties influence reaeration. Probably the most significant process implication drawn from this study is the relationship between energy dissipation and stream reaeration.

Energy dissipation was found to be an excellent indication of reaeration. This relationship supports the earlier conclusions drawn by Krenkel and Orlob (1962) and Dobbins (1964). Many previous investigators have commented on the close relationship which should exist between turbulence and reaeration. However there has been considerable disagreement as to the best method of estimating turbulence in a stream or river. Krenkel and Orlob (1962) and Dobbins (1964) argue that turbulence and reaeration can best be predicted and explained through an energy balance approach. They feel that the source of energy for the turbulence process is related to and derived from the loss in potential energy as the water flows downslope. The data from this study support this conclusion.

Two additional points should be considered when using an energy balance approach to explain the reaeration process. First, the loss in potential energy can be calculated for any given stream reach. However, the amount of potential energy lost is not directly equivalent to the amount of energy dissipated by turbulence. Some of the potential energy is expended in overcoming the frictional resistance of the channel bottom. Second, the loss in potential energy may not equal the amount of energy dissipated by turbulence, even after accounting for the frictional losses. Some of the potential energy may be converted to kinetic energy instead of being dissipated by turbulence. This would be the case if the velocity at the end of a

measured reach were greater than the velocity at the beginning of the reach. The higher velocity at the end of the reach would indicate that some of the potential energy had been expended to increase the kinetic energy. Apparently this point was overlooked by Krenkel and Orlob (1962). Problems of this type would not be encountered in the flume experiments used by Krenkel and Orlob to develop their reaeration equation. In the flume studies, the velocity was always uniform throughout the channel length. However some consideration should have been given to this question when recommending their flume equation for predicting reaeration in natural streams.

The literature equations underpredicted reaeration. Most of these literature equations were developed for and based on reaeration in larger river systems. This implies that the reaeration process in a large river is not the same as the process in a small turbulent stream. This is probably the case. However turbulence is probably the controlling factor in both situations.

Turbulence facilitates reaeration in two ways. Turbulence may increase the effective area for oxygen transfer. This occurs through deformation of the stream surface and by entrainment of air bubbles.

Air bubbles were present in the plunge pools beneath the two waterfalls of the study stream. As expected the highest reaeration rates occurred in these reaches. The turbulence which existed in the shallow, swift-moving riffle sections was probably sufficient to

entrain air bubbles. The reaeration rates in these reaches were also quite high. Oxygen transfer from entrained air bubbles would explain why the actual reaeration rates were higher than those predicted by the existing literature.

Most of the previous investigators have specified that their equations were not intended for predicting reaeration under conditions of a "heaving surface" (O'Conner and Dobbins, 1958) or "white water" (Churchill, Elmore and Buckingham, 1962). These terms refer to extreme states of turbulence where significant quantities of oxygen could be transferred to the water by entrained air bubbles.

Turbulence also promotes reaeration through vertical mixing. This maximizes the probability that oxygen deficient water will contact the atmosphere. The degree to which turbulent mixing extends through the bulk of the fluid is depth dependent. This is best appreciated by looking at the influence of depth (H) in Tables 6-1 and 6-2. The influence of depth on reaeration is significantly different in a small turbulent stream than in a large river. All of the literature equations include depth as a predictive variable. Table 6-3 shows that in a small turbulent stream depth is one of the least significant variables. Apparently, depth becomes a significant variable only at the lower levels of turbulence typically found in deep, slow-moving rivers. The turbulence encountered in a small mountain stream is probably so great that it penetrates uniformly through the entire depth

and thus is not depth dependent. O'Conner and Dobbins (1958) suggest such a relationship by proposing two different reaeration formulas. Equation 6-7 was to predict reaeration for depths less than five feet. Equation 6-9 was for streams with depths in excess of five feet. Table 6-1 shows that equation 6-7 predicted reaeration in the study stream more accurately than did equation 6-9. Thus, the difference in reaeration between large rivers and small streams is related to the degree to which turbulence extends through the bulk of the fluid.

