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Title: MODELING THE IMPACT OF LOGGING DEBRIS ON THE DISSOLVED OXYGEN BALANCE OF SMALL MOUNTAIN STREAMS

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Previous research has indicated that considerable amounts of finely divided slash accumulate in small mountain streams following timber harvesting. The subsequent biological decomposition of this organic matter can result in the reduction of dissolved oxygen (DO) concentration to levels as low as 0.6 ppm. These DO levels are lethal to the salmon and trout fry which inhabit these waters during the summer months.

The purpose of this study was to verify the results of previous research on the biochemical oxygen demand (BOD) characteristics of Douglas-fir needles, western hemlock needles, and red alder leaves, and to develop an oxygen balance model to describe the effect of logging debris BOD on the dissolved oxygen of small mountain streams.

The concept of the leaching process was examined and refined. As a result K_4, the leaching rate constant, was introduced to describe the addition of leachate to the stream. This necessitated redefining K_1, the first order decay rate constant, as it had been reported in previous studies. Slash may be described by its ultimate oxygen demand, or potential BOD, and two rate constants, K_1 and K_4. Manometric BOD tests were run for 20-day periods to estimate the potential BOD and the rate constants K_1 and K_4. The 20-day ultimate BOD, K_1 and K_4 estimates at 20°C were: 139 mg O_2 / gm (dry weight), 0.266 and 0.189 per day for Douglas-fir needles, 183 mg O_2 / gm (dry weight), 0.202 and 0.089 per day for western hemlock needles, and 226 mg O_2 /gm (dry weight), 0.121 and 0.141 per day for red alder leaves.

Laboratory experiments designed to examine the possibilities of occurrence and impact of nitrification on the oxygen demand curve concluded that even though it is possible to observe nitrification under special conditions, coastal streams are generally too nitrogen and carbon poor for nitrification to exert a significant oxygen demand. Even with the addition of large amounts of vegetation to the stream water, an available form of nitrogen must be added to produce detectable amounts of nitrogenous oxygen demand. It was found that the presence of nitrification cannot be detected in small mountain streams by nitrate analysis because of rapid fixation of nitrogen. Thus, the

nitrification process appears to be of more concern for evaluating nutrient pathways and fixation than oxygen demand.

The leachate of the Douglas-fir, western hemlock, and red alder vegetation was analyzed for sugar and phenol concentration and rate of leaching. The vegetation was gas sterilized using ethylene oxide. The sterile vegetation was placed in autoclaved aliquots of water and sealed. The leachate of each vegetation type was analyzed at periodic intervals over 90 days, and the leachate expressed in glucose and gallic acid equivalents. Of the 120 samples treated, only one showed signs of contamination. The sugar and phenolic release rate constants for Douglas-fir and western hemlock were very similar; 0.049 and 0.062 for Douglas-fir sugars and phenols respectively and 0.060 and 0.046 for western hemlock sugars and phenols. All of these rates are less than the leaching rates derived by BOD experimentation. These differences may be due to biological interaction in the leaching process. The leaching rate constants for red alder leaves were much higher than the experimental rates, 0.88 and 0.187 for sugars and phenols respectively.

The mean maximum sugar concentration was: 117 mg glucose equivalent / gm (dry weight) for Douglas-fir needles, 86 mg glucose equivalents / gm (dry weight) for western hemlock needles, 121 mg glucose equivalents / gm (dry weight) for red alder leaves, and 50 mg glucose equivalent/gm (dry weight) for dissected Douglas-fir twigs.

No rate constant was computed for Douglas-fir twigs because of the lack of fit to an exponential function.

Experimentation of the effects of temperature showed a significant response of potential BOD, K_1 and K_4 to variations in water temperature from 15 to 27°C. In general, ultimate BOD, K_1 and K_4 values increased with increasing water temperature.

The effect of mass concentration on ultimate BOD, K_1 and K_4 were negligible for vegetative loadings of 4 to 16 gm, fresh weight per liter.

The second phase of this study involved constructing a finite difference model of the dissolved oxygen balance in small mountain streams. A fixed cell approach was used to demonstrate the negligible effects of dispersion on pollutant and oxygen concentration. This then allowed the use of the much more efficient Lagrange variation of the finite difference modeling technique, utilizing a moving cell. By using available data and estimates of the distribution of slash in a simulated stream following clearcutting along 991 ft of the stream, DO levels as low as 5.72 ppm were computed for the simulated stream 5890 ft below the top of the clearcut. At these DO concentrations, salmon and trout fry experience "critical" stress conditions. Further model runs indicated that by reducing the length of the clearcut or by removing a portion of slash this condition could be avoided.

Modeling the Impact of Logging Debris on the Dissolved Oxygen Balance of Small Mountain Streams

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INTRODUCTION

The Pacific Northwest depends heavily on its renewable natural resources, timber and fish, for jobs, food, building materials, and export commodities. These two resources provide the raw materials for the harvesting and processing industries upon which the economy of the Pacific Northwest is based. Both industries are increasing their harvest yields because of rapidly increasing demand for wood products and high protein food. Intensive resource management has become the watch-word of the decade.

The coastal watersheds of Oregon and Washington contain the most productive timberlands in the Pacific Northwest. These watersheds also give rise to numerous streams and rivers which are the spawning and rearing grounds of salmon and trout. The quality of the water of these streams is paramount to fish survival and productivity. Water quality, in turn, can be significantly affected by land management practices.

The concern for water quality has been accelerated in recent years by two factors. First, there has been an alarming decline of the salmon runs caused by pollution and obstruction of the waterways. And second, water quality has become a focal point of an aroused public

concerned for the environment. Their concern is for the preservation of the quality of sportfishing areas as well as concern for the welfare of the fisheries industry. Because of the decline of the salmon runs on the Columbia and Willamette rivers, the fisheries and sportsmen have begun to rely more heavily on the coastal streams to produce as much or more than before. This added pressure means that coastal watershed managers will have to maintain the naturally high quality of the water in these streams to maximize their fish producing capacity.

Because most coastal watersheds are primarily managed for timber production, public and private foresters must work with the fishery biologists and managers, and the public, to evaluate the impact of timber harvest practices on water quality and yield. Such evaluations may necessitate restricted use of some harvest techniques and the development of new ones.

Water quality includes stream water temperature, turbidity, sedimentation, nutrient and oxygen concentration, and obstructions to fish migration resulting from logging debris and road failures. All of these parameters are important in evaluating man's impact on water quality. Of these water quality parameters, dissolved oxygen concentration is one of the most important, yet least studied, aspects.

One of the most controversial timber harvest techniques used in the Pacific Northwest is clearcutting. Clearcutting involves the removal of all merchantable wood fiber from a forested area. It is

used extensively in even-aged management, because it is economical and in many cases enhances the prospects for regeneration of forest stands. On steep terrain, as is the case on much of the West Coast, high-lead logging is practiced. As much of a clearcut unit as possible is yarded to as few landings as possible, dictated by economics. In the past this has resulted in trees being felled or yarded across streams. Any time timber is felled in or across a stream, large amounts of finely divided material are deposited in the stream. Even the subsequent removal of the merchantable wood, large limbs, and twigs does not effectively reduce the volume of fine residue.

Most timber harvesting occurs during the warm, dry summer months. This also coincides with the minimum annual stream flow and maximum water temperatures. Finely divided logging debris deposited under these conditions is held in place, unable to be flushed through the system. In these areas of debris accumulation, the oxygen concentration can fall to levels lethal to salmon and trout fry which inhabit these streams during the summer. At dissolved oxygen (DO) levels less than 5 ppm, salmon and trout experience acute oxygen stress. DO levels below 1 ppm are lethal to these fish.

Finely divided logging residue contains numerous sugars and phenols which readily leach into the stream water. This leachate then undergoes biochemical oxidation. As this decomposible leachate is degraded, oxygen is consumed. Biochemical oxygen demand (BOD)

is the amount of oxygen required to completely decompose a given amount of organic matter under aerobic conditions.

Stream temperature often increases after clearcutting, and may cause some serious problems. The saturated oxygen concentration at $0^{\circ}C$ is 14.6 ppm and at $20^{\circ}C$ is 9.2 ppm. Both water temperature increases and BOD lower DO to levels which require salmon and trout fry to increase their body maintenance and metabolism functions. In extreme conditions, the fish may be killed outright by direct heat. At sublethal oxygen concentrations the fish experience oxygen stress. Under these conditions the fry require more food to compensate for the increase in metabolism, they cannot be competitive with more tolerant organisms, and they are more susceptible to disease.

The large amounts of debris that accumulate in combination with the significant increases in stream water temperature following clearcutting can result in serious dissolved oxygen depletion. The dissolved oxygen balance which exists in nature attempts to dynamically stabilize this drain by replenishing the water with atmospheric oxygen in proportion to the size of the dissolved oxygen deficit. Thus as oxygen is removed by the decay process, the stream's potential for reaeration increases. When the rate of oxygen depletion exceeds the reaeration rate, the oxygen concentration decreases. Conversely as the reaeration rate exceeds depletion the oxygen level increases.

If the organic loading of the stream water is very large, the rate of deoxygenation can exceed the reaeration rate for long periods of time (on the order of days to weeks). Under these conditions severe oxygen deficits coupled with fish mortality have been documented. Currently forest managers have no way of predicting the impact of logging debris on leachate and dissolved oxygen concentrations. A dissolved oxygen balance model would enable foresters to predict the on-site and downstream impact of debris loading and to locate areas where residue removal is necessary or alternate harvest techniques should be used.

For such a model to be developed to a point of reliable applicability, several parameters must be measured or estimated. To predict the impact of logging debris on dissolved oxygen in stream water, the potential BOD and rate of decay must be evaluated for the various species of fish and environmental conditions which occur. Reaeration potential of the stream must be predicted from field survey data of stream hydraulic characteristics. And the interaction of these two processes, also termed the dissolved oxygen balance, must be predicted with respect to both time and distance along the stream course.

Previous studies have been conducted to determine the potential BOD and rate constants of Douglas-fir (Pseudotsuga menziesii [Mirb.] Franco) needles, western hemlock (Tsuga heterophylla [Raf.]

Sarg.) needles and red alder (Alnus rubra Bong.) leaves in stream water. Attempts were also made 1) to estimate the concentration of sugars and phenols in leachate with respect to time, 2) to estimate the effect of temperature fluctuations on potential BOD and rate constants, 3) to determine if nitrification is a significant factor in the natural biodegradation of finely divided debris in streams, and 4) to estimate the effect of variations of mass concentration on the potential BOD and rate constants (Ponce, 1974).

The objectives of this study are twofold. One is to confirm and clarify the results of the previously mentioned studies on BOD characteristics of finely divided slash and to add to the body of BOD data available. The second objective is to develop the structure of an oxygen balance model for small forest streams which contain logging slash in some portion, by combining the results from previous BOD and reaeration studies. Such a model could ultimately be used to design harvest units which minimize fish mortality, and to locate problem areas where special stream cleanup is necessary. This model will be the basis for defining areas for continued research which could eventually lead to validation of the model.

LITERATURE REVIEW

Dissolved oxygen in stream water has long been recognized as a primary water quality parameter. Not only is it important to aquatic organisms to sustain life, but also it is the principal indicator of the degree of organic pollution present and a measure of the assimilative capacity of water bodies. Because it is a good indicator of water quality, minimum dissolved oxygen (DO) levels are closely regulated by environmental quality acts. There also is currently a trend toward closer DO monitoring and stricter enforcement of environmental standards on forest operations. Because most of the aquatic organisms of value to man are aerobic and rely upon oxygen for their existence, deoxygenation of their habitat is almost prohibitory.

Deoxygenation can occur whenever organic matter is added to a body of water containing microorganisms. Biological decomposition results in the oxidation of the organic matter to carbon dioxide and water. Microorganisms extract the oxygen required for this process from the water. Should this condition continue until the oxygen concentration becomes zero, all aerobic organisms would suffocate. When this occurs, the body of water is said to have gone anaerobic.

Because of the importance and complexity of the role of oxygen in the aquatic environment, a brief review of the following areas is in order. The importance of oxygen to the survival of salmon and trout

is examined to determine minimum levels of dissolved oxygen for acceptable fish habitat. A general review of the oxygen balance will introduce the sections on biological oxidation and associated biochemical oxygen demand (BOD) of finely divided slash, and a discussion of the reaeration process in small mountain streams. This review of the dissolved oxygen balance and its components with their relation to forest water quality will show how vegetative and hydraulic characteristics can be used to estimate oxygen balance parameters.

Modeling the impact of pollution on the dissolved oxygen balance will be examined to review previous attempts to bring the various components of the dissolved oxygen balance together into descriptive and predictive models. This section is presented in detail in order to lay a proper foundation for the presentation of a dissolved oxygen model for small mountain streams.

The Importance of Dissolved Oxygen to Salmonoids

Coho salmon and trout of the Pacific Northwest spawn in the coastal streams of Oregon and Washington during the late fall and early winter. The fry hatch and live in the streams at least one year before migrating to the ocean. During the year the fish require high levels of dissolved oxygen for metabolism and maintenance of all body functions. This makes them particularly vulnerable to adverse changes in dissolved oxygen.

The summer months are particularly hazardous for the young fry. Pacific Northwest summers are warm and dry, averaging less than 4 inches of rainfall on the western side of the Coast Range (Johnsgard, 1963) and less than 2 inches on the east side (Bates, 1969) during the three summer months of June, July and August. Stream flow during this period falls to minimum levels for up to 90 days, before the first fall freshet flushes the stream systems. This season also coincides with the height of the timber harvest activities, often resulting in large deposits of fine logging debris in the streams. Two resulting hazards to fish of particular concern are increases in water temperature and biochemical oxygen demand (BOD). Both of these conditions cause physiological stress to the salmon and trout fry, either directly or indirectly.

If the stream water temperature increases and the dissolved oxygen concentration is depressed, fish metabolism and body maintenance functions must increase. If the changes are great enough, the fish may die due to direct heat or suffocation. Indirectly, the salmon and trout fry cannot effectively compete with other more tolerant species of fish, due to decreased activity and growth. Under stress or sublethal conditions, the fry are also more susceptible to parasites and disease (Lantz, 1971).

As a general "rule of thumb," it is recommended by fishery biologists that dissolved oxygen (DO) levels not be allowed to drop

below 5 ppm for fish in warm water habitats and 6 ppm for fish in cold water habitats. Oxygen concentrations from 2.5 to 5 ppm are generally accepted as the sublethal region for salmonoids. At sublethal levels the fish experience critical stress conditions and rapid decreases in weight gain (Herrmann, Warren and Doudoroff, 1962).

These low DO levels may also obstruct fish migration as well as produce severe stress conditions. Shelford and Allee (1913) and Jones (1952) have noted avoidance and violent reactions of over 16 species of fish native to North America to water containing 0.5 to 3.5 ppm oxygen. Whitmore, Warren and Doudoroff (1960) demonstrated pronounced avoidance of water containing less than 4.5 ppm oxygen by chinook and coho salmon. Temperature and length of exposure to low DO levels can cause significant variations in fish reactions to sublethal conditions. Davison et al. (1959) found that coho salmon could survive water with only 2 ppm oxygen for 30 days at 18°C.

The lethal limits of salmonoids is approximately 1 to 2 ppm oxygen. Herrmann, Warren and Doudoroff (1962) have shown that food consumption and growth of juvenile coho salmon decline rapidly as the DO drops from 5 to 1.8 ppm oxygen at 20°C. At the 2 ppm level, food consumption approached zero and weight loss was significant. Below 1 ppm, death occurred.

As shown by this review, response of salmon and trout fry to low dissolved oxygen concentrations is variable depending on water

temperature and length of exposure. This makes it very difficult for one to set fixed critical oxygen levels. Some aquatic biologists (Doudoroff and Shumway, 1967) insist that any decrease in dissolved oxygen below saturation is detrimental and therefore unacceptable. This line of reasoning can be further supported by research conducted by Hermann, Warren and Doudoroff (1962), which found a gradual decline in food consumption and weight gain as the DO concentration dropped from 8.3 to 5.5 ppm at 20°C for juvenile coho salmon.

Complete protection of all waters from logging debris and resulting deoxygenation of stream waters cannot be entirely justified. First, complete protection would require extremely expensive stream cleanup. Second, private landowners often cannot afford to leave buffer strips on the very small headwater streams. And third, if the DO level is not allowed to drop to levels of critical oxygen stress, the fish are not severely handicapped allowing some leeway for forest land managers. Such stringent standards requiring complete protection is both impractical and unnecessary.

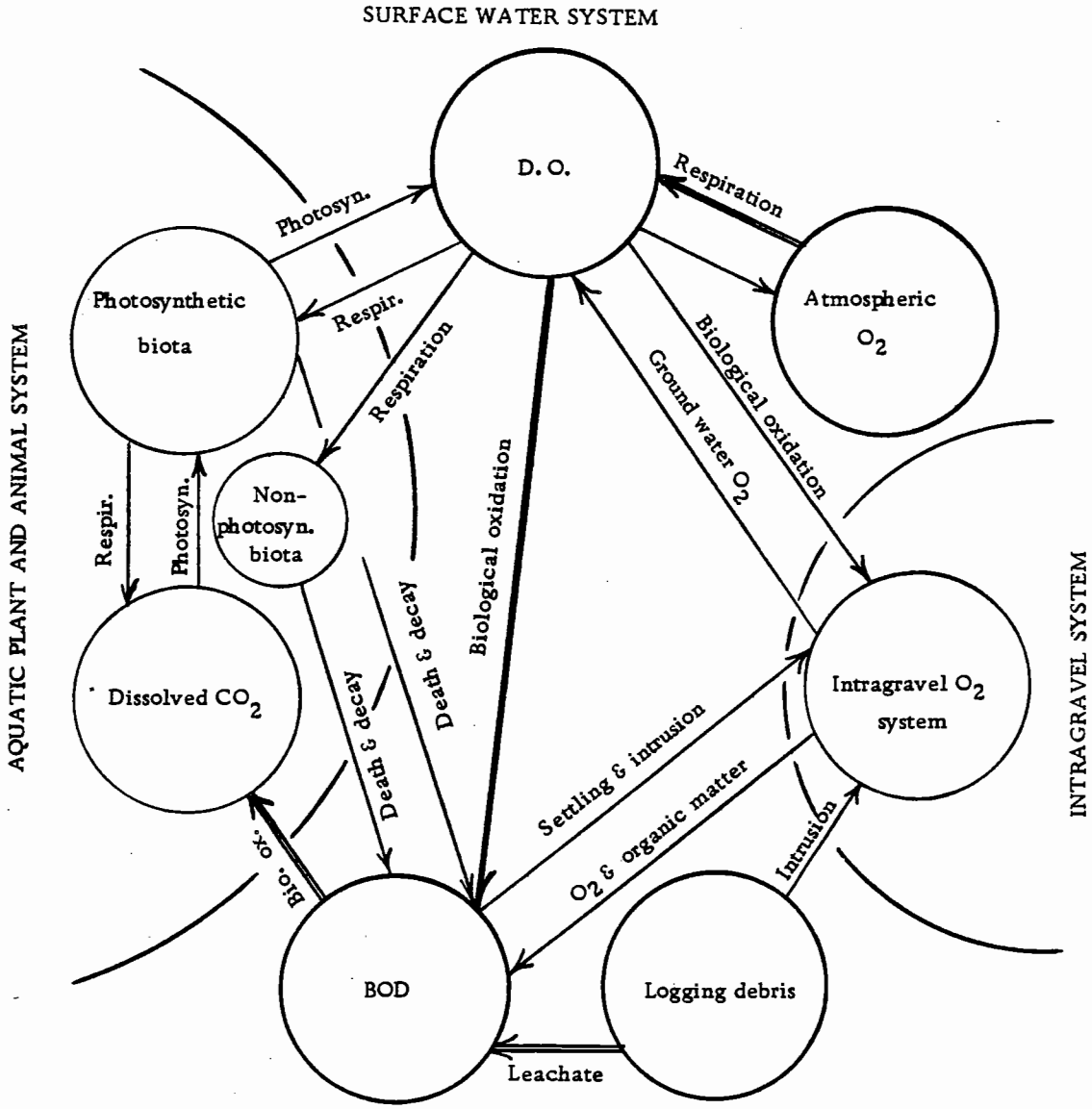
In conclusion, it is evident that as the DO level of a body of water declines, fish attempt to avoid regions of low oxygen concentration. After prolonged exposure at suppressed oxygen levels, salmon and trout exhibit decreases in growth and food consumption. These reactions tend to increase as the dissolved oxygen concentration approaches zero. In general, salmonoids experience critical oxygen

stress from approximately 6 to 2 ppm oxygen and die at levels below 1 to 2 ppm. The 6 ppm DO level will be referred to loosely as the "critical oxygen level" and 1 ppm as the "lethal limit."

The Dissolved Oxygen Balance

Oxygen exists as a solute in water in very small amounts even at saturation (9.2 ppm at 20°C and 760 mm Hg). As oxygen is removed from water (deoxygenation), the oxygen concentration drops below the saturation level. The difference between the saturation concentration and the oxygen concentration is defined as the oxygen deficit. The subsequent rate at which oxygen is replenished by the diffusion of oxygen from the atmosphere is proportional to the oxygen deficit. This process of entrainment and absorption of oxygen is termed reaeration. The combined process of deoxygenation and reaeration is the oxygen balance.

The dissolved oxygen balance for a compartmental unit of water may be conceptualized as a container of constant volume, in and out of which oxygen is being continually removed and replenished. The relationships which exist between the various components of the stream water in the container can be illustrated by Figure 1. Numerous studies have been undertaken to describe either the whole system or parts of it. However, the majority of these studies have been on large, slow streams or rivers with pollutant loads from either



$$C_nH_aO_bN_c + (n + \frac{a}{4} - \frac{b}{2} - \frac{3}{4}c)O_2 \text{ ——— } nCO_2 + (\frac{a}{2} - \frac{3}{2}c)H_2O + cNH_3$$

Leachate BOD + oxygen $\xrightarrow{\text{organisms}}$ carbon dioxide + water + ammonia

Figure 1. The surface water system of small mountain streams and its interaction with the intragravel system and aquatic organisms.

industry or domestic sewage. For this reason the major components of the oxygen balance (dissolved oxygen, logging debris and BOD) will be reviewed first with respect to the dissolved oxygen exchange system in general, and secondly with respect to the BOD of logging debris and reaeration of small mountain streams.

The Dissolved Oxygen Exchange Process

Dissolved oxygen levels in streams tend to balance or reach an equilibrium because of a series of interrelated exchange processes. As dissolved oxygen is used by respiring organisms, a reaeration gradient and photosynthesis by plants allows the stream water to be resupplied with oxygen. Basic to the dissolved oxygen balance are the physiochemical laws governing the solubility of oxygen.

The solubility of oxygen in water is determined by the ability of the water to hold oxygen. This ability is proportional to the partial pressure of oxygen on the open water surface given by Henry's law:

$$C_s = K_H P_{(oxygen)} \tag{1}$$

where C_s is the solubility or saturation concentration of oxygen in water in ppm, K_H is Henry's gas law constant for oxygen at the pressure in ppm-atm, and $P_{(oxygen)}$ is the partial pressure of oxygen exerted on the free water surface in atm. Since the solubility of oxygen in water is proportional to the partial pressure of oxygen

above the water, then by the ideal gas laws the solubility of oxygen in water is inversely proportional to temperature.

The solubility of oxygen in pure water varies from 14.6 ppm at 0°C to 5.6 ppm at 50°C and 760 mm Hg pressure (APHA, 1971). Churchill et al. (1962) developed an empirical equation to estimate the saturation concentration of oxygen, C_s (ppm), at a given temperature, T (°C):

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

The major source of oxygen to the stream is the atmosphere. Atmospheric oxygenation or reaeration is the predominent method of replenishing stream water with oxygen. Green plants capable of photosynthesis are another potential source of oxygen as well as a sink. During daylight hours, phytoplankton, benthic algae, and rooted aquatic plants can produce significant inputs of oxygen with midday oxygen concentrations as high as 150 percent of the saturated oxygen concentration (Gameson and Barrett, 1958). Aquatic plants can also be a significant sink of dissolved oxygen during the night when respiration reaches a maximum (Warren et al., 1964). These studies were all done on large lakes and slow flowing streams where, under certain conditions, algae and plants may be the dominant factor in the oxygen balance. Sedell (1974) indicates that aquatic organisms, including fish, do not produce detectable differences in the dissolved

oxygen concentration of turbulent mountain streams. This is the case both before and immediately after clearcutting. Plant and animal respiration and photosynthesis, however, do significantly affect carbon dioxide concentrations, nitrate levels, conductivity and pH. Thus, the impact of photosynthesis and respiration on the dissolved oxygen balance of small mountain streams of steep gradient appears negligible.

The intragravel and the ground water systems also may add to or take oxygen from the surface water system. However, the fluid flow through and between these systems is extremely small, on the order of 2 to 5 cm/hr compared to 500 to 1500 cm/sec for the surface flow (Narver, 1971). The dominant method of oxygen transport between the surface water and the intragravel water appears to be molecular diffusion. Using the theoretical solution to Fick's law of hydrodiffusion for quiescent bodies of water (Krenkel and Orlob, 1962):

$$K_d = \frac{2 D_m t}{4 h^2} \tag{3}$$

K_d, the oxygen transfer rate for intragravel depth of 16 cm, is 0.00267 1/day. D_m is molecular diffusivity in cm^2/sec, t is time in sec/day and h is depth in cm. Molecular diffusivity can be computed by the method of Negulescu and Rojanski (1969):

$$D_m = 2.037\ (1.037)^{(T-20)}\ (10^{-5}) \tag{4}$$

where T is temperature in ^{o}C. Although this analysis was intended for

quiescent bodies of water, Streeter and Kehr (1935) found these equations to be in very close agreement with studies of channel flow at very low flow rates. The reaeration rate constant for a stream with a velocity of 0.6 ft/sec, slope of 0.05 ft/ft, at 14°C is 103 1/day. Thus the intragravel-surface water exchange is approximately 10^5 times smaller than the atmospheric reaeration rate. However, the surface-intragravel flow constitutes one of the major sources of oxygen to the intragravel system.

Deoxygenation of a body of water may be affected by any one of several means, including chemical oxidation, respiration of aquatic organisms or biological decomposition of organic matter. Of these, biological decomposition is the most prevalent process acting in mountain streams. The amount of oxygen required by microorganisms to completely decompose a given amount of organic matter to carbon dioxide, water and ammonia is defined as biochemical oxygen demand (BOD). This is not synonymous with complete oxidation since decomposable refers only to organic matter which can serve as a nutrient source for bacteria. BOD is generally given as mg of oxygen required to stabilize a liter or gram dry weight of pollutant (mg/l, ppm, or mg/gm). In a sense, BOD is a measure of the strength of a pollutant in oxygen demand equivalents.

BOD is the major oxygen sink in small, unpolluted mountain streams. On forested watersheds, significant levels of BOD are

usually a result of deposition of fine logging debris in the stream channel. Lammel (1973) found that up to 17.16 tons of organic debris per 100 ft of stream length were added following conventional felling. Of this, 0.292 tons per 100 ft were fine debris 0 to 3 cm in size. For a stream 5 ft wide with a flow of 0.25 ft^3/sec this would produce a BOD load of approximately 115,210 ppm. This estimate was based on a theoretical distribution of slash on a 30-ft wide stream bed, an assumption that the species involved was western hemlock, and that the ultimate BOD is 183 mg O_2/gm. (See Appendix II for computations.)

The dissolved oxygen balance is a cause and effect type of dynamic equalizing process. Any time decomposable organic matter such as finely divided logging debris enters a stream, it begins to decay resulting in an oxygen deficit, tipping the balance. Simultaneously reaeration begins to replenish oxygen to the water in proportion to the oxygen deficit.

The resulting profile of the DO concentration with time may be described mathematically as a set of flow rate equations in differential form. The combination of these equations constitutes an equation of the dissolved oxygen sag curve. An example of a dissolved oxygen sag profile is shown in Figure 2.

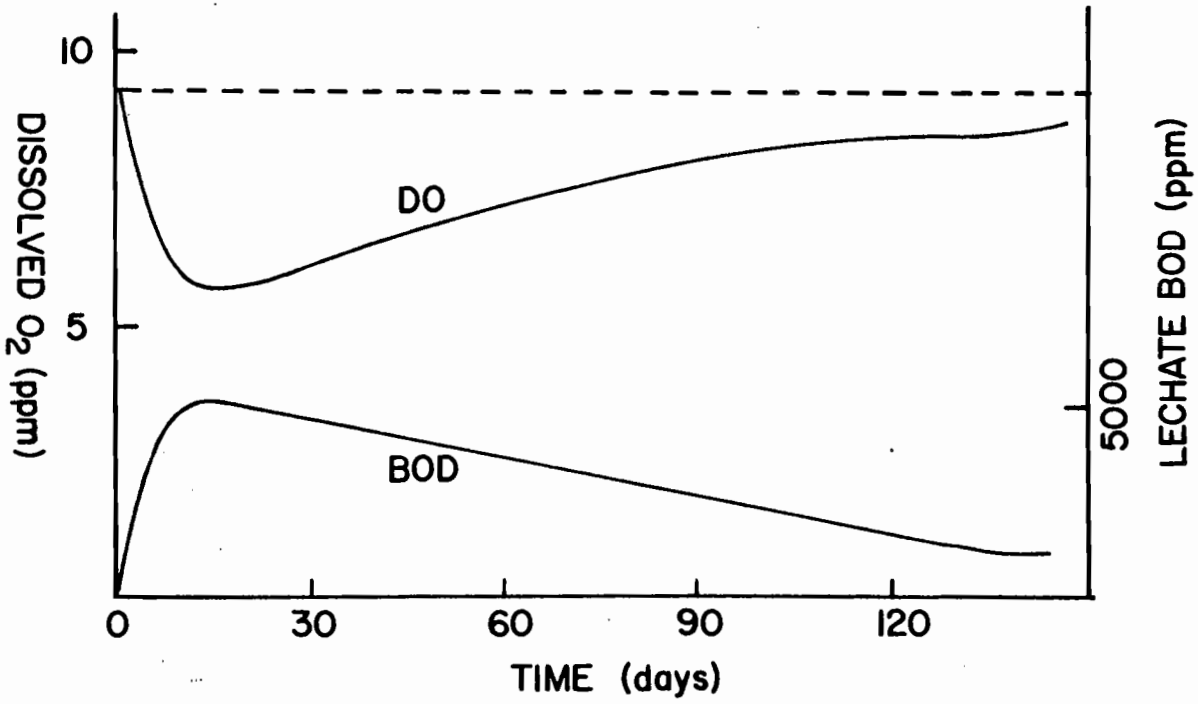


Figure 2. A typical dissolved oxygen sag curve.

Logging Debris and BOD

The pollutant inputs to a small forest stream during timber harvesting are quite often logging residues consisting of logs, splinters, limbs, twigs and needles or leaves. Hall and Lantz (1969) have found that dissolved oxygen concentrations in streams following clear-cutting may drop as low as 0.6 ppm before the fall freshets flush the fine debris from the stream channel. An examination of the constituents of logging debris and its BOD characteristics follows.

