Dissolved Oxygen in the Tualatin River, Oregon, During Winter Flow Conditions, 1991 and 1992

Water-Supply Paper 2465–A

Prepared in cooperation with
Unified Sewerage Agency of
Washington County

U.S. Department of the Interior
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By VALERIE J. KELLY

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CONVERSION FACTORS

[SI = International System of units, a modernized metric system of measurement]

Multiply	By	To obtain
A. Factors for converting SI metric units to inch/pound units		
	Length	
centimeter (cm)	0.3937	inch (in)
millimeter (mm)	0.03937	inch
meter (m)	3.281	foot (ft)
	1.094	yard (yd)
	Volume	
milliliter (mL)	0.001057	quart (qt)
liter (L)	1.057	quart
liter	0.2642	gallon (gal)
	Mass	
gram (g)	0.03527	ounce (oz avoirdupois)
kilogram (kg)	2.205	pound (lb avoirdupois)
	Temperature	
degree Celsius (°C)	Temp degree F = 1.8 (Temp degree C) + 32	degree Fahrenheit (°F)
B. Factor for converting inch/pound units to SI metric units.		
	Volume per unit time (flow)	
cubic foot per second (ft^3/s)	0.02832	meter per second (m^3/s)
acre	4,047	cubic meter (m^3)
C. Factors for converting SI metric units to other miscellaneous units		
	Concentration, in water	
milligrams per liter (mg/L)	1	parts per million (ppm)
nanograms per liter (ng/L)	1	parts per trillion (ppt)
nanograms per liter	0.000001	parts per million
	Concentration, in bed sediment	
micrograms per kilogram (µg/kg)	1	parts per billion (ppb)
micrograms per kilogram (µg/kg)	0.001	parts per million
	Concentration, in tissue	
micrograms per gram (µg/g)	1	parts per million

Electrical conductivity is measured as specific electrical conductance, in units of microsiemens per centimeter (µS/cm) at 25 degrees Celsius.

Dissolved Oxygen in the Tualatin River, Oregon, during Winter Flow Conditions, 1991 and 1992

By Valerie J. Kelly

Abstract

During the winter season, defined as November 1 to April 30, four wastewater treatment plants in the Tualatin River Basin discharge about 10,000 to 15,000 pounds per day of biochemical oxygen demand, both carbonaceous and nitrogenous, to the river. These loads often increase substantially during storms when streamflow is also increased. Another issue concerns the discharge of about 2,000 pounds per day of ammonia from the treatment plants during the early winter season, when streamflow is frequently less than the average winter flow. This study focused on the capacity of the river to assimilate oxygen-demanding loads under winter streamflow conditions during the 1992 water year, with an emphasis on peak-flow conditions in the river, as well as winter base-flow conditions during November 1992.

Concentrations of dissolved oxygen throughout the main stem during the winter remained high relative to the State standard for Oregon of 6 milligrams per liter, except during periods of streamflow less than 500 cubic feet per second and temperatures greater than 10 degrees Celsius (°C). The most important factors controlling oxygen consumption during winter low-flow conditions (streamflow from 500 to 2,000 cubic feet per second) were carbonaceous biochemical oxygen demand from wastewater treatment plants and tributaries, and input of oxygen-depleted waters from tributaries. In spite of increased ammonia loads, nitrification was not significant because of the cold water temperature and reduction in residence time. During peak-flow conditions (streamflow greater than 2,000 cubic feet per second), oxygen depletion was negligible; the effect of increased oxygen-demanding loads was minimized by the greatly reduced travel time and increased dilution associated with the increased streamflow.

During winter low-flow conditions, the combined mean load of carbonaceous biochemical oxygen demand from the wastewater treatment plants was essentially equivalent to the combined mean load from the tributaries. The total mean load increased by twofold during peak-flow conditions, relative to winter low-flow conditions. This increase in carbonaceous biochemical oxygen demand was exclusively from the tributaries, because the combined mean load from the wastewater treatment plants actually decreased.

During the base-flow period in November 1992 (streamflow less than 300 cubic feet per second, conditions that would be expected to occur once every 3 to 4 years), concentrations of dissolved oxygen at river mile 3.4 consistently fell below 6 milligrams per liter. A hydrodynamic water-quality model was used to identify the processes depleting dissolved oxygen, including sediment oxygen demand, nitrification, and carbonaceous biochemical oxygen demand. Sediment oxygen demand was the most significant sink for oxygen during this period, accounting for more than 50 percent of the oxygen demand. Nitrification was important, but not as significant, accounting for nearly 30 percent of the oxygen demand. The effect of carbonaceous biochemical oxygen demand, about half of which was from the

wastewater treatment plants, was slight due to the low rate of in-river decay. These results suggest that further reductions in loads of carbonaceous biochemical oxygen demand from the treatment plants during the base-flow period would have little effect on oxygen concentrations in the river.

Hypothetical scenarios were simulated to evaluate the effect of different loading strategies employed by the treatment plants in the basin during winter base-flow conditions. Streamflow and temperature were determined to be significant factors governing the capacity of the river to assimilate oxygen-demanding loads. For the range of loads simulated, at water temperatures between 12 to 13°C, streamflows near 350 cubic feet per second are required to maintain dissolved oxygen concentrations in the river above the State standard. When water temperatures increase to 18°C, streamflows must increase to 500 cubic feet per second to maintain concentrations above the standard.

INTRODUCTION

The Tualatin River Basin (fig. 1) is located in the rapidly growing Portland metropolitan area of northwestern Oregon. Wastewater treatment for the population of approximately 300,000 people living in the basin is provided by the Unified Sewerage Agency (USA) of Washington County, Oregon, which maintains four major wastewater treatment plants (WWTP) in the basin. During the summer season, from May through October, excessive growth of phytoplankton and depleted dissolved oxygen (DO) concentrations in the river have historically been a problem, especially during periods of low-flow and high-light conditions. As a consequence, the effluent from the WWTPs is carefully monitored and regulated during this period. Water quality in the river during the winter season, from November through April, has not been well studied. Little is known about the effect of WWTP discharges into the Tualatin River during these winter months.

Several characteristics inherent to the winter season might play important roles in determining water-quality conditions in the river. In particular, large storms, characteristic of the area during the winter, may have an effect on concentrations of DO in the river. During these storms, loads of biochemical oxygen demand (BOD), including carbonaceous and nitrogenous oxygen demand, from the WWTPs sometimes increase greatly; the capacity of the river to assimilate the increased oxygen demand during these storms is unknown. In addition, the WWTPs discharge large quantities of ammonia as nitrogen (ammonia) relative to other sources to the river throughout the winter season. Streamflow in the river during the early winter season, however, is frequently less than usual winter streamflow, and is comparable to base flow. Furthermore, DO concentrations less than the State standard for Oregon of 6 mg/L (milligrams per liter) have been observed in the lower Tualatin River during November. The increased ammonia load may exceed the assimilative capacity of the river at base flow, causing oxygen depletion through the process of nitrification.

Because continued population growth in the Tualatin Basin is anticipated, USA will require information concerning the effect of increased carbonaceous biochemical oxygen demand (CBOD), ammonia, and other loads during the winter season in order to make effective management decisions regarding wastewater treatment in the basin. This study was funded in 1991 by a cooperative agreement between USA and the U.S. Geological Survey (USGS) in an effort to address these concerns regarding winter flow conditions in the Tualatin River.

Purpose and Scope

The purpose of this report is to provide an understanding of the capacity of the Tualatin River to assimilate oxygen-demanding material during the winter season, and to provide a basis for evaluating the relative significance of the various factors that affect oxygen concentrations in the river during this period. Specifically, the report is intended to (1) describe the ambient conditions in the Tualatin River and its major tributaries from November 1991 to April 1992 and during November 1992, with an emphasis on documenting conditions upstream and downstream from the major WWTP and nonpoint or tributary sources of CBOD and ammonia, (2) present an analysis of the major processes controlling DO in the river and the effect of streamflow and temperature on those processes, (3) evaluate the effects of various WWTP loading scenarios on DO concentrations, especially during the early winter base-flow period, as

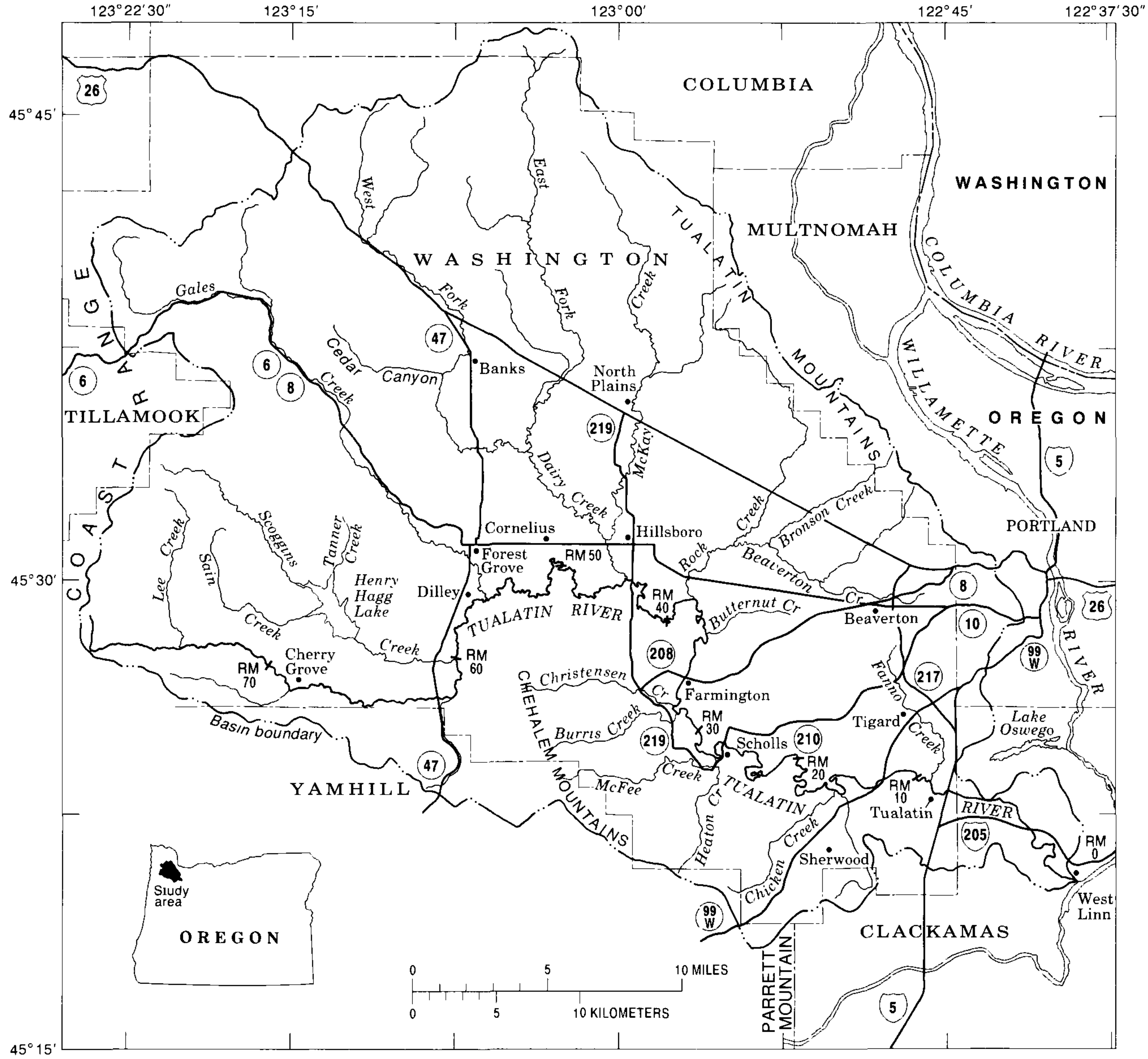


Figure 1. Drainage area of Tualatin River Basin in northwestern Oregon.

an aid to management decisions in the basin, and (4) provide background data on other water-quality parameters, especially total suspended solids (TSS), which provide a context for the analysis of water quality during the summer season, and which are valuable to management of water quality.

The analysis and discussion presented here are partly derived from historical hydrologic and water-quality data, and data collected during the Summer Tualatin River Program, an ongoing USGS study of water quality in the Tualatin River during summer base-flow conditions. Primarily, however, this report focuses on data collected during a winter sampling program that included the following elements:

- Weekly monitoring of the Tualatin River and its major tributaries during November to April, 1991–92, with daily sampling during the major storm of the season, and near-daily analyses of effluent loading from the four WWTPs; and
- Intensive sampling of the river, tributaries, and WWTPs, averaging 4 to 5 times per week, during November 1992.

Data from the November 1992 sampling program was used to calibrate the water-quality model CE-QUAL-W2 (Cole and Buchak, 1995), which was then used to evaluate the relative significance of the various deoxygenation processes during this period. The model also was used to examine the effect of a range of hypothetical loading and flow conditions on DO in the river.

Acknowledgments

The author gratefully acknowledges the staff of Unified Sewerage Agency for their cooperation, laboratory analyses, and technical assistance. In particular, Gary Krahmer, William Gaffi, John Jackson, Janice Miller, and Jan Wilson provided crucial support that was vital to the success of this project. In addition, Jerry Rodgers of the Oregon Water Resources Department provided necessary hydrologic data.

TUALATIN RIVER BASIN

The Tualatin River Basin is a subbasin of the Willamette River, Oregon, and drains an area of 712 mi^2 (square miles) in Washington County, west of Portland, Oregon (fig. 1). The basin is bounded on the west and northwest by the Coast Range, on the east and northeast by the Tualatin Mountains, and on the south by the Parrett and Chehalem Mountains. The direction of flow is generally from NW to SE for 79.4 miles through a variable watershed. Although the headwaters are in the steep, forested Coast Range, the river flows for most of its length through a relatively low gradient area characterized by rolling forest and croplands, interspersed with an increasingly urbanized region, before the confluence with the Willamette River at West Linn, Oregon. The length of river encompassed by this study extended from the USGS streamflow-gaging station near Dilley (14203500) at river mile (RM) 58.8 to the river mouth at Weiss Bridge, RM 0.2, just downstream from the USGS streamflow-gaging station (14207500) at West Linn, RM 1.8.

Climate in the basin is characterized by wet winters and dry summers, with at least 70 percent of the annual precipitation, which averages close to 45 in/yr (inches per year) (National Oceanic and Atmospheric Administration, 1991), occurring as rain between November and April. The winters are mild with mostly cloudy skies. Temperatures are rarely below freezing.

Five major tributaries empty into the Tualatin River: Scoggins, Gales, Dairy, Rock, and Fanno Creeks. Dairy Creek is the largest tributary, draining an area approximately one-third of the total area of the Tualatin River Basin (table 1). The Dairy Creek subbasin is largely agricultural land, with only a few small towns located within it. The combined drainage areas of the other major tributaries, in addition to the upper Tualatin River near Dilley, constitute an additional one-half of the basin area. Characteristics of these basins range from predominantly forested lands with little development (upper Tualatin River, Scoggins and Gales Creeks) to mostly urban subbasins (Rock and Fanno Creeks). About 17 percent of the Tualatin River Basin is drained by smaller tributaries (table 1). Chicken Creek, a minor tributary which flows into the Tualatin River at RM 15.2, also was sampled during the winter season, November 1991 to April 1992, to provide some information on the characteristics of this small tributary.

Table 1. Physical characteristics of the main stem and major tributaries of the Tualatin River, Oregon
[Tributary drainage area data from Scientific Resources, Inc., 1990; --, not applicable or data not available]

Stream name	Stream length (miles)	Drainage area (square miles)	Confluence to Tualatin River (river mile)
Tualatin River near Dilley	20.6	125	--
Scoggins Creek	17.5	45	60.0
Gales Creek	28.0	87	56.7
Dairy Creek	--	225	44.8
West Fork	21.5	92	--
East Fork	21.5	68	--
McKay Creek	24.0	64	--
Rock Creek	19.0	76	38.1
Fanno Creek	14.0	34	9.3
Minor tributaries	--	118	--
Tualatin River at mouth	79.4	712	--

Channel Morphology

On the basis of channel morphology and hydraulic characteristics, the Tualatin River can be divided into four ecologically distinct reaches (table 2). These include a shallow, fast-moving mountain reach at the headwaters, which slows and meanders considerably as the river flows out of the mountains. For about 40 percent of its length, the river is comparable to a reservoir, characterized by a relatively deep and wide channel and slow-moving water. A short series of shallow pools and riffles occurs at the lower end of the river, just before it flows into the Willamette River. The diversity of physical characteristics associated with these reaches is reflected in very different biological habitat conditions for aquatic organisms. These conditions, in turn, greatly affect the water quality within each reach by either facilitating or inhibiting the growth of certain kinds of organisms such as nitrifying bacteria. Travel times among these reaches vary greatly and will be discussed in greater length in the section entitled "Historical Flow Conditions."