BOD Considerations

The objective of this discussion is to demonstrate the combined effects of biochemical oxygen demand (BOD) and reaeration on the dissolved oxygen concentration in a typical mountain stream. The source of the BOD could be organic leachates and decomposing debris from slash which accumulated in the stream during the logging operation. For demonstration purposes, several simplifying assumptions have been made. A uniform set of characteristics was assumed for the stream. A velocity of 0.5 ft/sec and a channel slope of 0.04 ft/ft were used. The reaeration coefficient was assumed to be constant throughout the reach. The value chosen for the reaeration coefficient was 68.2 1/days. This was the mean value of the reaeration rate calculated for the study stream. A point source of BOD was also assumed.

The BOD rate constant (k_1) was assigned a value of 0.30 1/days. Research is presently underway to determine the BOD from floating logs. The value used for the BOD rate constant was based on the initial results from this study (Atkinson and Schaumburg, 1970). A BOD rate constant has not been determined for finely-divided, organic debris normally found on logged watersheds. The BOD rate constant for floating logs was considered to be the best available estimate. One would suspect this estimate to be conservative. The water temperature in the stream was assumed to be 15.1° C. The saturation value at this temperature is 10.00 mg/l. Three different values were used for the initial or ultimate BOD concentration (L_a). These were 1,000, 100 and 10 mg/l. With these assumptions, equation 2-12 was used to predict the oxygen deficits at various points along the stream. The deficits were then subtracted from the saturation value to obtain the predicted dissolved oxygen concentrations. The results of these calculations are shown in Figure 7-1.

Several significant conclusions can be drawn from the oxygen sag curves shown in Figure 7-1. First, it demonstrated the technique to be used in predicting the effects of logging on downstream oxygen concentrations. When methods are available to predict the initial or ultimate BOD from slash accumulations, this technique will find application in forest management decisions. Despite the assumptions used to derive these curves, the effects of slash accumulation