Until recently, very little was known about the volume and size distribution of organic matter in streams before and after timber harvesting. Lammel (1973) attempted to quantify these parameters. He was primarily interested in determining how much debris is natural, how much is deposited following various logging techniques and at what stages in the harvest process, and what is the impact of the stream environment, particularly physical barriers to fish migration and disturbance of spawning grounds. Lammel (1973) concludes that organic matter on the stream bed increases after felling and is subsequently reduced following yarding, in some cases to levels below the prelogging values. This appears to be true for material greater than 1 cm. However, the amount of finely divided debris less than 1 cm, principally twigs, was significantly increased following all harvest operations except in two of six units, one where felling was

cable assisted and another where a wide buffer strip (150 ft) was effective. Lammel's study was the most precise and extensive body of data found. For this reason, it will be used to estimate the organic loading of small mountain streams following harvest operations.

The characteristics of the BOD of finely divided slash have only recently been examined in any detail. Chase and Ferullo (1957) investigated the effects of maple and oak leaves, and pine needles in water under laboratory conditions. Slack (1964) examined the BOD concentrations existing in streams but not in relation to the amount of organic matter. He found dissolved oxygen concentrations from 0.09 ppm for isolated pools to 10.3 ppm for main streams. Slack and Feltz (1968) followed the decrease in oxygen concentration of a small stream from mid-September to October and found a high degree of correlation between the decline of dissolved oxygen and leaf fall. As the leaf fall rate increased from 2 to 12 $gm/m^2/day$, the DO level dropped from 8 to less than 1 ppm. Later, in November, the DO concentration increased to more than 11 ppm following the first fall freshet.

Schaumburg (1970) reports that log leachate BOD ranges from 0.1 gm/ft^2 of surface area for 120-year-old Douglas-fir with bark intact to 1.4 gm/ft^2 for ponderosa pine with the bark removed. Should a log 12 inches in diameter be lengthwise along the thread of a stream 3 inches deep, 0.52 ft^2 would be submerged per linear foot

of stream. This log then may exert a potential BOD of up to 680 mg of oxygen demand per linear foot. Using Lammel's data (1973), and the estimation of Overton et al. (1973) of the ratio of needle mass to stem mass for old-growth Douglas-fir, up to 67.88 lb of fine debris per lineal foot would be deposited on the stream bed 19 ft wide. Assuming a stream width of 5 ft, this would amount to 17.86 lb of submerged debris per linear foot of stream (see Appendix II for calculations). If the debris is western hemlock, the potential BOD is 183 mg/gm or a potential BOD of 49,655 mg of oxygen demand per linear foot of stream. The leachate BOD rate constant (K_1) for logs is 0.19 to 0.25 per day (Atkinson, 1971) as compared to 0.202 per day for western hemlock needles. However, the leaching rate for needles is approximately 0.089 whereas the leaching rate for logs is estimated to be several orders of magnitude smaller. Thus, it may be assumed that 1) the BOD of larger debris is smaller and exerted over a much longer term, and 2) the probability of large amounts of submerged surface area occurring is remote. For these reasons, the leachate of debris larger than 3 cm is not considered to exert a significant oxygen demand when compared to that exerted by the finely divided debris during periods of less than 90 days following timber felling.

Hall and Lantz (1969) noted low DO levels in a coastal stream of the Pacific Northwest following clearcutting. From June to July 1966 the dissolved oxygen concentration fell to levels lethal to salmon and

trout. Levels as low as 0.6 ppm were reported.

Ponce (1974) made the first attempt to describe the potential BOD of finely divided slash and its characteristics. He examined the ultimate BOD exerted by various species of vegetation in stream water over periods of 5 to 90 days. Since he did not account for the separate processes of leaching and subsequent decay of leachate, his decay rate constant is not K_1 as reported, but is an approximation of the combined effects of K_1 and K_4. This slightly erroneous estimate will be referred to as K_D. L_u is defined as the ultimate BOD previously referred to as the potential BOD. Ponce's BOD results are given in Table 1.

Table 1. The ultimate BOD (L_u) and rate constants (K_D) of Douglas-fir, western hemlock and red alder needles (after Ponce, 1973).

Vegetation type	L_u (mg O_2/gm)	K_D (1/days)
Douglas-fir	110	0.125
Western hemlock	116	0.064
Red alder	286	0.046

Ponce also reports that he found no indication of the occurrence of nitrification. Had nitrification been indicated, the BOD curve would have been a two-stage curve, complicating the oxygen demand analysis considerably.

The concentration of the organic loading, however, does not appear to have a significant effect on potential BOD on a per gm weight of vegetation basis. However, temperature fluctuations seem to affect both the ultimate BOD and the decay rate constant (Ponce, 1974). This finding is further supported by the work of Zanoi (1967) when he studied the effects of temperature on the BOD and rate constants for sewage. Ponce's temperature tests were for only 5 days and he simulated a diurnal temperature fluctuation observed in clearcut watersheds. Thus, his estimates for long-term BOD and K_D are inconclusive.

Ponce (1974) also attempted to examine the chemical composition and quantity of debris leachate. His methods of sterilization, however, failed to completely stop biological degradation of the leachate, especially the fungal activity.

Ponce's study was an important beginning in quantifying the characteristics of the BOD of finely divided slash. His thesis (Ponce, 1974) has provided the basis for most of the biological experimentation conducted for this study and a portion of the data used.

All previous investigations except Schaumburg (1970) and Atkinson (1971) have assumed that the addition of leachate to the stream water is a slug or constant load (Ponce, 1973; Chase and Ferullo, 1957). Upon greater inspection, it is evident that the rate of leachate addition to a stream is not a slug load nor a constant addition

of pollutant. Therefore the assumption that the decay of logging debris BOD is a first order decay is violated.

The process of the leaching of pollutant from slash may be conceived as a linear first order decay reaction represented by:

$$\frac{dS}{dt} = -K_4 S \tag{5}$$

where S is the potential BOD remaining in the slash (ppm oxygen demand), K_4 is the leaching rate (per day, base e) and t is time in days. Solving for S at any time t:

$$S = S_o e^{-K_4 t} \tag{6}$$

where S_o is the initial BOD concentration in the slash. By substituting (6) back into (5), the rate at which pollutant is leached from the slash at any time t is:

$$\frac{dS}{dt} = -K_4 S_o e^{-K_4 t} \tag{7}$$

Conversely, the rate of addition of pollutant to a body of water is equal but opposite to the leaching rate of pollutant from slash:

$$\text{(addition rate)} = K_4 S_o e^{-K_4 t} \tag{8}$$

Simultaneously, the decay of pollutant may be represented as:

$$\text{(decay rate)} = -K_1 L \tag{9}$$

where L is the concentration of pollutant BOD in ppm and K_1 is the

BOD decay rate constant (1/day). Applying a mass balance to equations (8) and (9), the rate of change of pollutant (L) with time is equal to the rate of addition plus the rate of decay or:

$$\frac{dL}{dt} = K_4 S_o e^{-K_4 t} - K_1 L \tag{10}$$

where dL/dt is the rate of change of pollutant with time (ppm/day). Upon integrating equation (10), it can be shown that the BOD concentration (L) of a completely mixed body of water at any time is:

$$L = \left(L_o - \frac{K_4 S_o}{K_1 - K_4}\right) e^{-K_1 t} + \frac{K_4 S_o e^{-K_4 t}}{K_1 - K_4} \tag{11}$$

where L_o is the initial pollutant concentration at time equal to zero (see Appendix III).

Atmospheric Reaeration

The reaeration process of small mountain streams is perhaps one of the most difficult parameters of the oxygen balance to estimate. Nearly every hydraulic characteristic of a stream may vary by a factor of 10 to 100 over a distance of less than 10 ft. Small pools and riffles of varying size and length are very characteristic of small mountain streams. For this reason, the atmospheric reaeration rate constant (K_2), which is highly dependent upon hydraulic characteristics

of a stream, may take on values from 1 to 5,000 over very short distances (Holtje, 1971).

Even Holtje's (1971) study of Berry Creek shows large amounts of deviation of predicted K_2 values from measured values, particularly for measured K_2 values between 20 and 200 (1/day). Since Berry Creek is very representative of coastal streams and Holtje's survey data was readily available, various stream segments of Berry Creek have been combined to make a simulated stream for testing the oxygen balance model in this study. For this reason Holtje's model for predicting atmospheric reaeration has been used as a best available estimate. The reaeration rate constant K_2 (1/day) at any temperature T (^{o}C) may be estimated by:

$$K_{2_T} = 1.016^{(T-20)} (181.6\ E - 1657\ S_L + 20.87) \tag{12}$$

where E is energy dissipation, and S_L is the channel slope in ft/ft. Energy dissipation may be expressed as:

$$E = S_L \cdot U \cdot g \tag{13}$$

where U is the average flow velocity in ft/sec and g is the gravitational acceleration constant (32.174 ft/sec^2).

The atmospheric reaeration portion of the oxygen balance may be represented mathematically as:

$$\text{(reaeration rate)} = -K_2 D \tag{14}$$

where K_2 is the reaeration rate constant from equation (12) and D is the dissolved oxygen deficit, defined as the algebraic difference between the saturated oxygen concentration and the actual oxygen concentration at any time. In other words, reaeration is a reduction of the oxygen deficit in proportion to the deficit. The complete oxygen balance equation incorporates deoxygenation and reaeration. This is most often referred to as the oxygen sag equation.

The Dissolved Oxygen Sag Analysis

The oxygen sag is a term introduced by Streeter and Phelps (1925). It is the basis for calculating the capacity of a body of water to assimilate organic wastes. The simplest of all pollutant loadings is the slug load. Slug loads occur when a readily decomposable waste is instantaneously injected into a completely mixed body of water. The pollutant removes oxygen from the water at a rate equal but opposite to the BOD rate of decay. Equation (9), when expressed as deoxygenation, becomes:

$$\text{(rate of deoxygenation)} = K_1 L \tag{15}$$

Thus, the complete equation for the change in oxygen deficit with time is:

$$\frac{dD}{dt} = K_1 L - K_2 D \tag{16}$$

Upon integrating, this equation becomes:

$$D = \frac{K_1 L_o}{K_2 - K_1} (e^{-K_1 t} - e^{-K_2 t}) + D_o e^{-K_2 t} \tag{17}$$

where D_o is the initial oxygen deficit at time equal to zero. Equation (17) is often referred to as the Streeter-Phelps equation. Closer examination reveals that by differentiating equation (17) and solving for t when dD/dt equals zero yields:

$$t_c = \frac{1}{K_2 - \frac{1}{K_1}} \quad \ell n \left\{ \frac{K_2}{K_1} \left[1 - \left(\frac{K_2}{K_1} - 1 \right) \frac{D_o}{L_o} \right] \right\} \tag{18}$$

where t_c is the time of the critical point or maximum deficit in days (Fair, 1939). The maximum deficit (D_c) is:

$$D_c = \frac{K_1 L_o}{K_2} \quad e^{-K_1 t_c} \tag{19}$$

The oxygen sag equations rapidly become more complex as the number of inputs or outputs of the oxygen balance increase or become functions of time.

Camp (1965) has solved for an oxygen balance which includes a constant addition of pollutant with time, a settling rate of BOD, and reaeration by photosynthesis. These equations are of particular use in slow flowing streams or estuaries where 1) the flow velocity is small, yet 2) there is complete vertical mixing, 3) water temperature is constant, 4) photosynthesis is a significant factor, 5) BOD from benthic anaerobic decomposition is added to the overlying water,

and 6) dispersion is negligible. Under these conditions the pollutant concentration (L) at any time (t) is:

$$L = \left[L_o - \frac{p}{K_1+K_3} \right] e^{-(K_1+K_3)t} + \frac{p}{K_1+K_3} \tag{20}$$

where p is the rate of addition of BOD in ppm/day and K_3 is the settling rate of BOD to the bottom. It should be noted that the rate of pollutant addition is a constant and cannot be a function of time in this equation. The dissolved oxygen deficit (D) under these conditions may be given by:

$$D = \frac{K_1}{K_2-K_1-K_3} \left[L_o - \frac{p}{K_1+K_3} \right] \left[e^{-(K_1+K_3)t} - e^{-K_2 t} \right]$$

$$+ \frac{K_1}{K_2} \left[\frac{p}{K_1+K_3} - \frac{\alpha}{K_1} \right] (1-e^{-K_2 t}) + D_o e^{-K_2 t} \tag{21}$$

where α is the rate of oxygen production by photosynthesis.

In a situation where logging debris is to be considered, equation (11) replaces (20) and the resulting solution for the dissolved oxygen deficit (D) becomes:

$$D = \frac{K_1}{K_2-K_1} \left(L_o - \frac{K_4 S_o}{K_1-K_4} \right) (e^{-K_1 t} - e^{-K_2 t})$$

$$+ \frac{K_1}{K_2-K_4} \left(\frac{K_4 S_o}{K_1-K_4} \right) (e^{-K_4 t} - e^{-K_2 t}) + D_o e^{-K_2 t} \tag{22}$$

This equation, when correctly used, can be an excellent estimator of the oxygen deficit of small mountain streams under special conditions.

These conditions include 1) uniform steady flow, 2) constant water temperature, 3) negligible longitudinal dispersion, and 4) a uniform debris loading distribution.

Stream Water Temperature

Since all rate constants, ultimate BOD, and saturated oxygen concentration are dependent on water temperature, a brief review of stream water temperature is in order. During a four-year study of three coastal watersheds of the Pacific Northwest, Brown and Krygier (1970) found monthly maximum stream temperature variations between winter and summer as large as 6.1°C and diurnal fluctuations as large as 3.9°C for forested watersheds.

The variation in stream water temperature through a clearcut at any instant in time is proportional to the radiation incident to the stream surface but inversely proportional to the volume of water in the stream channel at that instant. The change in water temperature of a stream through a clearcut may be expressed as:

$$\Delta T = \frac{\text{Rad} \times A_s}{Q} \times 0.000267 \qquad (23)$$

where ΔT is the change in water temperature in °F, Rad is net thermal radiation in BTU/ft^2 min, A_s is the surface area of the stream in ft^2, and Q is the discharge of the stream in cfs (Brown, 1969). This estimate of water temperature is applicable for clearcut streams less

than 2,000 ft in length and where the stream channel is not solid rock. Solid rock channels may cause errors in the estimate of net solar radiation (Q) on the order of 15 to 20 percent, due to conduction into the stream bed (Brown, 1972).

It is evident from this review that the oxygen balance parameters of a mountain stream are extremely variable. This dilemma is compounded by the lack of accurate and adequate data. The preceding portion of this literature review has given some indication of the vast complexity and intricacies of the dissolved oxygen balance in small mountain streams which are subjected to organic loading from timber harvesting. The examination of the oxygen sag equation for debris loading points out the severe limitations of analytic solutions for dissolved oxygen in mountain streams. Computer simulation may solve these problems. Computer modeling is a useful and powerful tool due to its immense ability to perform long and numerous calculations relatively error free in fractions of a second.

Modeling

Before attempting to model or simulate anything, one should first start with the most general definitions and concepts and narrow these areas to the case at hand, developing all concepts and assumptions in a concise, orderly manner. The following discussion on modeling will use concepts and terms outlined by Fishman (1973).

These modeling principles are basic to understanding the modeling approach in a correct manner.

A model is any contrived representation of a system. A system consists of two or more units interacting through distance, time or both. In a sense, even models are systems which are subject to modeling. Models may be pictorial scale models, analogs (electrical), or symbolic models which are mathematical, logical representations. Symbolic models are of special interest in this study. Symbolic models may be analytic equations or numerical techniques for solving problems. So far, we have only discussed analytical models of the dissolved oxygen balance. The real world system of interest is a physical system which is tangible and whose characteristics are measurable.

Because models are representations of real world systems, they can never be exactly like the real system unless they are an exact duplicate. This extreme is contrary to the homomorphic nature of modeling. The purpose of modeling is to reduce a large number of variables to a manageable amount of data items. Thus, it is necessary to set realistic and logical constraints on models. These constraints are termed assumptions and occur everywhere the model deviates from the real system.

Not all assumptions can nor will be recognized because their numbers approach infinity with increasing attention to detail. The

limit of such an approach is that these complexities and uncertainties are beyond the human ability to comprehend. The degree to which a system can be comprehended as a whole is defined as perspective, and the degree of attention paid to detail is termed resolution. Thus, the greater the perspective the lower the resolution and vice versa. What degree of resolution and perspection must the model meet to be of use?

Limitations consist of defining what portion of the real world is to be considered, or the scope of the model. This includes temporal as well as spatial delineation. Defining and describing the effects of the models' limitations are possibly the most crucial steps in modeling. By limiting the real world to an isolated system we can set the scope of the model, and in a sense "trade-off" detail and high resolution for a greater perspective. Such a simplification reduces a very complex system to a model of greater perspective and more manageable size.

A systems model is much like a window for the modeler to look at the real world. He must decide on the dimensions of the window (scope) and how clear the glass must be (resolution). This may be termed the model specifications. Before the model specifications can be set, one must determine 1) the definition of the problem, 2) the system of interest, 3) which parameters are to be measured, and/or

predicted, and 4) how precise and accurate must the parameters be measured and predicted.

Once the scope and degree of resolution of the model have been set, the next step is to build a conceptual model. This is a descriptive model in which the modeler attempts to depict events, activities, and states of the real system by narration, equations, graphs and charts. The system is divided into entities or parts. All of the physical dimensions or attributes of an entity at a given point in time and space define the state of the entity and collectively the states of the entities define the state or level of the system. A change in state is defined as an event and the operations which change one state to another are termed activities. Finally, the orderly sequence of events in a continuous time frame is a process (Fishman, 1973).

In a sense, the computer model is a simulation of the conceptual model. The computer simulation is a representation of a system by a symbolic model that can be manipulated easily and yields numerical results (Fishman, 1973). Here, the conceptual model is translated into a computer language.

In both the formulation of the conceptual and computer models, our degree of resolution has been reduced by conceptual and programing assumptions. Conceptual assumptions occur whenever detail is forfeited for greater perspective in simplifying or approximating the real system. Programming assumptions occur when approximating

techniques are used to solve mathematical equations or reduce the number of computations to produce an economically feasible simulator. The modeling procedure may be viewed as the flow chart in Figure 3.

The preceding literature review of the impact of logging debris on the dissolved oxygen balance of small mountain streams has stated the problem and sets the stage for developing a model. The parameters of particular interest are the levels of dissolved oxygen and pollution in small mountain streams with respect to time and distance downstream. The expected precision for predicting these parameter levels is to the nearest foot along the stream course and to 0.1 ppm for the DO and pollutant. The desired accuracy is to estimate the parameter concentrations within $\pm$ 5 percent of the actual measured values. Any inputs or outputs which cannot cause variations of this magnitude have been termed minor influences. As previously discussed, photosynthesis, ground water and intragravel flow influences, and plant and animal respiration fall into this category.

When conceptualizing the system, it can be broken down into entities such as the surface water system divided into compartments of water, the logging residue, the dissolved oxygen, and the leachate BOD. The attributes of these entities include such terms as velocity, width, discharge, concentration, and temperature. Collectively, the body of attributes of an entity is its state. For example, the entity dissolved oxygen has concentration, reaeration rate, deoxygenation

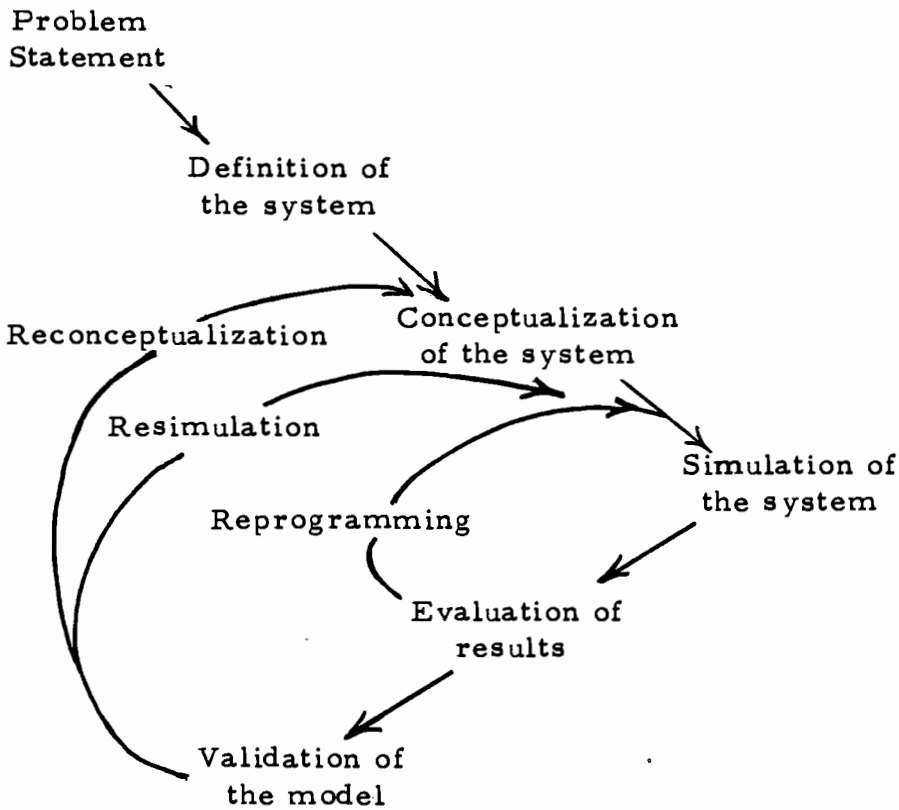


Figure 3. The modeling procedure.

rate, and oxygen deficit as its attributes. The surface water system state is defined at any point in time and space by the states of the critical entities, oxygen, pollution, and water temperature. The interactions of biochemical decomposition and stream reaeration with respect to temperature and hydraulic stream characteristics have been defined as the dissolved oxygen process or DO balance.

The simulation of the surface water system in which the dissolved oxygen process operates has been approached from many aspects in the past. As previously described, Streeter and Phelps (1925), Fair (1939), and Camp (1940) have proposed analytic solutions for estimating the oxygen sag process. However, these methods are ineffective in view of the rapid changes in water temperature and hydraulic conditions found in mountain streams.

Bella and Dobbins (1968) refined the finite difference technique from the finite element technique widely used by structural engineers and ground water hydrologists. This technique uses fixed changes in distance (Δx) and time (Δt) which do not approach zero as do the dx and dt of differential equations. The Δx and Δt are made physically small enough so that by summing the new equation with Δx and Δt over time and distance, a close approximation of the integration of complex differentials can be made.

For a stream, the surface water system can be thought of as discrete volumes of water bounded by cross sectional interfaces of

area A and length dx (Figure 4). These cells are fixed along the stream course. A mass balance may be performed for each cell yielding the one-dimensional stream equation for oxygen. The change in oxygen concentration (C) with time (t) is:

$$\frac{\partial(AC)}{\partial t} = \partial \frac{(A\, D_L \frac{\partial c}{\partial x})}{\partial x} - \partial \frac{(AUC)}{\partial x} - K_1\, A\, L(t)$$

$$+ K_2\, A\, (C_s - C) + AC \qquad (24)$$

where D_L is the longitudinal dispersion coefficient, U is the flow velocity, and L(t) is the pollutant concentration as a function of time (Bella and Dobbins, 1968). The first term represents the effect of longitudinal dispersion. The second stands for the rate of addition and subtraction of pollutant due to stream flow, termed advection. The third term represents the rate of deoxygenation by biochemical decomposition of an organic pollutant. The fourth stands for the rate of atmospheric reaeration. And the last term is termed the rate of addition or subtraction of dissolved oxygen along the stream due to the minor influences.

Rather than using the available integration approximations, such as Simpson's rule or the trapezoid rule, which cannot be used with partial differentials, the finite difference techniques replace the differential elements with finite difference terms and summations are made over distance and time. This technique avoids the necessity of

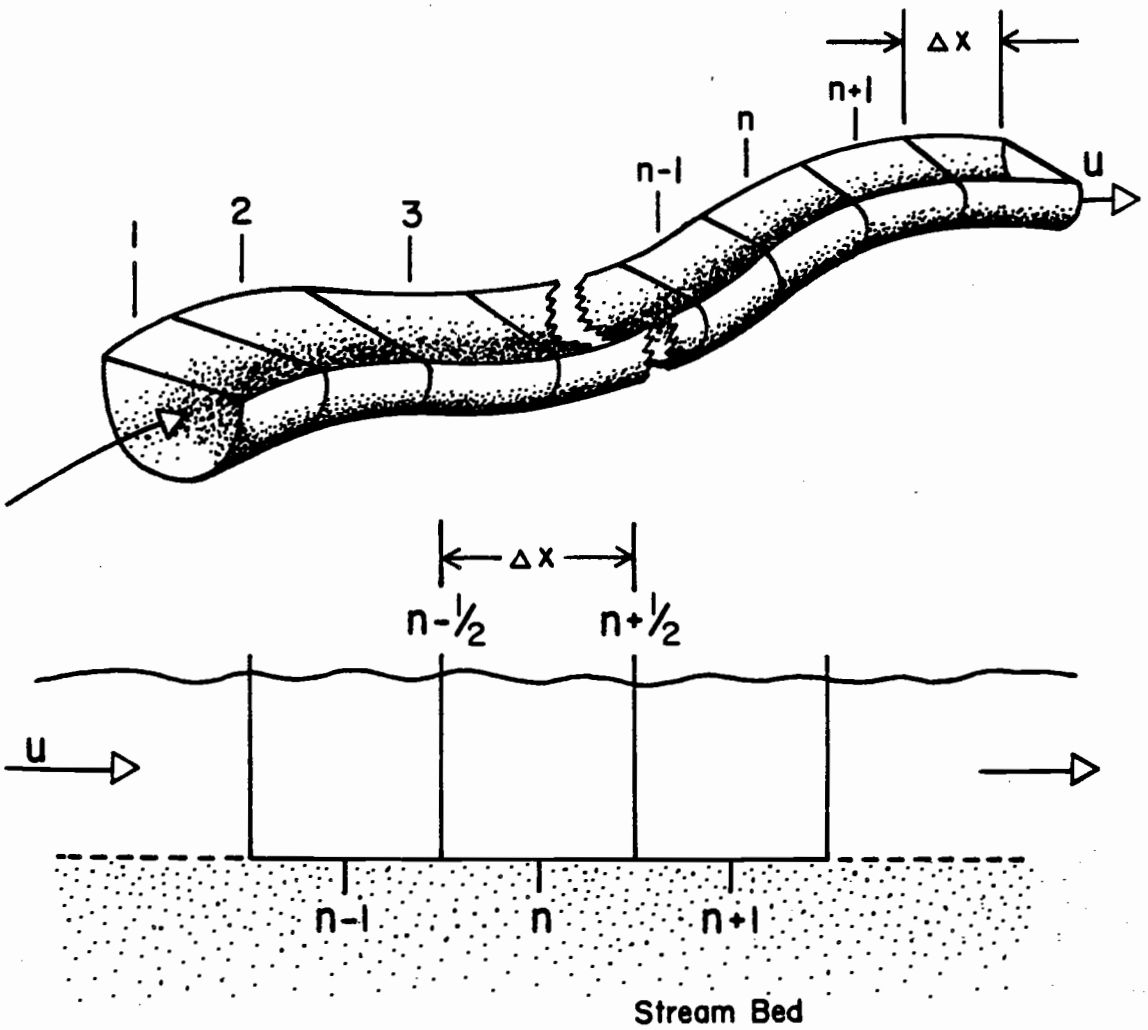


Figure 4. The finite-difference conceptualization of the surface water stream system.

manipulating complex differential equations and is easier to visualize in physical terms.

A second method of modeling streams is a derivation of the finite difference technique called the Lagrange method, also tenderly referred to as "oxcart." This method looks at a differential cell of fixed volume which moves through the stream system at varying intervals of time (Bella, 1967). This method is much more economical to use if the dispersion effect may be neglected and the rate of polluted addition with time is known.

Both of these techniques can handle the variation of water temperature, discharge, and hydraulic characteristics with time and distance. Both were used in this study to estimate the impact of logging debris on dissolved oxygen. Their development and results are presented in following sections.

METHODS

Sampling Methods

The stream water and vegetation samples were collected using the sampling methods outlined by Ponce (1974).

The stream water samples were collected in carboys at the Oak Creek bed load experiment station about 1/2 mile north of the Pacific Cooperative Water Pollution Laboratory. The samples were gathered the same day experiments were initiated.

The vegetation samples were collected the day before experimentation initiation so that the samples could be sorted and proportionately mixed for maximum consistency and minimum bias. The material was stored in sealed plastic bags at 4°C until used.

The Douglas-fir samples were collected approximately 1/4 mile northwest of the old grader shack on Oak Creek (NE 1/4, SE 1/4, Sec. 18, T 11 S, R 5 W, W.M.). Three trees were selected and limbs cut from the north and south aspects of each tree. The limbs were taken to the Forest Research Laboratory where the needles and twigs were removed and segregated by age and aspect and placed in plastic bags. Smith (1970) reports that the average percentage of needles retained by Douglas-fir are: 30 percent current, 26 percent one-year, 20 percent two-year, 13 percent three-year, 8 percent four-year and

3 percent five-year. The needles were proportionately combined and thoroughly mixed.

Western hemlock samples were collected from behind Waldo Hall on the Oregon State University campus. The procedure for collection was identical to the method used in collecting Douglas-fir samples except 1) only current, one-, two- and three-year-old vegetation was collected and 2) the proportion of needle mix was: 46 percent current, 41 percent one-year, 11 percent two-year and 1 percent three-year-old growth (Smith, 1970).

The red alder vegetation samples were from near the Sulfur Springs gate of McDonald Forest (NE 1/4, SW 1/4, Sec. 5, T 11 S, R 5 W, W. M.). Three trees were selected and stacks of leaves were picked and placed in plastic bags. The leaves were then cut into 2 cm squares, being careful not to excessively damage the leaves.

Analytical Methods

Biochemical Oxygen Demand Analysis

All BOD determinations were made using the Hach manometric BOD apparatus. The apparatus consists of a 500 ml bottle connected to a closed-end mercury manometer. The sample to be tested is placed in the bottle with a magnetic stirring rod and sealed.

As the microorganisms oxidize the organic matter in the sample, the dissolved oxygen is depleted. The magnetic stirring bar continually agitates the sample allowing the enclosed air (21 percent oxygen) to replenish the depleted dissolved oxygen. As the oxygen is utilized a pressure deficit results and is registered on the mercury manometer. The addition of a few drops of 45 percent potassium hydroxide solution to a cup suspended above the sample removes any carbon dioxide produced by the oxidation of the organic matter. The absorption of carbon dioxide by the potassium hydroxide solution eliminates the positive gas pressure that would have been exerted by the carbon dioxide. Oxygen may be easily replenished to the system by simply breaking the seal between the manometer and the bottle. The test may again be resumed by resealing the system.

The Hach manometric BOD technique allows continuous monitoring of the oxygen demand exerted with time using only a few samples as compared to the standard dilution technique explained in Standard Methods (APHA, 1971). The Hach method also eliminates the need for diluting the samples.