Table 2. Physical characteristics of the Tualatin River, Oregon
[ft/mi, feet per mile]

Reach	River miles	Change in bed elevation (feet)	Approximate streambed slope (ft/mi)
Headwater	79.4–55.3	1,790	74.3
Meander	55.3–33.3	28	1.27
Backwater	33.3–3.4	2.3	.077
Riffle	3.4–mouth	45.6	13.4

The uppermost reach, extending for 24.1 miles in length from the headwaters to just downstream from the confluence with Gales Creek, is characterized by a steep slope with correspondingly high water velocities. The character of the river throughout most of this reach is that of a pristine mountain stream, shaded by dense riparian vegetation for most of its length. At the lower end, the gradient flattens in the Patton Valley; this part of the reach is predominantly agricultural land. Two major tributaries enter the Tualatin River in this reach, Scoggins and Gales Creeks, as well as effluent from a small WWTP at Forest Grove. The flow level in Scoggins Creek is controlled by releases from Henry Hagg Lake (storage volume 53,600 acre-feet), which was formed behind Scoggins Dam in 1975 (fig. 1); releases are designed to maintain water levels in the river to satisfy various water rights during the summer base-flow season. Water quality, as defined by DO concentrations, in this reach is excellent, reflecting the relatively undisturbed nature of the subbasin.

The river changes character in the meander reach as the stream gradient decreases further, which occurs as the river flows into the Tualatin Valley, and the water volume increases due to the inflows from Dairy and Rock Creeks. The river attains nearly its full volume by the end of this 22-mile long reach; during the winter the streamflow at RM 33.3 is typically 80 to 90 percent of the streamflow at the mouth at West Linn. Within this meandering reach, the channel widens slightly, shading is less pronounced, and water velocities are decreased. During low flow, depositional zones form at the lower end of this reach and extensive amounts of organic material accumulate on the streambed in many sections. Two WWTPs discharge effluent into the river in this reach—a small plant in Hillsboro and a large plant at Rock Creek. Water quality is slightly degraded, in terms of DO demand, in this reach as a consequence of agricultural and urban development in this part of the basin. Serious DO depletion, however, does not generally occur.

Water from the backwater or reservoir reach is diverted near the lower end of the river into Lake Oswego via the Lake Oswego Canal at RM 6.7. The water level in the lower river is controlled for purposes of this diversion by the Lake Oswego Corporation (LOC) diversion dam, located on a natural geologic sill at RM 3.4. DO concentrations in this section of the river can be seriously affected by its backwater characteristics, which partially result from the presence of the dam and extends for nearly 30 miles upriver. Fanno Creek flows into the Tualatin River in this reach, in addition to effluent from the large WWTP at Durham. The river deepens and velocities are further reduced as the water moves through this reach, even at the higher flow levels typical during the winter. Considerable deposits of organic material collect in depositional zones during low flows, exerting a significant sediment oxygen demand. The stream gradient is essentially flat throughout, although the streambed is uneven, forming occasional deep pools which can thermally stratify during periods of warm weather.

The riffle reach below the LOC dam is characterized by a narrowed channel, steep gradient, and increased water velocities. Comparatively little deposition occurs in this part of the river, and the water is considerably shallower than in the adjacent upstream reach. Many rocks in the streambed are exposed. DO concentrations usually increase in this reach due to increased reaeration.

Wastewater Treatment in the Basin

The four major WWTPs in the Tualatin River Basin vary considerably both in size and their affect on the river quality (table 3). The two smaller plants, located in Forest Grove and Hillsboro, release effluent to the river only during the winter season, November 1 to April 30. From May through October, effluent from these plants is diverted to land for irrigation purposes. The other two plants, at Rock Creek and Durham, serve larger populations and discharge treated effluent to the river throughout the year. The two smaller plants use primary and secondary treatment. The larger plants use primary and secondary treatment during the winter, as well as advanced tertiary treatment during the summer.

Table 3. Characteristics of major wastewater treatment plants in the Tualatin River Basin, Oregon (flow data based on wastewater treatment plant data from November to April for sampling period noted; maximum capacity data from Unified Sewerage Agency, 1990; population data from Unified Sewerage Agency)
[Mgal/d, million gallons per day]

Wastewater treatment plant name	Discharge point (river mile)	Sampling period	Daily mean effluent flow (Mgal/d)	Peak effluent flow (Mgal/d)	Maximum capacity for secondary treatment (Mgal/d)	Population served (1990)
Forest Grove	55.2	3/89–11/92	5.14	14.7	10	14,000
Hillsboro	43.8	11/87–11/92	3.42	7.78	12.8	19,100
Rock Creek	38.1	3/88–11/92	20.5	54.8	77	135,000
Durham	9.3	1/87–11/92	18.4	42.7	40	142,000

Daily mean effluent flows tend to be fairly constant for all the WWTPs, increasing slightly as the river streamflow increases. Historically, peak effluent flows occur concurrently with a rapid increase of streamflow in the river or during a peak-flow stage, generally during periods of high-intensity rainfall (fig. 2). Increased effluent flows under these conditions result from precipitous increases of influent flow that may exceed the plant treatment capacity under extreme conditions. These greatly increased influent flows are caused by inflow from inadvertent storm sewer connections, as well as infiltration of elevated ground water through leaky sewer pipes. A significant period of inflow and infiltration can decrease the efficiency of wastewater treatment, with a concomitant discharge of increased waste loads to the river. Because inflow and infiltration always occur during high streamflow in the river, however, the effects of dilution and decreased travel times would be expected to lessen the potential effect of increased loads on the river system.

Removal of ammonia from effluent by nitrification is maintained throughout the warm summer months at the Rock Creek WWTP; proposed treatment design improvements at the Durham plant include nitrification as well. As water temperatures

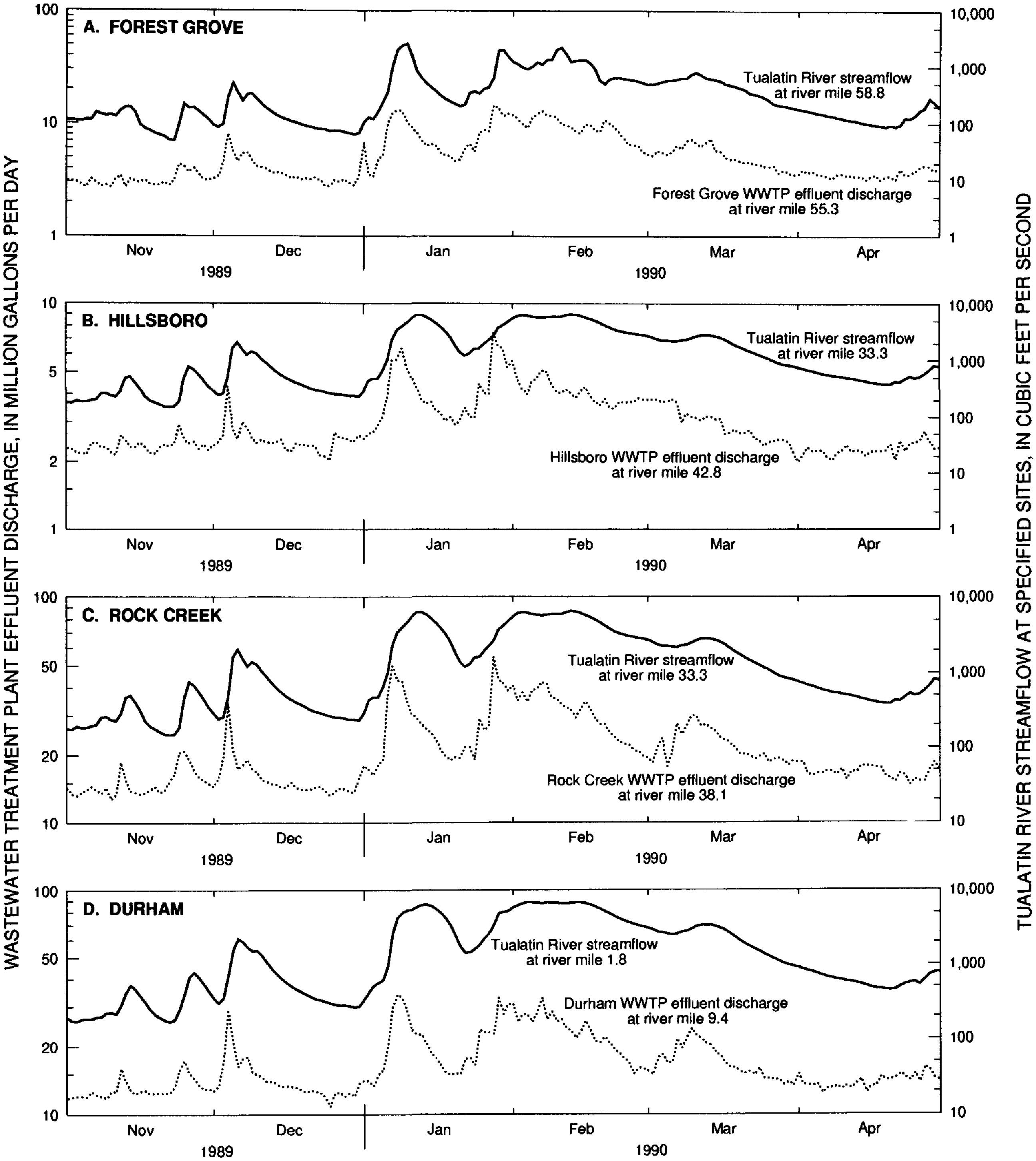


Figure 2. Relation between historical daily mean effluent discharge from wastewater treatment plants and daily mean streamflow in the Tualatin River, Oregon, at selected river miles, for the winter season (November to April), 1990 water year. (A) Forest Grove, (B) Hillsboro, (C) Rock Creek, and (D) Durham.

cool at the end of the summer, however, the nitrification process becomes progressively less efficient and eventually ceases completely (Thomann and Mueller, 1987). At the same time, at the beginning of November, the two smaller WWTPs begin discharging their effluent, containing large quantities of ammonia, directly into the river. Consequently, loads of ammonia in the river dramatically increase in November and remain high throughout the winter season.

Historical Flow Conditions

Flows in the Tualatin River exhibit a definite seasonal pattern, illustrated by a plot of mean monthly streamflow at West Linn (14207500) for the 17-year period beginning with the start of flow regulation by Scoggins Dam, from October 1, 1975 to September 30, 1992 (fig. 3). Mean monthly flows range from about 150 ft^3/s (cubic feet per second) in July and August to about 3,000 ft^3/s during January and February. Although November is considered part of the winter season for regulatory purposes, the mean flow for November during this period is intermediate between the extremes of the summer and winter flow conditions. In actuality, the mean monthly flow of slightly more than 1,000 ft^3/s for November more closely resembles the mean monthly flow for May than flows of any other winter months.

Mean monthly values are indicative of long-term trends; however, they can mask extended periods of extreme low or high flows that can be significant on

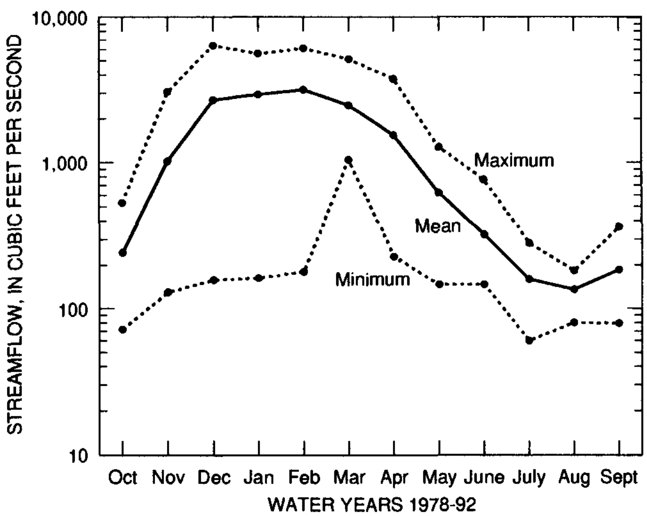


Figure 3. Mean, maximum, and minimum monthly streamflow in the Tualatin River at West Linn, Oregon, 1976–92 water years.

a short term basis. To evaluate this possibility, 7-day and 14-day consecutive low-flow streamflow duration curves were compared for the winter season (November to April) for the periods before and after flow augmentation from Hagg Lake began in 1975 (fig. 4). Prior to the completion of Scoggins Dam (data from 1942–75), extended periods of low flow during the winter were typical. About 75 percent of the mean 7-day low flows and more than 50 percent of the 14-day mean low flows were less than 300 ft^3/s. After flow regulation began (data from 1976–92), the mean low flows were increased; nonetheless, about 50 percent of the time, the 7-day mean low flow was less than 300 ft^3/s and the 14-day mean low flow was less than 500 ft^3/s. These data indicate that extended low-flow periods in the Tualatin River during the winter are still common.

To assess the streamflow characteristics of each winter month during the 17-year period since flow regulation began, categories of flow were defined: less than 300 ft^3/s, 300–500 ft^3/s, 500 to 2,000 ft^3/s, and greater than 2,000 ft^3/s. In the Tualatin River, the lower flows generally are not observed during the winter season, except during November (fig. 5). Daily mean flows in November were less than 500 ft^3/s nearly 50 percent of the time, compared to about 10 percent of the time for the other winter months. Flows were in he highest category less than 20 percent of the time during November. In contrast, flows were greater than 2,000 ft^3/s about 50 to 60 percent of the time for the remaining winter months, with the exception of April, when streamflow typically remained greater than 500 ft^3/s.