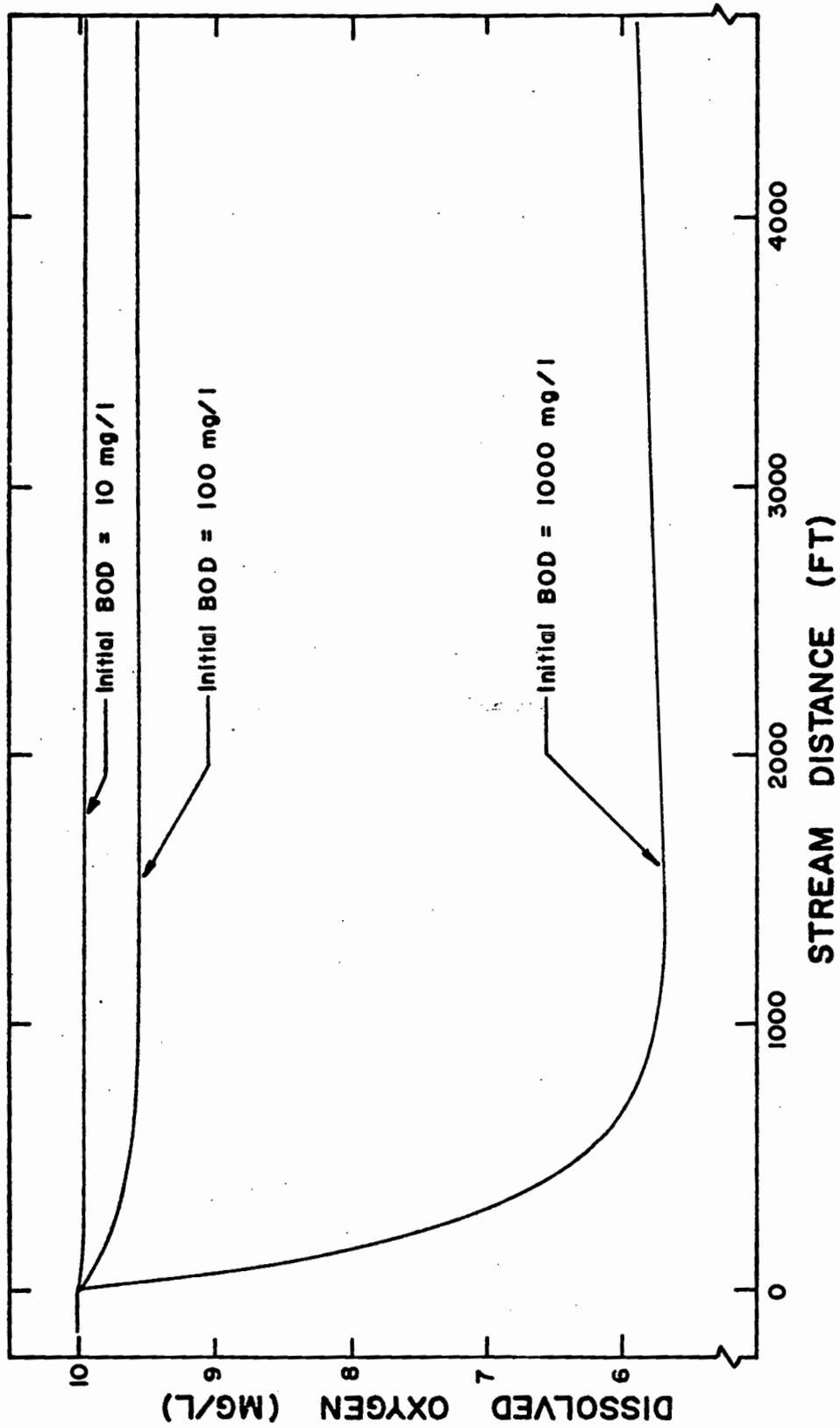


Figure 7-1. Hypothetical oxygen sag curves for a small stream using different initial BOD concentrations resulting from accumulated logging debris.

can be implied. Figure 7-1 shows that with an initial BOD of 1,000 mg/l, the minimum oxygen concentration was 5.70 mg/l. McGahey (1968) points out that normal domestic sewage has a five-day 20° C BOD of 200 to 250 mg/l.

Compare this information with the observations of Hall and Lantz (1969). They reported dissolved oxygen concentrations as low as 0.6 mg/l from a stream draining a clearcut watershed. Only a combination of high water temperatures, low reaeration rates, and high BOD from slash accumulations could produce such a low dissolved oxygen concentration.

The low oxygen concentrations reported by Hall and Lantz (1969) occurred on Needle Branch. This stream was one of three used in a study of the effects of logging on aquatic resources. Hypothetical oxygen sag curves were developed for Needle Branch using unpublished data collected by Brown (1969) and Hall and Lantz (1969). These data show Needle Branch to have a slope of 0.0144 ft/ft, a velocity of 0.05 ft/sec and a stream temperature of 27° C. The saturation value of oxygen at 27° C is 7.87 mg/l. Using these stream characteristics the reaeration coefficient predicted by equation 6-15 was 1.43 1/day. A point source of BOD and a BOD rate constant of 0.30 1/day were assumed. The same techniques used in calculating the oxygen sag curves in Figure 7-1 were used to develop the curves shown in Figure 7-2. Figure 7-2 shows the hypothetical oxygen sag curves for Needle