The Oak Creek stream water is assumed to have a sufficient viable bacteria population to consume the organic matter and to contain the necessary trace salts and nutrients for microorganism growth and reproduction. Thus, no bacteria seed or nutrient solution is added to tests run on stream water samples.

Mass Concentration Dependency Tests. Four levels of debris loading were tested to determine if the BOD concentration (mg O_2/gm dry weight) was dependent on mass concentration. The tests were conducted on Douglas-fir, western hemlock and red alder leachate at concentrations of 1.0, 2.0, 3.0 and 4.0 gm fresh weight per 244 ml of stream water.

The leachate was obtained by drawing samples from bottles, continuously mixed and open to the air, at regular 2-day intervals. Each sample was filtered through a glass fiber filter (grade 934AH). The sample bottle was rinsed with distilled water to remove any large organic particles. The sample was returned to the bottle and sealed in the Hach BOD apparatus for 20 days at 20°C.

Leachate BOD and K_1 Determination. The determination of BOD concentration and decay rate constant (K_1) of Douglas-fir, western hemlock, and red alder leachate were conducted with the mass concentration dependency test. The purpose of this test was to determine the BOD concentration of leachate in stream water with time and to test for variations in K_1 with time.

As previously mentioned, the vegetation-stream water samples were filtered after 2, 4, 6, 8 and 10 days. The filtrate contained the amount of leachate which had leached from the needles or leaves during the time period minus the amount which had decayed during the same time period. The samples were subjected to Hach manometer analyses at 20°C and 20 days duration.

The manometric data were then fit to the curve:

$$L_{(20)} = L_u (1 - e^{-K_1 t}) \tag{25}$$

where $L_{(20)}$ is the quantity of oxygen used in 20 days (mg O_2/gm dry wt), L_u is the potential demand of the leachate or the asymptotic limit of the BOD curve (mg O_2/gm dry wt), K_1 is the BOD decay rate constant (1/day) and t is time (days). This fit was accomplished using a least-squares subroutine available on the CDC 3300 computer at Oregon State University.

Determination of K_4. The leaching rate constant (K_4) of Douglas-fir, western hemlock and red alder vegetation was determined by combining the results of leachate BOD concentration and K_1 tests with manometer BOD data collected by Ponce (1974). The data collected by Ponce represented the oxygen demand exerted over time by the portion of the leachate which decayed during that time period. If the leachate BOD concentration at time t is added to the BOD exerted up to time t, the result is the total amount of leachate leached from the vegetation up to time t. These calculated data points were then fit to the curve:

$$S_t = S_o (1 - e^{-K_4 t}) \tag{26}$$

where S_t is the pollutant (mg O_2/gm dry wt) which has been leached from the slash over the time interval t = 0 to t = t, S_o is the initial pollutant in the slash (mg O_2/gm dry wt), K_4 is the leaching rate

constant (1/day), and t is the time (days). The computed leachate data were fit to equation (26) using the same least squares subroutine referred to in the preceding section.

Temperature Dependency Test. The purpose of this test was to determine the effect of temperature on K_1, K_4 and BOD concentration. K_1, K_4 and potential BOD concentration were evaluated at 15°, 20° and 27°C. These tests were made to examine the possible effects of the extreme temperature differential found in streams through clear-cuts on the leaching rate and subsequent decay of leachate. The tests were conducted using the same procedure outlined for the leachate BOD concentration tests.

Nitrification

The purpose of this experiment was to verify the findings of previous studies that nitrification does not have a significant influence on the BOD of small mountain streams.

Nitrification is the oxidation of ammonia to nitrite and nitrate. Nitrification in small mountain streams has not been extensively documented. In fact, literature concerning nitrification in small mountain streams is nearly nonexistent. Nitrification of sewage was first demonstrated in 1877 by Schloesing and Muntz. Since then, the occurrence of nitrification has been found in a wide variety of polluted waters rich in nitrogen compounds.

Nitrification occurs when nitrifying bacteria oxidize ammonia and nitrite in the presence of oxygen. The two reactions take the following form:

$$2NH_3 + 3O_2 \longrightarrow 2NO_2^- + 2H^+ + 2H_2O \tag{27}$$

$$2NO_2^- + O_2 + 2H^+ \longrightarrow 2NO_3^- + 2H^+ \tag{28}$$

Because oxygen is utilized in the process, nitrification may be a significant sink of oxygen in the dissolved oxygen process in streams.

The primary source of oxygen for nitrification in stream water is the dissolved oxygen. As the nitrifying process proceeds, the dissolved oxygen is depleted. This effect, if it exists, must be accounted for in the dissolved oxygen balance. Nitrification may enter into manometric BOD determinations as a form of interference. With the use of 'N-serve', nitrification can be inhibited so that the carbonaceous oxygen demand can be isolated. The oxygen demand of nitrifying bacteria generally is not significant until after 8 to 10 days because of the relatively low initial population levels.

Several factors or conditions must be met for nitrification to occur. First, nitrifying bacteria must be present in the stream water or in the organic matter added to the water. Second, if the bacteria are present, there must also be a surplus of readily available nitrogen compounds. A surplus means that there is excess nitrogen beyond the nitrogen demand of the carbonaceous bacteria. Available nitrogen

compounds refers to nitrogen which is in relatively accessible form such as urea, ammonia salts, or ammonium carbonate. Relatively unavailable and unaccessible forms of nitrogen include plant and animal protein such as RNA, DNA, amino acids and protein. Even though forest litter is composed of 1.5 percent nitrogen by dry weight, much of it may not be in a readily accessible form for nitrification.

Twelve treatments were examined for the BOD exerted and the amount of nitrogen present in various forms over a 20-day period. The contents of each treatment are listed in Table 2.

The first four treatments examine the main effects of stream water alone on the nitrification process. Treatments 4 through 8 look at the nitrification properties of blended vegetation in distilled water. The last four treatments (9-12) were designed to evaluate whether or not nitrification can occur in stream water with an organic load of blended Douglas-fir needles.

The amount of ammonium chloride to be added was computed on the basis of two criteria; 1) the NH_4Cl level should not be toxic to the microorganisms in the system and 2) NH_4Cl levels should be great enough to provide enough NH_4^+ for significant nitrification. According to these guide lines, no more than 18.6 ppm NH_4Cl should be added (see Appendix IV for calculations). If 18.6 ppm NH_4Cl is used and the reaction were to go to completion, the resultant oxygen demand

Table 2. The design matrix of the experiment on the significance of nitrification in stream water.

	Treatment											
	1	2	3	4	5	6	7	8	9	10	11	12
Oak Creek stream water	x	x	x	x					x	x	x	x
Double distilled water & basic salts					x	x	x	x				
Douglas-fir vegetation				x	x	x	x	x	x	x	x	x
Ammonium chloride			x	x			x	x			x	x
Nitrifying bacteria seed		x		x		x		x		x		x

would be 25 ppm. This would also result in the production of 21.5 ppm NO_3^-.

The basic salt solution in Table 3 was added to the double distilled water to provide a known level of essential minor elements as constituents of the salts and to act as a buffering agent.

Table 3. Composition of the double distilled water and basic salt solution.

Dipotassium phosphate (K_2HPO_4)	0.02175 gm
Calcium chloride ($CaCl_2 \cdot 6H_2O$)	0.0275 gm
Magnesium sulfate ($MgSO_4 \cdot 7H_2O$)	0.0225 gm
Disodium hydrogen phosphate heptahydrate ($Na_2HPO_4 \cdot 7H_2O$)	0.0334 gm
Ferric chloride ($FeCl_3 \cdot 6H_2O$)	0.00025 gm
Potassium dihydrogen phosphate (KH_2PO_4)	0.0085 gm
Double distilled water (deionized)	1,000 ml

The nitrifying bacteria seed was a 375 mg portion of freeze-dried Nitrosomonas and Nitrobacter or approximately 10^5 to 10^6 cells/liter.

The 12 treatments were placed in one-liter beakers arranged on magnetic stirring plates. Two 244 ml samples of each treatment were placed in Hach manometer bottles at 20°C, and the BOD followed for 20 days. Two hundred ml samples were drawn on days 0, 5, 10, 15

and 20. These samples were filtered and analyzed for organic nitrogen, NH_4^- N, NO_2^- N, and NO_3^- N. The suspended solids were analyzed for organic nitrogen. Ammonia nitrogen concentration was determined using distillation followed by nesslerization, Standard Methods sections 132 and 212 (APHA, 1971). Nitrate-nitrogen con-concentration was determined using both the zinc reduction method and the Brucine method, Standard Methods sections 213A and 213C (APHA, 1971). Nitrite-nitrogen was determined using section 134 of Standard Methods (APHA, 1971). Organic nitrogen was determined by distillation and nesslerization of the filtrate followed by a total Kjeldahl analysis plus a microkjeldahl analysis of the suspended solids remaining in the filters.

Leachate Composition Tests

The purpose of this test was to observe the rate and amount of sugars and phenols leached with time from Douglas-fir, western hemlock and red alder vegetation. The results of this experiment will be used to help describe the process of the leaching of organic substrate from fine debris.

Previous attempts to separate the leaching process from the biological decomposition of leachate were inconclusive due to incomplete sterilization of the leaching samples (Ponce, 1974). Ponce (1974) used 2.706 ppm $HgCl_2$ to poison the stream water and

vegetation. $HgCl_2$ has been shown to be a good bacteriocide but not an effective fungicide (Schaumburg and Atkinson, 1970). Because fungal colonies did develop in the tests conducted by Ponce (1974), quite probably some of the sugars and phenols were assimilated by the organisms resulting in erroneously low values.

The test conducted in this study utilized a procedure of gas sterilization by ethylene oxide. Ethylene oxide has been extensively used to sterilize wood (Smith, 1967) and soil samples (Bartlett and Zelazny, 1967). The ethylene oxide gas is capable of penetrating wood and moisture with relative ease, providing almost complete sterilization.

Thirty samples (1 gm) of each species (Douglas-fir, western hemlock and red alder) and dissected Douglas-fir twigs were placed in open-mouthed, plastic vials (5 dram). The vials were placed on wire racks which were in turn placed inside three layers of plastic garbage bags (20 gal capacity). A shallow pan of water was placed in the bags to maintain a high humidity in the enclosure. The high humidity tended to maintain the freshness of the vegetation and increase the sterilization effectiveness of the ethylene oxide. A 21-ounce canister of 11 percent ethylene oxide was opened inside the bags. The bags were immediately sealed and left for 24 hours under a well ventilated hood. The samples were then moved to a clean transfer room.

Clean bottles (125 ml capacity) and caps were filled with 100 ml of distilled water and autoclaved. The bottles were immediately sealed and transported to the transfer room. In the transfer room the vegetative samples were placed in the bottles and the bottles resealed.

Three samples of each type of vegetation were taken on day 1, 3, 6, 8, 10, 13, 15, 30, 60 and 90 following sterilization. The drawn samples were filtered and sent to the Forest Products Department of the School of Forestry, Oregon State University, for sugar and phenol analysis.

The phenol analysis results were expressed as gallic acid equivalents per gm (dry weight) of sample. The determination was made with phosphomolybdic-phosphotungstic acid reagent according to Singleton and Rossi (1965).

Sugar analysis results were expressed as mg glucose equivalents per gm (dry weight) of sample. The sugar determination was made with Somogyi reagent and 0.892 gm KSO_3/l reagent. Titrations were made with 0.012 N thiosulfate. This procedure has been described by Somogyi (1945) and Hodge and Hofreiter (1962).

Three determinations of both sugar and phenolic concentration were run for each sample. Duplicates for each single sample agreed within 5 percent for sugars and 10 percent for phenols.

Model Development

As previously mentioned in the review of literature, the technique of converting exact differential equations, such as equation (24), into integral approximations is termed the finite difference technique. This technique is most meaningful when the system is first defined in finite-difference terms and then solutions are derived using a mass balance approach. In this way the physical activities of the dissolved oxygen exchange process are expressed as finite-difference relations. These partial solutions may be combined as a group to describe the spatial and temporal variations of the dissolved oxygen balance.

The finite-difference conceptualization of the actual physical stream is basic to understanding the modeling terminology. The surface waters of a stream may be thought of as a series of completely mixed cells each of length Δx. As shown in Figure 4 the cells are numbered from upstream to downstream, with the center of the cell as the reference point. It follows that the spatial location of the interface between two cells, n and n + 1, is defined as n + 1/2. In general, when an attribute is spatially described by a whole number (n) it represents the average value of that attribute through the cell (n). Attributes which are designated by n + 1/2 or n - 1/2 are considered to be the mean value of that attribute at the respective interface. Although attributes may be averaged over the cell length, they are

never implicitly averaged over time. The values of attributes are recorded at the beginning time (t) and end time ($t + \Delta t$) of each interval of time (Δt).

The concentration of dissolved oxygen in any cell n at any time t may be expressed as $C(n, t)$. $L(n, t)$ is then the leachate concentration in cell n at time t. $A(n, t)$ represents the average cross sectional area of cell n at time t. Since mean water velocity, U, and the longitudinal dispersion coefficient, D_L, are always used as products with A, then $UA(n + 1/2, t)$ will be referred to as the total flow rate or discharge across the interface between cells n and n + 1 at time t, and $D_L A(n + 1/2, t)$ will be termed the value of the mean total dispersion coefficient at the interface of cells n and n + 1 at time t. The term $w(n + 1/2, t)$ is defined in the dispersion activity as a small volume of water which is transferred across the interface of cells n and n + 1 during an interval of time Δt.

By applying a mass balance analysis to each of the activities of the surface water system, it can be said that the mass per unit volume in cell n at time $t + \Delta t$ is equal to the mass per unit volume in cell n at time t plus the change in mass per unit volume during the time period Δt. In the following section dispersion, advection, leachate addition, decay of leachate, deoxygenation, and atmospheric reaeration solutions will be developed individually following roughly the same finite-difference derivation of Bella (1973).

Finite-Difference Equations

Longitudinal Dispersion, in finite-difference terms may be thought of as an exchange of equal volumes of water between adjacent cells. For any cell n, the mass of leachate leaving the cell during the time interval Δt may be given by:

$$w(n-1/2, t)\ L(n, t) + w(n+1/2, t)\ L(n, t) \tag{29}$$

The first term represents the efflux across the n - 1/2 interface to cell n - 1 and the second term is the efflux across the n + 1/2 interface to cell n + 1. Conversely, the mass of leachate entering cell n from adjacent cells during Δt is:

$$w(n-1/2, t)\ L(n-1, t) + w(n+1/2, t)\ L(n+1, t) \tag{30}$$

The summation of equations (29) and (30) equals the change in mass of leachate in cell n during the time period Δt due to dispersion. By applying the mass balance approach solving in terms of leachate concentration, the concentration of leachate in cell n at time $t + \Delta t$ is:

$$\begin{aligned} L(n, t + \Delta t) = {} & L(n, t) \\ & + \frac{w(n-1/2, t)}{A(n, t)\ \Delta x}\left[L(n+1, t) - L(n, t)\right] \\ & + \frac{w(n+1/2, t)}{A(n, t)\ \Delta x}\left[L(n+1, t) - L(n, t)\right] \end{aligned} \tag{31}$$

The longitudinal dispersion coefficient, D_L, may be defined as the product of the volume rate of water exchanged per unit of cross sectional area with the effective mixing length (Bella, 1973). In

finite-difference terms, the effective mixing length is Δx and the volume rate of water exchanged per unit of cross sectional area is:

$$\frac{w}{A\,\Delta t} \tag{32}$$

Thus,

$$D_L = \frac{w\,\Delta x}{A\,\Delta t} \tag{33}$$

and

$$w = \frac{D_L\,A\,\Delta t}{\Delta x} \tag{34}$$

Substituting equation (34) into equation (31) leads to:

$$\begin{aligned} L(n, t + \Delta t) = {} & L(n, t) \\ & + \frac{D_L A(N-1/2, t)\,\Delta t}{A(n, t)\;\Delta x^2}\left[L(n-1, t) - L(n, t)\right] \\ & + \frac{D_L A(N+1.2, t)\,\Delta t}{A(n, t)\;\Delta x^2}\left[L(n+1, t) - L(n, t)\right] \end{aligned} \tag{35}$$

To prevent instability errors it is necessary that the volume exchanged during any interval Δt not be greater than the total volume of any one cell. This leads to the requirement that:

$$\frac{D_L\,\Delta t}{\Delta x^2} < 1/2 \tag{36}$$

The longitudinal dispersion of dissolved oxygen is handled the same as the dispersion of leachate. The dispersion equation for

oxygen is the same as equation (35) with L replaced by C.

Advection is the transporting and diluting effect of water moving through the surface water cells from upstream to downstream on leachate and dissolved oxygen. This activity may be thought of as the transfer of a given volume of water from cell to cell down the stream. The volume of water transferred into cell n is equal to the product of the flow rate or discharge across the n - 1/2 interface with the time interval Δt:

$$UA(n-1/2, t)\ \Delta t \tag{37}$$

Since this volume comes from cell n - 1, then the leachate mass transferred into cell n during Δt is:

$$UA(n-1/2, t)\ \Delta t\ L(n-1, t) \tag{38}$$

Similarly, the amount transferred out of cell n to cell n + 1 is:

$$UA(n+1/2, t)\ \Delta t\ L(n, t) \tag{39}$$

The sum of equations (38) and (39) represents the change in leachate mass in cell n during Δt due to advection. By applying a mass balance to cell n and solving for leachate concentration:

$$L(n, t + \Delta t) = \frac{L(n, t)\ A(n, t)}{A(n, t + \Delta t)} + \frac{UA(n-1/2, t)\ L(n-1, t)\ \Delta t}{A(n, t + \Delta t)\ \Delta x} - \frac{UA(n+1/2, t)\ L(n, t)\ \Delta t}{A(n, t + \Delta t)\ \Delta x} \tag{40}$$

A stability requirement is that no more water can be transferred through a cell than there is in the cell, which leads to:

$$U \Delta t \leq \Delta x \tag{41}$$

Further, if

$$U \Delta t < \Delta x \tag{42}$$

a numerical mixing error is produced which must be corrected by a pseudo-dispersion coefficient given by:

$$D_p = \frac{U}{2} [\Delta x - U \Delta t] \tag{43}$$

(Bella, 1968)

Again, the effect of advection on the dissolved oxygen concentration is identical to equation (40) with L replaced by C.

Leachate addition to the stream is directly proportional to the amount of leachate remaining in the slash and the time interval Δt. In other words, the addition of leachate is equal to the decrease in the leachate level in the slash over the time period. The mass of leachate in the slash associated with cell n at time t is $S(n, t)\ A(n, t)\ \Delta x$. The change in slash leachate over Δt is given by:

$$-K_4(n, t) \Delta t [(1-\theta) S(n, t) A(n, t) \Delta x + \theta S(n, t + \Delta t) A(n, t + \Delta t) \Delta x] \tag{44}$$

where K_4 is the leaching rate constant and θ is a weighting function ($0 < \theta < 1$) used to approximate the average leachate mass in the slash over the interval Δt. When Δx and Δt are small enough to assume the change in leachate with time is linear, then $\theta = 0.5$. Applying the mass balance yields the mass of leachate remaining in the slash after Δt.

$$S(n, t + \Delta t)\, A(n, t + \Delta t)\, \Delta x = S(n, t)\, A(n, t)\, \Delta x \cdot \frac{[1 - .5\, K_4(n, t)\, \Delta t]}{[1 + .5\, K_4(n, t)\, \Delta t]} \tag{45}$$

The change in leachate mass in cell n is equal but opposite in sign to the change of leachate in slash. However, a portion of the leachate added to the cell n comes by way of addition to cell n - 1 and subsequent advection into cell n during the time interval Δt. Likewise, a portion of the leachate added to cell n during the time period Δt is advected to cell n + 1. Thus the change in leachate concentration over Δt due to leaching is:

$$\begin{aligned} L(n, t + \Delta t) = {} & \frac{L(n, t)\, A(n, t)}{A(n, t + \Delta t)} + \left(1 - \frac{U \Delta t}{2 \Delta x}\right) \\ & \cdot \left(\frac{S(n, t)\, A(n, t)}{A(n, t + \Delta t)}\right) \\ & \cdot \left(1 - \frac{[1 - .5\, K_4(n, t)\, \Delta t]}{[1 + .5\, K_4(n, t)\, \Delta t]}\right) \\ & + \left(\frac{U \Delta t}{2 \Delta x}\right) \frac{S(n-1, t)\, A(n-1, t)}{A(n, t + \Delta t)} \\ & \cdot \left(\frac{[1 - .5\, K_4(n, t)\, \Delta t]}{[1 + .5\, K_4(n, t)\, \Delta t]}\right) \end{aligned} \tag{46}$$

<u>Decay of leachate and deoxygenation.</u> The decay of leachate in the surface water is proportional to the amount present and the length of the time interval Δt. The mathematics involved in deriving the

effect of leachate decay are identical to the leaching effect. Thus, the change in leachate concentration due to biochemical decay is given by:

$$L(n, t + \Delta t) = \frac{(Ln, t)\ A(n, t)}{A(n, t + \Delta t)} \left(\frac{1 - .5\ K_1(n, t)\ \Delta t}{1 + .5\ K_1(n, t)\ \Delta t} \right) \tag{47}$$

The deoxygenation caused by the biochemical decay of leachate is equal to the change in leachate BOD concentration over Δt:

$$- \frac{L(n, t) A(n, t)}{A(n, t + \Delta t)} \left[1 - \left(\frac{1 - .5\ K_1(n, t)\ \Delta t}{1 + .5\ K_1(n, t)\ \Delta t} \right) \right] \tag{48}$$

Thus, the decrease in dissolved oxygen concentration over Δt in the mass balance equation is:

$$C(n, t + \Delta t) = \frac{C(n, t)\ \Delta(n, t)}{A(n, t + \Delta t)} - \frac{L(n, t)\ A(n, t)}{A(n, t + \Delta t)} \left[1 - \left(\frac{1 - .5\ K_1(n, t)\ \Delta t}{1 - .5\ K_1(n, t)\ \Delta t} \right) \right] \tag{49}$$

<u>Atmospheric reaeration</u> is the stabilizing effect which tries to force the dissolved oxygen concentration of a body of water to saturation. Reaeration is proportional to the deficit between $C(n, t)$ and the saturation concentration $C_s(n, t)$. The change in dissolved oxygen due to atmospheric reaeration during a time interval Δt is:

$$.5\ K_2\ (n, t)\ \Delta t \left\{ [C_s(n, t) - C(n, t)] + [C_s(n, t + \Delta t) - C(n, t + \Delta t] \right\} \tag{50}$$

By applying a mass balance, the dissolved oxygen concentration in

cell n at time $t + \Delta t$ may be computed from:

$$C(n, t+\Delta t) = \frac{C(n,t)(1-.5K_2(n,t)\Delta t) + .5K_2(n,t)\Delta t[C_s(n,t) + C_s(n,t+\Delta t)]}{[1 + .5\,K_2(n,t)\,\Delta t]} \quad (51)$$

Equations (35), (40), (46), (49) and (51) may be combined to form the finite-difference model of the impact of logging debris on dissolved oxygen of the surface waters of streams. This section is the central body of the simulation of the simultaneous occurrence of longitudinal dispersion, advection, leachate addition, leachate decay, deoxygenation, and atmospheric reaeration. However, these equations alone are useless until data from the real system are collected, predicted and manipulated to a usable form. The following is a description of the process of assimilation of available data and the assumptions involved.

Calculating Finite-Difference Variables and Constants

Stream water temperature. Potential BOD and all rate constants are dependent on stream water temperature. Water temperature in turn fluctuates both yearly and diurnally due to seasonal and daily changes in thermal radiation. Water temperature may significantly increase as a stream flows through a clearcut. This increase is proportional to the incident net solar radiation which strikes the water surface and inversely proportional to the flow rate or discharge.

Variations in stream water temperature may be described empirically by relationships based on previous temperature records of yearly and diurnal water temperature fluctuations. Mean monthly maximum and diurnal water temperature data of Flynn Creek (Brown and Krygier, 1970) in Oregon's Coast Range were used to develop two stream temperature functions to describe yearly and diurnal water temperature variation.

A sine curve was fit to the mean monthly maximum water temperatures of the Flynn Creek data. This relationship gives the mean monthly maximum water temperature as:

$$T_m = 4.5 \cos [t (1.99102 \times 10^{-7}) - 3.66519] + 52.5 \qquad (52)$$

where T_m is the mean monthly maximum stream temperature during month m in ^{o}F, t is time of the year in seconds to midmonth from 0600, January 1. All trigonometric functions presented in this text are in radians. Similarly diurnal water temperature variation about the mean monthly maximum was also fit to a sine curve. The water temperature at any time during the early morning hours (0400 to 1000) may be given as:

$$T_{am} = -1.8 \sin [t (1.09083 \times 10^{-4}) - 1.963495] + T_m \qquad (53)$$

where T_{am} is the morning water temperature in ^{o}F and t is time of the day in seconds from 0600 hours. Midday and evening (1000 to 0400 hours) water temperature is given as:

$$T_{pm} = 1.7 \sin[t(5.45415 \times 10^{-5}) - 0.981748] + T_m \qquad (54)$$

where T_{pm} is the afternoon and evening water temperature in ^{o}F and t is time of the day in seconds. These two functions, equation (52) and equations (53) and (54), and the data they were fit to are plotted in Figures 5 and 6.

This model has been designed to describe the maximum impact of slash on dissolved oxygen of small streams. This maximum occurs when water temperature is also maximized, which in turn is maximized by clear days and no rainfall. For these reasons no attempt was made to model the random effects of clouds and rainfall on stream water temperature. This is reasonable since the period of interest is June through September when there is very little rain and mostly clear weather prevails.

Equation (52) and equations (53) and (54) will be used to estimate the incoming stream water temperature at the top of a clearcut.

As previously reported, the change in stream water temperature through a clearcut may be estimated by equation (23) (Brown, 1969). This equation requires that the net solar radiation flux density (BTU/ $ft^2 \cdot$ min) be known. Maximum daily net solar radiation in a clearcut is a function of the daily maximum altitude (h) of the sun above the horizon. The altitude of the sun is, in turn, a function of latitude and time of year. The daily maximum altitude of the sun above the horizon

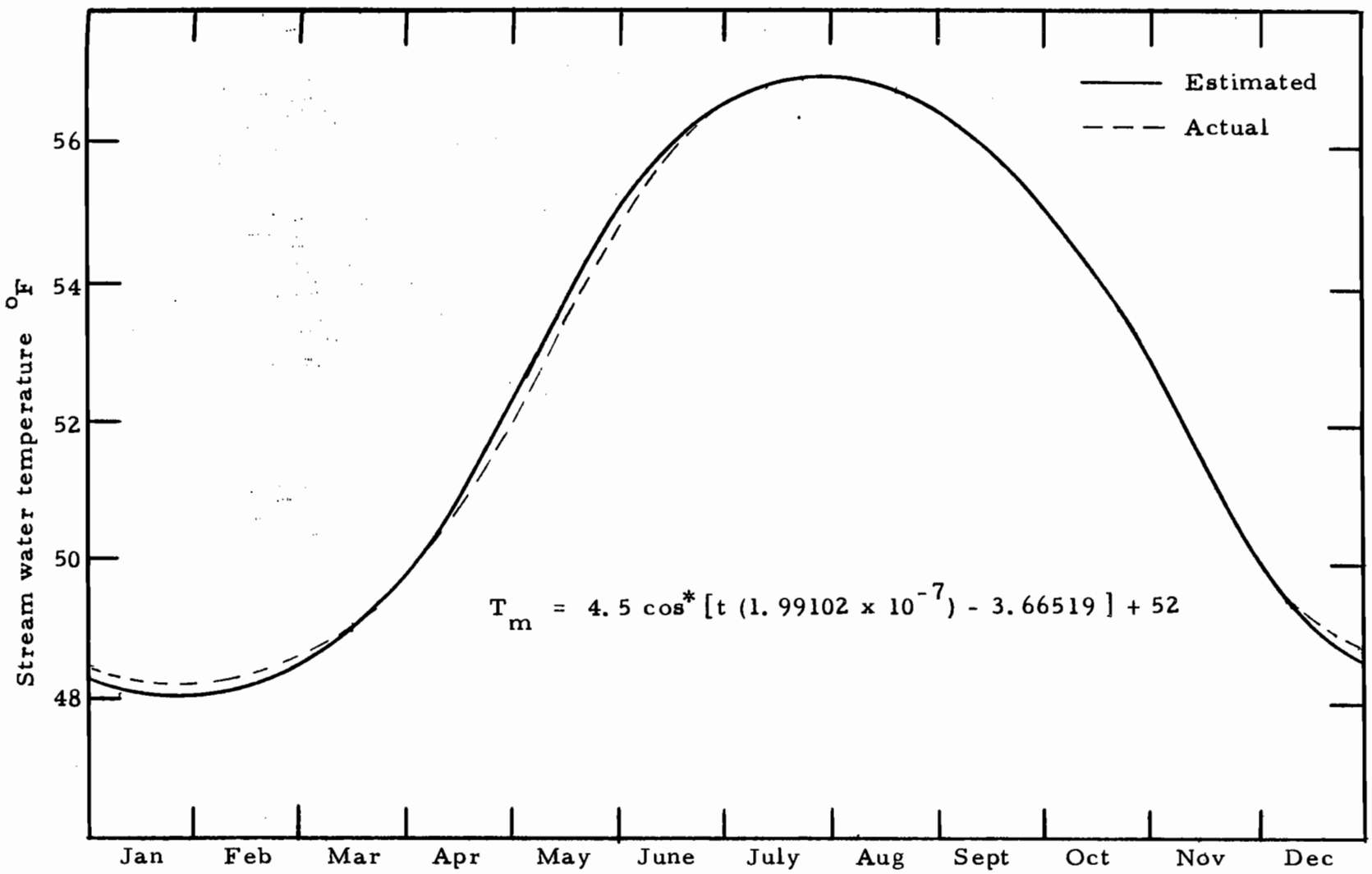


Figure 5. Mean maximum monthly stream temperature of Flynn Creek (after Brown and Krygier, (1970). * = trigonometric functions are in radians

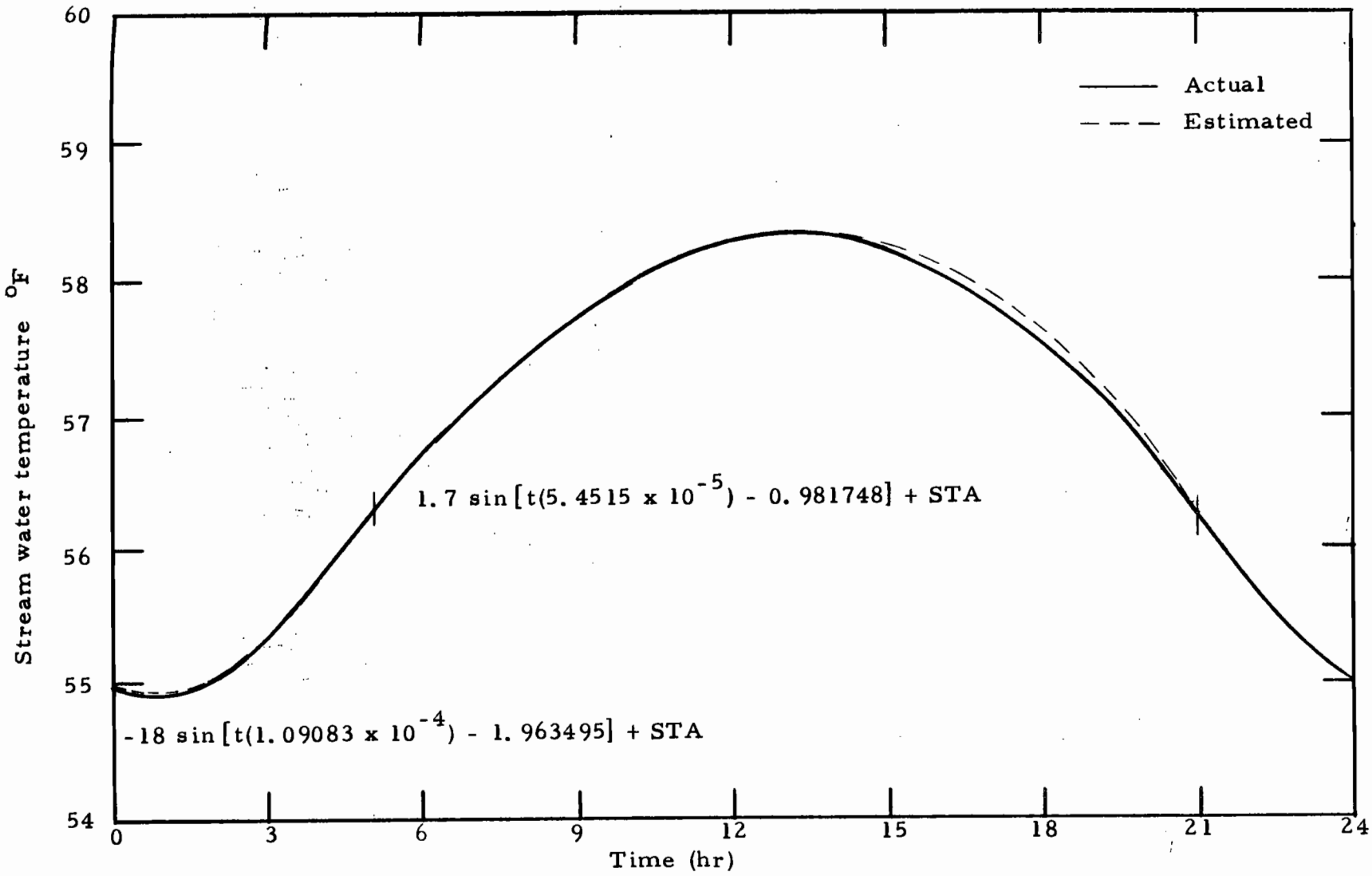


Figure 6. Diurnal fluctuation in stream water temperature of Flynn Creek (after Brown and Krygier, 1970). August 24, STA = 56.6°F.

at 45° N latitude may be estimated by:

$$h = 25 \cos [t(1.99102 \times 10^{-7}) - 2.9760] + 45 \tag{55}$$

where h is the maximum daily altitude of the sun in degrees and t is time in seconds of the year from 0600 January 1. Equation (56) is the result of a sine curve fit to data from a solar altitude table for 45° N (List, 1966). Equation (55) and the data it was fit to is plotted in Figure 7.