Approximate times of streamflow travel, based on equations derived from dye studies (Antonius Laenen, U.S. Geological Survey, written commun., 1994) are plotted in figure 6 for four flow categories at West Linn: 150 ft^3/s, 300 ft^3/s, 500 ft^3/s, and 2,000 ft^3/s. Observed values for travel time at flows that ranged from 207 ft^3/s at RM 33.3 to 266 ft^3/s at RM 1.8 are also plotted. Several cautions need to be kept in mind when interpreting these estimates of travel time. Considerable uncertainty in estimating travel time is caused by the variability in weir characteristics of the LOC dam, which are informally maintained and historically have not been well documented. Changes in these characteristics have a considerable effect on residence time, especially in the lower river. Although this uncertainty is most important during the summer season (May to October), it does apply to the winter as

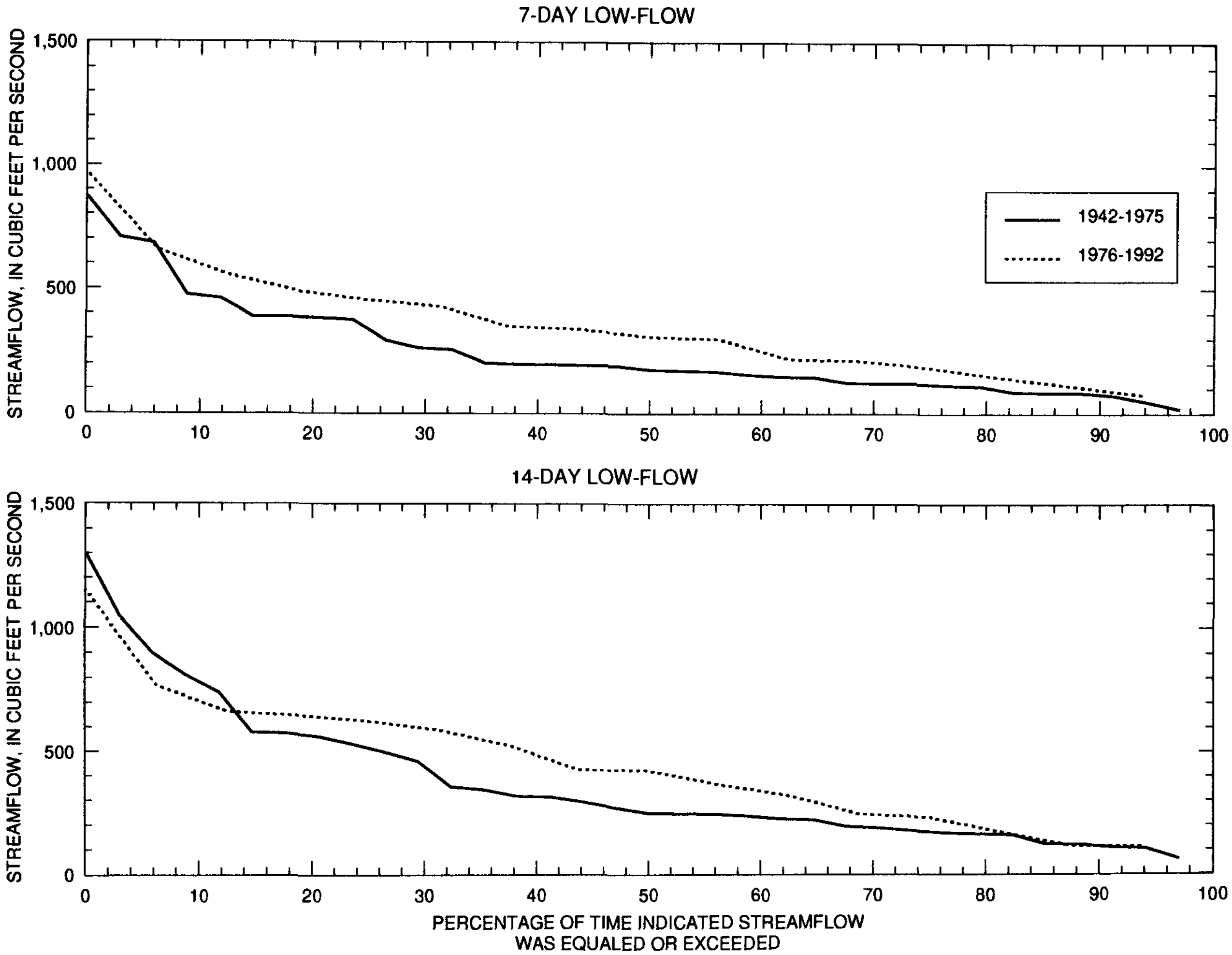


Figure 4. Streamflow-duration curves for the Tualatin River at West Linn, Oregon, for the winter season (November to April) for the period 1942–75, prior to flow augmentation from Henry Hagg Lake and after flow augmentation began, from 1976–92. (A) 7-day low flow and (B) 14-day low flow.

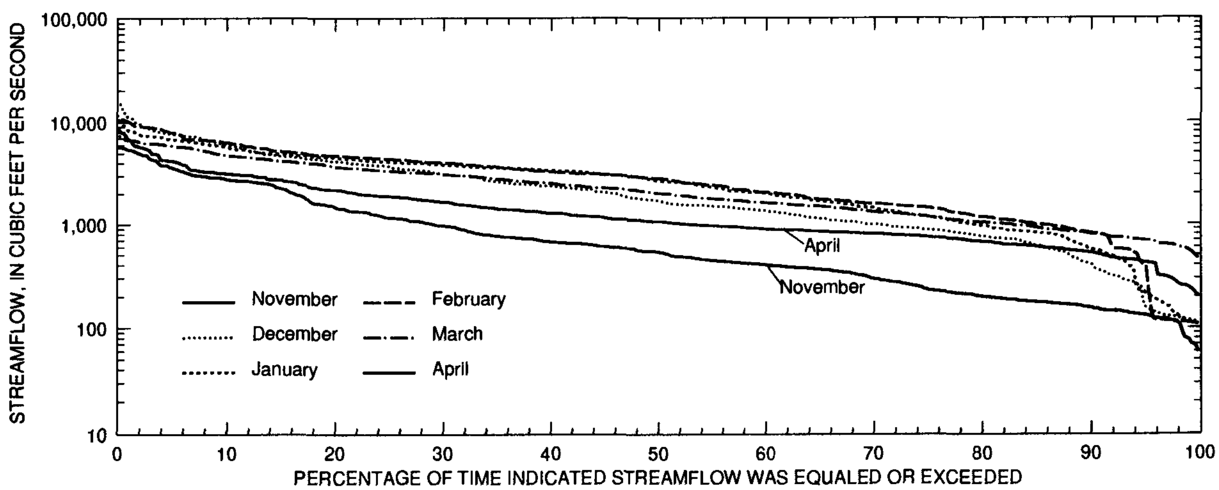


Figure 5. Streamflow-duration curves for the Tualatin River at West Linn, Oregon, for the winter season (November to April), 1976–92.

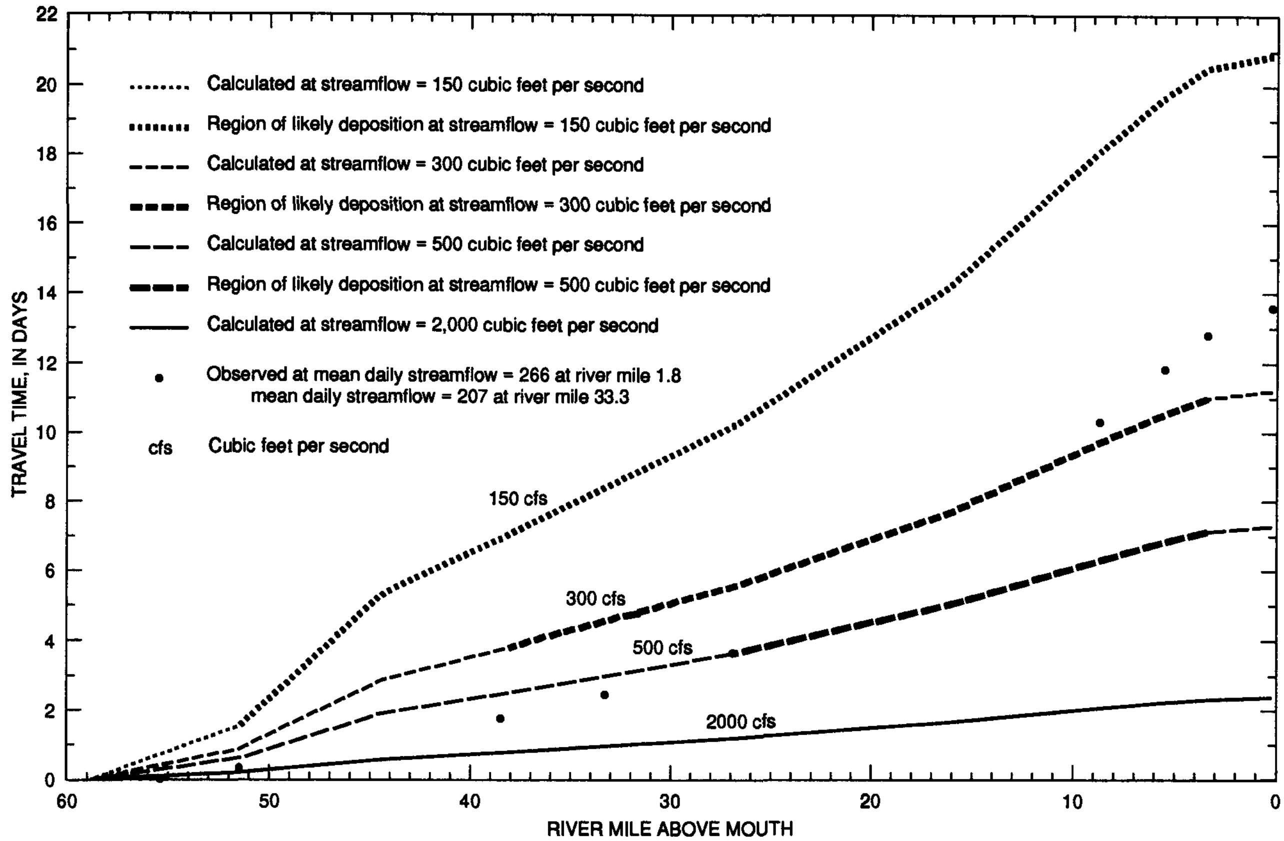


Figure 6. Calculated estimates of travel time in the Tualatin River, Oregon, for four selected values of constant streamflow, and observed travel times from dye studies (Antonius Laenen, written commun., 1994) under variable streamflow conditions. Estimated travel times were based on calculations derived from the dye studies and assumed constant streamflow below river mile 44.4, reduced by 50 percent upstream. Regions of likely deposition were determined by calculated velocities less than 0.6 feet per second.

well when base-flow periods (streamflow less than 300 ft^3/s) extend into November. In addition, in comparing the observed and calculated values, note that the observed data are from the summer season, when flow augmentation from Hagg Lake maintains flows in the upper river at levels comparable to flows in the lower river. During the winter base-flow period, however, flow augmentation does not occur, and flow in the upper river is typically about one-half that in the lower river. For the purpose of these calculations, flow was considered to be constant throughout the river length, reduced by 50 percent upstream of RM 44.4. The comparison was provided for perspective, and should not be assumed to represent identical flow conditions. For these ranges of flow, the difference in residence time is significant, ranging from approximately 21 days at the lowest flow to about 2 days at peak flow.

In addition, figure 6 indicates the regions of potential deposition in the study reach for the selected levels of streamflow. According to Velz (1984), deposition and accumulation of organic sludge material is likely when velocities are less than 0.6 ft/s (feet per second) for extended distances. The velocity at which scouring will occur depends upon the extent to which accumulated material is compacted, and is usually 1.0 ft/s or greater. According to this guideline, deposition is possible throughout most of the study reach when flows are 300 ft^3/s or less. At flows of 500 ft^3/s, deposition can occur throughout the lower river beginning near RM 27. In contrast, velocities are sufficient at peak flow to induce scour throughout most of the entire study reach.

Historical Temperature Conditions

Limited historical temperature data during the winter for the Tualatin River were available from USA for the period prior to this study, dating back to November 1986. The data were categorized for this analysis on the basis of the flow at West Linn (fig. 7). In general, mean temperatures were less than 10°C throughout the length of the study reach except when

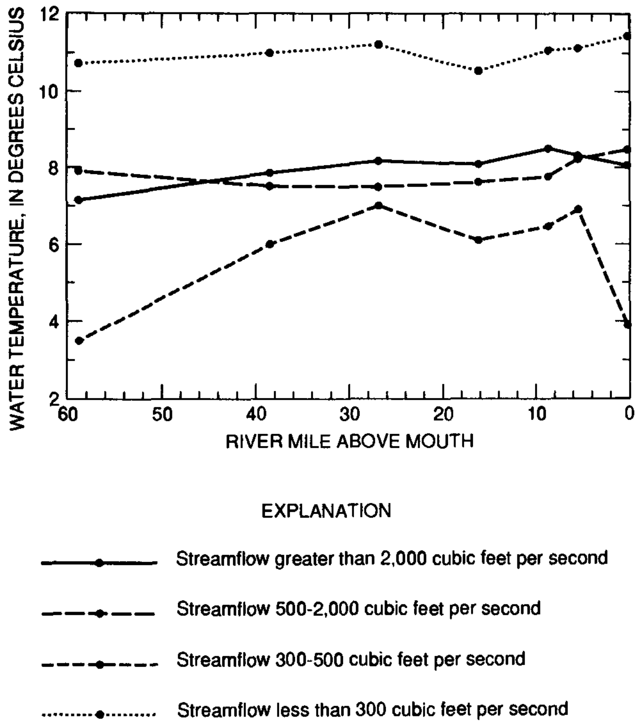


Figure 7. Historical mean water temperatures, categorized by selected ranges of streamflow in the Tualatin River at West Linn, Oregon, at selected sites in the main stem river during the winter season (November to April) 1987–91. Temperature data provided by Unified Sewerage Agency.

flows were less than 300 ft^3/s. Under these conditions, mean temperatures were higher than 10°C at all sampling sites. These higher temperatures reflect the increased residence times associated with base-flow conditions, as well as the tendency for winter low flow to occur early in the winter season, during the transition from the warmer summer flows.

Historical Water-Quality Conditions

By evaluation of historical water-quality conditions in the Tualatin River, its major tributaries, and wastewater treatment plant effluent in context with the flow and temperature conditions in the river, the effects of dilution, residence time, and decreased temperatures on deoxygenation processes can be considered. Historical water-quality data available from the Unified Sewerage Agency (USA) included field and laboratory measurements for main-stem and tributary sites. For the specific constituent method, see the section "Methods of Study." Streamflow and effluent discharge measurements were available only for the main stem and the WWTPs; consequently constituent loads from the tributaries could not be calculated. Loading data from the WWTPs, however, were extensive.

Figure 8 illustrates the historical mean concentrations of selected water-quality constituents for four ranges of streamflow in the main stem during the winter seasons of water years 1987 to 1991. The selected constituents are DO, 5-day carbonaceous biochemical oxygen demand (CBOD-5), TSS, and ammonia (as nitrogen). Figure 9 shows the corresponding data for the major tributaries. All the samples were taken during the day, generally between 8 a.m. and 4 p.m. DO concentrations in the Tualatin River during the winter months of this period were observed to drop considerably in the lower river when flows were less than 300 ft^3/s; the mean DO concentrations were less than the minimum State standard of 6 mg/L at some sites in the lower river. DO concentrations in the tributaries were also consistently lower at low flows. When streamflow in the river was greater than 300 ft^3/s, however, the mean DO concentrations did not fall below the State standard. Possible explanations for this oxygen decline include decreased reaeration in water moving at reduced velocities and longer residence times coupled with higher temperatures, which allow the various deoxygenation processes to have a greater effect.

Historical concentrations of CBOD-5 were highest in the river and tributaries at low flow, presumably because of dilution effects. Mean loads of CBOD-5 from the WWTPs over the same historical period, however, increased over the range of flows from two to fourfold, with the maximum loads increasing by more than one order of magnitude from the lowest to the highest flow range (table 4). In the case of effluent from the smaller WWTPs, the increased loads were due primarily to increased effluent flows, because mean concentrations of CBOD-5 were unchanged throughout the flow ranges. In the larger plants, however, CBOD-5 concentrations increased as effluent and river flow increased, suggesting that treatment was less complete.

Mean concentrations of TSS in the river were substantially higher at peak-flow periods, indicating that dilution effects were less important than increased loading when flow levels were high. Significant loading of TSS from the tributaries at high flows is suggested by mean TSS concentrations, which were generally highest at the high-flow level—particularly in Gales Creek. Mean loads of TSS from the WWTPs also tended to increase considerably as flow levels increased, with the exception of the Hillsboro plant.