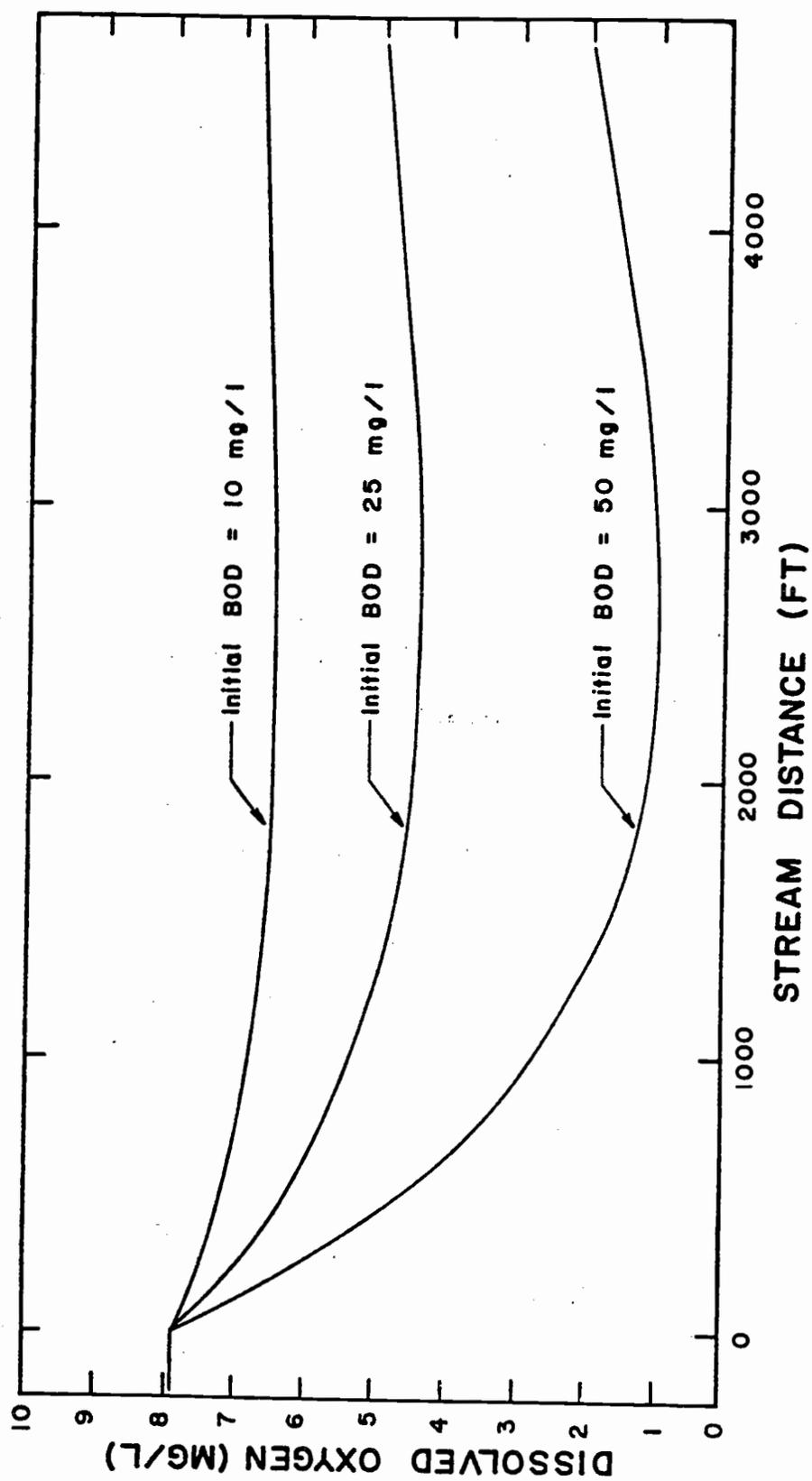


Figure 7-2. Hypothetical oxygen sag curves for Needle Branch using different initial BOD concentrations resulting from accumulated logging debris.

Branch using assumed initial BOD concentrations of 50, 25, and 10 mg/l.

Comparing Figures 7-1 and 7-2 emphasizes the importance of reaeration and stream temperature in maintaining acceptable dissolved oxygen levels. In Figure 7-1 high reaeration rates and cool stream temperatures offset an initial BOD concentration of 1000 mg/l. The minimum dissolved oxygen concentration under these conditions was 5.70 mg/l. In Needle Branch, the same minimum dissolved oxygen concentration could have resulted from an initial BOD of only 25 mg/l. The potential for natural purification was significantly less in Needle Branch because of the low reaeration rate and elevated stream temperature. An initial BOD concentration of 52 mg/l would have produced the 0.6 mg/l dissolved oxygen value reported for Needle Branch by Hall and Lantz (1969).

The possibility of BOD affecting the reaeration rates calculated for the study stream has been considered. If significant amounts of BOD had been present, the calculated reaeration rates would have been less than the actual rates. It was assumed that no BOD was present in the stream. Even though no measurements of BOD were made, it is unlikely that any significant amount existed. This conclusion was based on the measurements of stream dissolved oxygen before injecting the reducing agent. These measurements showed that the initial dissolved oxygen concentrations were at or very close to

the saturation value. If any BOD had been present, the initial oxygen concentrations would have been depressed below the saturation value.