Net daytime thermal radiation incident to a nonforested stream may be estimated by the fitted equation:

$$Rad = (0.07h + 0.3) \sin [t(7.272205 \times 10^{-5})] \tag{56}$$

where t is time in seconds of the day from 0600. Nocturnal radiation for the clearcut is given by:

$$Rad = 0.5 \sin [t(7.272205 \times 10^{-5})] \tag{57}$$

Net thermal radiation for a forested stream can be estimated by the fitted equations:

$$Rad = -0.75 \cos [t(1.45444 \times 10^{-4})] \tag{58}$$

for daytime hours (0900 to 1500) and

$$Rad = 0.3 \cos [t(4.84813 \times 10^{-5})] \tag{59}$$

for nighttime (1500 to 0900 hours).

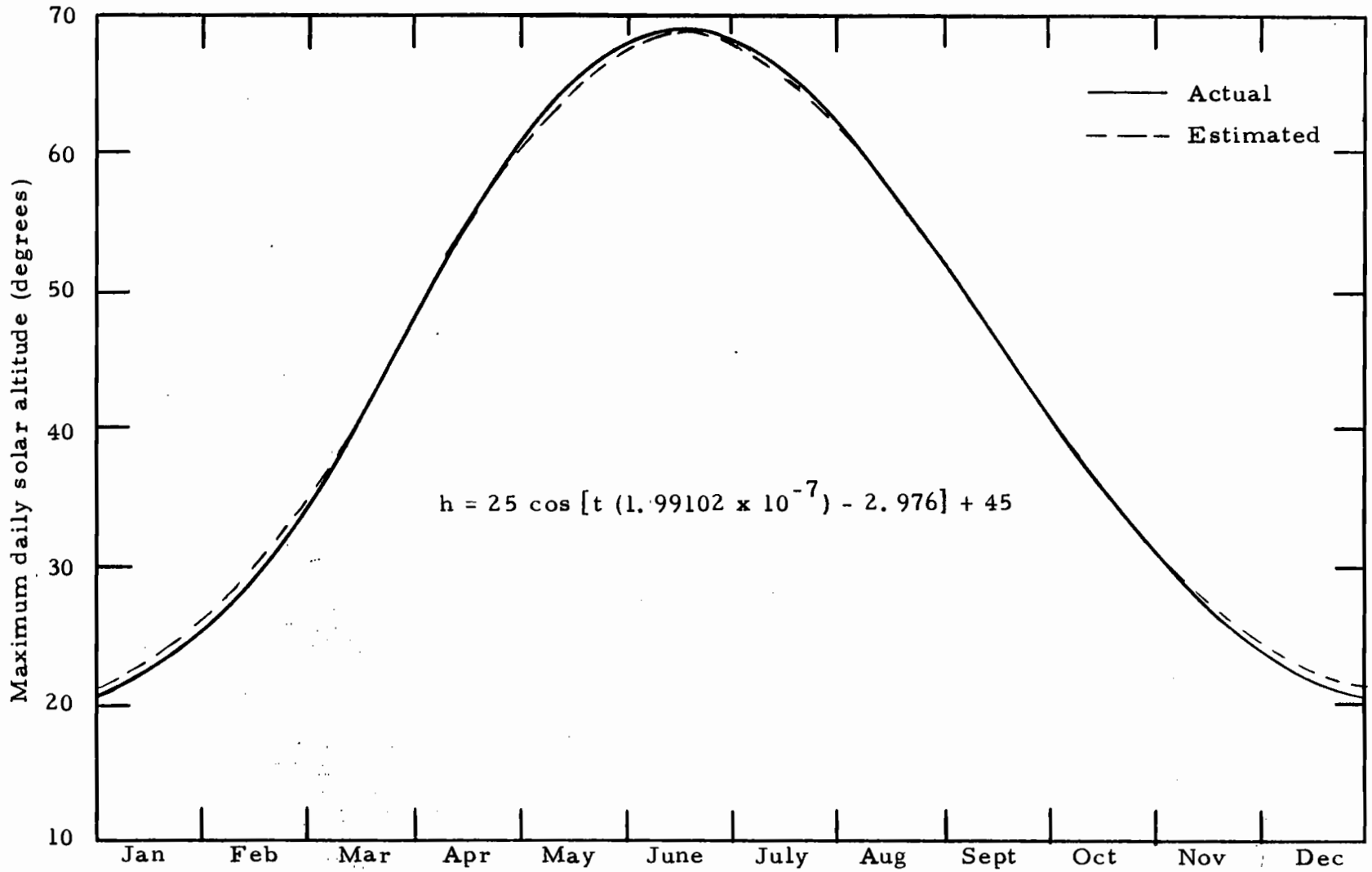


Figure 7. Maximum daily solar altitude (h).

Equations (56), (57), (58) and (59) were fit to net thermal radiation data by Brown (1969). Equations (56) through (59) and Brown's (1969) data are plotted in Figures 8 and 9.

These temperature and radiation functions incorporate an assumption that a body of water heats and cools linearly. This assumption is true when clearcuts are less than 2,000 feet long and there are negligible conductive errors due to solid rock stream bottoms (Brown, 1972). It also has been assumed that the slope of the stream channel is close enough to zero to neglect aspect and that the surrounding landforms do not significantly affect the heating or cooling of the streamwater.

Leachate BOD concentration and all rate constants are affected by streamwater temperature. The effects of water temperature on raw municipal sewage BOD and the first order reaction rate constants have been studied extensively by Zanoni (1967). He found that the first order reaction rate constants can be estimated at various temperatures by:

$$K_T = (0.796)(1.126)^{(T-15)}K_{20} \quad ; \text{ Range 2 to } 15^{\circ}C \tag{60}$$

$$K_T = (1.047)^{(T-20)}K_{20} \quad ; \text{ Range 15 to } 32^{\circ}C \tag{61}$$

$$K_T = (1.728)(0.985)^{(T-32)}K_{20} \quad ; \text{ Range 32 to } 40^{\circ}C \tag{62}$$

where K_T and K_{20} are the value of the reaction rate constant at temperature T and $20^{\circ}C$ respectively. Equations (60), (61) and (62)

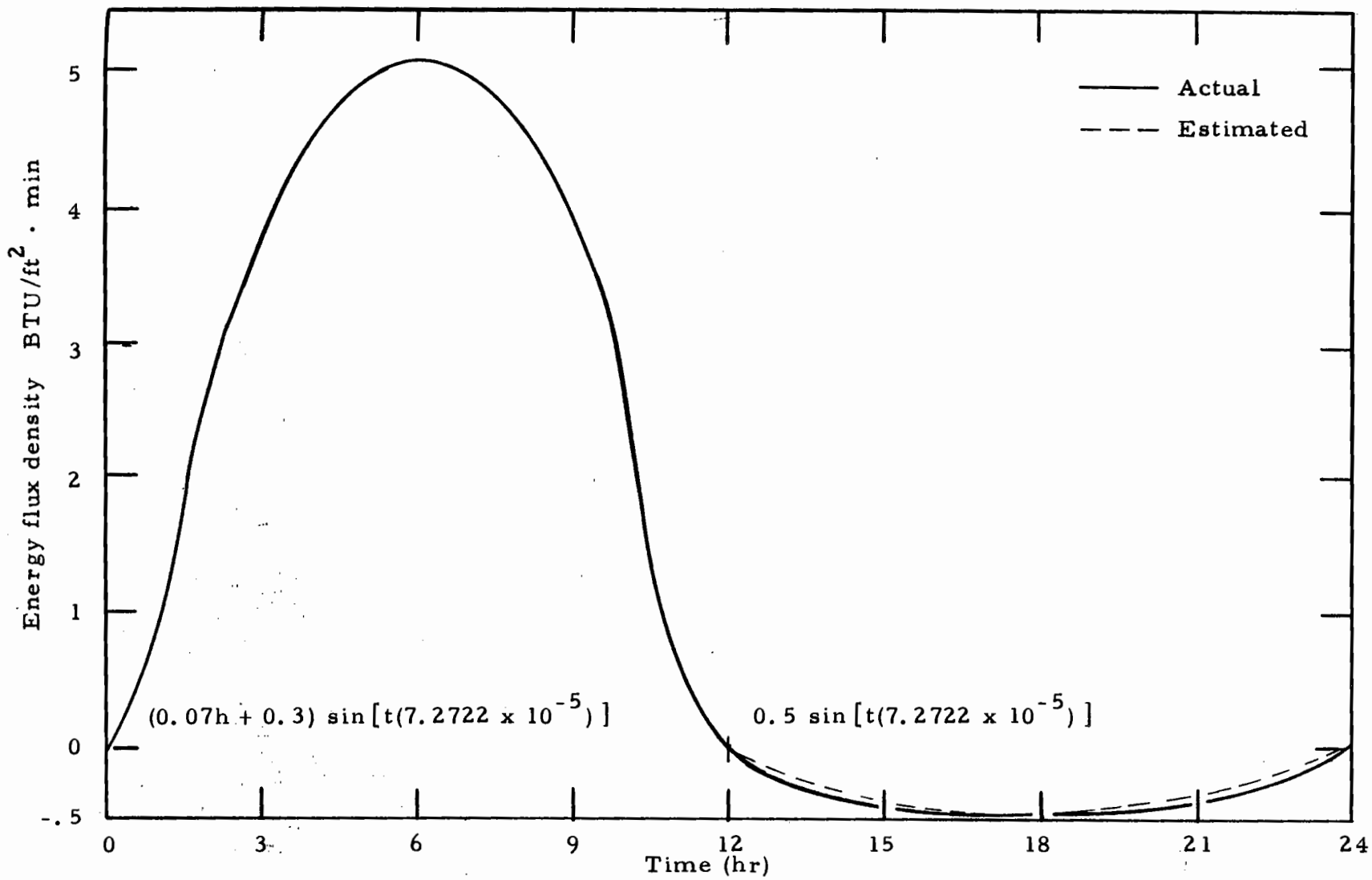


Figure 8. Net thermal radiation to a clearcut stream (after Brown, 1969).

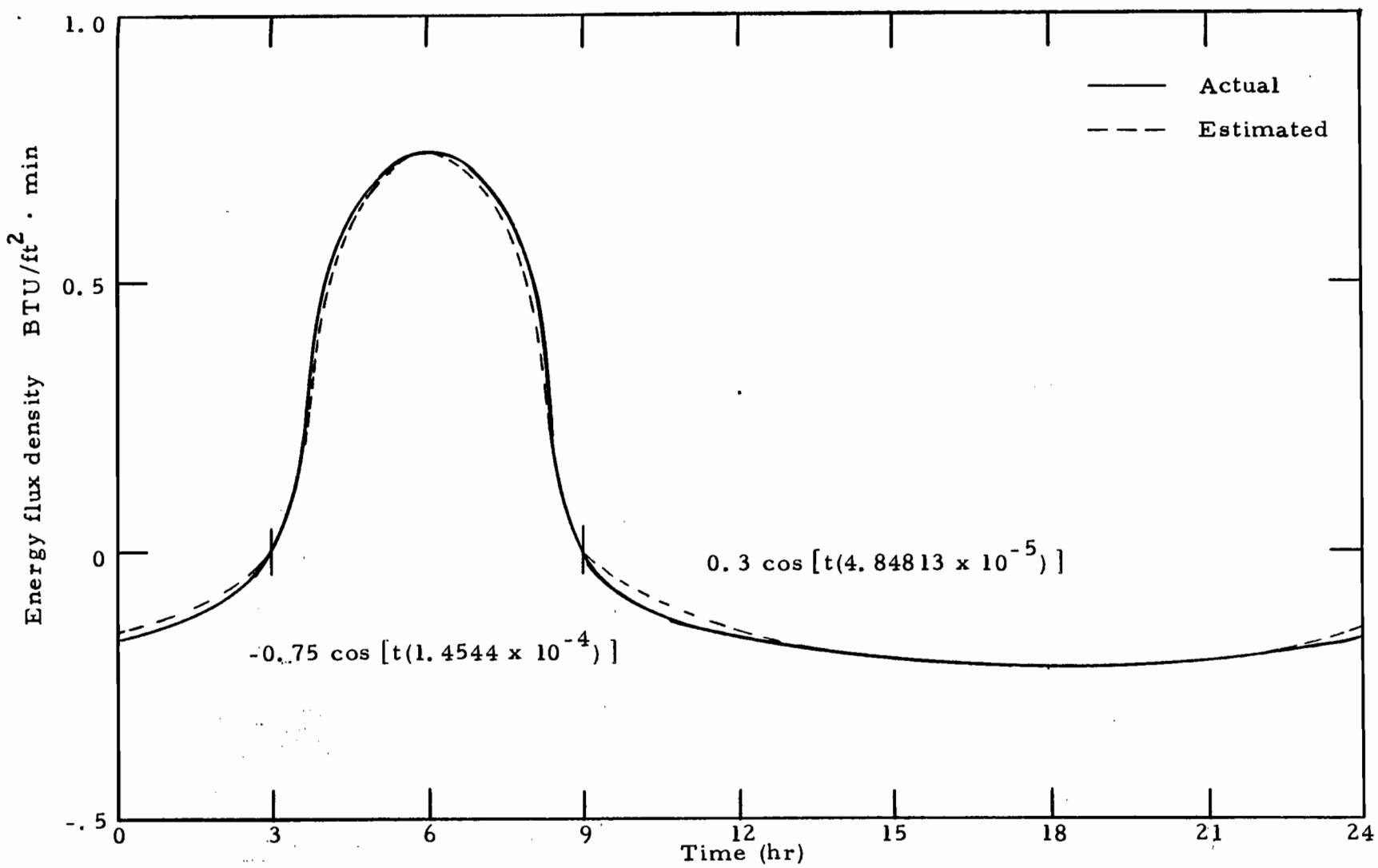


Figure 9. Net thermal radiation incident to a forested stream (after Brown, 1969).

will be used to adjust K_1 and K_4 values at 20°C determined in this study and by Ponce (1974).

The effect of water temperature on pollutant BOD concentration may be expressed by:

$$L_T = L_{20} [1 + 0.0113 (T-20)] ; \text{ Range 20 to } 35^\circ C \tag{63}$$

$$L_T = L_{20} [1 + 0.0033 (T-20)] ; \text{ Range 2 to } 20^\circ C \tag{64}$$

where L_T and L_{20} are the leachate BOD concentration at T and 20°C (Zanoni, 1967). Results from experimentation on the BOD of fine logging debris by this study and Ponce (1974) will be used to estimate L_{20}.

The variation of K_2 with water temperature is accounted for in equation (12) for predicting atmospheric reaeration.

The longitudinal dispersion coefficient for small streams may be estimated by:

$$D_L = 49.62 \, R_H^{.8333} \, U \eta \tag{65}$$

where D_L is the longitudinal dispersion coefficient in ft^2/sec, R_H is the hydraulic radius of the stream channel in ft, U is the mean stream velocity in ft/sec and η is the Manning's coefficient (Glover, 1964). This approximation will be used to estimate D_L for the simulated stream. A sample solution for a dispersion coefficient is given in Appendix V.

Input Data

The K_1 and K_4 values at 20°C used in this model were determined experimentally in the biological research portion of this study and by Ponce (1974). K_2 and D_L were estimated using equations (12) and (65) developed by Holtje (1971) and Glover (1964) respectively. The simulated stream was a composite of the various segments of Berry Creek studied by Holtje (1971). Holtje's survey data of Berry Creek provided the required hydraulic data for use in the model and in the estimation of K_2 and D_L.

The degree of logging debris loading in a hypothetical clearcut was based on studies conducted by Froehlich (1973) and Lammel (1973) on natural debris and logging residue in headwater streams. Several assumptions have been made to use their data in the dissolved oxygen model. Lammel assumed the impact zone of a stream to be 30 ft wide. It will be assumed that the distribution of slash across the stream bed and along the stream length was constant. Froehlich and Lammel did not include needles and leaves in their debris calculations. Overton et al. (1973), however, report that the estimated ratio of needle to stem mass for old-growth Douglas-fir is 0.17089. By applying this ratio to the total weight of organic matter deposited after logging, the amount of needles may be estimated. These seemingly gross assumptions appear to be the most reliable available and are used only as rough approximations.

Another reasonable assumption is that a harvest operation begins by felling the trees along the stream course first, then proceeding uphill. This assumption leads to two more conditions, 1) it can be assumed that the majority of logging residue is deposited in the stream during a relatively short time period (less than 8 hours), and 2) yarding will not begin for at least 2 to 3 weeks. These assumptions mean that the debris loading can be assumed to be nearly instantaneous and that the debris will not be disturbed for 2 to 3 weeks after deposition.

The streams to which this study applies are at low flow (0.1 to 5 cfs) during the summer harvest and cannot transport significant amounts of debris downstream. The depth of these streams varies from 0.1 to 2 feet and are 1 to 10 feet wide. Thus, logging debris larger than 3 cm is not as likely to be submerged as is debris less than 3 cm in size, all of which will be assumed to be submerged. Furthermore, research on leachate BOD from logs indicates that even if the larger debris is submerged, its effect on dissolved oxygen is much smaller and more long-term in nature than that of finely divided debris. Thus, it is assumed that the 20 to 90 day BOD of logging debris is predominantly due to needles and twigs less than 3 cm in size. A sample problem exemplifying the calculation of the loading of logging residue is presented in Appendix II.

Model I

Model I may be diagrammatically represented as the flow chart in Figure 10. Input data sets the start and end time, the flow rate or discharge, K_1 and K_4 at 20 ^{o}C and the conversion factor of mg BOD/ gm dry weight of slash. The stream to be simulated is input by segments of homogeneous hydraulic characteristics. The hydraulic characteristics entered are; average cross sectional area, mean velocity, length of the segment, average width, slope of the channel and hydraulic radius.

Conversion factors are calculated and constants for each segment (cell length, weight of slash, surface area and dispersion coefficient) initialized as the data for each segment is input. The stability checks indicated by equations (36) and (41) are also made at this time.

Initial stream temperature, mass concentration of slash, leachate concentration and dissolved oxygen concentration are initialized for each cell. For stream temperature, the temperature routine is called to calculate the stream water temperature at the start time for a forested stream using equation (52) and equations (53) and (54). The mass concentration of slash is determined from the weight of slash in the cell divided by the cell volume. The leachate concentration is set equal to zero and the dissolved oxygen concentration is set

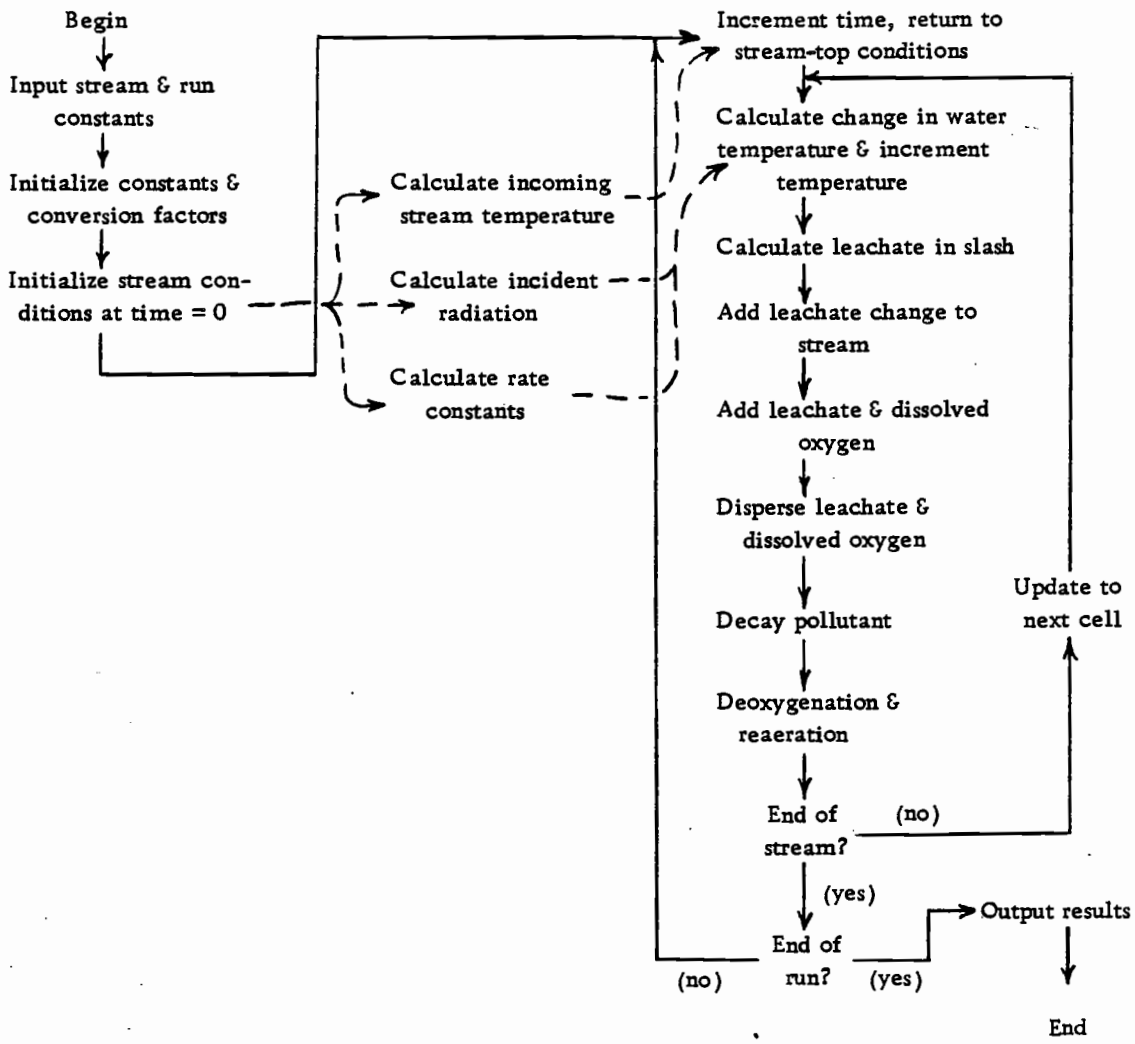


Figure 10. Operational flow chart of Model I.

equal to the saturated dissolved oxygen concentration at the given stream temperature.

The boundary conditions are accounted for by a precell and a post cell at the beginning and end of the stream. The precell has water temperature equal to the incoming stream temperature, saturated dissolved oxygen concentration, and zero leachate concentration. The postcell takes on the same values as the last cell of the stream.

The actual simulation of the stream begins by calculating the water temperature of the incoming water in the precell and the radiation flux density of the clearcut by equations (56) and (57). Time since 0600 January 1 and time of day since 0600 are incremented by Δt. If it is a new day, time of day is reset to zero. The change in stream temperature per cell is calculated from the radiation flux density for the segment and by equation (23).

Each cell of the stream is updated to $t + \Delta t$ by first adding the temperature change for the cell, correcting the reaction rate constants for temperature and evaluating each of the finite difference equations. The scheme of contributing concentrations for calculating the concentrations in cell n at time $t + \Delta t$ is shown in Figure 11. The concentrations in cell n at $t + \Delta t$ are computed from the concentrations in cells n - 1, n and n + 1 from the previous time period t and from the concentrations in cell n - 1 during the present time period $t + \Delta t$. By proceeding from upstream to downstream, then, the concentrations

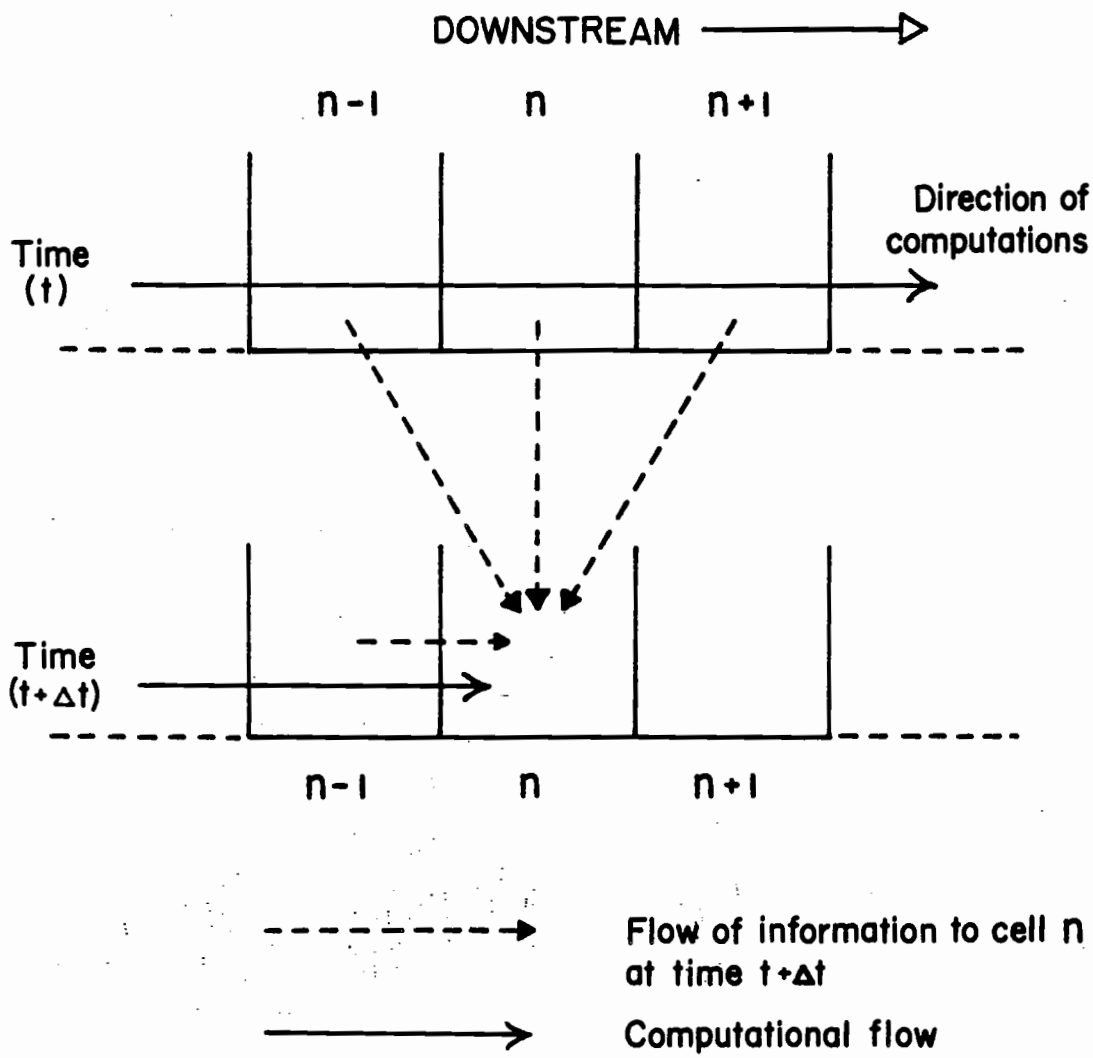


Figure 11. Computational and informational flow scheme for Model I.

of cell n at $t + \Delta t$ are always calculated from concentrations of previously determined cells. The following order was used to calculate the concentrations in cell n at $t + \Delta t$: change in leachate in the slash, equation (44); addition of leachate to the cell, equation (46); advection of the leachate, equation (40); advection of dissolved oxygen, equation (40a); dispersion of leachate, equation (35); dispersion of oxygen, equation (35a); decay of leachate, equation (48); and deoxygenation and atmospheric reaeration of dissolved oxygen, equations (49) and (51). This order is logical, but in some cases arbitrary; for example, the order of dispersion and advection. Any error contributed by altering the order are minute and time-compensating.

The simulation continues in a downstream-fashion until the end of the clearcut is reached. Equations (58) and (59) are then called to estimate the incident radiation for a forested stream. The simulation continues until the end of the stream is reached.

If the time is not greater than or equal to the end time, the present concentrations which are subscripted $t + \Delta t$ are changed to t, time is incremented by Δt and the simulation begins from the top of the clearcut again. If the time is greater than or equal to the end time, the output subroutine is called.