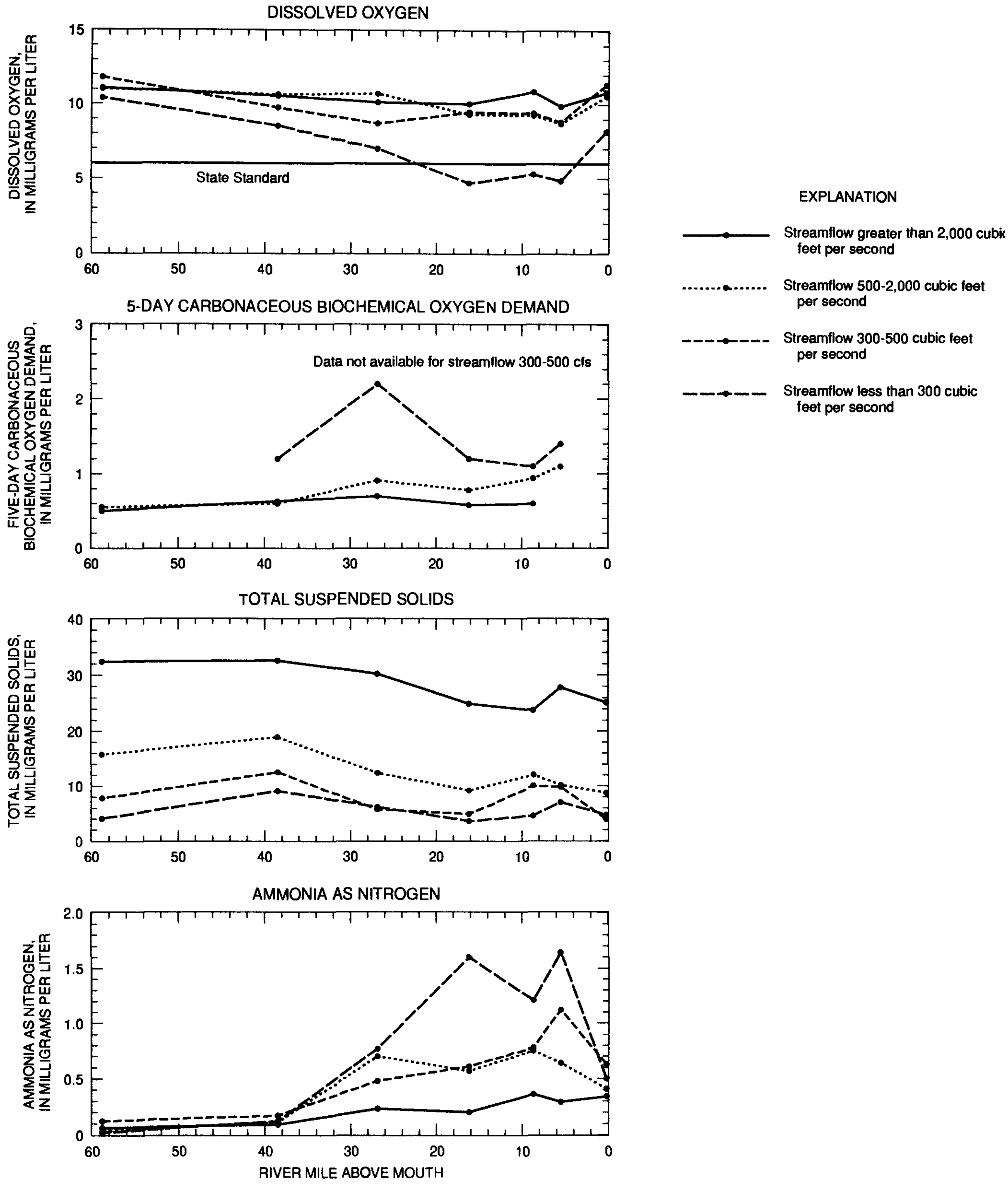


Figure 8. Historical mean concentrations of dissolved oxygen, 5-day carbonaceous biochemical oxygen demand, total suspended solids, and ammonia as nitrogen, categorized by selected ranges of streamflow in the Tualatin River at West Linn, Oregon, at selected sites in the main stem river during the winter season (November to April) 1987–91. Water-quality data provided by Unified Sewerage Agency.

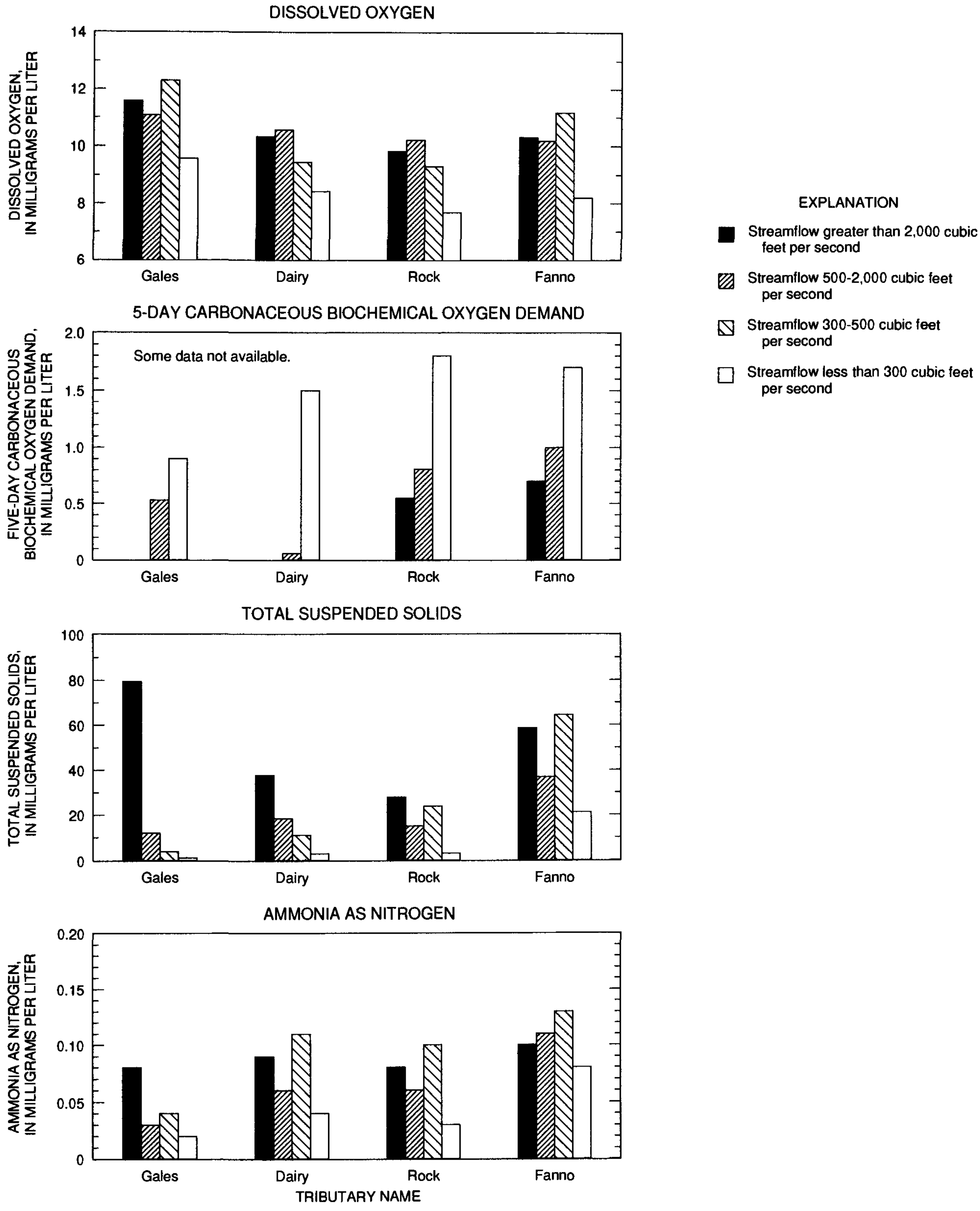


Figure 9. Historical mean concentrations of dissolved oxygen, 5-day carbonaceous biochemical oxygen demand, total suspended solids, and ammonia as nitrogen, categorized by selected ranges of streamflow in the Tualatin River at West Linn, Oregon, in the major tributaries to the main stem river during the winter season (November to April) 1987–91. Water-quality data provided by Unified Sewerage Agency.

Table 4. Historical mean and range of daily loads of 5-day carbonaceous biochemical oxygen demand (CBOD-5) and total suspended solids (TSS) from the wastewater treatment plants for selected ranges of streamflow in the Tualatin River at West Linn, Oregon (U.S. Geological Survey streamflow-gaging station 14207500) during the winter season, November to April 1987–91

[Loads in thousand pounds per day; N, number of observations]

Wastewater treatment plant	Characteristic	Range of streamflow in Tualatin River, in cubic feet per second											
		Less than 300			300–500			501–2,000			Greater than 2,000		
			Loads			Loads			Loads			Loads	
		N	Mean	Range	N	Mean	Range	N	Mean	Range	N	Mean	Range
Forest Grove	CBOD-5	14	0.13	0.10–0.21	15	0.15	0.084–0.27	55	0.20	0.058–3.1	56	0.40	0.12–3.6
	TSS	24	.28	0.12–0.59	28	.36	0.18–0.88	118	.57	0.079–17	124	.91	0.090–14
Hillsboro	CBOD-5	19	.088	0.045–0.20	17	.11	0.042–0.22	94	.11	0.029–0.60	95	.14	0.051–0.62
	TSS	25	.16	0.087–0.35	29	.16	0.068–0.45	116	.14	0.061–0.51	120	.17	0.059–0.72
Rock Creek	CBOD-5	22	.48	0.15–0.93	29	.81	0.24–4.1	167	1.0	0.18–6.3	145	1.2	0.25–3.4
	TSS	40	.27	0.062–0.72	51	.41	0.085–1.7	280	.72	0.073–8.0	231	.77	0.18–5.7
Durham	CBOD-5	42	.22	0.11–0.56	27	.33	0.087–0.84	195	.47	0.073–1.9	175	.74	0.19–4.2
	TSS	70	.24	0.099–1.4	54	.37	0.010–1.4	369	.60	0.073–3.9	332	1.1	0.19–8.6

Mean ammonia concentrations were highest in the lower main stem during low flows, whereas ammonia concentrations in the tributaries were lowest at low flow. In contrast, ammonia loads from the WWTPs changed relatively little across the range of flows (table 5). Across the ranges of streamflow, which span more than an order of magnitude, mean loads from the larger WWTPs increased only about twofold, and maximum loads were similar at all flow ranges. The loading patterns of nitrite-plus-nitrate nitrogen from the WWTPs also were generally similar across all flow ranges. These data suggest that WWTPs and treatment of ammonia and nitrite-plus-nitrate nitrogen were not significantly affected by the hydrologic conditions in the river.

METHODS OF STUDY

Streamflow Measurements

Continuous streamflow-gaging stations were maintained by the USGS in the upper Tualatin River at RM 58.8 (Tualatin River near Dilley, Oregon—14203500) and near the mouth at RM 1.8 (Tualatin River at West Linn, Oregon—14207500). In addition, a USGS streamflow-gaging station was maintained in Scoggins Creek, at RM 60.0 (Scoggins Creek below Henry Hagg Lake near Gaston, Oregon—14202980) and in Fanno Creek at RM 9.3 (Fanno Creek at Durham, Oregon—14206950). A continuous streamflow-gaging station was maintained during the winter by the OWRD (Oregon Water Resources Department) at RM 38.5 (Tualatin River at Rood Bridge at Hillsboro, Oregon—14206440) and at RM 33.3 (Tualatin River at Farmington, Oregon—14206500). In addition, OWRD maintained stations at several sites in the main stem (Tualatin River at Golf Course Road, Highway 219, and Elsner Road) and in the other major tributaries (Gales Creek, Dairy Creek, Rock Creek, and Chicken Creek); vertical staff plates were located at these stations and periodic discharge measurements were made by OWRD in order to develop rating tables. Incidental gage-height readings at these sites were made concurrent with water-quality samples and do not reflect daily mean streamflow.

Sampling and Analytical Procedures

During the winter season November 1991 to April 1992, the sampling program had two components: weekly monitoring of ambient conditions and daily sampling during storms. Analyses of main-stem and tributary samples were performed by the USA Water Quality Laboratory (WQL) for nitrogen species, 5-day BOD and CBOD, and others as shown in table 6. Data is stored in the USA computer database, Hillsboro, Oregon. Analyses of extended CBOD were also performed by USGS personnel in the Oregon District once a month, as described in the next section, "Biochemical Oxygen Demand."

Nine sites on the Tualatin River were sampled by USGS personnel. The four major tributaries and one minor tributary were sampled near their confluences with the main stem by USA personnel, as described in table 7 (see fig. 10 for location of sites). The sites were chosen on the basis of their location upstream and downstream from major WWTP and tributary sources, as well as accessibility for sampling from bridges. It was originally planned to sample during two major storms in the winter of 1991 to 1992, one early in the rainy season, when the water table was low, and a second later in the season, when the ground had become saturated. The year was unusually dry, however, and few major storms occurred; only one storm was targeted for additional samples, from January 29 to February 7, 1992.

A total of 32 sampling trips were made during the winter 1991 to 1992, and 15 sampling trips during November 1992. Vertically and horizontally integrated samples were collected, sampling from the surface to the bottom at 5 to 10 points in a cross section (depending on width of river), using a USGS D77 aluminum sampler. These samples were composited in a churn splitter and dispensed directly into bottles, which were field rinsed with native water before being filled. Filtered samples from the churn splitter for analysis of soluble orthophosphate were obtained using 12-ml plastic syringes and Gelman nylon Acrodisc filters (0.45 micrometer pore size). Samples were immediately placed on ice and transported to the USA WQL within 5 hours for analysis. WWTP samples were 24-hour flow-weighted composite samples collected daily by personnel from the respective WWTP; analyses were performed at the Rock Creek WWTP Laboratory for the three plants

Table 5. Historical mean and range of daily loads of ammonia as nitrogen (NH_4) and nitrite-plus-nitrate nitrogen (NO_2+NO_3) from the wastewater treatment plants for selected ranges of streamflow in the Tualatin River at West Linn, Oregon (U.S. Geological Survey streamflow-gaging station 14207500) during the winter season, November to April 1987–91

[Loads in thousand pounds per day; N, number of observations; --, no data]

Wastewater treatment plant	Constituent	Range of discharge in Tualatin River, in cubic feet per second											
		Less than 300			300–500			501–2,000			Greater than 2,000		
			Loads			Loads			Loads			Loads	
		N	Mean	Range	N	Mean	Range	N	Mean	Range	N	Mean	Range
Forest Grove	NH_4	0	--	--	0	--	--	0	--	--	0	--	--
	NO_2+NO_3	0	--	--	0	--	--	0	--	--	0	--	--
Hillsboro	NH_4	5	0.25	0.15–0.31	5	0.20	0.072–0.30	27	0.25	0.11–0.40	27	0.26	0.18–0.33
	NO_2+NO_3	5	.080	0.010–0.17	5	.057	0.010–0.13	24	.061	0.004–0.25	26	.074	0.013–0.23
Rock Creek	NH_4	40	1.1	0.072–4.3	51	1.9	0.004–3.7	282	2.3	0.021–4.4	231	1.9	0.047–4.0
	NO_2+NO_3	6	.57	0.13–0.93	6	.48	0.14–0.69	47	.65	0.065–2.9	38	.97	0.16–6.5
Durham	NH_4	37	.93	0.46–2.0	33	1.2	0.45–2.0	217	1.5	0.37–4.1	190	1.6	0.30–3.4
	NO_2+NO_3	0	--	--	2	.64	0.28–1.0	13	.40	0.017–1.1	16	.28	0.076–0.63

Table 6. Methods of water analysis

[USA, Unified Sewerage Agency; WWTP, wastewater treatment plant; USGS, U.S. Geological Survey; EPA, Environmental Protection Agency; rev, revised; ed, edition; ASF, automated-segment flow colorimetry; µm, micrometer]

Measurement	USA Water Quality Laboratory	Durham WWTP Laboratory	Rock Creek WWTP Laboratory	USGS National Water Quality Laboratory	USGS Oregon District Laboratory
Chemical oxygen demand	EPA 600/4-79-020 (rev 3/83) #410.4 (colorimetric)	EPA 600/4-79-020 (rev 3/83) #410.3 (titrimetric)	EPA 600/4-79-020 (rev 3/83) #410.3 (titrimetric)	Analysis not done	Analysis not done
Biochemical oxygen demand —5 day	Standard Methods 16th ed. #507 (probe)	Standard Methods 16th ed. #507 (probe)	Standard Methods 17th ed. #5210 (electrode)	Analysis not done	Standard Methods 17th ed. #5210
Carbonaceous biochemical oxygen demand —5 day	Standard Methods 16th ed. #507 (probe) Hach Formula 2533 inhibitor	Standard Methods 16th ed. #507 (probe) Hach Formula 2533 inhibitor	Standard Methods 17th ed. #5210 (electrode) Hach Formula 2533 inhibitor	Analysis not done	Standard Methods 17th ed. #5210 Hach Formula 2533 inhibitor
Total suspended solids	Standard Methods 16th ed. #209C (gravimetric 103°C)	Standard Methods 16th ed. #209C (gravimetric 103°C)	Standard Methods 17th ed. #2540D (gravimetric 103°C)	Analysis not done	Analysis not done
Ammonia as nitrogen	Standard Methods 16th ed. #417G (colorimetric)	Standard Methods 16th ed. #417E (potentiometric)	EPA 600/4-79-020 (rev 3/83) #350.1 (colorimetric)	0.45 µm filter, colorimetric, salicylate-hypochlorite, ASF	Analysis not done
Total Kjeldahl nitrogen as nitrogen	EPA 600/4-79-020 (rev 3/83) #351.2 (colorimetric)	EPA 600/4-79-020 (rev 3/83) #351.2 (colorimetric)	EPA 600/4-79-020 (rev 3/83) #351. 2 (colorimetric)	Colorimetric, salicylate-hypochlorite, ASF	Analysis not done
Nitrite-plus-nitrate nitrogen	EPA 600/4-79-020 (rev 3/83) #353.2 (colorimetric cadmium reduction)	EPA 600/4-79-020 (rev 3/83) #353.2 (colorimetric cadmium reduction)	EPA 600/4-79-020 (rev 3/83) #353.1 (colorimetric hydrazine reduction)	0.45 µm filter, cadmium reduction, hydrazine reduction-diazotization method (colorimetric)	Analysis not done
Total phosphorus	EPA 600/4-79-020 (rev 3/83) #365.4 (colorimetric)	EPA 600/4-79-020 (rev 3/83) #365.4 (colorimetric)	EPA 600/4-79-020 (rev 3/83) #365.4 (colorimetric)	Potassium persulfate wet digestion, phosphomolybdate method (colorimetric)	Analysis not done
Soluble ortho-phosphate as phosphorus	EPA 600/4-79-020 (rev 3/83) #365.1 (0.45 µm filter, colorimetric)	EPA 600/4-79-020 (rev 3/83) #365.1 (0.45 µm filter, colorimetric)	EPA 600/4-79-020 (rev 3/83) #365.4 (0.45µm filter, colorimetric)	0.45 µm filter, colorimetric, phosphomolybdate, ASF	Analysis not done