Management Recommendations

The purpose of studying reaeration in small mountain streams is to be able to predict a stream's potential for natural purification. By knowing the reaeration capabilities of a stream, the effect of organic loading on the dissolved oxygen content can be predicted. This has been demonstrated in the previous section. The management objective is to prevent oxygen depletions which would endanger the anadromous fish population. This can be accomplished by removing or diminishing the sources of organic loading and promoting conditions favorable to reaeration.

The stream-side buffer strip is one of the most practical methods of preventing serious oxygen depletions. A buffer strip is an area of undisturbed vegetation bordering a stream which flows through a clearcut watershed. This management technique accomplishes several purposes. A buffer strip acts as a trash rack and prevents large debris from accumulating in the stream channel. Thus, some of the sources of organic loading are removed or at least diminished. This also reduces the chances of the stream becoming choked with finely-divided organic debris. Large accumulations of organic debris can reduce the flow velocity and often the surface area of the stream.

Figure 6-3 shows that reaeration rates decrease as the velocity is reduced. If the water is forced to seep through accumulations of finely-divided debris, it may have no direct contact with the air. Reaeration would certainly be suppressed under such conditions.

Water temperature regulation is another recognized benefit resulting from buffer strips (Brown, 1969). Riparian vegetation left to shade the stream prevent the large increases in water temperature observed on clearcuts without buffer strips. Cooler water temperatures provide a more optimum environment for fish. Lower temperatures also promote higher oxygen concentrations. Table E-1 shows that the lower the water temperature the higher the saturation value of oxygen. Subtracting any given oxygen deficit from an elevated saturation value yields a higher resulting dissolved oxygen concentration.

In cases where no buffer strip is left, some form of channel clearance is recommended. Removal of the larger slash will reduce some of the sources of BOD. It will also help prevent the formation of debris dams. Water seeping through closely packed debris dams has little or no potential for reaeration. Furthermore, the rate of organic leaching would be greatly accelerated under such conditions. By removing debris which would otherwise impede the flow of water, higher reaeration rates will result from increases in velocity and turbulence.

The effects of logging on stream dissolved oxygen would be most severe immediately following logging. Most of the floatable organic debris would be washed from the watershed by the high winter flows common in the Oregon Coast Range. The effects of debris can be severe, but in most cases they are temporary. However in logged watersheds where the larger debris has not been removed from the channel, high winter flows can create massive debris dams. Oxygen depletions could then be expected below these debris dams the following summer.

Suggestions for Further Research

This research has been directed towards methods of estimating reaeration in small mountain streams. Some general watershed management recommendations have been indicated. Further research is required to understand and predict the effect of logging on the dissolved oxygen content of these mountain streams. Research is presently underway to determine the biochemical oxygen demand from various types of logging debris. Initial estimates have already been made for the BOD rate constant (k_1). This study provides a method for estimating the reaeration coefficient (k_2). If the initial or ultimate BOD (L_a) can be predicted, then equation 2-12 can be used to predict the effect of logging on downstream oxygen concentrations.

Figure 7-1 showed a hypothetical solution to equation 2-12 for

various downstream points using assumed values for the initial BOD. Some method of estimating the amount of debris or BOD per foot of logged stream channel is needed. With this information, the predictive equation could be worked backwards for any given stream. Given the stream characteristics and the lowest acceptable oxygen concentration, a maximum allowable length could be set for a cutover stream channel.

Further evaluation of the proposed reaeration equation (6-15) is also recommended. Reaeration data from other streams should be collected to supplement the data points shown in Figure 6-2. Better definition of the regression line especially at the higher values would be helpful. If such studies are conducted the continued use of the YSI dissolved oxygen meter is recommended. This technique of measuring dissolved oxygen proved invaluable in this study. A specially constructed BOD probe is available for the oxygen meter. A probe such as this could be used effectively in the previously recommended BOD research.