The output consists of monitoring ten points along the stream course for dissolved oxygen concentration, leachate concentration, leachate remaining in the slash and water temperature. The output

is displayed in tabular form on the line printer and as line printer graphs.

Each component and subroutine of Model I was checked independently against hand-calculated results. For example, the incoming water temperature and change in stream temperature through a clear-cut were first calculated and simulated for a simple stream of constant hydraulic characteristics. By comparing these results, the temperature functions were made computationally correct. The addition of leachate, decay of leachate, deoxygenation, and reaeration functions were tested separately, and in combined form for a completely mixed, constant temperature, one-cell model. Slug loads, constant loads, leachate loading and various combinations of these loadings were simulated and checked against the results from equations (11) and (22).

The next debugging step was a one-cell model with all of the model components except advection and dispersion. This one-cell model was then expanded to a complete multicellular stream with advection and longitudinal dispersion. The complete model was then subjected to a sensitivity analysis to determine the effect of various lengths of Δt and Δx and different hydraulic dimensions on dispersion and leachate concentration. Further runs were made, varying the longitudinal dispersion coefficient directly, to determine its impact on leachate concentration.

Longitudinal Dispersion and the Lagrange Technique

Longitudinal dispersion can produce significant changes in leachate concentration only when a steep concentration gradient exists. Because the actual value of D_L is uncertain and can only be approximated by equation (65), it is appropriate to determine the magnitude of error in leachate concentration if longitudinal dispersion is neglected.

Dobbins (1964) examined the effects of longitudinal dispersion on steady and uniform stream flow. He used the one-dimensional stream equations for leachate concentration and dissolved oxygen deficit (equation 24) as ratios against themselves with and without dispersion. Dobbins (1964) found that the ratio (R) between equations with and without dispersion were functions of the dimensionless parameter $U^2/(2\ K\ D_L)$. As shown in Table 4, the effect of dispersion is negligible for values of $U^2/(2\ K\ D_L)$ which exceed 10.

Table 4. The relationship of $U^2/(2\ K\ D_L)$ and R (after Dobbins, 1964).

$U^2/2KD_L$	1	10	20	40	100	200	∞
R	0.732	0.954	0.976	0.988	0.995	0.997	1.000

The minimum value of $U^2/(2\ K\ D_L)$ for the streams considered in this study is greater than 130. Thus, it will be tentatively assumed that the effect of longitudinal dispersion is negligible.

It can be seen from equation (43) that by making $\Delta t = \Delta x/U$, the pseudodispersion due to advection becomes zero. This condition leads to the conclusion that the factors affecting any given cell are all downstream effects. That is, the concentration of cell n is never affected by the concentration of cell n + 1 either at time t or $t + \Delta t$.

By assuming that longitudinal dispersion is negligible, we can now consider a small slice of stream of constant volume $A\ \Delta x$ which moves downstream with velocity U. Such a completely mixed cell would experience no net advection nor dispersion. This eliminates equations (35) and (40) from the finite difference model. As this cell moves downstream, leachate is added and decays resulting in an oxygen deficit and subsequent reaeration.

The finite difference equation of leachate addition may be solved exactly for any finite time interval by solving equation (6) at time t and $t + \Delta t$ and taking the difference. Inspection shows that half of the leachate is added between n - 1 and n - 1/2 from the slash between n - 3/2 and n - 1/2. The other half is added between n - 1/2 and n from the slash between n - 1/2 and n + 1/2 (see Figure 12). Thus, the addition of leachate to cell $A\ \Delta x$ during Δt is:

$$\Delta L = \frac{1}{2} S_{o(n-1)}(e^{-K_4 t} - e^{-K_4(t+\Delta t)}) + \frac{1}{2} S_{o(n)}(e^{-K_4 t} - e^{-K_4(t+\Delta t)}) \qquad (66)$$

where ΔL is the change in leachate concentration, $S_{o(n-1)}$ is the

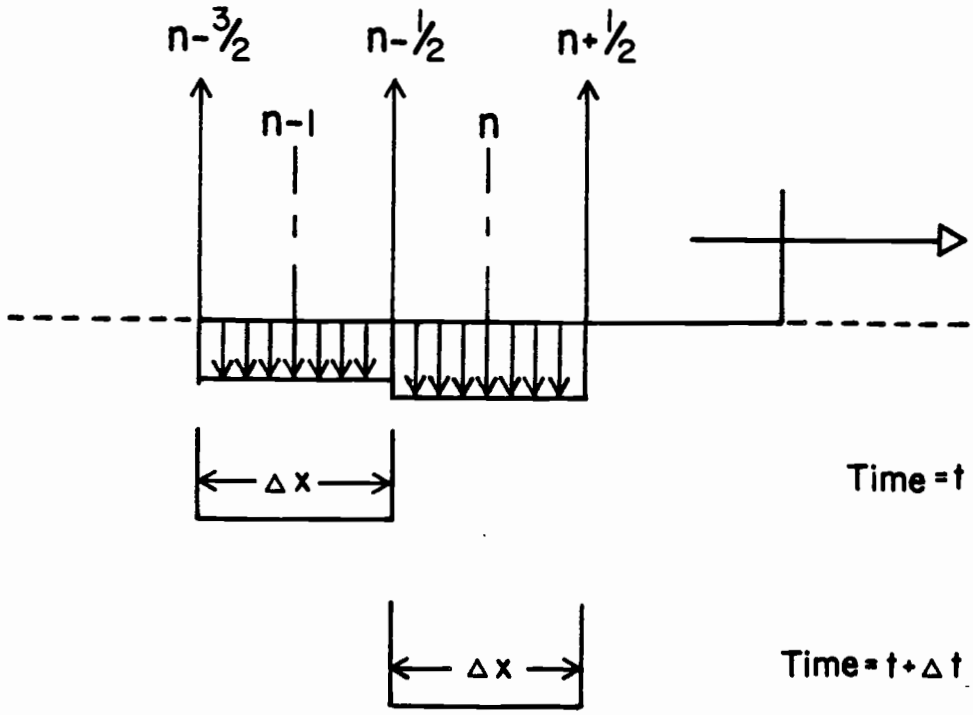


Figure 12. The addition of leachate to cell A Δx, moving from n - 1 to n during Δt.

leachate remaining in the slash in unit n - 1 and $S_{o(n)}$ is the leachate remaining in the slash in unit n. This method of modeling enables one to simulate the state of the surface water system at any time without advancing from time t = 0 to time t by increments of Δt. That is, in Model I the entire stream is simulated from t to t + Δt with all of the cells updated to the same time. However, this second technique allows a slice of water to move downstream at velocity U incrementing time as it goes, meaning the downstream concentration is dependent only on the concentration of the water immediately upstream of a given point at any time and the beginning concentration may be derived from analytic solutions. This permits one to choose any start time for routing a slice downstream as compared to Model I where the start time was always fixed at t + Δt. This second technique of stream simulation was developed by Bella (1967) and is termed the Lagrange method. This new model will be referred to in this study as Model II.

Model II

The flow chart for Model II is shown in Figure 13. Model II begins by reading input constants (begin time, end time, discharge, 20°C rate constants and slash loading), and calculating constants. Next, the survey data for the stream is read in by segments. Constant flow is checked, surface area of the segment is calculated, the BOD

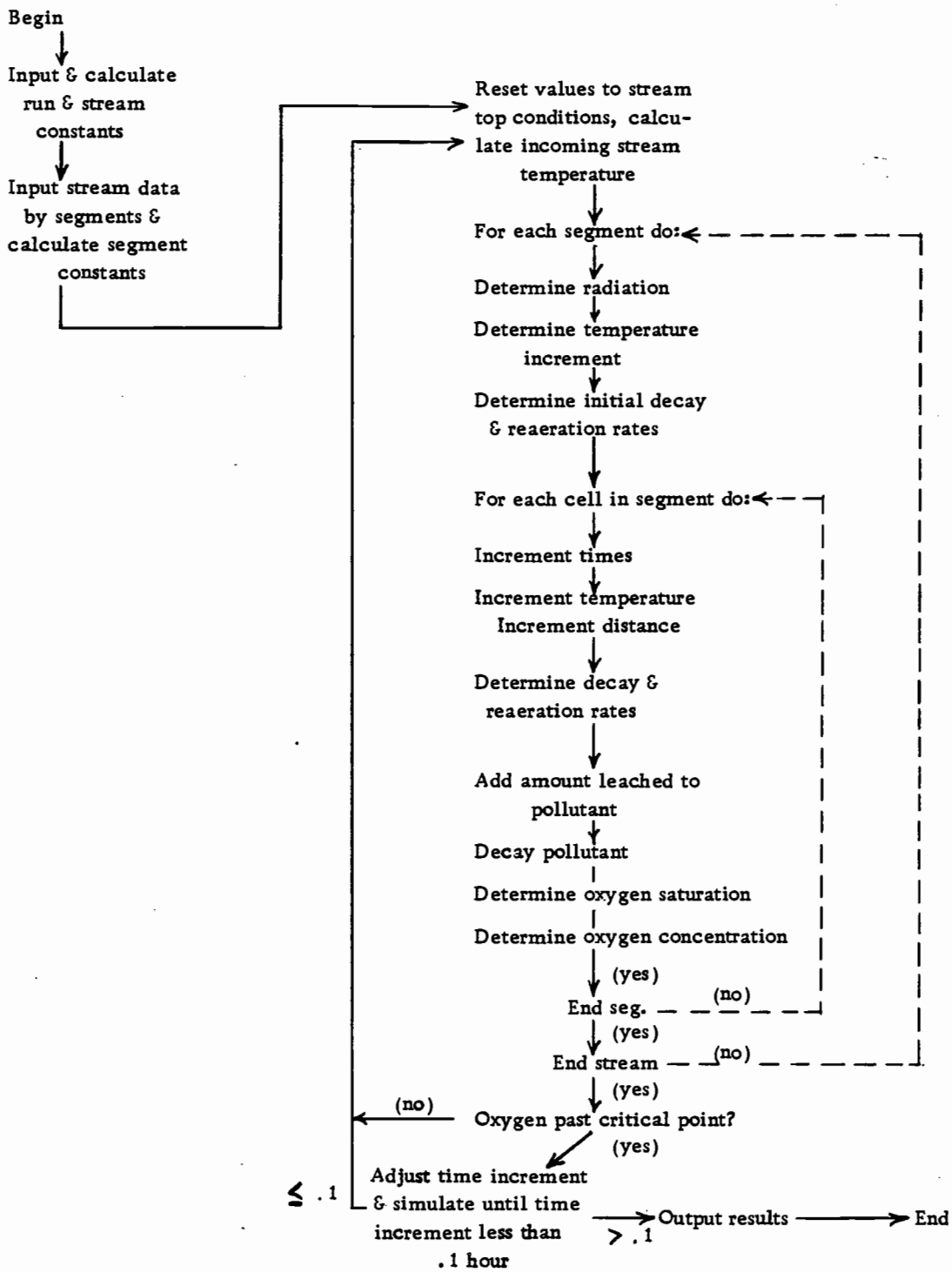


Figure 13. Flow chart of Model II.

concentration of the slash is determined and the volume of the segment is checked to see if it is a minimum. Δx and Δt are calculated by dividing the smallest segment volume into 10 parts and making $\Delta t = \Delta x/U$. Segment constants are also calculated at this time.

The stream simulation begins by calculating the mean maximum monthly stream temperature, equation (54), and by calculating the maximum solar altitude for the day, equation (55). The incoming stream temperature is calculated using either equation (53) or (54).

The radiation subroutine is called and the temperature change per Δt is calculated. The average temperature of the first cell is determined. The reaction rate routine is called to calculate the reaction rate constants.

The addition of leachate is calculated by equation (66). The leachate is added and decayed. The dissolved oxygen concentration is deoxygenated and atmospherically reaerated. The resulting dissolved oxygen concentration is checked to see if it is the lowest for the run. If it is, then the distance, time and concentration of oxygen are stored. Time and distance are incremented and the cycle of addition, decay, deoxygenation and reaeration begins again.

If the end of the stream is reached, the minimum oxygen concentration for the run is checked to determine if the minimum is below the minimum for the previous run. If the minimum concentration is still decreasing, begin time is incremented and the stream is

simulated for the next run. If the minimum DO concentration for the present run is greater than the DO concentration for the previous run, then the begin time increment is divided in half and time incrementation is reversed. This minimum-seeking continues until the begin time increment is less than 0.1 hour. The begin time is backed up 0.5 hours and incremented by 0.1 hours for 10 runs. The dissolved oxygen concentration, concentration of leachate, stream water temperature, time and distance are recorded for each Δx and Δt of the ten runs and output in tabular and graphic form on the line printer.

Model II was run for the same simple stream of constant hydraulic dimensions used to test Model I. This was done without the sensitivity search for the minimum oxygen concentration. Instead, begin time was incremented by 5 second increments for 200 runs. The results of Model I were compared to those of Model II. Model II was then expanded to include the sensitivity search for the minimum dissolved oxygen concentration and plotting the profile of the minimum point with respect to time and distance.

A sensitivity analysis of Model II included adding and subtracting slash to determine the loads which would result in critical oxygen concentrations. Later, length of the clearcut was manipulated to examine its effect on dissolved oxygen concentration.

RESULTS

BOD Test Results

Mass Concentration Dependency

The effect of mass concentration on the leaching rate constant (K_1) and the mean potential leachate BOD concentration (L) was examined to determine if varying degrees of organic loading would have a significant effect on leaching and the decay process. Four grams of fresh weight vegetation per liter of stream water was the standard concentration used throughout this study. This test compared the BOD exerted by the standard mass concentration used in previous tests to organic loads two to four times greater.

The mean K_1 and potential leachate BOD concentration (L) for varying organic loads are summarized in Table 5.

The term L is the potential BOD concentration of the leachate averaged over ten days of leaching time for Douglas-fir and western hemlock and 15 days for red alder. The high BOD value for western hemlock at the 16 gm/l concentration may be due to variations in the sample vegetation or an isolated BOD test bottle. Test results on all vegetation in this study consistently exhibited high variation.

An F test was used to test the null hypothesis that the mean values of K_1 and L are not significantly different at the various levels

Table 5. The mean values of the BOD rate constant (K_1) and potential BOD of the leachate (L) of Douglas-fir, western hemlock needles and red alder leaves at different mass concentrations.

	Mass concentration (gm fresh wt/l)			
	4	8	12	16
Douglas-fir				
L (mg/gm dry wt)	24.32	23.94	26.40	27.48
K_1 (1/day)	0.278	0.184	0.315	0.287
Western hemlock				
L (mg/gm dry wt)	17.61	11.46	17.86	31.68
K_1 (1/day)	0.248	0.118	0.165	0.277
Red alder				
L (mg/gm dry wt)	33.79	42.36	38.75	43.83
K_1 (1/day)	0.143	0.106	0.102	0.133

of mass concentration for any of the species tested. The null hypothesis could not be rejected at the 1.0 or 5.0 percent level of significance. This test was for all means, even the ones which seem to be exceptionally different. Each species was tested separately.

Leachate BOD, K_1 and K_4

Leachate BOD and K_4 values were determined from test data fit to equation (26). The K_1 values were from leachate BOD tests fit to equation (25). The results of these determinations are presented in Table 6.

Table 6. Experimentally determined values of L_u, K_1 and K_4 at 20°C.

	L_u (mg O_2/gm, dry wt)	K_1 (1/day)	K_4 (1/day)
Douglas-fir needles	138.99	0.266	0.189
Western hemlock needles	182.59	0.202	0.089
Red alder leaves	226.16	0.121	0.141

These values of L_u compare with ultimate BOD calculation by Ponce (1974) of 110 mg O_2/gm (dry weight) for Douglas-fir needles, 166 mg O_2/gm (dry weight) for western hemlock needles and 286 mg O_2/gm (dry weight) for red alder leaves.

It should be noted that, as shown in Table 6, the leaching rate constant K_4 need not be greater than the decay rate constant K_1. In fact, K_1 and K_4 values are independent of each other.

The amount of leachate leached from the slash with time was determined by adding the BOD concentration of the leachate in the stream water to the oxygen demand exerted for various lengths of time. The results are the values of L_u and K_4. The values of leachate BOD concentration and its associated decay rate constant K_1 were determined from filtrate of vegetation which had been leaching for various lengths of time. The results of these tests are shown in Table 7.

The difference in leachate BOD concentration with time was significantly different at the 1 percent significance level. The leachate concentrations for Douglas-fir and western hemlock vegetation increase by 457 and 449 percent respectively from day 2 to day 10. The leachate concentration of red alder leaves remains somewhat constant from day 2 to day 8 but decreases significantly by day 15.

The decay rate constant of the leachate was found to not significantly vary with time for any of the species tested. This conclusion was reached from an F-test on the means of the K_1 values at various times.

Table 7. The leachate BOD concentration and decay rate constant (K_1) for different points of time after leaching.

	Time of leaching (days)						
	2	4	6	8	9	10	15
Douglas-fir							
L (mg O_2 /gm dry wt)	8.87	7.44	24.61	46.28	-	40.54	-
K_1 (1/day)	0.515	0.319	0.151	0.220	-	0.125	0
Western hemlock							
L (mg O_2 /gm dry wt)	7.29	17.07	13.14	-	27.99	32.76	-
K_1 (1/day)	0.231	0.205	0.178	-	-.212	0.185	-
Red alder							
L (mg O_2 /gm dry wt)	49.97	39.39	38.17	43.20	-	-	27.68
K_1 (1/day)	0.158	0.152	0.100	0.095	-	-	0.101

Temperature Dependency

A preliminary investigation of the effect of streamwater temperature on potential BOD, K_1 and K_4 yielded the results presented in Table 8.

The experimental data correlates very poorly with the prediction equations of Zanoni (1967). This experiment was run with very few sample BOD tests. Such a test made it impossible to statistically analyze results. Further, no regressions were run on the temperature data because an insufficient number of points were available to construct a significant regression equation. In general, however, the potential BOD and leachate decay rate constants tend to increase in value with increasing water temperatures.

Nitrification Results

The results of the 12 nitrification experiments on blended Douglas-fir vegetation are given in Table 9.

The BOD associated with the addition of 375 mg of freeze-dried bacteria appears to be approximately 180 mg O_2/liter. The Douglas-fir vegetation alone accounts for approximately 130 mg O_2/gm (dry weight). The mass concentration of vegetative loading was approximately 0.5 gm (dry weight)/liter. The combination of vegetation and bacteria yielded about 115 mg O_2 demand/liter. This is only

Table 8. The effect of different stream water temperatures on the potential BOD (L_u), the decay rate constant (K_1) and the leaching rate constant (K_4) of Douglas-fir, western hemlock and red alder vegetation.

	Water temperature (°C)			
	15	20	27	35
Douglas-fir				
L_u (mg O_2/gm dry wt)	138.75 (136.70)[a]	138.99	140.98 (149.98)	234.35 (162.55)
K_1 (1/day)	0.184 (0.212)	0.266	0.196 (0.367)	-
K_4 (1/day)	0.235 (0.150)	0.189	0.330 (0.261)	-
Western hemlock				
L_u (mg O_2/gm dry wt)	111.31 (179.58)	182.59	148.91 (197.03)	262.24 (213.54)
K_1 (1/day)	0.294 (0.161)	0.202	0.153 (0.279)	-
K_4 (1/day)	0.120 (0.071)	0.089	0.161 (0.123)	-
Red alder				
L_u	-	226.16	-	309.86 (264.49)

[a]Predicted values using 20°C data and Zanoni's (1967) temperature corrections (equations 60-64).

Table 9. The results of 12 different treatments to determine the effect of nitrification on Douglas-fir vegetation and leachate decay.

Treatment	20-Day BOD exerted (mg O_2/gm)	20-Day change in:				
		NH_4-N (mg/l)	NO_3-N (mg/l)	organic N (mg/l)	total N (mg/l)	Suspended solids (gm/l)
Stream water	20.29[a]	-	-0.019	+0.015	-0.004	-
Stream & bacteria	188.43[a]	-	-	-	-	0.0429
Stream & NH_4Cl	5.21[a]	-1.730	-0.004	-0.030	-1.76	-
Stream, bact & NH_4Cl	176.81[a]	-1.750	-0.004	-1.110	-2.87	+0.0262
Vegetation[b]	129.22	+0.130	-0.044	+0.590	+0.677	+1.521
Vegetation & bacteria[b]	227.53	+0.050	-0.022	+0.214	+0.224	+1.473
Vegetation & NH_4Cl	186.15	-5.270	-0.014	+5.285	+0.001	+1.676
Veg, bact, NH_4Cl[b]	216.10	-5.150	-0.038	+5.410	+0.222	+1.557
Stream & vegetation	130.24	+0.130	-0.060	+0.035	+0.105	+1.879
Stream, veg, bact	257.43	+0.135	-0.060	+0.011	+0.086	+1.574
Stream, veg, NH_4Cl	197.05	-5.442	-0.060	+5.646	+0.144	+1.737
Stream, veg, bact, NH_4Cl	184.76	-5.210	-0.062	+5.462	+0.190	+1.337

[a] mg O_2/l

[b] In distilled H_2O solution

40 mg of O_2 greater than the vegetation alone, indicating an interaction between the bacteria and vegetation.

When ammonium chloride was added to streamwater without vegetation, significant drops in total nitrogen occurred over a 20-day period. When vegetation was present no decreases in total nitrogen were observed.

In treatments where ammonium chloride was added to solutions containing vegetation, nearly all ammonium chloride ammonia was converted to organic nitrogen with no decrease in total nitrogen.

Nitrite nitrogen (NO_2-N) never occurred at detectable levels during the 20 days. By day 10, nitrate nitrogen (NO_3-N) had fallen to levels below detection.

Total suspended solids, however, increased by 1.34 to 1.88 gm without corresponding increases in organic or total nitrogen, whenever vegetation was included in the treatment. The small changes, less than 0.1 ppm, are most likely errors which can be attributed to laboratory technique and limited sensitivity of the tests performed.

Leachate Composition Results

The leachate composition tests were designed to compare the leaching of two leachate components, sugars and phenols, under sterile conditions with the results obtained from the leachate BOD analysis. The leachate data were fit to equation (26). The results of

this analysis are shown in Table 10.

Table 10. The ultimate sugar and phenol concentrations (S_o) and leaching rate constants (K_4) from 90-day data.

	Sugars (mg/l glucose equiv.)	Phenols (mg/l Gallic acid equiv.)
Douglas-fir		
S_o (mg/l)	117.04	15.99
K_4 (1/day)	0.049	0.062
Western hemlock		
S_o (mg/l)	86.22	16.16
K_4 (1/day)	0.060	0.046
Red alder		
S_o (mg/l)	136.35	47.72
K_4 (1/day)	0.880	0.187

The leaching rate constant of western hemlock and red alder sugars is greater than that for phenols. However, the leaching rate constants of red alder are ten times greater than those of western hemlock and Douglas-fir.

The limit of the red alder phenol leachate curve is nearly three times larger than that for Douglas-fir and western hemlock. The sugar leachate limits are roughly the same for the three species. The leachate leaching rate constants for Douglas-fir, in this test, are about 1/4 the magnitude of those rate constants determined experimentally (K_4 = 0.189). The leaching rate constants of western hemlock sugars and phenols are 67 and 52 percent of those determined

from BOD tests (K_4 = 0.089). The leachate rate constant for red alder phenols is only 133 percent of the BOD test result where K_4 was found to be 0.141 per day. The leaching rate of red alder sugars is 7.27 times larger than the BOD test analysis indicates.

Modeling Results

The modeling results are presented as a comparison of the hand-calculated solutions versus the simulated output for various stream conditions and include a sensitivity analysis of model parameters. The hand-calculated solutions range from water temperature prediction to solutions for the cycle of leachate addition, decay, deoxygenation and atmospheric reaeration. Sensitivity analysis was applied in order to evaluate the impact of dispersion, varying the organic loading and manipulating the size of the clearcut on the dissolved oxygen regime of the simulated stream.

The test results of the stream water temperature function of Model I and Model II are given in Table 11. The dates and time of day were chosen to represent a varied, but representative, cross-section of the possible conditions which occur during July.

The model output and hand-calculations agreed consistently within $\pm$ 0.01 ^{o}C. This indicates that the models' stream temperature functions are computationally correct. Computationally correct means that the temperature function for Model I and Model II

Table 11. Hand-calculated stream temperatures, in parentheses, versus temperature estimated by simulation.

Date	Time	Incoming water temp. (°C)	Water temp. at end of 700 ft clearcut (°C)	Water temp. 200 ft below clearcut (°C)
7-1	0600	12.62 (12.62)	12.62 (12.62)	12.53 (12.53)
7-1	1400	14.06 (14.06)	23.38 (23.38)	23.60 (23.60)
7-7	1000	13.30 (13.28)	22.53 (23.52)	22.75 (22.74)
7-7	1800	14.59 (14.59)	14.59 (14.59)	14.50 (14.50)
7-13	1800	14.68 (14.69)	14.68 (14.69)	14.59 (14.60)
7-14	0200	13.94 (13.95)	13.04 (13.06)	12.88 (12.89)
7-19	2200	14.61 (14.61)	13.71 (13.72)	13.55 (13.55)
7-20	0600	12.91 (12.91)	12.91 (12.91)	12.82 (12.82)
7-25	0600	12.95 (12.95)	12.95 (12.95)	12.86 (12.86)
7-25	1400	14.39 (14.40)	23.27 (23.27)	23.49 (23.49)
7-31	1000	13.50 (13.51)	22.18 (22.17)	22.40 (22.39)
7-31	1800	14.81 (14.82)	14.81 (14.82)	14.73 (14.73)

1) follows the designed sequence of computations correctly, 2) calculates these computations correctly and 3) the results approximate the actual conditions of Flynn Creek within 1 percent of published data on streamwater temperature for this stream.

Table 12 shows how the simulated results of the cycle of leachate addition, decay, deoxygenation and atmospheric reaeration compare to hand calculated analytic solutions of a completely mixed body of water at 14°C. The functions tested in these runs are used in both Model I and Model II. In this test situation the body of water is not a stream but a completely mixed container of water. Here again, the results of the finite difference equations are easily within 1 percent of the analytic solutions. Six more runs were made with various combinations of loading. The results of these runs were comparable with the results presented in Table 12. The results of the oxygen balance functions were compared only to hand-calculated results of known analytic solutions of completely mixed containers of water and not to actual stream conditions.

It should be noted that negative dissolved oxygen concentrations were listed in Table 12. This means that the system went anaerobic. When this condition occurs, the analytic equations and Models I and II no longer are applicable since they are aerobic system estimators. During model runs described later, simulation will terminate if DO becomes zero.

Table 12. Results of analytic solutions of leachate concentration (L) and dissolved oxygen concentration (C), in parentheses, versus finite difference model results (K_1 = 0.16; K_2 = 103.01; K_4 = 0.13).

Time since initial loading (days)	Slug load 6,000 ppm		Constant load 800 ppm/day		Leaching addition 20,310 ppm	
	L (ppm)	C (ppm)	L (ppm)	C (ppm)	L (ppm)	C (ppm)
1	5113	2.33	739	9.12	2284	6.74
	(5113)	(2.31)	(739)	(9.12)	(2284)	(6.74)
2	4357	3.51	1368	8.15	3952	4.14
	(4357)	(3.48)	(1369)	(8.14)	(3952)	(4.14)
3	3713	4.51	1904	7.31	5129	2.31
	(3713)	(4.48)	(1906)	(7.31)	(5129)	(2.31)
4	3164	5.36	2361	6.60	5917	1.08
	(3164)	(5.34)	(2364)	(6.60)	(5917)	(1.08)
5	2696	6.09	2751	5.99	6400	0.33
	(2696)	(6.07)	(2753)	(5.99)	(6400)	(0.32)
6	2297	6.71	3083	5.48	6646	-0.06
	(2297)	(6.69)	(3086)	(5.47)	(6646)	(-0.06)
7	1158	7.24	3365	5.02	6710	-0.16
	(1958)	(7.22)	(3369)	(5.03)	(6710)	(-0.16)
8	1668	7.69	3606	4.66	6637	-0.05
	(1668)	(7.67)	(3610)	(4.66)	(6637)	(-0.05)
9	1422	8.07	3812	4.32	6463	0.21
	(1422)	(8.05)	(3815)	(4.34)	(6463)	(0.22)
10	1211	8.40	3987	4.07	6216	0.60
	(1211)	(8.38)	(3991)	(4.06)	(6217)	(0.60)

The next step was to combine all of the corrected model components plus the advection and dispersion functions which make up Model I to simulate leachate concentration in a stream. Model I was then used to simulate 200 ft of a small stream with the following constant hydraulic characteristics: velocity $U = 0.6$ ft/sec, time increment $\Delta t = 16.66$ sec, distance increment $\Delta x = 10$ ft, leachate BOD concentration of the slash $S_o = 20,470$ ppm, cross-sectional area $A = 0.42$ ft^2, stream discharge $Q = 0.25$ cfs, channel slope $S_L = 0.05$ ft/ft and hydraulic radius $R_H = 0.0822$ ft. The estimated longitudinal dispersion rate (D_L) using equation (65) is 0.1485 ft^2/sec. The results of the simulation using Model I for such a stream, where felling began July 1, are listed in Table 13.

Leachate BOD concentration tends to approach an apparent steady state after initial loading. Leachate addition during this small time span is nearly a constant rate ranging from 0.073008 to 0.072996 ppm/sec.

The longitudinal dispersion rate was increased in subsequent runs until $D_L = 0.89$ or about six times the estimated value for this stream from equation (65). All other hydraulic parameters were held constant. The results of this run are shown in Table 14. The value of D_L used in this run was 2.7 times the largest estimated value of D_L for any of the hypothetical stream segments to be used in the final Model II runs.

Table 13. Simulation results of Model I for Δt = 16.66 sec, Δx = 10 ft, U = 0.6 ft/sec, S_o = 20.470 ppm, A = 0.42 ft^2, Q = 0.25 cfs, S_L = 0.05 ft/ft, K_4 = 0.0822 ft and D_L = 0.1485 ft^2/sec.

Time since loading (sec)	Leachate concentration (mg O_2/l)	
	10 ft into the clearcut	200 ft into the clearcut
0	0.0	0.0
100	0.320	4.03
200	0.321	7.84
300	0.321	11.77
350	0.321	13.89
400	0.321	14.31
800	0.321	14.61

Table 14. Results of complete Model I with Δt = 16.66 sec, Δx = 10 ft, U = 0.6 ft/sec, S_o = 20,470 ppm and D_L = 0.89 ft^2/sec.

Time since loading (sec)	Leachate concentration (mg O_2/l)	
	10 ft into the clearcut	200 ft into the clearcut
0	0	0
100	0.428	3.646
200	0.430	7.166
300	0.430	10.69
350	0.430	12.38
400	0.430	13.64
800	0.430	14.62
1000	0.430	14.62

To test the theoretical reasonableness of Model I, longitudinal dispersion was forced to zero and the results of Table 15 compared to hand calculated solutions from equation (11) at 333.3 sec, the time at which a moving cell would reach the 200 ft point of the stream after starting from the top of the clearcut.