Table 7. Sampling sites in the Tualatin River, Oregon, and its major tributaries for all field and laboratory water-quality analyses, and selected sites for extended analysis of carbonaceous biochemical oxygen demand (CBOD)

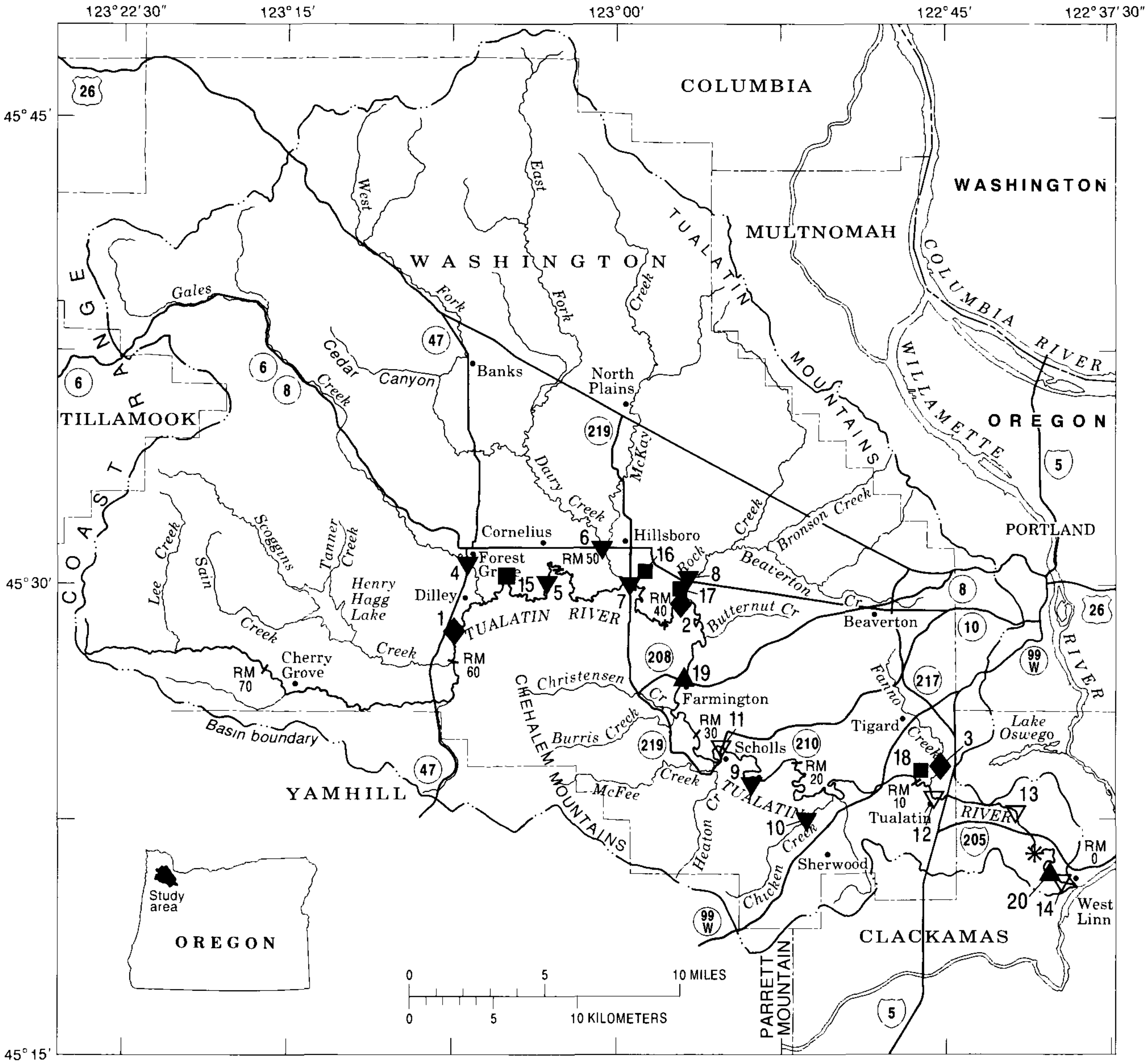
Sampling site	Tualatin river mile	Extended CBOD
Tualatin River near Dilley	58.8	X
Gales Creek at new Highway 47	56.7	X
Tualatin River at Golf Course Road	51.5	
Dairy Creek at Highway 8	44.8	X
Tualatin River at Highway 219	44.4	
Tualatin River at Rood Road	38.5	X
Rock Creek at Highway 8	38.1	X
Tualatin River at Highway 210 near Scholls	26.9	
Tualatin River at Elsner Road	16.2	X
Chicken Creek on Scholls-Sherwood Road	15.2	
Fanno Creek at Durham Road	9.3	X
Tualatin River at Boones Ferry Road	8.7	X
Tualatin River at Stafford Road	5.5	
Tualatin River at Weiss Bridge	.2	X

located in the upper river basin, and at the Durham WWTP Laboratory for the Durham plant.

During the winter, November 1991 to April 1992, field measurements of pH, water temperature, specific conductance, and DO were taken at tributary sites by USA personnel with separate instruments as follows: pH, Orion model 250; water temperature and specific conductance, YSI model 3000; DO, YSI model 57 and 58. All other field measurements were made using a Hydrolab multiparameter water-quality sensor. Readings were taken at three locations in the cross section, approximately 25 percent, 50 percent, and 75 percent from the left bank, with vertical profiles (surface, mid-depth, and near the bottom) measured when the depth was sufficient. Field instruments were calibrated prior to each sampling trip, and post-calibrations were done within 1 day after every trip. Whenever the post-calibration showed the instrument to require adjustment, the difference from the correct value was always within the variability inherent to the measurement. Calibration for pH and specific conductance was checked against standards that bracketed the anticipated values in the river; calibration for DO was done using the air-calibration technique. In addition to the weekly field measurements, continuous measurements of water quality were obtained from a four-parameter field mini-monitor that was installed and maintained at RM 3.2, just below the diversion dam (fig. 10). This location was chosen as the site where DO depletion was likely to be most severe.

The analysis of historical loading patterns to the Tualatin River from the WWTPs used unpublished data from the USA data base. The WWTP data were based on samples composited over a 24-hour period on a flow-weighted basis, as in the present study, and were analyzed by the WWTP laboratories, as described above. River and tributary data were available from surface-grab samples collected at the midpoint of flow at a limited number of sites, generally once per month during the winter months, beginning in November 1986. These samples were collected and analyzed by the USA WQL.

Although the previously used sampling procedures for the river and tributaries are not identical to those used in this study, the data were considered comparable for dissolved constituents. Martin and others (1992) compared surface-grab and cross-sectionally integrated stream sampling methods and found no significant difference for constituents in the dissolved phase, including nutrients. Significant underrepresentation of suspended sediment concentrations, particularly in the >62 micrometer diameter size fraction, however, was observed for the surface-grab sampling method. This negative bias suggests that the historical data for suspended solids, while not directly analogous to suspended sediment, cannot be fully compared with the data from the present study. Median percent differences between the two methods were reported by Martin and others (1992, p. 872) to



EXPLANATION

Water-quality sampling site, with continuous recording gage (See Table 7.)

1 Tualatin River near Dilley
2 Tualatin River at Rood Road
3 Fanno Creek at Durham Road

Wastewater treatment plant, with continuous record of discharge (See Table 3.)

15 Forest Grove
16 Hillsboro
17 Rock Creek
16 Durham

Water-quality sampliug site, with partial record of streamflow (See Table 7.)

4 Gales Creek at new Highway 47
5 Tualatin River at Golf Course Road
6 Dairy Creek at Highway 8
7 Tualatin River at Highway 219
8 Rock Creek at Highway 8
9 Tualatin River at Elsner Road
10 Chicken Creek on Scholls-Sherwood Road

Minimonitor site (See page 18.)

Water-quality sampling site, with uo streamflow record (See Table 7.)

11 Tualatin River at Highway 210 near Scholls
12 Tualatin River at Boones Ferry Road
13 Tualatin River at Stafford Road
14 Tualatin River at Weiss Bridge

Continuous recording gaging station (See page 15.)

19 Tualatin River at Farmington
20 Tualatin River at West Linn

Figure 10. Sampling stations in the Tualatin River, Oregon, and major tributaries during the winter season (November to April) 1991 to 1992 and November 1992.

be 17 percent for sediment <62 micrometers and 51 percent for sediment > 62 micrometers. Suspended-sediment concentrations in the Tualatin River at West Linn for the period 1974–92 (U.S.Geological Survey, WATSTORE database) have averaged 88 percent finer than 62 micrometers, (mean concentration 22 mg/L, mean flow 1,360 ft^3/s). The preponderance of fines in the suspended sediment in the Tualatin River indicates that the negative bias for suspended solids in the historical data could be about 20 percent.

Biochemical Oxygen Demand

As part of the ambient monitoring program, 5-day BOD and CBOD analyses were performed weekly by the USA WQL on all the main-stem and tributary samples. The WWTP laboratories maintained their schedule of analysis according to their respective permit regulations; generally 5-day BOD and CBOD analyses were performed 3 to 5 times per week, depending on the laboratory. During the storm sampling, analyses were performed daily for all river and tributary samples as well as samples from the larger WWTPs. These analyses were less frequent for the smaller plant samples. During the November 1992 sampling period, 5-day BOD and CBOD analyses were done twice weekly on all river and tributary samples, and daily on the WWTP samples. CBOD samples were treated with 2-chloro-6-(trichloromethyl) pyridine (TCMP), which inhibits nitrification, providing a direct measure of carbonaceous biochemical oxygen demand.

In addition to the ambient monitoring of 5-day BOD and CBOD performed by the WQL, samples were collected for extended CBOD analyses from selected river and tributary sites and analyzed by USGS personnel at the Oregon District laboratory. These extended analyses were done monthly for selected sites during the winter ambient sampling period, and two times for WWTP effluent samples during the storm sampling. Extended CBOD analyses were done twice during the November 1992 sampling period, during the first and third weeks of the month; samples were collected from main-stem and tributary sites (table 7) as well as the four WWTPs. The WWTP effluent was inoculated with bacterial seed by WWTP personnel during the winter sampling; in November 1992, WWTP samples were seeded by USGS personnel with river water. Samples were incubated at 20°C for 25 to 30 days; initial readings were taken within 2 to 7 hours after collection and at intervals of 1 to 4 days throughout the incubation.

Deoxygenation rates (k) and ultimate CBOD concentrations were estimated using an in-house computer program based on Lee's graphical method (Velz 1984). The Lee's grid method (Lee, 1951) uses a series of graphs, which are constructed for a range of k values on the basis of Phelps' law as follows:

$$L_t = L_u \times 10^{-kt} \qquad (1)$$

where

L_u is the initial ultimate oxygen demand at time zero, and

L_t is the oxygen demand at time t.

The graphs are constructed so that a time series of CBOD values, when plotted on the graph for the appropriate rate, produces a straight line. The ultimate CBOD concentration is obtained by extending the line to the right to "ultimate" time. The program computes regression correlation coefficients from linear regression analysis of time-series data from the extended analysis. These correlation coefficients allow an initial "best fit" choice of the rate constant; the estimated k values and ultimate concentrations are then corroborated by visual inspection of Lee's grid plots.

Analysis of the dynamics of CBOD within a river system requires knowledge of the ultimate CBOD load. Because extended CBOD analyses were generally only performed on a monthly basis, it was necessary to estimate the ultimate CBOD concentrations from the 5-day CBOD measurements which were made more frequently. The ratio between the ultimate and 5-day CBOD concentrations is not uniform, however, depending on the rate of decay. The calculated relation between the rate of decay, the demand measured after 5 days, and the ultimate CBOD is summarized in table 8. For the purpose of this study, the 5-day CBOD values were converted to ultimate CBOD concentrations using the appropriate ratio, based on the rates observed at a site during a particular month.

For the November base-flow analysis, extended CBOD analyses were done for WWTP effluent samples using river seed rather than the laboratory seed, which was used for 5-day analysis. For these samples, the decay rates were comparable to the decay rates observed throughout the winter season with the exception of the peak-flow periods (generally k =

Table 8. Calculated relation between 5-day carbonaceous biochemical oxygen demand (CBOD-5) and ultimate CBOD as a function of *k* (base 10 at 20 degrees Celsius)

Rate of decay (*k*)	Ratio of ultimate CBOD to 5-day CBOD	Percent of ultimate CBOD expended after 5 days
0.01	9.20	11
.02	4.86	21
.03	3.42	29
.04	2.71	37
.05	2.29	44
.06	2.01	50

0.02–0.3 day^{-1} at 20°C). It is important to note that, when rates of decay are low, the choice of an appropriate rate constant becomes critical. For example, the calculated value for ultimate CBOD based on a 5-day CBOD = 4.0 mg/L is 19.4 mg/L using $k = 0.02$, and 36.8 mg/L using $k = 0.01$. As a consequence, the ultimate CBOD loads that have been calculated in this study should be considered as estimates. In this report, ultimate CBOD will be referred to as CBOD; when 5-day CBOD is discussed, it will be referred to as CBOD-5.

Sediment Oxygen Demand

In-situ measurements of sediment oxygen demand (SOD) were made at selected sites on several days during October 1992 as part of the Summer Tualatin Project. In these measurements, a sealed bottomless chamber was placed over a known sediment area and DO concentrations were monitored within the volume of water above the sediment layer (method adapted from Larry Caton, Oregon Department of Environmental Quality, written commun., 1992). The chambers were constructed with a cutting edge at the sediment-water interface, which effectively sealed the bottom area of 0.225 m^2 (square meters) of sediment from the rest of the river. Each chamber contained a water volume of 51 liters, which was circulated by a submersible bilge pump with a capacity of 0.15 liters per second. The pump removed water near the top of the chamber and discharged it through three vertical diffusers within the chamber, maintaining velocities within the chamber at about 0.05 ft/s. This speed is comparable to base-flow velocities in the river. The chambers were observed, after the addition of dye, to be well mixed after 3 minutes of pump operation. Measurements of DO were obtained using a Hydrolab water-quality sensor sealed in the center of the chamber.

Two chambers were carefully deployed approximately 10 feet apart at each site; disturbance of the sediment layer during placement was minimized by allowing water to pass through two 10 cm ports in the chamber lid while the chamber was lowered through the water column and deployed. Sediment within the chambers was allowed to settle for 15 minutes after deployment. Outside water was then circulated into and out of the chamber for about 15 minutes to equilibrate conditions within the enclosure; the chambers were then sealed and stabilized for 10 minutes before measurements began. Measurements were taken every 10–15 minutes for about 2 hours. A "blank" chamber, sealed at the bottom, provided a measurement of oxygen demand within the water column only, which was observed to be negligible over the 2-hour period. SOD was calculated as grams of O_2 per square meter per day.