Little is now known about the reaeration process in a natural stream when entrained air bubbles are present. From an academic standpoint, this process certainly needs further definition. However, the predictive value of such research is questionable. If research of this sort is to be used as a predictive tool, some method of determining the size and distribution of entrained air bubbles will have to

be developed.

An alternative approach to this study should be considered. The approach used in this research was to study one stream which could be subdivided into 16 hydraulically different reaches. An alternative would be to choose 16 different streams, each of which exhibited a different hydraulic condition. In either case, the degree of confidence in extending the results to other streams would be quite similar.

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APPENDIX A

NOTATION

A	area	ft ² or cm ²
C	dissolved oxygen concentration	mg/l
C _i	concentration of oxygen at the liquid interface	mg/l
C _s	saturation value of oxygen	mg/l
D	oxygen deficit	mg/l
D _a	initial oxygen deficit	mg/l
D _m	molecular diffusivity	ft ² /sec
e	mathematical constant	none
E	energy dissipation	ft ² /sec ³
f	frictional resistance coefficient	none
Fr	Froude number	none
g	gravitational acceleration constant	ft/sec ²
H	stream depth	ft
H _t	total head	ft
K ₁	BOD rate constant (base e)	1/day
K ₂	reaeration coefficient (base e)	1/day
K _g	mass transfer coefficient through gas film	none
K _L	mass transfer coefficient through liquid film	none
K _o	orifice correction	none
k ₁	BOD rate constant (base 10)	1/day
k ₂	reaeration coefficient (base 10)	1/day
L	BOD concentration	mg/l
L _a	initial BOD concentration	mg/l
L _f	liquid film thickness	ft
N	solution normality	equiv/mole
n	Manning roughness coefficient	none
P	barometric pressure	mm of Hg
P _g	concentration of oxygen in main body of gas	mg/l
P _i	concentration of oxygen at the gas interface	mg/l
P _o	oxygen from photosynthesis	mg
p	pressure of saturated water vapor	mm of Hg
Q	discharge	ft ³ /sec
R ₂	hydraulic radius	ft
R ²	multiple correlation coefficient	none
Re	Reynolds number	none

R_s	oxygen used for plant respiration	mg
r	simple correlation coefficient	none
r_n	surface renewal rate	sec
S	channel slope	ft/ft
T	temperature	$^{\circ}\text{C}$
T_e	absorption time	sec
t	time	sec or day
V	velocity	ft/sec
V^*	friction velocity	ft/sec
V_o	volume of liquid undergoing reaeration	ft^3
W	stream width	ft
W_t	weight of solute gas	lbs
z	elevation head	ft
γ	specific weight	lbs/ft^3
ν	kinematic viscosity	ft^2/sec
π	mathematical constant	none
a		
b		
c		
d	empirical coefficients or	none
n_b	constants	
θ		

APPENDIX B

DERIVATION OF THE OXYGEN DEFICIT EQUATION
(EQUATION 2-12)

Given:
$$\frac{dL}{dt} = -K_1 L \quad (1)$$

$$\frac{dD}{dt} = K_1 L - K_2 D \quad (2)$$

Solving (1) for L:

$$\frac{dL}{L} = -K_1 dt \quad (3)$$

$$\int_{L_a}^L \frac{dL}{L} = -K_1 \int_0^t dt \quad (4)$$

In (4), $L_a = L$ at $t = 0$:

$$\int_{L_a}^L \frac{dL}{L} = \ln L - \ln L_a = \ln \frac{L}{L_a} \quad (5)$$

$$\int_0^t dt = t \quad (6)$$

Substituting (5) and (6) into (4):

$$\ln \frac{L}{L_a} = -K_1 t \quad (7)$$

Exponentiating both sides:

$$e^{\ln \frac{L}{L_a}} = e^{-K_1 t} \quad (8)$$

$$\frac{L}{La} = e^{-K_1 t} \quad (9)$$

$$L = La e^{-K_1 t} \quad (10)$$

Substituting $La e^{-K_1 t}$ for L in (2):

$$\frac{dD}{dt} = K_1 La e^{-K_1 t} - K_2 D \quad (11)$$

Equation (11) may be solved for D by using a general solution presented in Differential Equations by Agnew (1960) on page 71.