Table 15. Results of complete Model I with Δt = 16.66 sec, Δx = 10 ft, U = 0.6 ft/sec, S_o = 20,470 ppm and D_L = 0.0 ft^2/sec.

Time since loading (sec)	Leachate concentration (mg O_2/l)	
	10 ft into the clearcut	200 ft into the clearcut
0	0.0	0.0
100	0.355	4.26
200	0.355	8.52
300	0.355	12.79
350	0.355	14.58

The hand calculated solution of equation (11) is 14.61 mg O_2/l. This compares closely with the steady state levels given in Tables 13-15. These results also indicate that: 1) longitudinal dispersion has the greatest impact when the stream is initially adjusting to the leachate addition and the concentration gradients are steepest; 2) with no longitudinal dispersion, the quasi-steady state concentration is reached much faster than with dispersion and 3) the quasi-steady state concentrations are within 0.27 percent of each other.

The test runs for Model I were run with $\Delta t = \Delta x/U$ to make the pseudo-dispersion effect of advection in equation (43) nearly equal to zero. This was done to separate the advection error from the longitudinal dispersion effect.

Model II was first used to simulate the leachate concentration pattern in the same stream of constant hydraulic dimensions used to examine Model I. Model II utilizes a control volume concept rather than a fixed Δx and Δt. The volume of the smallest segment is computed from values for cross sectional area and length and divided by 10. This volume is termed the control volume. This makes the slice of water or control volume which is routed through the stream a constant volume throughout the stream and allows at least 10 time iterations for each segment of the stream. The value of Δx for a segment is found by dividing the control volume by the average cross sectional area of the segment; Δt is then $\Delta x/U$. For this stream the control volume was 0.42 ft^3, $\Delta x = 1.0$ ft and $\Delta t = 1.666$ sec. The simulation results of Model II are given in Table 16.

The quasi-steady state concentration shown in Table 16 closely agrees with those obtained using Model I (Tables 13-15) and hand calculated evaluations of equation (11). Even though the results were comparable, the runs for Model I required 35.0 computer seconds while a comparable run for Model II required only 6.67 computer seconds. Model II is also more flexible in that the start time for

routing the volume through the stream is left to the the user.

Table 16. Results of Model II with U = 0.6 ft, S_o = 20.4 ppm, Δx = 1.0 ft, Δt = 16.66, A = 0.42 ft^2, S_L = 0.050 ft/ft and R_H = 0.0822 ft.

Time since loading (sec)	Leachate concentration (mg O_2/l)	
	1 ft into the clearcut	200 ft into the clearcut
0	0.0	0.00
350	0.120	14.61
400	0.120	14.61
800	0.120	14.59
1000	0.120	14.58

Model II was then used to simulate the hypothetical stream described in Appendix VI. A clearcut was chosen which would contain 1000 ft of stream or less to the nearest whole stream segment. Thus the clearcut used included 991 ft of the hypothetical stream. This run included the minimum-seeking function for DO. A minimum dissolved oxygen concentration of 5.72 ppm was found to occur at the 5890-foot point and 7.3 hours after initial organic loading. These results appear to be reasonable because the estimated point of minimum oxygen concentration for a simple slug load of 200 ppm from the bottom of the clearcut using equation (18) was calculated to occur at approximately 7200 seconds of travel time below the bottom of the clearcut or at approximately 4300 ft. This approximation would place

the occurrence of the minimum DO concentration 5291 feet below the top of the clearcut. Secondly, these oxygen concentrations are not extraordinary when compared to minimums of 0.60 ppm dissolved oxygen found by Hall and Lantz (1969).

The results of the simulation of this 6000 ft hypothetical stream with 991 ft of clearcut are depicted in Figure 14 and 14a. The DO curve with distance is very similar to the DO sag curve shown in Figure 2. Stream temperature increases through the clearcut, begins to level off but does not drop as it flows on through the forest. This is expected from the results of Brown et al. (1971). Also evident from Figure 14a are the diurnal fluctuations of DO and water temperature with time.

The slash loading of the hypothetical stream was increased from 3.74 to 10.0 lb/ft^2 of stream surface. This resulted in a minimum of 3.7 ppm of DO at the 5890 foot point and 7.3 hours after falling. These results are graphically represented in Figures 15 and 15a. The point and time the minimum oxygen level occurred on the stream were found to be independent of the degree of slash load. The value of the minimum DO level, however, is not independent of slash loading. It was also found that the minimum daily DO was continuously below the "critical oxygen level" (6.0 ppm) for 3 days following felling with a slash load of 3.74 lb/ft^2 and 13 days with 10 lb/ft^2.

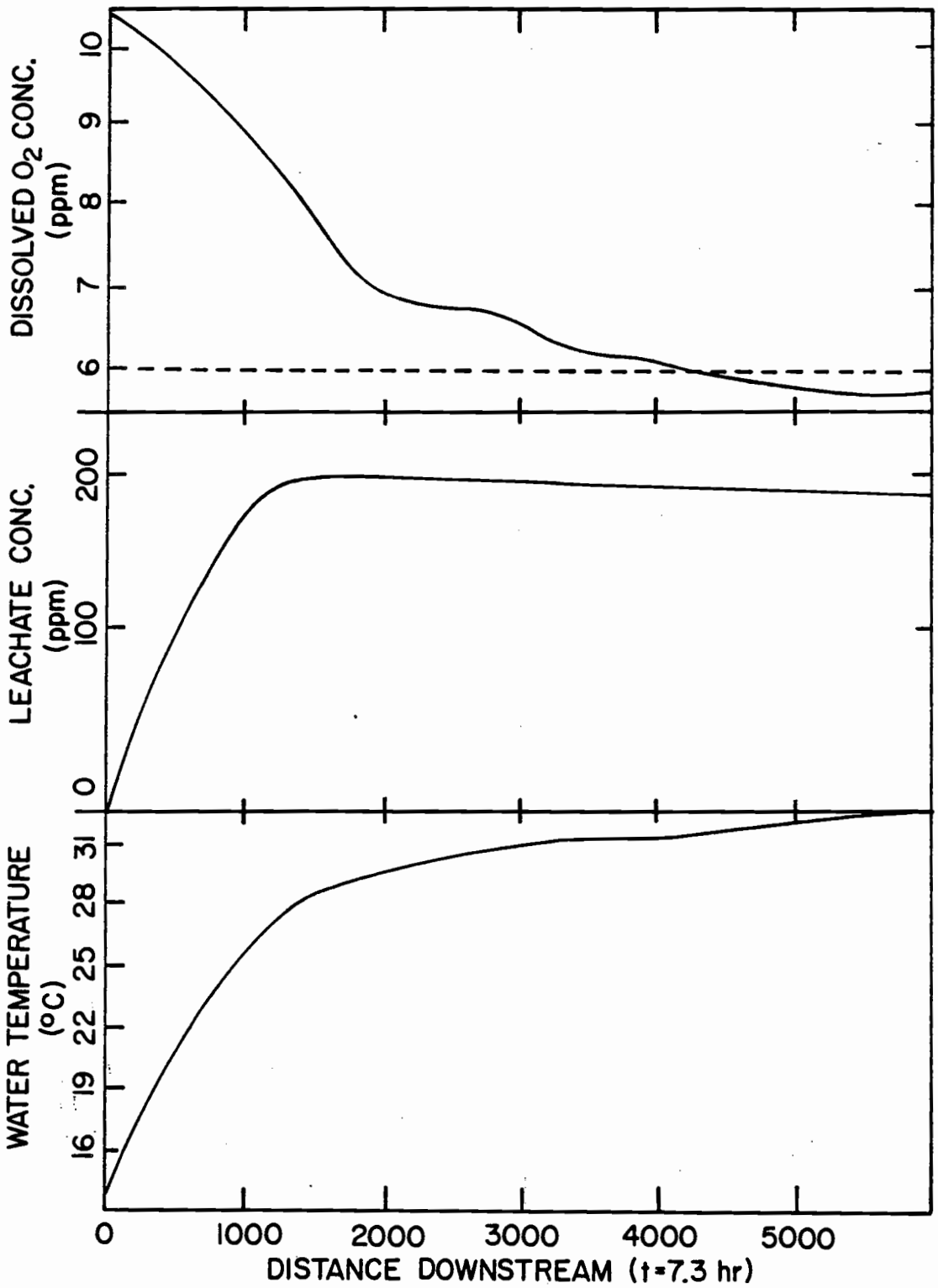


Figure 14. Simulation results of Model II, 991 ft clearcut, 3.74 lb slash/ft^2.

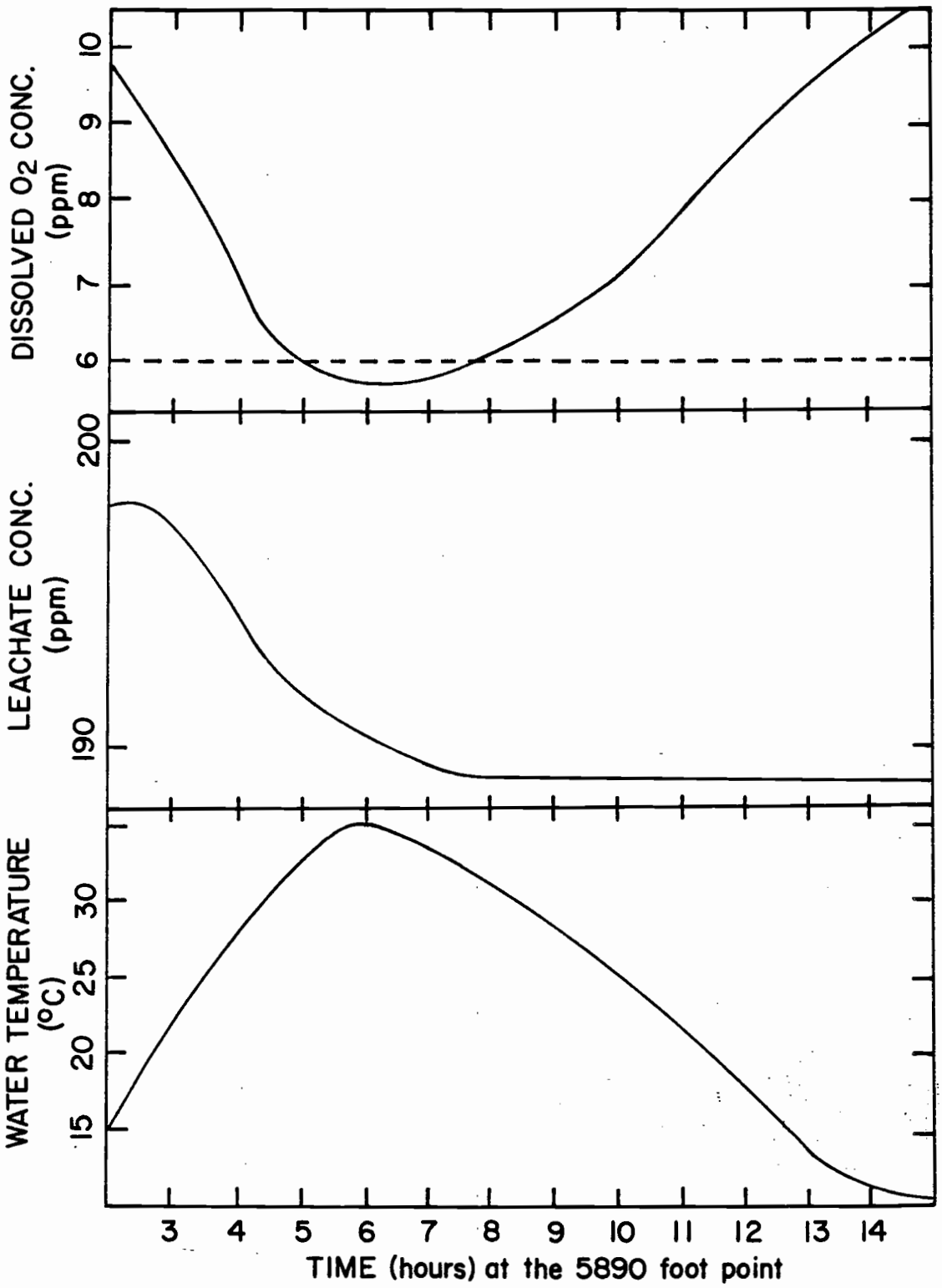


Figure 14a. Simulated results of Model II, 991 ft clearcut, 3.74 lb slash/ft^2.

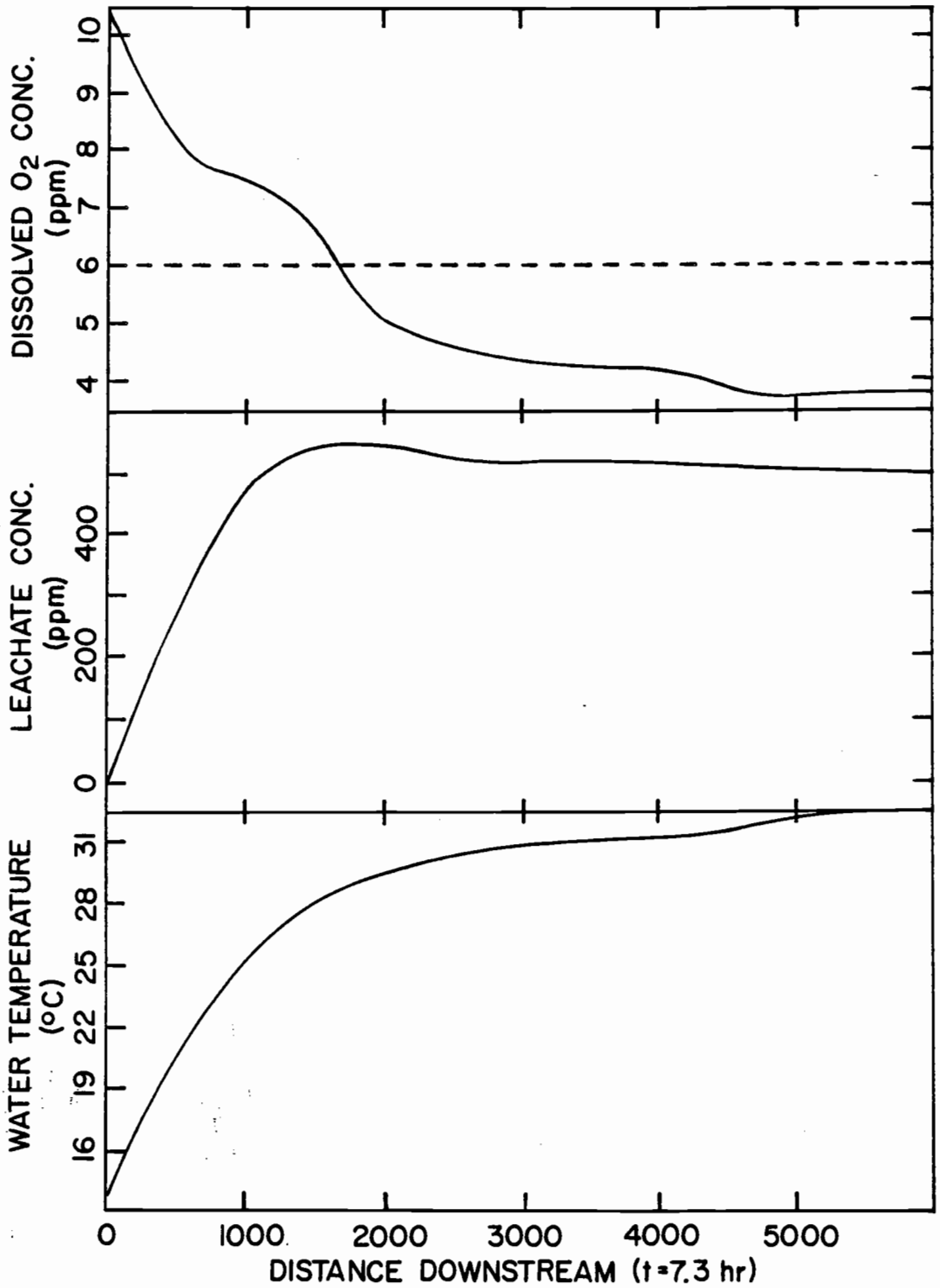


Figure 15. Simulation results of Model II, 991 ft clearcut, 10.0 lb slash/ft^2.

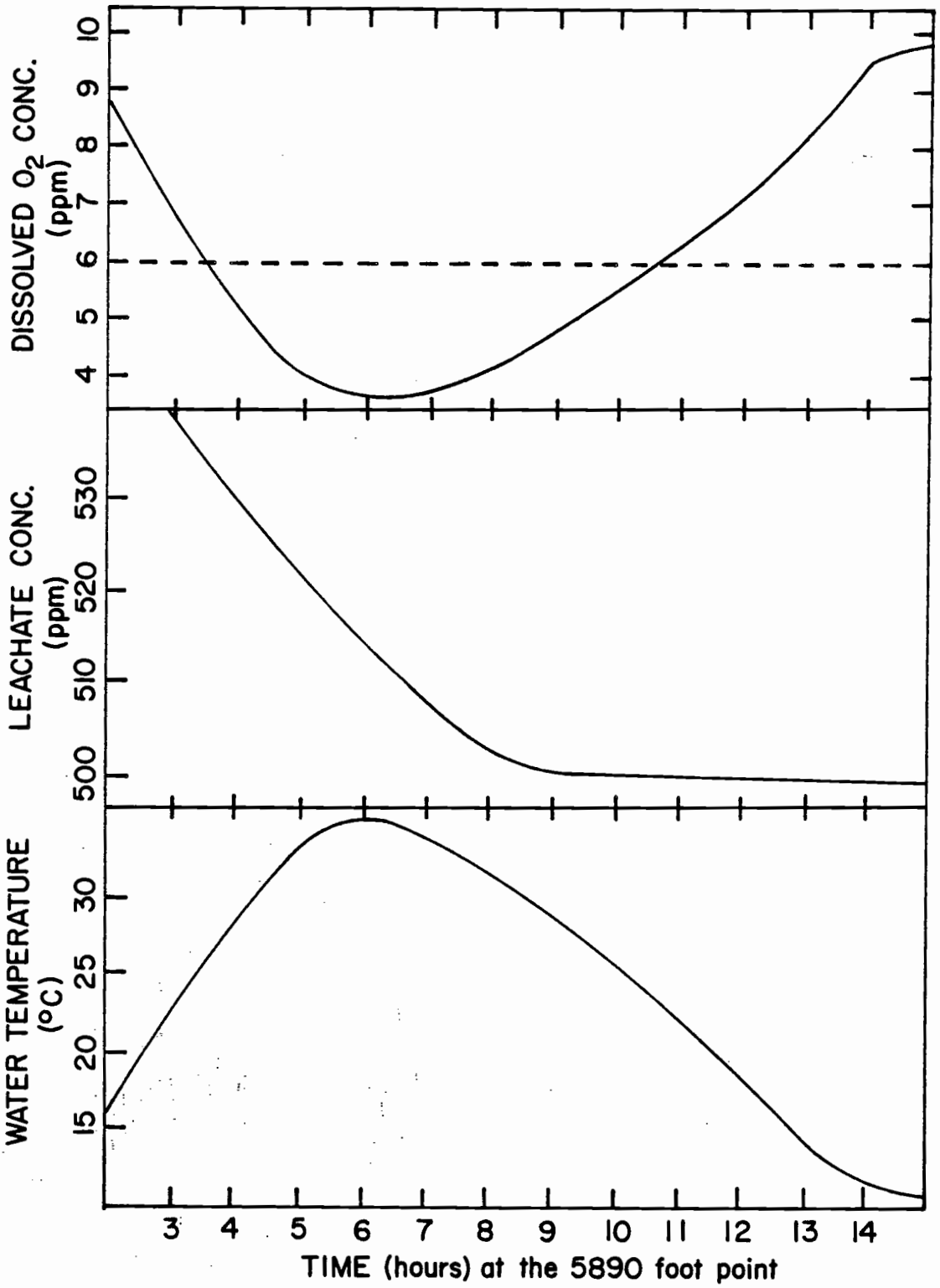


Figure 15a. Simulation results of Model II, 991 ft clearcut, 10.0 lb slash/ft^2.

A third case was modeled where the clearcut was increased from 991 to 1473 ft with 3.74 lb of slash/ft^2. A clearcut length of 1473 ft was chosen because we wished to increase the 991 clearcut length by about 50 percent (to the nearest whole stream segment) in order to examine the effect of this increase on the leachate concentration, DO and temperature. Figures 16 and 16a show that the location of the minimum DO point on the stream, the timing and the magnitude of the minimum DO when compared to Figures 14 and 15 are dependent on clearcut length. The minimum DO level of 4.6 ppm occurred at 6000 ft, 7.4 hours after felling. For this case the minimum daily DO concentration remained below 6.0 ppm for 18 days following clear-cutting. The actual DO sag did not occur on the modeled portion of the stream but is estimated to occur at about the 6500 foot point.

A simulation run was also made on the 991 ft clearcut with 10.16/ft^2 of slash load where the stream temperature was not allowed to increase above the incoming streamwater temperature. This test was designed to isolate the effect of leachate decay and resulting deoxygenation from the impact of large increases in streamwater temperature. The resultant minimum DO concentration was 8.28 ppm. The saturated oxygen concentration for the same point was 10.16 ppm. The minimum point of DO occurred at 7.6 hours at 4891 ft from the top of the clearcut.

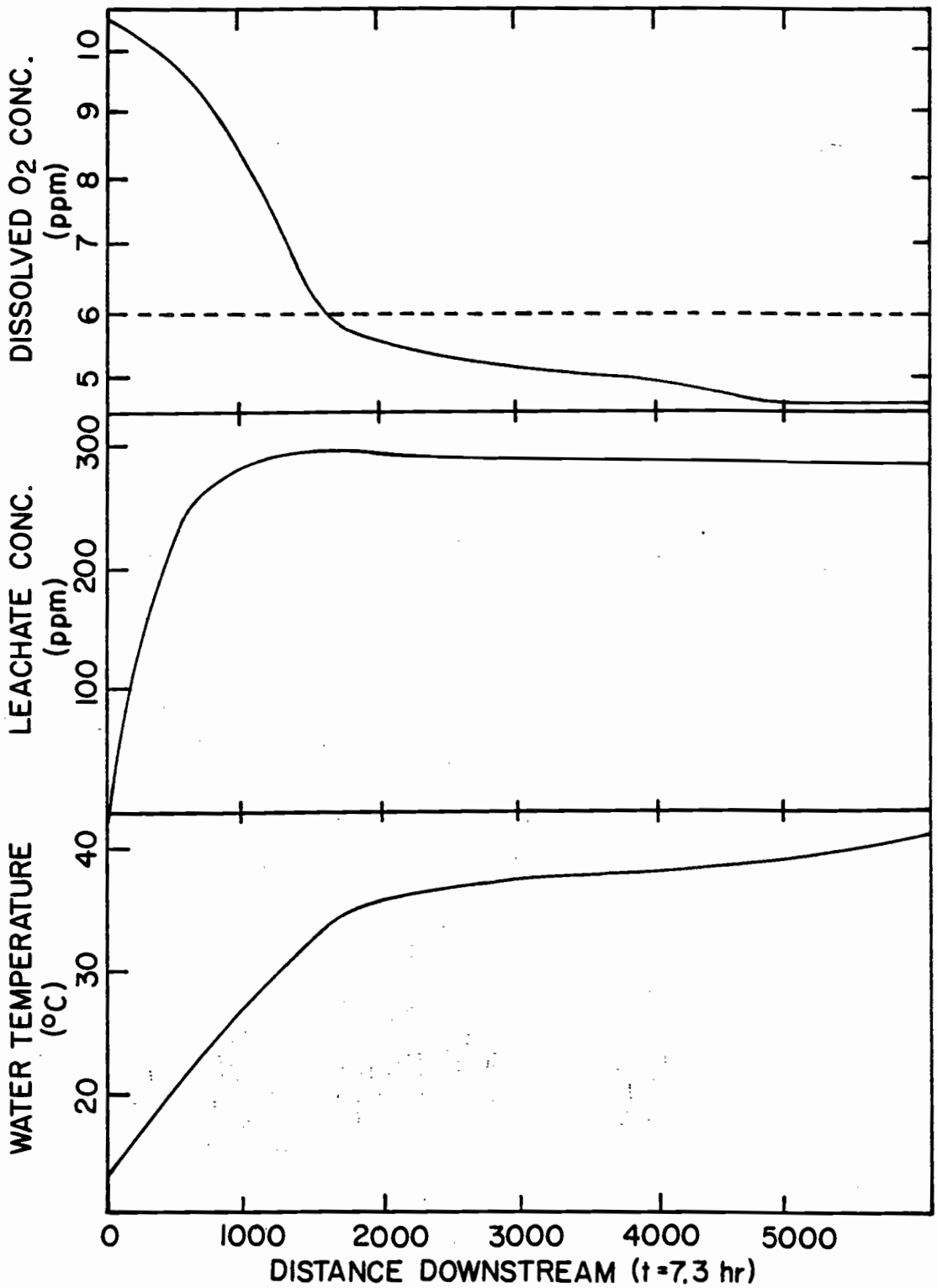


Figure 16. Simulation results of Model II, 1473 ft clearcut, 3.74 lb slash/ft^2.

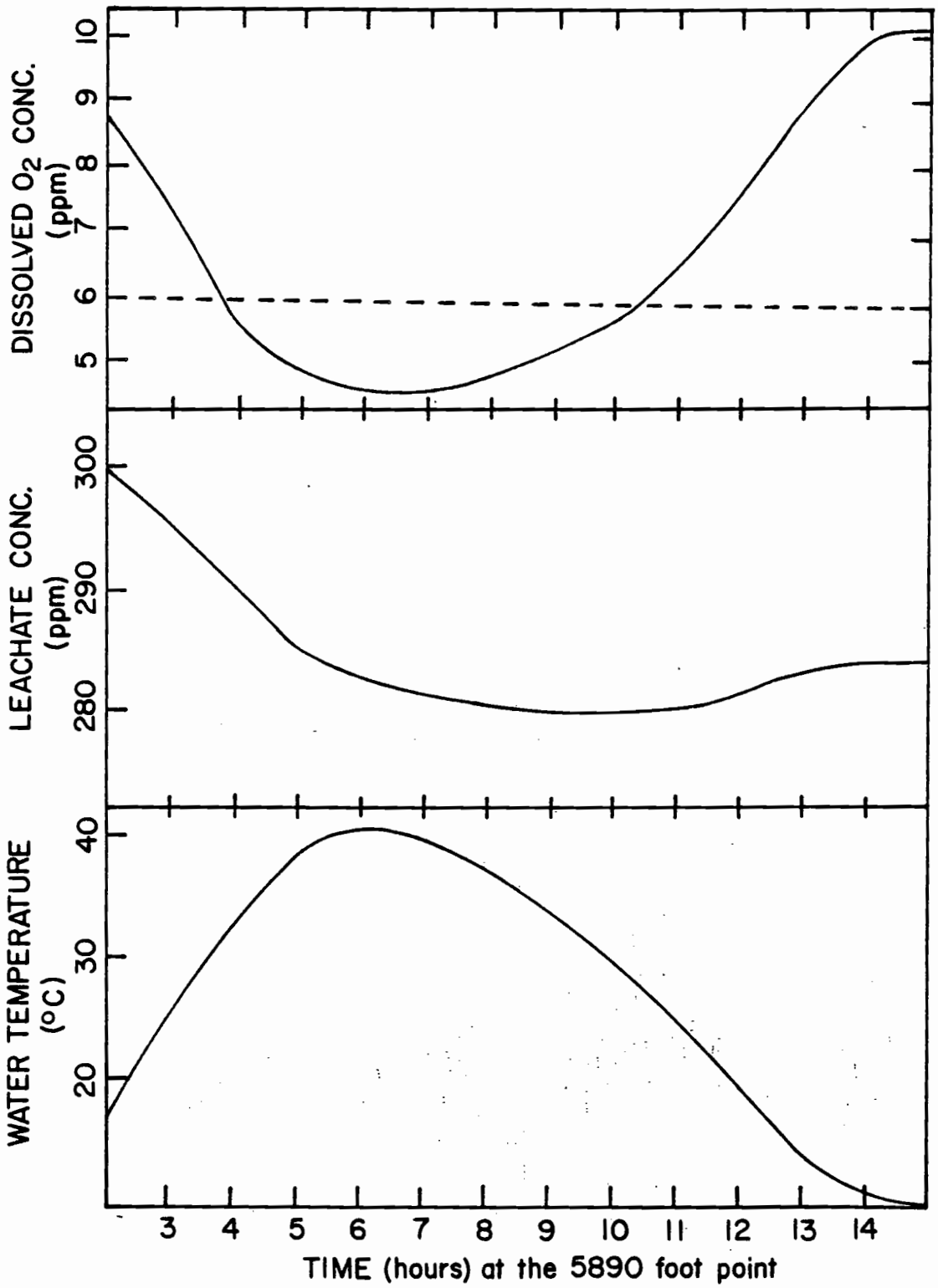


Figure 16a. Simulation results of Model II, 1473 ft clearcut, 3.74 lb slash/ft^2.

DISCUSSION AND CONCLUSIONS

The objectives of this project were to examine the impact of logging debris on dissolved oxygen concentration in small mountain streams. The subject matter has been presented in two major categories; biological testing of the leaching and decay of leachate from logging residue, and modeling the impact of logging debris on the dissolved oxygen balance of small mountain streams. The biological tests were necessary to evaluate the biochemical oxygen demand, the leaching rate and the way in which fine logging debris oxidizes with time in order that oxygen depletion parameters for the oxygen balance model of small mountain streams could be determined.

Biochemical Oxygen Demand Relations

From the results of the mass concentration dependency test presented in Table 5, it appears that potential BOD of leachate (mg O_2/gm dry weight) is not dependent on mass concentration of the slash. Potential BOD of leachate and the decay rate constant (K_1), however, are dependent on species. The leachate decay rate constant (K_1) also is independent of mass concentration.

The experimentally determined values of the potential BOD of leachate (L_u), the decay rate constant (K_1) and the leaching rate constant (K_4) are independent of one another. The value of K_1 may be

greater than K_4 or vice versa, and the values of K_1 and K_4 do not depend on or affect S_o (Table 6).

The tests on leachate concentration with time demonstrate that the difference between the leachate that has decayed and the leachate which remains in the stream water is an important and dynamic parameter of leachate BOD. Table 7 illustrates that the red alder samples had high initial levels of reserved leachate which decreased with time. Douglas-fir and western hemlock needle solutions had low initial leachate reserves which increased during the first 8 to 10 days and then gradually decreased with time. The leachate decay rate constants computed at various times through these tests indicate that the leachate composition is rather consistent with time, or at least the decay characteristics do not vary significantly with time.

The potential BOD of leachate in slash (S_o), values of K_1 and K_4 appear to be dependent on stream water temperature (Table 8). Because only a few complete tests were run at only three different temperatures, it is difficult to draw many conclusions. However, the experimental data seem to deviate significantly from predictions made by using Zanoni's (1967) temperature functions for raw municipal sewage.