QUALITY ASSURANCE

The quality-assurance program was administered by USGS Oregon District personnel, who provided quality assurance for all three USA laboratories, as well as the USGS National Water Quality Laboratory (NWQL) in Denver, Colorado. Laboratory analytical accuracy, precision, comparability, and reproducibility from week to week were assessed using blank and synthetic reference samples, prepared weekly at the USGS Oregon District Laboratory and submitted to the USA WQL, Durham WWTP Laboratory, and Rock Creek WWTP Laboratory. The samples were prepared using glass-distilled/deionized water and reagent-grade nutrient salts and spanned three ranges of nutrient levels: low, mid, and high, in addition to blanks. In this manner, the full analytical range of the laboratories was tested. In addition, split samples from the churn were collected weekly from one site on the Tualatin River and submitted to the three USA laboratories and the NWQL for analytical interlaboratory comparisons. Because the WQL was responsible for most of the chemical analyses for this study, field duplicate samples were also collected from this site once per week and submitted to assess the analytical precision. Finally, field duplicate measurements of CBOD-5 were made as part of the extended

CBOD analysis by the USGS Oregon District Laboratory; these data provide an estimate of the accuracy of the CBOD-5 data from the USA laboratories. Samples included effluent from the four WWTPs as well as five sites on the Tualatin River and the four major tributaries (table 7).

Data from the analysis of blank samples indicate that laboratory contamination was not a major problem for the analysis of river water and WWTP effluent, which typically contain concentrations from at least two to several hundred times the minimum reporting limit. Of the 25 blanks, eight (32 percent) contained measurable concentrations of ammonia, two (8 percent) contained measurable concentrations of nitrite-plus-nitrate nitrogen and orthophosphate (as phosphorus), and one (4 percent) contained measurable concentrations of total phosphorus. All of these measurements, however, were at or close to the minimum reporting limit. All other samples contained undetectable analyte concentrations.

An assessment of analytical accuracy was performed using data from the synthetic-reference- sample program, with a percentile analysis of the results for each laboratory for each constituent and the range in actual constituent concentrations. To provide some perspective on the data, relative to the concentration of sample submitted, the results are expressed as relative percent difference (100× [observed-actual]/actual). These data are presented in table 9, along with the ranges in concentration for each level of reference sample submitted during the study period. The number of observations ranged from 22 to 27. Samples in the low concentration range show the greatest level of relative bias; the WWTP laboratory data had a positive bias for total phosphorus, orthophosphate, and ammonia at the low level that was pronounced in some cases. As these laboratories analyze WWTP effluent only, however, it is unlikely that nutrient concentrations in the low range would be encountered during the course of their normal operation. The WQL data were within 10 percent relative bias, as defined by the 25th and 75th percentiles, for all constituents except total phosphorus at the low range of concentrations, where a stronger negative bias (about 20 percent) was observed. Concentrations of total phosphorus within this low range are occasionally observed in the Tualatin River. Note also that the concentrations of phosphorus in these synthetic samples were dissolved, rather than associated with particulate material; the negative bias for a total (that is, digested) phosphorus analysis may be even more significant. Data in the mid- and high concentration ranges for all laboratories and constituents indicated very little bias, with median values on the order of 5 percent in the mid-concentration range and less than 2.5 percent in the high concentration range. With the exception of total phosphorus, therefore, data from the synthetic-reference-sample program demonstrated comparability for the concentration ranges typically analyzed by each laboratory.

Analysis of the data from the split-sample program provides additional verification of the interlaboratory comparability and includes the NWQL in addition to the three USA laboratories. These data reflect random sampling errors as well as sample preparation and laboratory analysis. The absolute value of the difference from the mean as relative percent was computed for each analysis ($100\times|\text{observed - mean}|/\text{mean}$); the number of observations for each constituent was 21 to 23. Median values of the percent difference from the mean are presented for each laboratory in table 10, along with the 25th to 75th percentile range and the concentration range for each constituent in the samples analyzed. According to American Public Health Association and others (1989), acceptance limits for nutrient and other inorganic replicate values are within 25 percent for low-level concentrations (less than 0.20 mg/L) and 10 percent for high-level concentrations (greater than 0.20 mg/L). The median values for nutrients for these data groups were generally about 10 percent or less, with the greatest variation shown for orthophosphate analysis, ranging up to 33 percent for the Rock Creek WWTP Laboratory. Median values for total solids (roughly analogous in synthetic samples to TSS) ranged from about 5 to 34 percent. According to the acceptance standards described above, however, most of the data can be considered to be comparable among the four laboratories, although there is more uncertainty for the orthophosphate and TSS data.

Precision analysis of the field-duplicate samples performed by the WQL was done by computing an absolute difference ($|\text{difference}|$ /mean), expressed as percent; the number of analyses ranged from 23 to 26. Median values, as well as the 25th to 75th percentile range, are presented in table 11 and indicate a high level of precision. Values are closely clustered and

Table 9. Accuracy data for reference samples analyzed from November 1991 to April 1992 and during November 1992

[Relative difference ([observed-actual]/actual) reported as percent; number of observations, 22–27; USA, Unified Sewerage Agency; WWTP, wastewater treatment plant; mg/L, milligrams per liter; ammonia, ammonia as nitrogen; nitrate, nitrite-plus-nitrate nitrogen; phosphate, soluble orthophosphate as phosphorus; Total P, total phosphorus]

			Relative difference, in percent								
			USA Water Quality Laboratory			Durham WWTP Laboratory			Rock Creek WWTP Laboratory		
			Percentile			Percentile			Percentile		
Constituent	Range	Concentration (mg/L)	25th	50th	75th	25th	50th	75th	25th	50th	75th
Ammonia	Low	0.03–0.13	-6.0	0	4.9	35.6	56.7	65.1	-6.8	1.7	11.1
	Mid	0.22–1.34	-9.6	-3.4	1.4	2.8	5.8	10.8	-4.8	-3.0	-2.0
	High	1.48–9.23	-6.4	-1.8	1.4	-.2	2.5	5.0	-3.3	-2.2	-1.1
Nitrate	Low	0.09–0.23	-6.7	1.6	1.0	-2.4	4.8	8.7	-1.8	4.8	9.2
	Mid	0.79-1.64	-4.8	-1.7	-.3	-.8	1.0	4.6	-1.3	1.2	3.7
	High	4.51–8.24	-3.6	-2.5	1.1	-3.5	1.9	5.2	-.3	1.2	2.6
Phosphate	Low	0.02–0.05	-8.4	-2.4	1.7	7.7	15.3	30.1	-34.6	-18.6	2.6
	Mid	0.08–0.38	-5.4	-3.6	0	-1.8	1.3	3.3	-7.6	-5.5	-1.6
	High	1.00–2.29	-4.4	-2.4	.7	-1.4	-.4	2.5	-7.4	-2.1	.3
Total P	Low	0.04–0.06	-35.0	-21.5	-2.1	9.0	18.7	29.5	-3.2	9.0	28.2
	Mid	0.12–0.38	-12.4	-5.5	2.6	0	3.4	5.5	-3.1	.2	4.8
	High	1.21–3.11	-8.2	-.7	1.4	-.1	2.2	4.1	-4.0	-.9	0

Table 10. Interlaboratory comparability data for split samples analyzed from November 1991 to April 1992 and November 1992

[Absolute difference, |(observed-mean)|/mean; number of observations, 21–23; USA, Unified Sewerage Agency; WWTP, wastewater treatment plant; mg/L, milligrams per liter; ammonia, ammonia as nitrogen; nitrate, nitrite-plus-nitrate nitrogen: phosphate, soluble orthophosphate as phosphorus; Total P, total phosphorus]

		Absolute difference, in percent							
		USA Water Quality Laboratory		Durham WWTP Laboratory		Rock Creek WWTP Laboratory		National Water Quality Laboratory	
		Percentile		Percentile		Percentile		Percentile	
Constituent	Range in sample concentration (mg/L)	50th	Range 25th–75th	50th	Range 25th–75th	50th	Range 25th–75th	50th	Range 25th–75th
Ammonia	0.05–1.22	6.3	3.8–14.3	13.8	6.6–24.3	4.2	1.4–9.3	9.3	7.3–18.5
Nitrate	0.15–3.85	2.5	2.0–3.9	7.8	4.2–11.7	1.7	0.7–4.4	3.5	2.4–5.1
Phosphate	0.04–0.23	13.0	8.2–19.1	5.0	3.0–20.2	33.3	18.0–48.2	21.6	11.6–27.5
Total P	0.07–0.40	4.6	3.4–10.0	4.6	1.8–11.0	5.9	3.7–10.9	8.8	5.3–20.0
Total solids	46–291	5.3	2.3–9.4	28.2	6.4–34.4	11.3	4.3–18.8	10.3	4.3–15.2

Table 11. Analytical precision data for field-duplicate samples analyzed by the Unified Sewerage Agency (USA) Water Quality Laboratory from November 1991 to April 1992 and November 1992

[Absolute difference, |(observed-mean)|/mean; number of observations, 23–26; mg/L, milligrams per liter; ammonia, ammonia as nitrogen; nitrate, nitrite-plus-nitrate nitrogen; phosphate, soluble orthophosphate as phosphorus; Total P, total phosphorus]

Constituent	Range in sample concentration (mg/L)	Absolute difference, in percent		
		Percentile		
		25th	50th	75th
Ammonia	0.06–1.191	1.4	2.4	3.7
Nitrate	0.99–3.52	.4	.9	2.0
Phosphate	0.05–0.23	.9	2.0	3.4
Total P	0.09–0.35	3.1	6.2	9.4
Total solids	92–170	0	1.8	5.3

generally vary about 2 to 3 percent. The maximum difference is for total phosphorus, but is less than 10 percent.The field duplicate CBOD-5 data were evaluated in a similar fashion, using the absolute difference between the various laboratory values (table 12). The main-stem and tributary samples comprised 56 observations, and 24 samples came from the WWTPs. Analyses of both stream water and WWTP samples had less precision than did the previous analytes, with a median value of close to 20 percent absolute difference. This analysis illustrates the difficulty of precisely measuring CBOD-5.

To summarize, quality-assurance data from the synthetic-reference-sample program and the split-sample program demonstrate that the nutrient data from the various USA laboratories are relatively unbiased and comparable, with a few exceptions. Low-level total phosphorus data (less than 0.06 mg/L) from the WQL was found to be negatively biased by about 20 percent, although bias in the analysis for the mid- and high concentration ranges was low (generally less than 10 percent). Interlaboratory variability was observed to be approximately 10 to 30 percent for orthophosphate, and TSS data were similarly variable. For the other analytes, variability among the four laboratories was found to be acceptable, within 10 percent. Precision for nutrient and TSS analyses done by the WQL

Table 12. Precision data for field-duplicate samples of 5-day carbonaceous biochemical oxygen demand from December 1991 to April 1992 and November 1992

[Absolute difference, |(observed-mean)|/mean; mg/L, milligrams per liter]

Category of sample	Number of observations	Range in sample concentration (mg/L)	Absolute difference, in percent		
			Percentile		
			25th	50th	75th
Main stem and tributary	56	0.10–3.9	0	22.2	48.1
Wastewater treatment plant	24	1.1–6.9	9.6	20.2	38.1

was excellent, most within 5 percent. The CBOD-5 data should be interpreted with caution, because of a median value of 20 percent difference between the duplicate analyses. Considering the use of monthly rate estimates to convert CBOD-5 to ultimate CBOD (described in the section "Biochemical Oxygen Demand"), the values for CBOD loading should be regarded only as estimates.

FACTORS AFFECTING DISSOLVED OXYGEN

The concentration of DO measured in a stream at any given time is the result of a complex interplay of physical characteristics and biological processes. The physical characteristics, such as channel morphology, streamflow, and temperature, provide the structural framework within which the biological processes occur. Dissolved oxygen can be introduced into a water body by several processes, including reaeration from the atmosphere and aquatic-plant or algal photosynthesis. The most important of the biological processes that consume DO during the winter season are the decomposition of organic material and the nitrification of ammonia, referred to as carbonaceous and nitrogenous biochemical oxygen demand, respectively. Sediment oxygen demand is a combination of these two processes occurring in the bottom sediment, using oxygen from the overlying water.

Physical Characteristics

The rate of streamflow, depending on the channel morphology, determines the river's carrying capacity for DO, as a function of the volume of water. As a consequence, when streamflow is high, the river has a larger potential reserve of oxygen, in contrast to low flow, when this reserve declines. Channel characteristics and streamflow also affect the significance of deoxygenation processes in the river system by affecting the residence time for a parcel of water. Long residence times allow deoxygenation processes to become important sinks for DO in the river. When residence times are greatly reduced, however, these deoxygenation processes are less significant. In addition, the streambed provides habitat for aquatic organisms, and consequently the nature of the channel determines the character of the biological community within a river reach.

Temperature affects the concentration of DO in water primarily as a function of oxygen solubility (that is, the saturation potential). The solubility of oxygen in water increases as the temperature decreases; water can hold up to twice the quantity of DO during the cold winter season that it can carry during the summer. Biological processes of decay are also affected by temperature. An increase in temperature increases the rate of deoxygenation reactions, up to the point at which protein denaturation and death occurs in the bacteria mediating the reaction. Similarly, a decrease in temperature causes the decay rates to decrease. At some point, generally around 5 to 10°C, the bacteria will cease multiplying or go dormant and the reactions will essentially stop altogether.

Oxygen Sources

Reaeration, the process in which gases interact with the atmosphere and either enter or leave the water to achieve saturation equilibrium, can be both a source and a sink for DO. It is included here as a source because reaeration is the only significant source that can offset the various deoxygenation reactions occurring in the Tualatin River during the winter. The rate of reaeration is proportional to the oxygen deficit in a given volume of water and becomes highly variable in a changing DO regime. The equation generally used to describe the process is:

$$dC/dt = K_2 (C_s - C) \qquad (2)$$

where

C is the DO concentration in the water, in mg/L;

C_s is the saturation concentration of DO at the appropriate temperature, in milligrams per liter;

K_2 is the reaeration coefficient; and

t is travel time, in hours (Velz, 1984).

The reaeration coefficient is dependent on numerous physical characteristics, including channel depth, slope and irregularity, as well as flow velocity. The channel characteristics and the velocity of flow determine the extent of internal mixing, which affects the rate of dispersion of oxygen under the water surface. Molecular diffusion is the process by which oxygen gas is transferred across the air-water interface, and the rate is proportional to the gradient in concentration between the two points. Mixing is important because it determines the extent of contact renewal between the atmosphere and the water surface, as well as the extent of dispersion between saturated and undersaturated parcels of water in the water column. The equation used in this study to calculate the reaeration coefficient is as follows:

$$K_2 = (20.2 \times U^{0.607}) / D^{1.689} \qquad (3)$$

where

K_2 is the reaeration coefficient, base "e" at 20°C;

U is the mean velocity, in feet per second; and

D is the mean depth, in feet (Bennett and Rathbun, 1972, p. 58).

Mean depth is calculated as the ratio between volume and surface area of a river reach. The reaeration coefficient is temperature dependent; corrections were made to K_2 using the following equation:

$$(K_2)T = (K_2)_{20} \times 1.0241^{T-20} \qquad (4)$$

where

(K_2)T is the reaeration coefficient at temperature T; and

$(K_2)_{20}$ is the reaeration coefficient at 20°C (McCutcheon, 1989).

Photosynthesis, the process by which carbon dioxide is used by plants for growth, is an important source of DO during the summer. Respiration, on the other hand, is the process in which DO is consumed in the utilization of organic material for energy and can be an important sink for DO. During warm summer months, when there is abundant light and adequate nutrients are available, significant communities of aquatic plants can develop in a river system. Large quantities of DO can be introduced to the aquatic system during the day under these conditions. Photosynthesis and respiration occur concurrently during light periods; at night, when photosynthesis ceases, respiration can result in significant decreases in DO. The result is a distinct diel variation in the DO concentration. In addition, algal communities under eutrophic conditions exhibit cycles of increase and decline in biomass that commonly are associated with light availability. DO concentrations reflect these cycles, ranging from conditions of supersaturation (during peak periods of primary production or growth of algal biomass) to severe oxygen depletion (when the algae die and subsequently decompose). In the winter, when light levels are reduced, photosynthetic production of oxygen in some rivers typically becomes insignificant.