$$\frac{dy}{dx} + my = e^{kx} \quad (12)$$

The solution to (12) is:

$$y = \frac{1}{k-m} e^{kx} + C e^{-mx} \quad (13)$$

By algebraic manipulation, (11) can be arranged in the form of (12) as:

$$\frac{d}{dt} \frac{D}{K_1 La} + K_2 \frac{D}{K_1 La} = e^{-K_1 t} \quad (14)$$

Using the solution to (12):

$$\frac{D}{K_1 La} = \frac{1}{-K_1 + K_2} e^{-K_1 t} + C e^{-K_2 t} \quad (15)$$

$$D = \frac{K_1 La}{K_2 - K_1} e^{-K_1 t} + C' e^{-K_2 t} \quad (16)$$

In (15) and (16), C and C' are arbitrary constants and $C' = C - K_1 La$. Since K_1 and La are constants, they can be included in the arbitrary constant C' . This can be done without affecting the ultimate solution. The value of C' can be determined by solving (16) for C' at $t = 0$:

$$\begin{aligned} Da &= \frac{K_1 La}{K_2 - K_1} e^0 + C' e^0 \\ &= \frac{K_1 La}{K_2 - K_1} + C' \end{aligned} \quad (17)$$

$$C' = Da - \frac{K_1 La}{K_2 - K_1} \quad (18)$$

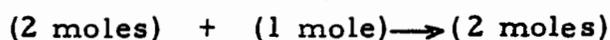
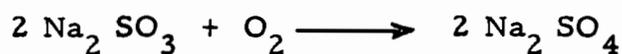
In (18), Da is the oxygen deficit at t equal to 0. Substituting the value of C from (18) into (16) yields:

$$\begin{aligned} D &= \frac{K_1 La}{K_2 - K_1} e^{-K_1 t} + \left[\frac{Da - K_1 La}{K_2 - K_1} \right] e^{-K_2 t} \\ &= \frac{K_1 La}{K_2 - K_1} e^{-K_1 t} + Da e^{-K_2 t} - \frac{K_1 La}{K_2 - K_1} e^{-K_2 t} \\ &= \frac{K_1 La}{K_2 - K_1} \left[e^{-K_1 t} - e^{-K_2 t} \right] + Da e^{-K_2 t} \end{aligned} \quad (19)$$

APPENDIX C

DERIVATION OF SODIUM SULFITE CONCENTRATION EQUATION

The basic chemical balance for the sulfite-oxygen reaction is:



The equivalent weight of Na_2SO_3 per mole of O_2 is then:

$$\begin{aligned} \text{equivalent weight of } \text{Na}_2\text{SO}_3 &= \frac{2 \text{ (mole weight } \text{Na}_2\text{SO}_3)}{\text{mole weight } \text{O}_2} \\ &= \frac{252 \text{ gm } \text{Na}_2\text{SO}_3}{32 \text{ gm } \text{O}_2} = 7.86 \text{ gm } \text{Na}_2\text{SO}_3/\text{gm } \text{O}_2 \end{aligned}$$

To reduce the oxygen concentration in the stream to a given level below saturation (D), an equivalent amount of sulfite must be added at a rate proportional to the amount of oxygen in the water and the flow rate of the stream. The term D is defined as:

$$\begin{aligned} D &= \text{oxygen deficit below saturation value} \\ &= \text{saturation value (mg/l)} - \text{desired minimum dissolved oxygen concentration (mg/l)} \end{aligned}$$

Beginning with a mass balance equation, an equation can be derived to calculate the required concentration of sodium sulfite reducing

agent.

amount Na_2SO_3 injected = amount O_2 reduced (D) x equivalent

wt. Na_2SO_3 or:

$$\text{conc. of } \text{Na}_2\text{SO}_3 \left(\frac{\text{gms}}{\text{liter}} \right) \times \frac{1000 \text{ mg}}{\text{gm}} \times \frac{\text{injection rate } \left(\frac{\text{mls}}{\text{min}} \right)}{1000 \text{ mls}}$$

$$= D \left(\frac{\text{mg}}{\text{liter}} \right) \times \text{flow } \left(\frac{\text{ft}^3}{\text{sec}} \right) \times \frac{60 \text{ sec}}{\text{min}} \times \frac{28.3 \text{ liters}}{\text{ft}^3} \times \frac{7.86 \text{ gm } \text{Na}_2\text{SO}_3}{\text{gm } \text{O}_2}$$