It is very apparent from the BOD test results that the Hach manometric technique is very imprecise. Variation as large as 10 to 15 percent were considered to be excellent results. Major problems

were encountered with this technique. Vegetative samples were so small that large variations in leaf and needle composition resulted. It was very difficult to obtain a good seal between the sample bottles and manometers causing erroneously low exerted BOD values. The Hach manometers and bottles are not large enough to run tests for long periods of time without breaking the fragile seal to replenish the oxygen and reset the manometers. And, the manometric technique in its present design does not allow for continuous recording of exerted BOD. Although the Hach manometric apparatus is not well suited to measure BOD of logging debris, the manometric technique is still the best known method and could be redesigned to yield excellent results. Another promising technique is the oxygen gas analyzer which would allow for larger, more representative samples and continuous data recording.

Nitrification

The results of the nitrification experiments in Table 9 indicate that both a readily available source of carbon and nitrogen must be present in the stream water for nitrification to occur. The treatments of stream water and ammonium chloride and stream water, ammonium chloride and bacteria showed drops in total nitrogen in the system. This could be interpreted to mean that the $NH_4^+ \longrightarrow NH_3$ (gas) until an equilibrium was reached. Also, total organic nitrogen

remained constant throughout the 20-day period indicating that the bacteria did not have enough carbonaceous substrate to grow or reproduce. The BOD test further supports this hypothesis in that the treatment of stream water, ammonium chloride and bacteria had a 20-day BOD 170 mg O_2/l greater than the treatment without bacteria. The only way such a BOD could have been exerted is if the bacterial seed began feeding upon itself or was consumed by other organisms.

The other four treatments with ammonium chloride registered increases in organic nitrogen equal to the decreases in ammonia-nitrogen. Of these treatments, the ones seeded with bacteria experienced more rapid conversion of the NH_4^+ to organic nitrogen. The differences in BOD between samples with and without bacteria, which contained vegetation, distilled water, and ammonium chloride, is roughly equal to the BOD predicted for nitrification in Appendix IV. These results indicate that when both vegetation and ammonium were available to bacteria, nitrification did occur.

In natural streams, both available nitrogen and organic matter are limiting for nitrification. With the addition of organic matter or available nitrogen alone, nitrification cannot occur to a significant extent. However, when both are added, nitrification may play a significant role in the BOD and biological activities.

The marked increase in suspended solids with time, even though there was no corresponding decrease in nitrogen forms, may

be explained by the biosynthesis of dissolved solids by organisms with a high carbon to nitrogen ratio.

Thus, the nitrification process in small mountain streams appears to raise more questions of nutrient pathway or cycling and fixation than those relating to oxygen demand.

Leachate Relations

The results of the leachate composition tests (Table 10) indicate that there is significant difference between the leaching rates determined under "natural" and sterile conditions. This difference may be due to the interaction of microorganisms with the leaf and needle structure which may either increase or decrease the leaching rate.

The leaching rates of Douglas-fir and western hemlock sugars and phenols appear to be much smaller under sterile conditions. Conversely, the leaching rate of red alder sugars and phenols are much larger under sterile conditions. This leads to a hypothesis that microorganisms interact with Douglas-fir and western hemlock needles to increase the leaching rate and react with red alder leaves to decrease the leaching rate.

Recommendations for Model Use

The purpose of this modeling effort has been to construct a computer simulation of a small mountain stream which describes the

dissolved oxygen concentration, leachate level and water temperature with time and distance through a clearcut where slash has been allowed to accumulate in the stream channel. This model was designed to be used by researchers to examine the sensitivity of DO concentration to several real-world input parameters including stream channel hydraulic dimension, leaching rates of slash, decay rates of leachate, reaeration rates, and the distribution of slash following clearcut harvesting.

The final product, Model II, which utilizes the Lagrange technique has been shown to be both computationally accurate and flexible. Just as important, Model II is economically feasible to use both for experimentation and research and later as a resource management tool. However, Model II in its present state should not be used to make predictions upon which important management decisions are to be based, since this model has not been fully validated by real-world data.

The output results of Model II appear to be reasonable in that they compare closely to hand-calculated approximations of the timing and location of the DO minimum. Also, the magnitude of the minimum is within reason when compared to the data gathered by Hall and Lantz (1969). Incoming water temperature data compare well with data collected by Brown and Krygier (1970) for Flynn Creek. Further, the input data required are relatively easy to gather. Such data as

cross sectional area, mean velocity, segment length, stream width, channel slope and hydraulic radius may be gathered by a reconnaissance survey of the stream of interest.

Model II has already been used to make preliminary investigations and observe the characteristics of the impact of logging debris on small mountain streams. The first of these tests were to determine the effect of longitudinal dispersion on leachate concentration. As shown in Tables 13 and 14, longitudinal dispersion tends to 1) increase leachate concentrations in a stream segment where concentrations are initially low and decrease the concentration of segments with high leachate levels until the concentration gradient has stabilized, 2) decrease the rate at which the leachate concentration reaches its limits, and 3) have the greatest impact when the concentration gradient is steepest.

Further manipulations of stream data have shown that 1) the timing and magnitude of the point of minimum DO is very dependent on clearcut length and the resulting high stream temperatures, 2) the amount of slash loading is important but does not have nearly as large an impact as clearcut length, 3) DO concentration fluctuates diurnally with stream water temperature and 4) these fluctuations may depress the daily minimum DO concentration to "critical" levels for several days.

A simulated run with the stream water temperature maintained at the incoming water temperature at the top of the clearcut, resulted in a dissolved oxygen deficit of only 1.88 ppm and a minimum DO of 8.28 ppm. This analysis tends to support a hypothesis that the large increases in stream water temperature and its effect on the leachate and dissolved oxygen parameters is largely responsible for the extremely low levels of DO resulting from clearcutting.

It should be noted that the simulated stream was 3 to 6 ft wide, 0.1 to 0.9 ft deep and had a discharge of 0.25 cfs. The change in stream temperature with time and distance was assumed to be linear and a function of stream width, discharge and thermal radiation. As previously mentioned the effects of conduction and nonlinearity were not accounted for in this model. From Figures 14-16, it is evident that conduction and nonlinearty are significant since the simulated maximum temperatures of 35° to 40°C are much larger than the 31°C reported by Brown and Krygier (1970) for Needle Branch. These errors may slightly change the magnitude and location of the maximum oxygen deficit.

The next step in refining this dissolved oxygen model for management use is to validate its predictions against a real-world system. First, a sensitivity analysis must be conducted on all model parameters to determine their relative importance and how closely they must be measured or estimated. These parameters include

everything from estimating the distribution and amount of logging debris in the stream to measuring average mean velocity of a stream segment to predicting the effect of stream temperature on the rate constants K_1, K_2, and K_4.

This type of analysis would lead to a full scale test of the model on a real-world system. Any such testing must eventually include the complete process of harvesting and monitoring the model parameters and comparing predicted to actual results for trial validation. Methods which may be less expensive and not require such a concentrated research effort would consist of two smaller projects. One study should be made to determine the distribution and amount of submerged slash deposited during logging operations. A second project could check the accuracy of the simulated results presented in this study by loading known amounts of finely divided slash to a stream such as Berry Creek.

After validation this model could be used by resource managers to estimate the impact of various timber harvesting techniques and their application on the dissolved oxygen of small mountain streams. Model II may eventually be used to predict 1) if a planned harvest unit which would result in downstream DO levels below 6.0 ppm, 2) how long a clearcut could be made without stream cleanup and still keep DO concentrations above 6.0 ppm or 3) how much fine debris would have to be removed from a clearcut stream to avoid "critical" oxygen levels of less than 6.0 ppm.

A great deal of care and discretion must be applied when using this model or any model to predict the impact of manipulating a real-world system. Only after exhaustive testing and trial use can a model such as Model II become a standard management tool. It should be noted that even though Model II does show great potential for accurately simulating the dissolved oxygen balance of small streams, dissolved oxygen is not the only parameter of water quality nor is it the only threat to fish survival. Serious attention must still be paid to such factors as sedimentation, debris obstruction and stream water temperature.

SUMMARY

The results of this project have explained several areas of concern about the BOD decay process of Douglas-fir needles, western hemlock needles and red alder leaves. Further, this study has provided several conclusions about modeling the impact of fine logging debris on the dissolved oxygen balance of small mountain streams.

1. Leachate BOD (L_u) and the leachate decay rate constant (K_1) are not dependent on mass concentration of fine slash in stream water.
2. The biodegradation of finely divided logging debris is a two-stage process. First, the leachate enters the stream water in proportion to the amount of leachate remaining in the slash. And then this leachate decays at a rate proportionate to the amount in the stream water.
3. The leaching rate constant may be determined by adding the reserve leachate BOD concentration at various times to the BOD exerted up to those times and subsequently fitting these data to an exponential decay function.
4. The values of potential leachate concentration (L_u), the decay rate constant (K_1) and the leaching rate constant (K_4) appear to be dependent on stream water temperature.

5. Leaching samples of needles and leaves can be successfully sterilized using ethylene oxide gas sterilization techniques.

6. Nitrification does not appear to significantly influence the BOD-decay process of finely divided logging debris without the addition of available nitrogen.

7. Longitudinal dispersion is negligible under most small mountain stream conditions.

8. A finite-difference model can be successfully used to predict minimum DO levels following timber harvesting in an accurate, flexible and economical fashion.

9. This model may be used to predict the effects of increased loading and varying clearcut length on the dissolved oxygen concentrations of small streams.

10. The minimum level of DO resulting from the decay of finely divided logging debris is more dependent on length of the clear-cut and the resulting increases in stream temperature than debris concentration. This leads to the conclusion that the most feasible method of preventing low DO levels is by manipulating clearcut length and the use of buffer strips rather than debris removal.

SUGGESTIONS FOR FUTURE RESEARCH

Further research on the BOD and oxygen depletion parameters of logging debris should be carried on in several categories. In some cases existing prediction models and research may be modified or validated such as the prediction of the atmospheric reaeration rate constant or the quantification of 20°C K_1, K_4 and L_u values. Other areas of question may require original or intensified examination such as the effect of water temperature on rate constants and BOD or the distribution and quantity of submerged slash following timber harvest operations.

A reexamination or refinement of the existing atmospheric reaeration rate constant prediction models is needed to reduce the large variations between estimated and measured values and to produce a model which has been validated by tests on real-world systems. The prediction model currently used in Model II has never been validated on small turbulent streams.

Values of K_1, K_4 and L_u may be more accurately quantified by gas sterilization of vegetative samples, leaching them over long periods of time (20 to 90 days), and bacterial seeding of filtered samples to determine K_1 and L_u. The value of K_4 may be determined in a like manner by drawing samples at regular intervals to determine leachate concentration with time. The effect of temperature also may

be examined by repititions of these tests at various incubation temperatures.

More field work is needed to determine how submerged slash is distributed in small mountain streams, both across the width and longitudinally, following logging. The amount of fine slash reaching streams must also be quantified for a range of conditions in order to estimate the leachate concentration in the slash (S_o).

SUGGESTIONS FOR MODEL REFINEMENT AND VALIDATION

Model II has been checked and rechecked against hand-calculated solutions and available data to assure that it is computationally correct. Computationally correct means that the model correctly computes the programmed mathematical expressions and makes these computations in the right sequence as outlined in the flow chart presented in Figure 13. This leaves the correctness of the input data and the theoretical validity of the rate functions to be completed. Sensitivity analysis must be run for all input parameters to determine how accurately and precisely they must be examined to produce the desired accuracy and precision as outlined on p. 36.

The largest obstacle ahead is validation of this model. Model II must be compared to a real-world system and reevaluated. Possibly the entire modeling technique is inappropriate and reconceptualization is necessary. Or, perhaps only particular functions are erroneous and must be fine-tuned or resimulated. The process of model validation, illustrated in Figure 17, is a cycle of research and experimentation resulting in field observations of the real world, data analysis, studying model behavior and program synthesis, questioning existing hypotheses and assumptions, designing new experiments and continuing with more research and experimentation.

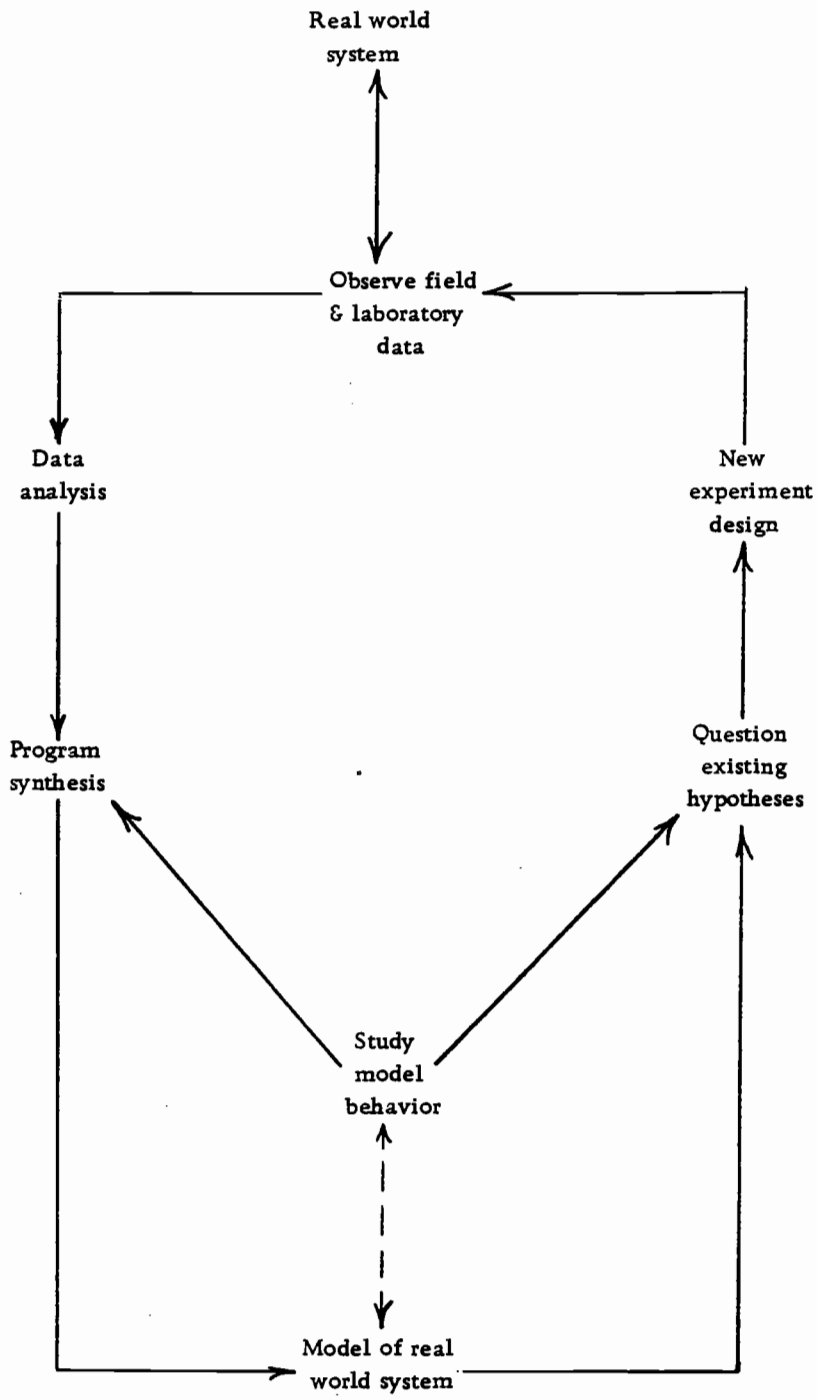


Figure 17. The process of model refinement and validation.

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APPENDICES

L = leachate BOD concentration in the stream, ppm O_2 demand

L_o = initial leachate BOD concentration in the stream, ppm O_2 demand

L_u = leachate BOD concentration per mass of slash, mg O_2/gm dry wt

$L_{(20)}$ = leachate BOD concentration per mass of slash exerted in 20 days, mg O_2/gm dry wt

η = Manning's roughness coefficient

p = rate of constant pollutant addition, ppm O_2 demand/day

P = wetted perimeter, ft

Q = discharge, cfs

Rad = net thermal radiation, BTU/ft^2 · sec

R_H = hydraulic radius, ft

S = leachate BOD concentration in slash per unit volume of stream water, ppm O_2 demand

S_L = slope of the channel bed, ft/ft

S_o = initial leachate BOD concentration in the slash per unit volume of stream water, ppm O_2 demand

t = time, day or sec

T = water temperature, ^oF

U = average mean flow velocity, ft/sec

μ^* = the shear velocity, ft/sec

w = effective volume of dispersion transfer, ft^3

W = weight of slash loading per square foot of stream surface area, lb/ft^2

x = unit of longitudinal length measured in the direction of stream flow, ft

APPENDIX II

DETERMINING SLASH AND LEACHATE CONCENTRATION IN A STREAM AFTER A CONVENTIONAL CLEARCUT

From Lammel (1973), the average width of the streamside zone for the small stream in the clearcut unit he studied was 19 feet wide and the predominent vegetation was western hemlock. The distribution of debris weight by size is given in Table 17.

Table 17. The weight of logging debris by size accumulating in stream channels after a conventional clearcut (after Lammel, 1973).

Debris size (cm)	Weight of debris (tons/100 ft of stream)		
	before felling	after felling	after yarding
0-1	0.049	0.212	0.149
1-3	0.155	0.284	0.218
3-10	0.527	0.928	0.802
greater than 10	5.730	22.194	4.370
Total organic debris	6.461	23.618	5.540

The total amount of fine debris (twigs) less than 3 cm in diameter added following felling was:

$$(0.212-0.049) + (0.284-0.155) = 0.292 \text{ tons/100 ft}$$

The total amount of organic debris added following felling was:

$$(23.618-6.461) = 17.157 \text{ tons/100 ft}$$

Overton et al. (1973) report that the ratio of needles to stem mass is approximately 0.17089. Thus, the amount of needles added to the stream after felling was:

$$(17.157)\ (0.17089) = 2.932 \text{ tons}/100 \text{ ft}$$

The total amount of needles and twigs, then, was

$$2.932 + 0.292 = 3.224 \text{ tons}/100 \text{ ft}$$

This may also be given as:

$$\frac{3.224\ (2,000)}{100 \times 19} = 3.394 \text{ lbs}/\text{ft}^2 \text{ of stream surface}$$

The potential leachate BOD of hemlock slash at 20°C is 0.18259 lb O_2/lb slash. The potential BOD of the computed slash load is:

$$3.394\ (0.18259) = 0.620 \text{ lb } O_2/\text{ft}^2$$

The leachate concentration of the slash may then be computed from the hydraulic dimensions of the stream segment by:

$$S_o = \frac{0.620\ B}{A} \tag{67}$$

or in general,

$$S_o = \frac{L_u \cdot W \cdot B}{A} \tag{68}$$

where S_o is the BOD concentration of leachate in the slash in lb O_2/ft^3, L_u is the potential leachate BOD of the slash in lb O_2/lb of slash, W is the dry weight of slash per ft^2 of stream surface area, B is the average width of the segment and A is the average cross sectional area of the segment.

APPENDIX III

DERIVATION OF LEACHATE BOD CONCENTRATION (L) AND DISSOLVED OXYGEN DEFICIT (D) FOR A COMPLETELY MIXED BODY OF WATER WITH TIME

The change in leachate concentration with time may be given by:

$$\frac{dL}{dt} = -K_1 L + K_4 S_o e^{-K_4 t} \tag{10}$$

Separating the variables yields:

$$\frac{dL}{dt} + K_1 L = K_4 S_o e^{-K_4 t} \tag{69}$$

By multiplying equation (69) by ($e^{K_1 t}$) leads to:

$$e^{K_1 t}\left(\frac{dL}{dt} + K_1 L\right) = K_4 S_o e^{-K_4 t} e^{K_1 t} \tag{70}$$

$$\frac{dL}{dt} e^{K_1 t} + K_1 L e^{K_1 t} = K_4 S_o e^{-K_4 t} e^{K_1 t} \tag{71}$$

The left side of equation (71) may be replaced by $(e^{K_1 t} L)'$:

$$(e^{K_1 t} L)' = K_4 S_o e^{-K_4 t} e^{K_1 t} \tag{72}$$

Integrating both sides and removing constants from the integral gives:

$$e^{K_1 t} L = \int K_4 S_o e^{-K_4 t} e^{K_1 t} dx \tag{73}$$

$$L = e^{-K_1 t} \int K_4 S_o e^{-K_4 t} e^{K_1 t} dx \tag{74}$$

$$L = K_4 S_o e^{-K_1 t} \int e^{t(K_1 - K_4)} dx \tag{75}$$

Integrating the right side results in:

$$L = K_4 S_o e^{-K_1 t} \left[\frac{e^{t(K_1 - K_4)}}{K_1 - K_4} + c \right] \tag{76}$$

$$L = \frac{K_4 S_o e^{-K_4 t}}{K_1 - K_4} + c K_4 S_o e^{-K_1 t} \tag{77}$$

When $t = 0$, $L = L_o$:

$$L_o = \frac{K_4 S_o}{K_1 - K_4} + K_4 S_o c \tag{78}$$

$$C = \frac{L_o}{K_4 S_o} + \frac{1}{K_4 - K_1} \tag{79}$$

Substituting (79) back into (77) leads to equation (11):

$$L = \frac{K_4 S_o e^{-K_4 t}}{K_1 - K_4} + \left(L_o - \frac{K_4 S_o}{K_1 - K_4} \right) e^{-K_1 t} \tag{11}$$

The change in dissolved oxygen concentration with time is given by:

$$\frac{dD}{dt} = -K_2 D + K_1 L \tag{16}$$

Substituting (11) into (16) yields:

$$\frac{dD}{dt} = -K_2 D + \frac{K_1 K_4 S_o}{K_1 - K_4} (e^{-K_4 t} - e^{-K_1 t}) + K_1 L_o e^{-K_1 t} \tag{80}$$

Separating the variables results in:

$$\frac{dD}{dt} + K_2 D = \frac{K_1 K_4 S_o}{K_1 - K_4} (e^{-K_4 t} - e^{-K_1 t}) + K_1 L_o e^{-K_1 t} \tag{81}$$

By multiplying (81) by ($e^{K_2 t}$), leads to:

$$e^{K_2 t}\left(\frac{dD}{dt} + K_2 D\right) = e^{K_2 t} \frac{K_1 K_4 S_o}{K_1 - K_4} (e^{-K_4 t} - e^{K_1 t})$$

$$+ L_o e^{-K_1 t} e^{K_2 t} \tag{82}$$

Upon integrating both sides and removing the constants:

$$e^{K_2 t} D = \int e^{K_2 t} \frac{K_1 K_4 S_o}{K_1 - K_4} (e^{-K_4 t} - e^{-K_1 t})\, dx$$

$$+ \int K_1 L_o e^{-K_1 t} e^{K_2 t}\, dx \tag{83}$$

$$D = \frac{K_1 K_4 S_o}{K_1 - K_4} e^{-K_2 t} \int e^{t(K_2 - K_4)} dx$$

$$- \frac{K_1 K_4 S_o e^{-K_2 t}}{K_1 - K_4} \int e^{t(K_2 - K_1)} dx$$

$$+ K_1 L_o e^{-K_2 t} \int e^{t(K_2 - K_1)} dx \tag{84}$$

Integrating the right side results in:

$$D = \frac{K_1 K_4 S_o}{K_1 - K_4} e^{-K_2 t} \left[\frac{e^{t(K_2 - K_4)}}{K_2 - K_4} + c_1 \right]$$

$$- \frac{K_1 K_4 S_o}{K_1 - K_4} e^{-K_2 t} \left[\frac{e^{t(K_2 - K_1)}}{K_2 - K_1} + c_2 \right]$$

$$+ K_1 L_o e^{-K_2 t} \left[\frac{e^{t(K_2 - K_1)}}{K_2 - K_1} + c_4 \right] \tag{85}$$

$$D = \frac{K_1 K_4 S_o e^{-K_2 t} e^{K_2 t} e^{-K_4 t}}{(K_1 - K_4)\ (K_2 - K_4)} + \frac{c K_1 K_4 S_o e^{-K_2 t}}{(K_1 - K_4)}$$

$$- \frac{K_1 K_4 S_o e^{-K_2 t} e^{K_2 t} e^{-K_1 t}}{(K_1 - K_4)\ (K_2 - K_1)}$$

$$+ \frac{K_1 L_o e^{-K_2 t} e^{K_2 t} e^{-K_1 t}}{(K_2 - K_1)} + K_1 L_o c_4 e^{-K_2 t} \tag{86}$$

where c is equal to $(c_1 - c_2)$.

Combining terms and simplifying leads to:

$$D = \frac{K_1 K_4 S_o e^{-K_4 t}}{(K_1 - K_4)(K_2 - K_4)} + \frac{c K_1 K_4 S_o e^{-K_2 t}}{(K_1 - K_4)}$$

$$- \frac{K_1 K_4 S_o e^{-K_1 t}}{(K_1 - K_4)(K_2 - K_1)} + \frac{K_1 L_o e^{-K_1 t}}{(K_2 - K_1)} + K_1 L_o c_4 e^{-K_2 t} \tag{87}$$

When $t = 0$, $D = D_o$:

$$D_o = \frac{K_1 K_4 S_o}{(K_1-K_4)(K_2-K_4)} + \frac{cK_1K_4S_o}{K_1-K_4} - \frac{K_1 K_4 S_o}{(K_1-K_4)(K_2-K_1)}$$

$$+ \frac{K_1 L_o}{K_2-K_1} + K_1 L_o c_4 \tag{88}$$

$$\frac{cK_1K_4S_o}{K_1-K_4} + K_1L_oc_4 = D_o - \frac{K_1 K_4 S_o}{(K_1-K_4)(K_2-K_4)}$$

$$+ \frac{K_1 K_4 S_o}{(K_1-K_4)(K_2-K_1)}$$

$$- \frac{K_1 L_o}{(K_2-K_1)} \tag{89}$$

Substituting (89) back into (87) leads to equation (22):

$$D = \frac{K_1K_4S_oe^{-K_4t}}{(K_1-K_4)(K_2-K_4)} - \frac{K_1K_4S_oe^{-K_1t}}{(K_1-K_4)(K_2-K_2)} + \frac{K_1L_oe^{K_1t}}{(K_2-K_1)}$$

$$+\left(D_o - \frac{K_1 K_4 S_o}{(K_1-K_4)(K_2-K_4)} + \frac{K_1 K_4 S_o}{(K_1-K_4)(K_2-K_1)}\right.$$

$$\left. - \frac{K_1 L_o}{(K_2-K_1)}\right) e^{-K_2t} \tag{90}$$

$$D = \frac{K_1 K_4 S_o e^{-K_4 t}}{(K_1 - K_4)(K_2 - K_4)} - \frac{K_1 K_4 S_o e^{-K_1 t}}{(K_1 - K_4)(K_2 - K_1)}$$

$$+ \frac{K_1 L_o e^{-K_1 t}}{(K_2 - K_1)} + D_o e^{-K_2 t}$$

$$- \frac{K_1 K_4 S_o e^{-K_2 t}}{(K_1 - K_4)(K_2 - K_4)} + \frac{K_1 K_4 S_o e^{-K_2 t}}{(K_2 - K_1)(K_1 - K_4)}$$

$$- \frac{K_1 L_o e^{-K_2 t}}{(K_2 - K_1)} \tag{91}$$

$$D = \frac{K_1}{(K_2 - K_4)} \left(\frac{K_4 S_o}{K_1 - K_4} \right) (e^{-K_4 t} - e^{-K_2 t}) + D_o e^{-K_2 t}$$

$$+ \frac{K_1}{(K_2 - K_1)} \left(L_o - \frac{K_4 S_o}{K_1 - K_4} \right) (e^{-K_1 t} - e^{-K_2 t}) \tag{22}$$

APPENDIX IV

DETERMINATION OF THE OXYGEN DEMAND OF NITRIFICATION OF AMMONIUM CHLORIDE

If ammonia is completely oxidized by microorganisms, it follows the pathway:

$$4\ NH_4^+\ O_2 \longrightarrow 4\ NH_3 + 2H_2O \qquad (92)$$

$$4\ NH_3 + 6\ O_2 \longrightarrow 4\ NO_2^- + 4\ H^+ + 4\ H_2O \qquad (27)$$

$$4\ NO_2^- + 2\ O_2 + 4\ H^+ \longrightarrow 4\ NO_3^- + 4\ H^+ \qquad (28)$$

or,

$$4\ NH_4^+ + 9\ O_2 \xrightarrow[\text{bacteria}]{\text{enzymes}} 4\ NO_3^- + 6\ H_2O + 4\ H^+ \qquad (93)$$

An oxygen demand of 25 mg O_2/l would be of sufficient magnitude to be detected. Thus, how much NH_4Cl is required to produce such an oxygen demand and what will be the resulting NO_3^- concentration if the NH_4Cl is completely oxidized? Oxygen reacts with ammonium in proportion to the following ratio:

$$\frac{9\ (32)\ \text{oxygen}}{4\ (18)\ \text{ammonium}} = \frac{25}{x}\ \begin{matrix}\text{mg oxygen}\\ \text{mg ammonium}\end{matrix}$$

Thus,

$$x \text{ mg of ammonium} = \frac{4\ (18)\ 25}{9\ (32)} = 6.25 \text{ mg}$$

The addition of 6.25 mg NH_4^+ or 18.56 mg NH_4Cl per liter could result in 25 mg O_2/l of nitrogenous demand if the NH_4^+ is completely

oxidized. This would also produce 21.525 mg/l NO_3^- or 4.86 mg/l NO_3-N.

APPENDIX V

DETERMINATION OF DISPERSION COEFFICIENTS

From Glover (1964), the longitudinal dispersion coefficient is highly dependent on the relationship of the hydraulic radius (R_H) and the shear velocity (μ^*). Glover (1964) has listed several values of $D_L/R_u\,\mu^*$ for varying hydraulic conditions the value given for small open channels and rectangular flumes varies from 5.86 to 13.0.

This leads to:

$$D_L = 13\,R_H\,\mu^* \tag{94}$$

μ^* may be estimated by:

$$\mu^* = \frac{U\sqrt{g}}{C_1} \tag{95}$$

where C_1 is the Chezy coefficient which may in turn be approximated by:

$$C_1 = \frac{1.486}{\eta}\,R_H^{1/6} \tag{96}$$

where η is Manning's roughness coefficient.

Substituting (95) and (96) back into (94) leads to:

$$D_L = \frac{13\,R_H^{5/6}\,U\sqrt{g}\,\eta}{1.486} \tag{77}$$

$$D_L = 49.62\,R_H^{5/6}\,U\,\eta \tag{65}$$

For the simulated stream of Appendix VI, with a Manning's

roughness coefficient of 0.04, the values of D_L range from 0.105 to 0.328 ft^2/sec. The value of Dobbin's (1964) dimensionless parameter to indicate the impact of dispersion, $U^2/(2\ K\ D_L)$ (Table 4), varies from 130.74 to 198.41. The ratio of oxygen or pollutant concentration with dispersion to the concentration without dispersion, R, is about 0.995 to 0.997. Thus, the effect of maximum longitudinal dispersion on concentration is less than 0.5 percent of conditions with no dispersion.