Oxygen Demand

Carbonaceous Biochemical Oxygen Demand

CBOD is a measure of the oxygen consumption that results from the oxidation of organic carbon by aquatic organisms, primarily heterotrophic bacteria. Assuming that DO is plentiful, the reaction is first order, and the quantity of oxidizable material is acted upon at a constant rate. "Phelp's Law" (Phelps, 1944) provides the basis for a quantitative analysis of the extent of carbonaceous deoxygenation as follows: "The rate of biochemical oxidation of organic matter is proportional to the remaining concentration of unoxidized substance, measured in terms of oxidizability." The mathematical expression becomes:

$$-dL/dt = KL \qquad (5)$$

which can be integrated and rearranged to obtain

$$log_e(Lt/L) = -Kt$$

or

$$log_{10}(Lt/L) = -0.434Kt = -kt$$

which becomes

$$L_t = L \times 10^{-kt}$$

where

L represents the initial oxygen demand at time zero (also known as the ultimate demand);

L_t represents the oxidizability remaining at time t; and

K (base "e"), or k (base 10), is the rate constant for deoxygenation (Velz, 1984).

The rate of oxygen depletion depends on the nature of the organic material (the amount of oxidizable carbon), with lower rates of decay associated with more refractory material. For rivers receiving sewage effluent that only has had primary treatment, rate constants are approximately 0.1 (base 10 at 20°C); as secondary treatment has become more prevalent, observed rate constants have decreased accordingly.

As is the case with many biological processes, temperature affects the carbonaceous deoxygenation rate. Equations that describe the effect of temperature on bacterial processes commonly have follow the general form:

$$k_1 = k_2 \times \theta^{(T1-T2)} \qquad (6)$$

where

k_1 is the rate of the reaction at temperature T_1,

k_2 is the rate of the reaction at T_2, typically equal to 20°C (Thomann and Mueller, 1987), and

θ is the temperature adjustment coefficient.

The value of θ varies somewhat and has been empirically derived for this type of reaction. For this study, laboratory rates of CBOD decay were adjusted using a θ of 1.047 (Bowie and others, 1985).

Nitrogenous Biochemical Oxygen Demand

Nitrification is the process by which ammonia (in the form of the ammonium ion) is oxidized to nitrite by *Nitrosomonas* bacteria and then to nitrate by *Nitrobacter* bacteria. The oxygen demand of this process can be significant because approximately 4.3 mg of oxygen is consumed per milligram of ammonium-N oxidized to nitrate-N (Thomann and Mueller, 1987; Velz, 1984). First-order kinetics are generally assumed for ammonia concentrations typical of natural waters, and the mathematical expression is identical to the equation describing biochemical oxygen demand:

$$-dN/dt = K_n N \qquad (7)$$

where

N is the concentration of ammonium-N in mg/L, and

K_n is the nitrification rate constant.

Nitrifying bacteria are slow growers and require substrate for attachment; in aquatic systems they are typically associated with shallow and rocky river reaches. They can become important in deep reaches, however, when the time of travel is adequate for the populations to develop and sufficient suspended particulates are present for substrate. These bacteria are sensitive to environmental conditions, especially temperature and pH; the optimum conditions are warm temperatures (28–36°C) and slightly alkaline pH (approximately 8.5). Within the temperature range of 10 to 30°C, the effect of temperature on the rate of nitrification is assumed to be the same as that previously described for CBOD decay. For temperatures less than 10°C, however, the reproduction of nitrifying bacteria is adversely affected. As a result, the rate of nitrification is generally considered to decrease rapidly below 10°C and be essentially zero at 5°C (Thomann and Mueller, 1987).

Sediment Oxygen Demand

Processes that consume oxygen at the sediment-water interface include:

(1) The microbial decay of allochthonous organic matter (matter that originated outside the system, such as leaf litter) or autochthonous organic matter (matter that originated within the system, such as algae) that has been deposited on the bottom;

(2) Nonbiochemical oxidation-reduction reactions; and

(3) Respiration by benthic invertebrates.

Factors that affect SOD include temperature, the characteristics of the sediments and the biological benthic community, and the water velocity near the sediment-water interface. Water depth is also a factor in controlling the effect of SOD on the overlying water column, as illustrated by the general equation for SOD:

$$dC/dt = -SOD/H \qquad (8)$$

where

C is the concentration of DO in the water column;

SOD is the quantity of oxygen consumed per unit bottom area per unit time;

H is the mean water depth; and

t is time (Thomann and Mueller, 1987).

The effect of temperature on SOD was calculated for the purpose of this study according to the general equation for bacterial processes described above. A value for θ of 1.065 was used; below 10°C, it has been reported that SOD decreases significantly, and is essentially zero at 5°C (Thomann and Mueller, 1987).

DISSOLVED OXYGEN REGIME DURING WINTER FLOW CONDITIONS

Streamflow

Streamflow near the mouth of the Tualatin River at West Linn (14207500) during the water year 1992 exhibited the typical seasonal pattern of summer low flow and winter high flow (fig. 11). During the winter, streamflow in the river varied greatly, ranging from less than 200 ft^3/s in November 1991 to 5,000 ft^3/s during the storm sampled in early February 1992 (fig. 12). After mid-November, however, flows were within the 500 to 2,000 ft^3/s range throughout most of the season; this flow range was considered to be the winter low-flow range for the purpose of this study. Streamflow at RM 33.3 (14206500) was approximately 90 percent of the flow at the mouth throughout the winter. The tributary flow patterns during the winter sampling period were similar to those of the main stem (figs. 13 and 14), as were the effluent flows from the WWTPs (fig. 15). Generally, under winter low-flow conditions, the total effluent discharge accounted for less than 10 percent of the streamflow in the Tualatin River at West Linn. Estimated travel times for the length of the study reach for these flow conditions, although variable due to the unsteady flow, were about 2 to 6 days (fig. 6).

Peak flows greater than 2,000 ft^3/s occurred during several storms. The stormflow period that occurred January 29 to February 5, 1992, was the

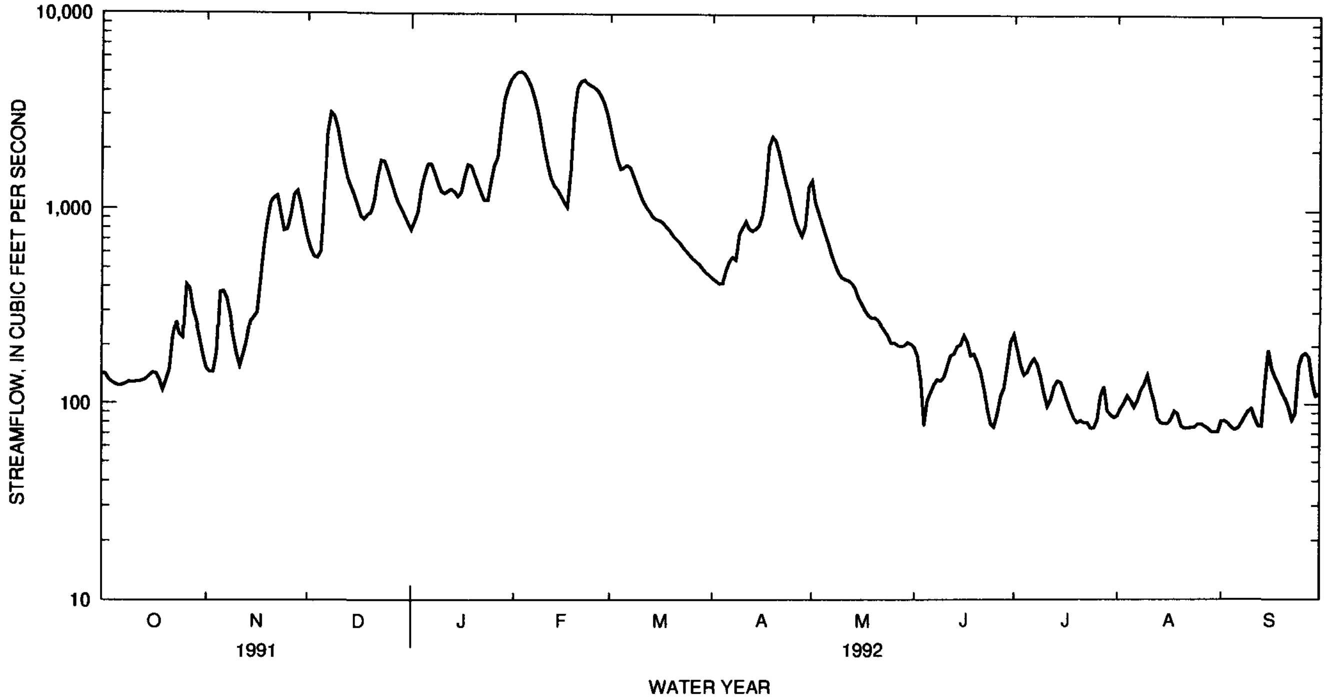


Figure 11. Daily mean streamflow in the Tualatin River at West Linn, Oregon, during the 1992 water year.

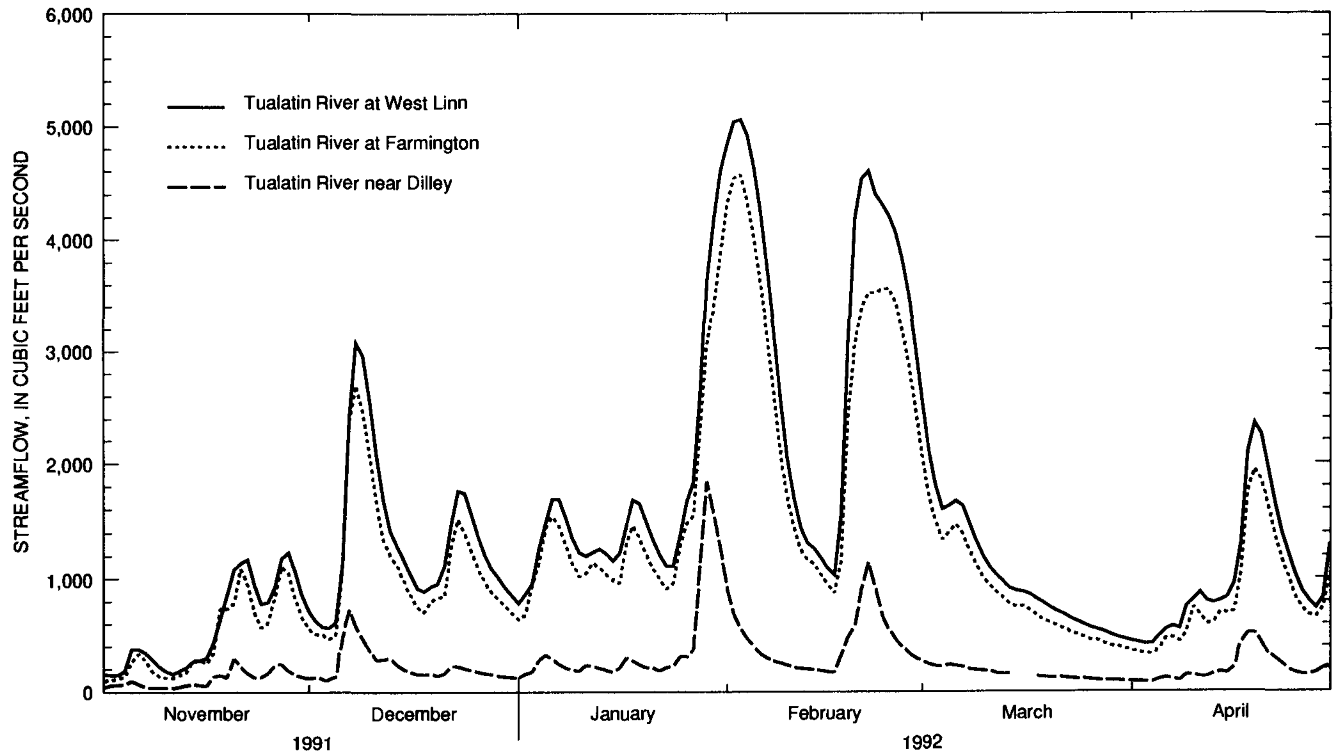


Figure 12. Daily mean streamflow in the Tualatin River, Oregon, at selected sites, November 1991 to April 1992.

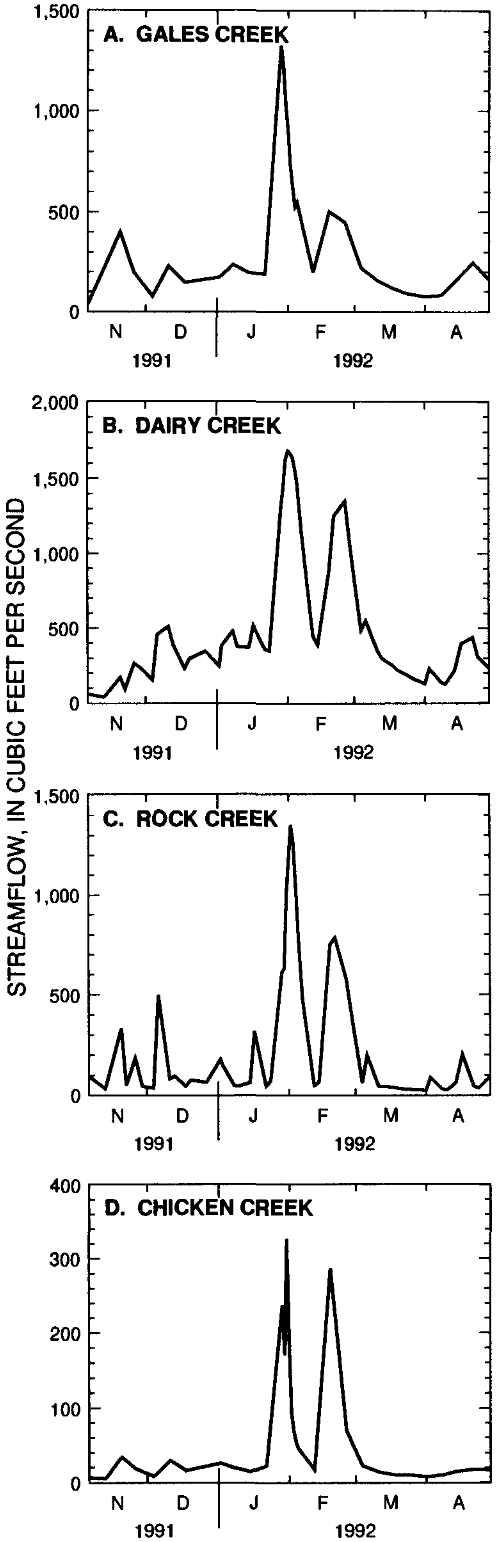


Figure 13. Instantaneous streamflow in tributaries to the Tualatin River, Oregon, November 1991 to April 1992. Measurements of stage converted to streamflow on the basis of rating curves maintained by the Oregon Water Resources Department (written commun., 1992). (A) Gales Creek, (B) Dairy Creek, (C) Rock Creek, and (D) Chicken Creek.

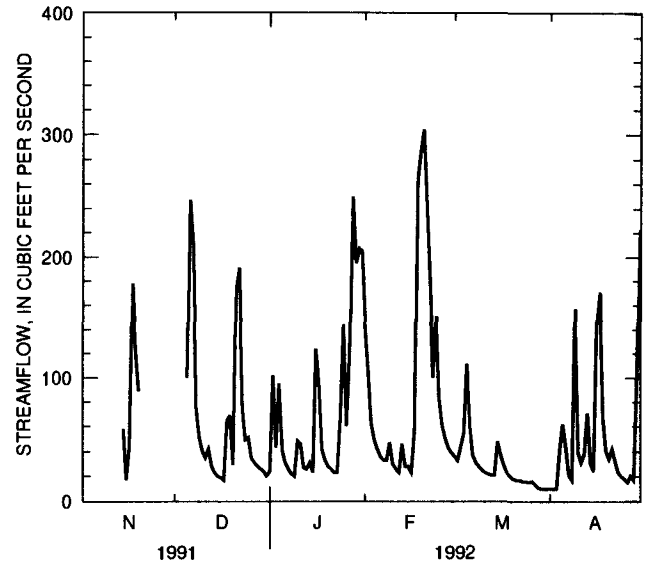


Figure 14. Daily mean streamflow in Fanno Creek at Durham, Oregon, November 1991 to April 1992. Some data missing in November and early December 1991.

first major storm of the season and was sampled extensively. Streamflow in the Tualatin River and the major tributaries increased by nearly fivefold during this storm; the residence time for the study reach during this period was estimated to be about 1 to 2 days (fig. 6). The increase in effluent discharge from the various WWTPs was less than in the river and tributaries by about one-half. Although large increases in effluent discharge coincided with peak streamflow in the river, the flow capacity for secondary treatment was not exceeded in any WWTPs during this sampling period. As a consequence, the WWTPs maintained the capacity to effectively treat the increased flow.