Cancelling terms and rewriting:

$$\text{required concentration of } \text{Na}_2\text{SO}_3 \left(\frac{\text{gms}}{\text{liter}} \right) =$$

$$\frac{(1.335 \times 10^4) \times D \left(\frac{\text{mg}}{\text{liter}} \right) \times \text{flow (cfs)}}{\text{injection rate } \left(\frac{\text{mls}}{\text{min}} \right)}$$

APPENDIX D

The Bernoulli equation can be used to prove constant injection rate. The injection rate depends only on the elevation head and the orifice characteristics, and not on the level of the liquid in the container. Consider the basic Bernoulli equation as presented by Albertson, Barton and Simons (1960):

total head (H_t) = pressure head + velocity head + elevation head

$$\text{or } H_t = \frac{P}{\gamma} + \frac{V^2}{2g} + z \quad (1)$$

Where H_t is the total head in lbs/sq ft, P is the pressure in lbs/sq ft, γ is the specific weight in lbs/cu ft, V is the velocity in ft/sec, g is the acceleration due to gravity in ft/sec², and z is the elevation in feet.

The Bernoulli equation can in the following form because the energy or total head must be the same at all points in the system:

$$H_t (\text{Point 1}) = H_t (\text{Point 2}) \quad (2)$$

$$\text{or } \frac{P_1}{\gamma} + \frac{V_1^2}{2g} + z_1 = \frac{P_2}{\gamma} + \frac{V_2^2}{2g} + z_2 \quad (3)$$

P_1/γ and P_2/γ are equal and drop out of the equation because atmospheric pressure operates at both Point 1 and Point 2. There is no

velocity head at Point 1, so $V_1^2/2g$ is zero. Since z_2 is the elevation head datum, it also is zero.

Collecting terms and rewriting, the equation becomes:

$$z_1 = \frac{V_2^2}{2g} \quad (4)$$

or
$$V_2 = \sqrt{2g z_1} \quad (5)$$

The discharge from the orifice can be specified by applying the continuity equation ($Q = A \cdot V$) and an orifice correction (K_o):

$$Q = K_o A V = K_o A \sqrt{2g z_1} \quad (6)$$

where A is the cross-sectional area of the orifice in sq ft. The pressure head (or the level of the liquid inside the container) has no effect on the discharge from the orifice since the discharge (Q) is dependent only on the characteristics of the orifice and the elevation head (z_1).

Table E-1. Saturation values of dissolved oxygen in pure water at 760 mm Hg pressure.

Temperature (Deg. C)	Truesdale ¹ (mg/l)	Churchill ² (mg/l)	Standard Methods ³ (mg/l)
0	14.16	14.65	14.6
1	13.77	14.25	14.2
2	13.40	13.86	13.8
3	13.05	13.49	13.5
4	12.70	13.13	13.1
5	12.37	12.79	12.8
6	12.06	12.46	12.5
7	11.76	12.14	12.5
8	11.47	11.84	11.9
9	11.19	11.55	11.6
10	10.92	11.27	11.3
11	10.67	11.00	11.1
12	10.43	10.75	10.8
13	10.20	10.50	10.6
14	9.98	10.26	10.4
15	9.76	10.03	10.2
16	9.56	9.82	10.0
17	9.37	9.61	9.7
18	9.18	9.40	9.5
19	9.01	9.21	9.4
20	8.84	9.02	9.2
21	8.68	8.84	9.0
22	8.53	8.67	8.8
23	8.38	8.50	8.7
24	8.25	8.33	8.5
25	8.11	8.18	8.4
26	7.99	8.02	8.2
27	7.86	7.87	8.1
28	7.75	7.72	7.9
29	7.64	7.58	7.8
30	7.53	7.44	7.6

¹ Truesdale et al. (1955)

² Churchill, Buckingham and Elmore (1962)

³ American Public Health Association (1960)