Table 18. Characteristics of each segment of a hypothetical stream with a total length of 1473 feet and a clearcut length of 991 feet.

Segment number	Cross sectional area (sq ft)	Average mean velocity (ft/sec)	Segment length (ft)	Average segment width (ft)	Channel slope (ft/ft)	Hydraulic radius (ft)
			Slash load 3.7241 lb/ft^2			
1	.64103	.39	22.40	3.00481	.00240	.19078
2	.29762	.84	14.40	4.05844	.20920	.08135
3	.50000	.50	10.80	5.76923	.00520	.08471
4	.58140	.43	11.90	6.70841	.02110	.08499
5	.27473	.91	7.30	2.57555	.11570	.10007
6	.39063	.64	16.80	4.18527	.01780	.09021
7	.58140	.43	11.90	6.70841	.02110	.08499
8	.39063	.64	16.80	4.18527	.01780	.09021
9	.39063	.64	16.80	4.18527	.01780	.09021
10	.50000	.50	19.40	9.37500	.05290	.05288
11	.29762	.84	14.40	4.05844	.20920	.07135
12	.14045	1.78	18.40	2.34082	.36760	.05770
13	.43103	.58	17.00	4.97347	.04980	.08440
14	.14045	1.78	18.40	2.34082	.36760	.05770
15	.78125	.32	10.80	5.85938	.00010	.12879
16	2.50000	.10	26.40	4.31034	.00010	.46735
17	.29762	.84	11.40	4.05844	.20920	.07135
18	.50000	.50	10.80	5.76923	.00520	.08471
19	.51020	.49	10.00	5.46647	.02670	.09094
20	.29762	.84	14.40	4.05844	.20920	.07135
21	.64103	.39	22.40	3.00481	.00240	.19087
22	.58140	.43	11.90	6.70841	.02110	.08499
23	.50000	.50	10.80	5.76923	.00520	.08471
24	.29762	.84	11.40	4.05844	.20920	.07135
25	.29762	.84	14.40	4.05844	.20920	.07135
26	.27473	.91	7.30	2.57555	.11570	.10007
27	.64103	.39	19.10	4.18060	.00130	.14492
28	2.50000	.10	26.40	4.31034	.00010	.46735
29	.29762	.84	14.40	4.05844	.20920	.07135
30	.64103	.39	22.40	3.00481	.00240	.19087

Table 18. Continued

Segment number	Cross sectional area (sq ft)	Average mean velocity (ft/sec)	Segment length (ft)	Average segment width (ft)	Channel slope (ft/ft)	Hydraulic radius (ft)
			Slash load 3.7241 lb/ft^2			
31	.29762	.84	11.40	4.05844	.20920	.07135
32	2.50000	.10	26.40	4.31034	.00010	.46735
33	.51020	.49	10.00	5.46647	.02670	.09094
34	.29762	.84	11.40	4.05844	.20920	.07135
35	.58140	.43	11.90	6.70841	.02110	.08499
36	.64103	.39	19.10	4.18060	.00130	.14492
37	.43103	.58	17.00	4.97347	.04980	.08440
38	.39063	.64	16.80	4.18527	.01780	.09021
39	.43103	.58	17.00	4.97347	.04980	.08440
40	.51020	.49	10.00	5.46647	.02670	.09094
41	.29762	.84	11.40	4.05844	.29020	.97135
42	.51020	.49	10.00	5.46647	.02670	.09094
43	.29762	.84	11.40	4.05844	.20920	.07135
44	2.50000	.10	26.40	4.31034	.00010	.46735
45	.14045	1.78	18.40	2.34082	.36760	.05770
46	.39063	.64	16.80	4.18527	.01780	.09021
47	.58140	.43	11.90	6.70841	.02110	.08499
48	.43103	.58	17.00	4.97347	.04980	.08440
49	.78125	.32	10.80	5.85938	.00010	.12879
50	.43103	.58	17.00	4.97347	.04930	.08440
51	.43103	.58	17.00	4.97347	.04980	.08440
52	.29762	.84	11.40	4.05844	.20920	.07135
53	.50000	.50	10.80	5.76923	.00520	.08471
54	.29762	.84	14.40	4.05844	.20920	.07135
55	.39063	.64	16.80	4.18527	.01780	.09021
56	.40323	.62	21.00	5.04032	.04000	.07810
57	.83333	.30	21.30	5.00000	.00510	.15835
58	.29762	.84	14.40	4.05844	.20920	.07135
59	.83333	.30	21.30	5.00000	.00510	.15835
60	.78125	.32	10.80	5.85938	.00010	.12879

Table 18. Continued

Segment number	Cross sectional area (sq ft)	Average mean velocity (ft/sec)	Segment length (ft)	Average segment width (ft)	Channel slope (ft/ft)	Hydraulic radius (ft)
			Slash load 3.7241 lb/ft^2			
61	.58140	.43	11.90	6.70841	.02110	.08499
62	.29762	.84	11.40	4.05844	.20920	.07135
63	.43103	.58	17.00	4.97347	.04980	.08440
64	.29762	.84	11.40	4.05844	.20920	.07135
65	.14045	1.78	18.40	2.34082	.36760	.05770
66	.40323	.62	21.00	5.04032	.04000	.07810
			Slash load 0 lb/ft^2			
67	.29762	.84	11.40	4.05844	.20920	.07135
68	.78125	.32	10.80	5.85938	.00010	.12879
69	.14045	1.78	18.40	2.34082	.36760	.05770
70	.50000	.50	19.40	9.37500	.05290	.05288
71	.64103	.39	22.40	3.00481	.00240	.19087
72	.29762	.84	11.40	4.05844	.20920	.07135
73	.83333	.30	21.30	5.00000	.00510	.15835
74	.40323	.62	21.00	5.04032	.04000	.07810
75	.83333	.30	21.30	5.00000	.00510	.15835
76	.39063	.64	16.80	4.18527	.01780	.09021
77	.29762	.84	11.40	4.05844	.20920	.07135
78	.78125	.32	10.80	5.85938	.00010	.12879
79	.39063	.64	16.80	4.18527	.01780	.09021
80	.51020	.49	10.00	5.46647	.02670	.09094
81	.51020	.49	10.00	5.46647	.02670	.09094
82	.29762	.84	11.40	4.05844	.20920	.07135
83	.50000	.50	10.80	5.76923	.00520	.08471
84	.43103	.58	17.00	4.97347	.04980	.08440
85	.14045	1.78	18.40	2.34082	.36760	.05770
86	.64103	.39	22.40	3.00481	.00240	.19087
87	.29762	.84	14.40	4.05844	.20920	.07135
88	.29762	.84	14.40	4.05844	.20920	.07135
89	.64103	.39	22.40	3.00481	.00240	.19087
90	.29762	.84	14.40	4.05844	.20920	.07135

Table 18. Continued

Segment number	Cross sectional area (sq ft)	Average mean velocity (ft/sec)	Segment length (ft)	Average segment width (ft)	Channel slope (ft/ft)	Hydraulic radius (ft)
			Slash load 0 lb/ft^2			
91	.50000	.50	19.40	9.37500	.05290	.05288
92	2.50000	.10	26.40	4.31034	.00010	.46735
93	.51020	.49	10.00	5.46647	.02670	.09094
94	.50000	.50	10.80	5.76923	.00520	.08471
95	.64103	.39	19.10	4.18060	.00130	.14492
96	2.50000	.10	26.40	4.31034	.00010	.46735

```
      DEFINE COMMON
      COMMON/PT/ PTMAX(5,10),PTMIN(5,10),PTU(5,10),PTL(5,10)
      COMMON/PT/ IPTVAR(5),IPTSPL(5),IPTLET(5,10),IPTCOD(5,10)
      COMMON TIME,DAYT,DX,PL,BEGINTIM,STARTTIM,DXMIN,XTIME
      COMMON TEMP,RAD,J,CONVERT1,CONVERT2,RUNDXMIN,TIMEINC
      COMMON RK20,RK1,RK2,RK1INC,RK2INC,RK2CONST(400),DTEMP
      COMMON V1(10),V2(10),V3(10),DT,FLS(400),DISTMIN,RUNDMIN
      END
      PROGRAM MODEL2
C
C      AUTHORS:  JOHN BERRY AND DONNA WORDEN
C      DATE:  AUGUST 1, 1974
C      INSTITUTION:  OREGON STATE UNIVERSITY, CORVALLIS,
C                    OREGON:  SCHOOL OF FORESTRY
C      COMPUTER INSTALLATION:  CDC 3300 UNDER THE CONTROL
C                    OF THE OS-3 OPERATING SYSTEM.
C      FURTHER REFERANCE:
C        *MODELING THE IMPACT OF LOGGING DEBRIS ON THE
C         DISSOLVED OXYGEN BALANCE OF SMALL MOUTAION STREAMS*
C                                MASTER OF SCIENCE THESIS
C                                BY JOHN D. BERRY, M.S.
C
C
C THIS PROGRAM SIMULATES THE OXYGEN AND POLLUTION CONCENTRATION
C      IN A SMALL MOUTAIN STREAM AFTER LOGGING DEBRIS
C      RESULTING FROM CLEAR CUT HARVESTING OF TREES HAS BEEN
C      ADDED TO THE STREAM.
C  THE MODEL EMPLOYS THE LAGRANGE METHOD OF STUDYING A UNIT
C      VOLUME AS IT TRAVELS FROM THE BEGINNING OF THE CLEAR
C      CUT AREA TO THE END OF THE STREAM PORTION IN THE STUDY.
C
C        *** DICTIONARY OF TERMS ***
C
C     A(J)       AVERAGE CROSS SECTIONAL AREA OF CELLS IN SEGMENT J.
C     ASQ(J)     THE PRODUCT OF THE AVERAGE SURFACE AREA AND
C                TEMPERATURE CONVERSION TERM 1 DIVIDED BY FLOW,
C                FOR CELLS IN SEGMENT J.
C     B          AVERAGE STREAM WIDTH OF SEGMENT.
C     BEGINTIM   THE TIME AT WHICH THE FIRST CELL OF THE
C                STREAM IS SIMULATED, FOR EACH VISITATION TO
C                THE STREAM.
C     DAYT       SECOND OF THE DAY, 6 AM IS ZERO.
C     DIST       THE NUMBER OF FEET FROM FIRST CELL TO THE
C                END OF THE PRESENT CELL.
C     DISTINC    THE NUMBER OF FEET BETWEEN CELLS FOR THE
C                PLOT SAMPLE.
C     DISTMIN    THE DISTANCE AT WHICH MINIMUN OXYGEN CONCEN-
C                TRATION OCCURS WITHIN A VISITATION TO THE STREAM.
C     DT         TIME INCREMENT IN SECONDS.
C     DTEMP      THE CHANGE IN WATER TEMPERATURE BETWEEN CELLS.
C     DX(J)      THE LENGTH OF THE CELLS IN SEGMENT J.
C     NCELLS(J)    THE NUMBER OF CELLS CONTAINED IN SEGMENT J.
C     NSEG       THE NUMBER OF HOMOGENEOUS SEGMENTS IN THE
C                SIMULATED STREAM.
C     NUMRCALL   THE NUMBER OF CELLS BETWEEN RADIATION
C                COMPUTATION UPDATES.
```

```
C       OS          THE MASS OF OXYGEN AT SATURATION.
C       OX          THE MASS OF OXYGEN CURRENTLY IN THE UNIT VOLUME
C       OXMIN       THE MINIMUM OXYGEN CONCENTRATION WITHIN A
C                   VISITATION TO THE STREAM.
C       PL          THE MASS OF POLLUTANT LEACHATE IN THE UNIT VOLUME
C                   AT PRESENT.
C       PLADD1      THE MASS OF POLLUTANT LEACHATE ADDED TO THE
C                   UNIT VOLUME IN THE PREVIOUS TIME INTERVAL AND
C                   PREVIOUS CELL.
C       PLADD2      THE MASS OF POLLUTANT LEACHATE ADDED TO
C                   THE UNIT VOLUME IN THE CURRENT TIME INTERVALL AND
C                   PRESENT CELL.
C       PLLEACH1    THE PROPORTION OF POLLUTANT LEACHATE IN THE
C                   SLASH ADDED TO THE STREAM UP TO THE PREVIOUS TIME
C                   INTERVAL.
C       PLLEACH2    THE PROPORTION OF POLLUTANT LEACHATE IN THE
C                   SLASH ADDED TO THE STREAM UP TO THE PRESENT
C                   TIME INTERVAL.
C       PLO         THE ULTIMATE SLASH DEMAND.
C       PLOLD       THE MASS OF POLLUTANT LEACHATE IN THE UNIT VOLUME
C                   FROM THE PREVIOUS TIME INTERVAL AND
C                   PREVIOUS CELL.
C       PLS(J)      THE TOTAL MASS OF POLLUTANT LEACHATE AVAILABLE
C                   FROM SLASH IN THE CELLS OF SEGMENT J.
C       Q           CONSTANT FLOW RATE OF STREAM.
C       QTOL        THE TOLERANCE ALLOWED BETWEEN FLOW AND
C                   DATA READ IN.
C       R           AVERAGE HYDROLIC RADIUS OF GIVEN SEGMENT.
C       RAD         INCIDENT RADIATION OF SUN TO STREAM.
C       RK1         RATE OF DECAY OF POLLUTANT.
C       RK2         RATE OF OXYGEN REAREATION.
C       RK20        RATE OF POLLUTANT DECAY AT 20 DEGREES CENTIGRADE.
C       RK2CONST(J)  REPRESENTS THE REAREATION FACTOR WHICH
C                   STREAM SEGMENT CHARACTERISTICS CONTRIBUTE.
C       RK4         RATE OF POLLUTANT LEACHING.
C       RUNDMIN     THE DISTANCE AT WHIC RUNOXMIN OCCURS.
C       RUNOXMIN    THE MODEL APPROXIMATION TO THE VALUE OF OXYGEN
C                   AT THE CRITICAL POINT.
C       RUNTIME     THE NUMBER OF SECONDS PASSED SINCE SLASH
C                   WAS ADDED TO STREAM.
C       S           SLOPE OF STREAM SEGMENT.
C       STA         AVERAGE DAILY INCOMING STREAM TEMPERATURE
C                   FOR CANOPIED STREAM.
C       STARTTIM    READ IN AS DAY OF YEAR AT WHICH RUN IS TO BEGIN,
C                   CONVERTED TO SECONDS.
C       TEMP        TEMPERATURE OF WATER IN GIVEN CELL.
C       TIME        SECOND OF THE YEAR AT WHICH THE INFORMATION
C                   IS BEING COLLECTED.
C       U           AVERAGE VELOCITY OF STREAM SEGMENT.
C       W           WEIGHT OF SLASH PERFOOT IN SEGMENT.
C       XSEG(J)     THE LENGTH OF SEGMENT J IN FEET.
C       XTIME       THE APPROXIMATE TIME THAT OXYGEN ATTAINS THE
C                   CRITICAL VALUE.
C       Z           ALTITUDE OF SUN TO GROUND.
C            *** WARNING TO USER ***
C
```

```
C   ARRAYS OF SEGMENT INFORMATION MUST BE DIMENSIONED *AT    157
C     LEAST* AS LARGE AS THE NUMBER OF SEGMENTS.
C
C                 ****************
C
      INCLUDE COMMON
      DIMENSION NOX(400),ASQ(400),A(400),XSEG(400),DX(400)
C
C              INITIALIZE PLOT ROUTINE
C
      CALL PTSYM(1,1H0,1H1,1H2,1H3,1H4,1H5,1H6,1H7,1H8,1H9)
      CALL PTSYM(2,1H0,1H1,1H2,1H3,1H4,1H5,1H6,1H7,1H8,1H9)
      CALL PTSYM(3,1H0,1H1,1H2,1H3,1H4,1H5,1H6,1H7,1H8,1H9)
      CALL PTGRP(1,10,00,0.,0.)
      CALL PTGRP(2,10,10,0.,0.)
      CALL PTGRP(3,10,00,0.,0.)
C
C              INPUT AND CALCULATE CONSTANTS
C
      REWIND 1
      REWIND 2
      READ(1,1000) QTOL,STARTTIM
 1000 FORMAT(5F10.0)
      READ(1,1000)Q,RK20,RK4,PLO
      STARTTIM = STARTTIM* 86400.
      RK4 = RK4 / 86400.
      CONVERT1 = 5./9.
      CONVERT2 = -CONVERT1 * 32.
C
C              DEFINE CONSTANS FOR STREAM SEGMENTS
C
      DIST = 0.
      CLEAR = 0.
      VOL = 1000.
      J = 0
   70 J = J + 1
      READ(2,1000)A(J),U,XSEG(J),B,S,R,W
      IF(EOF(2)) GO TO 100
      IF(ABS(A(J) *U - Q) .GT. QTOL)  STOP 0001
C              NOTE-- STOP IS CONSTANT FLOW IS VIOLATED
C
      ASQ(J) = B * 2.67E-04 / C * CONVERT1
      RK2CONST(J) = 1.157407E-05 * ((5842.798 * U - 1657.) * S +
     1                20.87) / 1.016**20
      PLS(J) = W * B * PLC / A(J)
      IF(VOL .GT. A(J) * XSEG(J)) VOL = A(J) * XSEG(J)
      GO TO 70
  100 NSEG = J - 1
      VOL = VOL * .1
      DT = VOL / Q
      DO 105 J = 1, NSEG
        DX(J) = VOL / A(J)
        NOX(J) = XSEG(J) / DX(J)
        IF(FLOAT(NOX(J)) * DX(J) - XSEG(J) .GT. .5 * DX(J))
     1                          NOX(J) = NOX(J) + 1
        ASQ(J) = ASQ(J) * DX(J)
```

```
        RK2CONST(J) = RK2CONST(J) * DT
        DIST = DIST + FLOAT(NDX(J)) * DX(J)
      IF(PLS(J) .NE. 0.) CLEAR = CLEAR + FLOAT(NDX(J)) * DX(J)
  105 CONTINUE
      NUMRCALL = 180. / DT
      IF(NUMRCALL .EQ. 0) NUMRCALL = 1
      DISTINC = (DIST - 10.) / 9.
      JFLAG = -10
      UNLOG = DIST - CLEAR
      WRITE(4,1030) DIST,CLEAR,UNLOG,VOL,DT
 1030 FORMAT(≠0 STREAM LENGTH:≠,F9.2,≠ CLEAR OUT STREAM:≠,
     :     F9.2,≠  UNLOGGED STREAM:≠,F9.2/
     :     ≠0  UNIT VOLUME:≠,F9.2,≠  TIME INCREMENT:≠,F9.2/)
      RUNOXMIN = 1000.
      CALL RATEINTL
      BEGINTIM = STARTTIM
      TIMEINC = 21600.
      XTIME = 0.
C
C***********SET CONDITIONS BACK TO STREAM TOP CONDITIONS*********
C
  110 DIST = 0.
      DXMIN = 100.
      PLOTDIST = 0.
      TIME = BEGINTIM
C
C             -STA- IS AVERAGE DAILY STREAM TEMPERATURE
C
      STA = 4.5 * COS(TIME * 1.99102E-07 - 3.66519) + 52.5
      DAYT = TIME - STARTTIM - FLOAT((IFIX(TIME - STARTTIM)/
     :       86400) * 86400)
      TDAYT1 = DAYT
      IF(DAYT .LT. 18000.) TDAYT1 = DAYT + 86400.
C
C              COMPUTE INCOMING STREAM TEMPERATURE
C
      IF(DAYT .GE. 18000. .AND. DAYT .LE. 75600.) GO TO 114
C
C              EVENING THRU MORNING HOURS
C
      TEMP = (-1.8 * SIN(TDAYT1*1.09083E-04 - 1.963495) +
     :       STA) * CONVERT1 + CONVERT2
      GO TO 115
C
C              MORNING THRU EVENING HOURS
C
  114 TEMP = (1.7 * SIN(DAYT * 5.4515E-05 - .981748) +
     :       STA) * CONVERT1 + CONVERT2
  115 RUNTIME = TIME -STARTTIM
      PLLEACH2 = EXP(-RK4 * RUNTIME)
      PLADD2 = 0.
      PL = 0.
C
C             OXYGEN IS SATURATED AS ENTERS CLEARCUT
C
      OX = 9.14721E-04 - TEMP * (2.56099E-05 - TEMP *
```

```
     :     (4.98896E-07 - TEMP * 4.85541E-09))
      II = 1
      NRCALL = NUMRCALL
C
C*************SIMULATE STREAM OVER TIME AND DISTANCE **********
C
      DO 300 J = 1, NSEG
C
C             INCREMENT TEMPERATURE PER CELL IN J-TH SEGMENT
C
      DTEMP = RAD * ASQ(J)
      TEMP = TEMP - .5 * DTEMP
C
      CALL RATES
      NX = NDX(J)
      DO 200 I = 1, NX
      DIST = DIST + DX(J)
      TIME = TIME + DT
      DAYT = DAYT + DT
      RUNTIME = RUNTIME + DT
      IF(NRCALL .LT. NUMRCALL) GO TO 125
      NRCALL = 0
C
      TEMP = TEMP + .5 * DTEMP
      IF(DAYT .GE. 86400.) DAYT = DAYT - 86400.
      CALL RADATN
      DTEMP = ASQ(J) * RAD
      TEMP = TEMP -.5 * DTEMP
      CALL RATES
C
  125 NRCALL = NRCALL + 1
      TEMP = TEMP + DTEMP
C
      RK1 = RK1 * RK1INC
      RK2 = RK2 * RK2INC
C
C             COMPUTE POLLUTION IN CELL
C
      PLOLD = PL
      PLLEACH1 = PLLEACH2
      PLLEACH2 = EXP(-RK4*RUNTIME)
C
      PLADD1 = PLADD2
      PLADD2 = PLS(J) * (PLLEACH1 - PLLEACH2)
C
C             ADDITION OF POLLUTANT
C
      PL = PL + .5 * (PLADD1 + PLADD2)
C
C             DECAY OF POLLUTANT
C
      PL = PL * (1. - RK1) / (1. + RK1)
C
C             OXYGEN SATURATION AND CONCENTRATION
C
      OS = 9.14721E-04 - TEMP * (2.56099E-05 - TEMP *
```

```
     1      (4.96896E-07 -TEMP * 4.85541E-09))
C
      OX = (OX - (RK1 * (PLOLD + PL) - RK2 *
     1      (OS + OS - OX))) / (1. + RK2)
C
C              CHECK TO PLOT:
C
      IF(DIST .LT. PLOTDIST) GO TO 150
      V1(II) = OX * 16018.
      V2(II) = PL * 16018.
      V3(II) = TEMP
      II = II +1
      PLOTDIST = PLOTDIST + DISTINC
C
  150 IF(OX .GE. OXMIN) GO TO 200
      OXMIN = OX
      DISTMIN = DIST
  200 CONTINUE
      TEMP = TEMP + .5 * DTEMP
  300 CONTINUE
      CALL OUTPUT
      BEGINTIM = BEGINTIM + TIMEINC
      IF(JFLAG .GE. 0) GO TO 310
      IF(OXMIN .LT. RUNOXMIN) GO TO 305
C              NOTE-- SET JFLAG WHEN THE CONCENTRATION OF OXYGEN
C                APPEARS TO RISE, IE., PASSES THE CRITICAL POINT.
C
      JFLAG = 0
      GO TO 110
C
  305 RUNOXMIN = OXMIN
      RUNDMIN = DISTMIN
      GO TO 110
C
  310 JFLAG = JFLAG + 1
      IF(OXMIN .GE. RUNOXMIN) GO TO 315
C              NOTE-- RESET JFLAG IF HAVE NOT ATTAINED THE
C                CRITICAL POINT.
      JFLAG = 0
      RUNOXMIN = OXMIN
      RUNDMIN = DISTMIN
      IF(TIMEINC .EQ. 360.) XTIME = BEGINTIM
      GO TO 110
C
  315 IF(JFLAG .LT. 9) GO TO 110
      IF(XTIME .NE. 0. .AND. OXMIN .GE. 3.74578E-04)
     1          JFLAG = 8
      BEGINTIM = BEGINTIM - 12. * TIMEINC
      CALL PLOTEND
      WRITE(4,1050)
 1050 FORMAT(1H1)
      JFLAG = 0
      RUNOXMIN = 100.
      GO TO 110
      END
      SUBROUTINE PLOTEND
```

```
      INCLUDE COMMON
C
C             PRINT TITLES AND PLOTS AND
C              ADJUST BEGIN TIME AND TIME INCREMENT TO NARROW THE
C                TIME INTERVAL IN THE SEARCH THE CRITICAL POINT.
C
C              NOTE-- SIX HOURS IS 21600. SECONDS,
C                     ONE HOURS IS  3600. SECONDS,
C                     SIX MINUTES IS 360. SECONDS.
C                THEREFORE THE ACTUAL TIME OXYGEN ATTAINS THE
C                CRITICAL POINT IS WITHIN SIX MINUTES OF RUN
C                FINDINGS.
C
      OUTOX = RUNOXMIN * 16018.
      IF(TIMEINC .GT. 360.) GO TO 10
      WRITE(3,2013) OUTOX,RUNDMIN
 2013 FORMAT(1H1,T34,*DISSOLVED OXYGEN (PPM) CRITICAL VALUE *,
     :       *WITHIN SIX MINUTES *,E10.3,* AT *,E10.3,* FEET*/)
      TIMEINC = 86400.
      BEGINTIM = XTIME
      GO TO 50
C
   10 IF(TIMEINC .GT. 3600.) GO TO 20
      WRITE(3,2016) OUTOX,RUNDMIN
 2016 FORMAT(1H1,T36,*DISSOLVED OXYGEN (PPM) CRITICAL VALUE *,
     :       *WITHIN ONE HOUR *,E10.3,* AT *,E10.3,* FEET*/)
      TIMEINC = 360.
      GO TO 50
C
   20 IF(TIMEINC .GT. 21600.) GO TO 30
      WRITE(3,2018) OUTOX,RUNDMIN
 2018 FORMAT(1H1,T36,*DISSOLVED OXYGEN (PPM) CRITICAL VALUE *,
     :       *WITHIN SIX HOURS *,E10.3,* AT *,E10.3,* FEET*/)
      TIMEINC = 3600.
      GO TO 50
C
   30 WRITE(3,2019)
 2019 FORMAT(1H1,T30,*DISSOLVED OXYGEN (PPM) AT DAY INTERVALS *,
     :       *UNTIL CONCENTRATION EXCEEDS 5 PPM*)
      TIMEINC = - 1
C
   50 CALL PTOPT(1,3,100)
      WRITE(3,2020)
 2020 FORMAT(1H1,T55,*POLLUTANT IN STREAM (PPM)*)
      CALL PTOPT(2,3,100)
      WRITE(3,2030)
 2030 FORMAT(1H1,T59,*TEMPERATURE (C.)*)
      CALL PTOPT(3,3,100)
      IF(TIMEINC .LT. 0.) CALL EXIT
      REWIND 45
      REWIND 46
      REWIND 47
      IPTSPL(1) = IPTSPL(2) = IPTSPL(3) = 0
      RETURN
      END
      SUBROUTINE OUTPUT
```

```
      INCLUDE COMMON
      DIMENSION LET(5)
      LET(1) = 3HO
      LET(2) = 3HPL
      LET(3) = 3HTMP
      LET(4) = 3HM D
      TIMEOUT = (BEGINTIM - STARTTIM)/3600.
      OXM = OXMIN * 16018.
C
      WRITE(4,2000)TIMEOUT,LET(1),(V1(J),J=1,9,2),
     !                     LET(2),(V2(J),J=1,9,2),
     !                     LET(3),(V3(J),J=1,9,2),
     !                     LET(4),OXM,DISTMIN
 2000 FORMAT(* *,F8.1/(3X,A3,5E10.3))
C
      CALL PTSAV(1,TIMEOUT,V1(1),V1(2),V1(3),V1(4),V1(5),
     !                     V1(6),V1(7),V1(8),V1(9),V1(10))
C
      CALL PTSAV(2,TIMEOUT,V2(1),V2(2),V2(3),V2(4),V2(5),
     !                     V2(6),V2(7),V2(8),V2(9),V2(10))
C
      CALL PTSAV(3,TIMEOUT,V3(1),V3(2),V3(3),V3(4),V3(5),
     !                     V3(6),V3(7),V3(8),V3(9),V3(10))
C
      CALL PTEXP(1,2,TIMEOUT)
      CALL PTEXP(2,2,TIMEOUT)
      CALL PTEXP(3,2,TIMEOUT)
      RETURN
      END
      SUBROUTINE RATES
      INCLUDE COMMON
C
C        COMPUTE POLLUTION DECAY AND REAERATION RATES.
C
      IF (TEMP .GT. 15.) GO TO 192
      RK1 = RK20A * 1.126**TEMP
      RK1INC = 1.126**DTEMP
      GO TO 194
C
  192 IF (TEMP .GT. 32.) GO TO 193
      RK1 = RK20B * 1.047**TEMP
      RK1INC = 1.047**DTEMP
      GO TO 194
C
  193 RK1 = RK20C * .985**TEMP
      RK1INC = .985**DTEMP
  194 RK2 = RK2CONST(1) * 1.016**TEMP
      RK2INC = 1.016**DTEMP
      RETURN
C
      ENTRY RATEINTL
C
C             THIS PORTION OF -RATES- INITIALIZES TEMPERATURE
C               RANGE FACTORS FOR DECAY RATE COMPUTATIONS.
C
      RK20 = RK20 * .5  * DT
```

```
      RK20A = RK20*9.21296E-06/1.126**15
      RK20B = RK20*1.157407E-05/1.047**20
      RK20C = RK20*2.05787E-05/.985**32
      RETURN
      END
      SUBROUTINE RADATN
C
C        THIS SUBROUTINE DETERMINES THE
C        RADIATION ON THE CLEARCUT OR CANOPIED STREAM
C
      INCLUDE COMMON
      TDAYT=DAYT
      IF(DAYT.LT.10800.)TDAYT=TDAYT+86400.
      IF(PLS(J).EQ.0.) GO TO 70
C
C        COMPUTE RADIATION FOR CLEARCUT AREA
C
      IF ( DAYT .GE. 43200. )  GO TO 60
C
C        RADIATION FOR CLEARCUT IN DAY TIME, Z IS ALTITUDE OF THE SUN
C
      Z = 25. * COS(TIME * 1.99102E-07 - 2.976) + 45.
      RAD = (.07 * Z + .3) * SIN(DAYT * 7.272205E-05)
      RETURN
C
C        RADIATION FOR CLEARCUT AT NIGHT
C
   60 RAD=.5 * SIN(DAYT* 7.272205E-05)
      RETURN
C
C        COMPUTE RADIATION FOR SHADED STREAM
C
   70 IF ( DAYT .GT. 10800. .AND. DAYT .LT. 32400.) GO TO 80
C
C        COMPUTE RADIATION FOR SHADED STREAM AT EVENING THUR MORNING
C
      RAD = .3 * COS(4.84813E-05 * TDAYT)
      RETURN
C
C              MORNING THRU EVENING
C
   80 RAD = -.75 * COS(1.45444E-04 *DAYT)
      RETURN
      END
```