The pattern of flow during the storm varied among the tributaries and sites along the main stem (fig. 16). The hydrographs for Gales and Fanno Creeks (14206950), as well as the upper Tualatin River near Dilley (14203500), have sharp peaks that fall off rapidly. Samples were not taken from these streams during the rising limb of the hydrograph; constituent loads for these sites during the storm, therefore, are probably biased low. Streamflow in Dairy Creek, and to a lesser extent in Rock Creek and the main stem near the mouth (14207500), had a broader peak, which was sustained for 4 to 5 days before gradually declining. Overall, about 98 percent of the mean streamflow at the mouth of the Tualatin River was accounted for during the peak-flow period by the mean inflow from the major tributaries, while effluent discharge

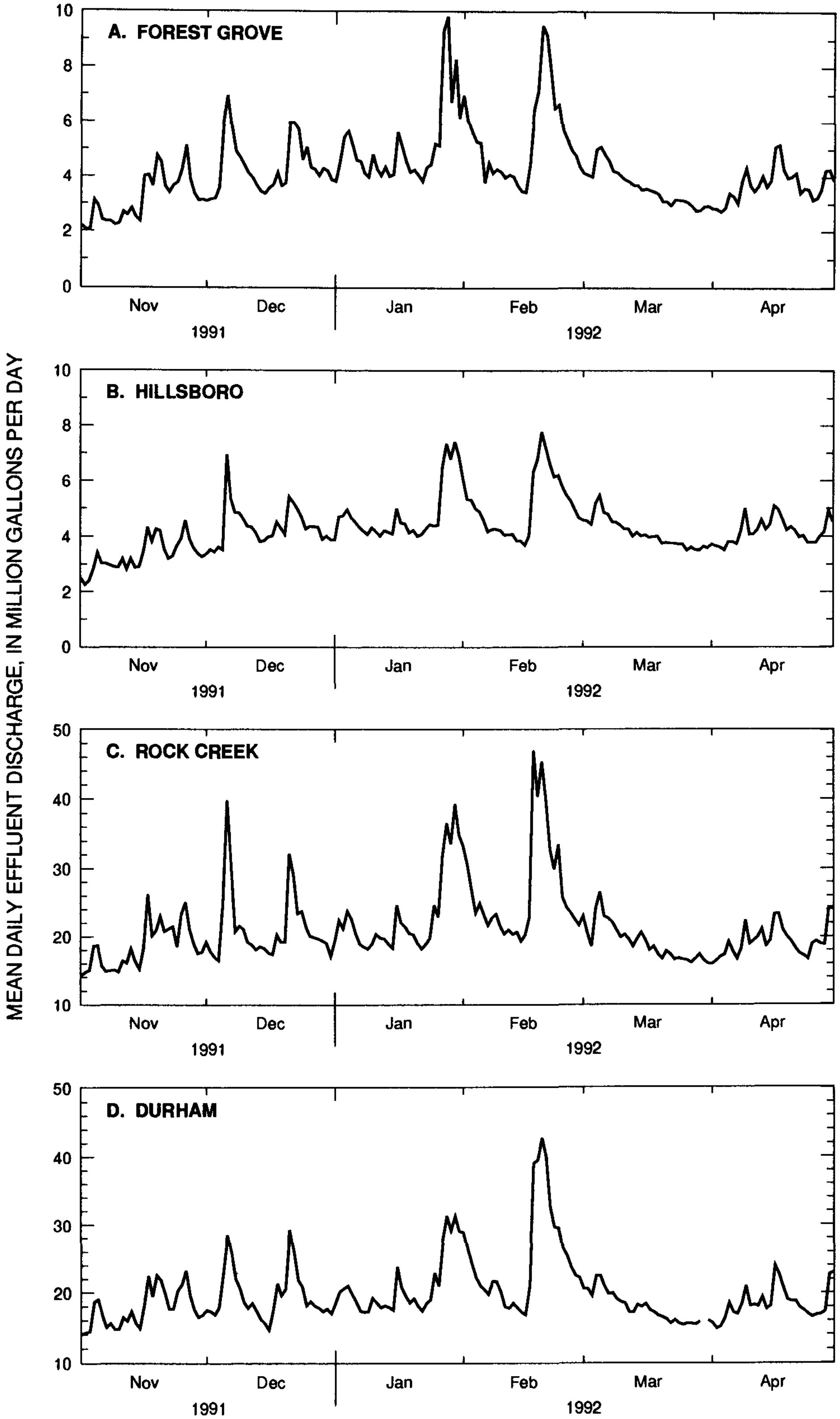


Figure 15. Daily mean effluent discharge to the Tualatin River, Oregon, from wastewater treatment plants, November 1991 to April 1992.

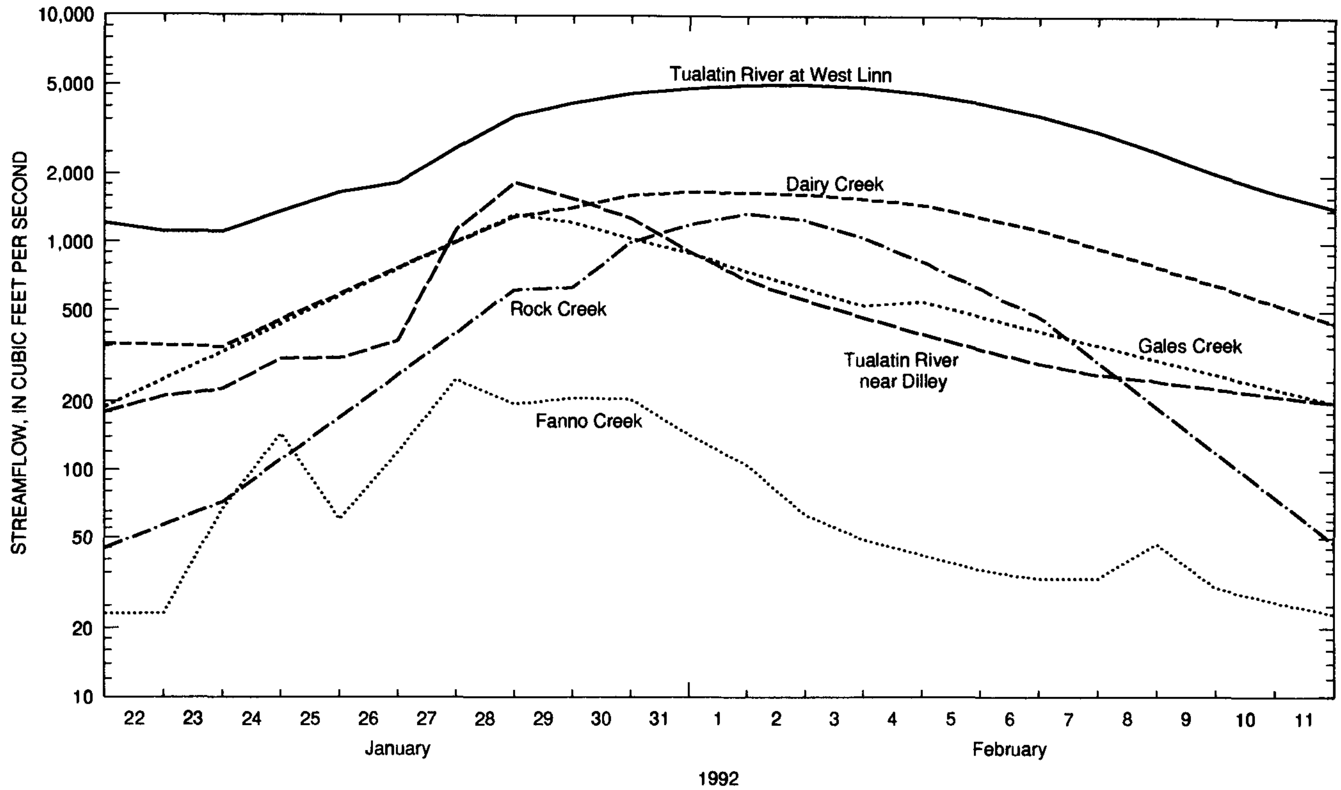


Figure 16. Daily mean streamflow in the Tualatin River at West Linn, Oregon, and near Dilley, Oregon, and instantaneous streamflow in the major tributaries, January 22 to February 12, 1992. Rating curves for tributary streamflow maintained by Oregon Water Resources Department (written commun., 1992).

from the WWTPs accounted for approximately 2 percent.

After a brief storm in early November 1992, the streamflow at the Tualatin River at West Linn remained less than 300 ft^3/s for about 14 days, with mean flow of 219 ft^3/s (256 ft^3/s if flow diverted by the Lake Oswego Canal is included) during the period November 6–20 (fig. 17). This volume of base flow would be expected to occur during November approximately every 3 to 4 years. Similarly, mean streamflow in the major tributaries during this period was low, about 5 to 10 times lower than mean winter low-flow conditions. Effluent discharge from the WWTPs during the base-flow period was also reduced, although less significantly—generally less than twofold. In contrast to the peak-flow conditions, the mean tributary input during base-flow conditions accounted for significantly less of the mean streamflow in the main stem, about 76 percent, while effluent discharge constituted about 24 percent of the mean main-stem streamflow. The travel time for the study reach was about 16 days during this period (fig. 6).

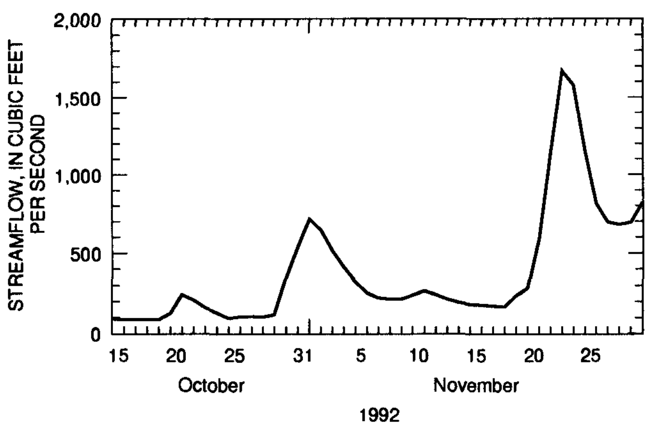


Figure 17. Daily mean streamflow in the Tualatin River at West Linn, Oregon, October 15 to November 30, 1992.

Water Temperature

Water temperature in the Tualatin River during the 1992 water year followed a distinct annual pattern (fig. 18). During the winter season, mean water temperature decreased significantly from the summer range of 20 to 25°C; values as low as 5°C were reached during the coldest months of December and January. The water temperature increased slightly during the peak-flow period in January-February, although it remained below 10°C until March. From November 1991 to April 1992, temperature change was slight as the water flowed downstream (fig. 19).

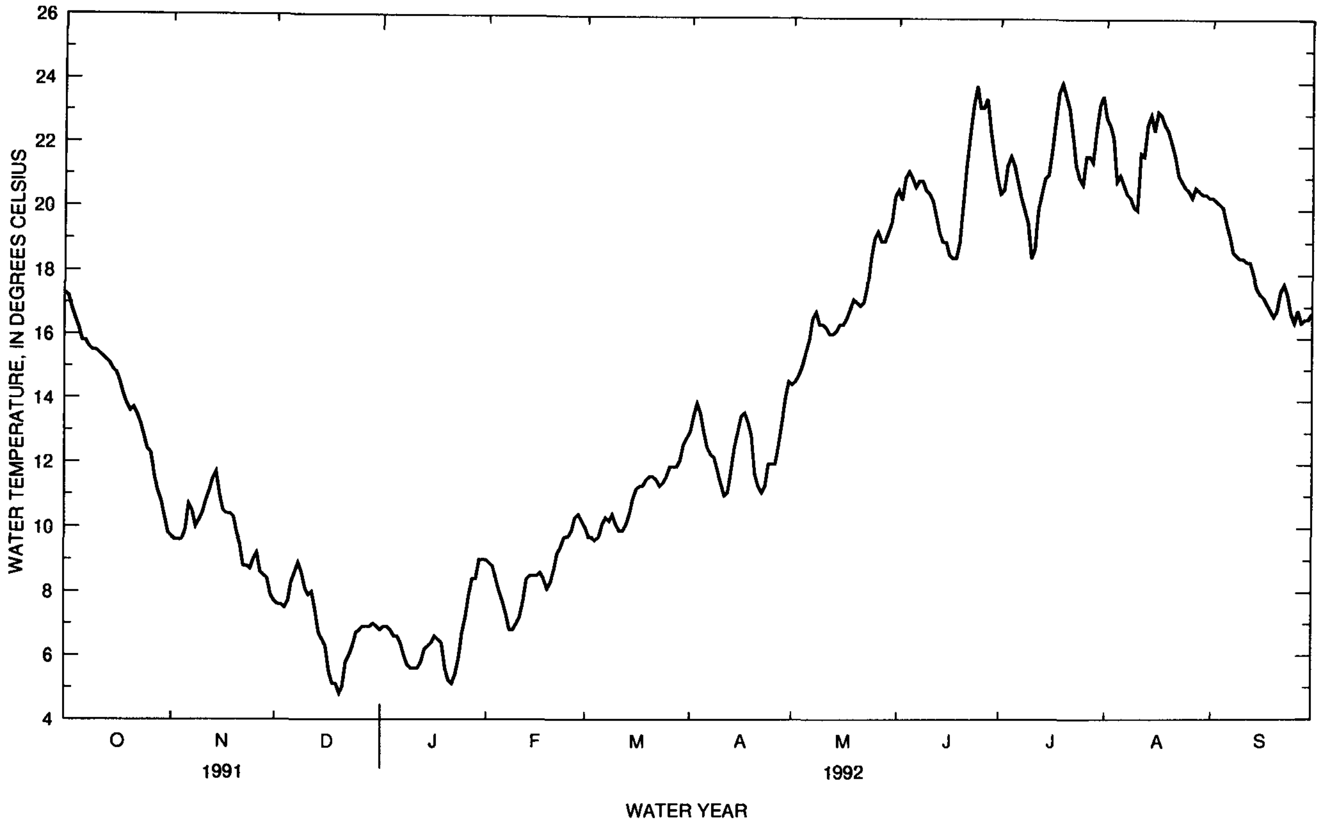


Figure 18. Daily mean water temperature of the Tualatin River, Oregon, at river mile 3.4 during 1992 water year.

During the month of November 1991, water temperature ranged from approximately 8 to 12°C, intermediate between the winter and summer extremes. During November 1992, the temperatures were greater than 10°C in the lower river until the end of the base-flow period (fig. 20).

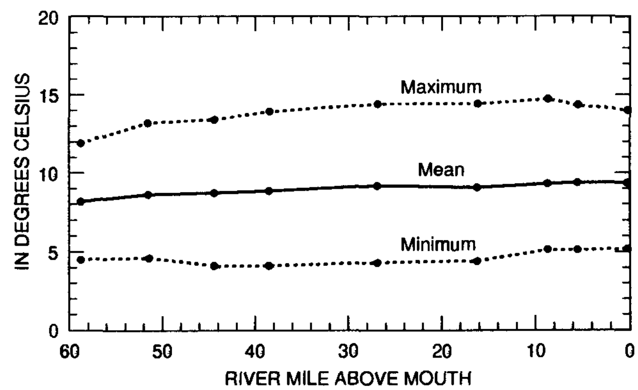


Figure 19. Mean, minimum, and maximum water temperatures for selected sites in the Tualatin River, Oregon, November 1991 to April 1992.

Carbonaceous Biochemical Oxygen Demand Decay Rates and Concentrations

The rates of CBOD decay (base 10 at 20°C) observed in the laboratory during the winter season 1991–92 and during November 1992 are presented in table 13. Low decay rates were obtained for the main stem, ranging from 0.01 to 0.04 day^{-1}; in the tributaries the rates tended to be slightly higher, though still low (0.0–0.06 day^{-1}). Rates of decay in the effluent from the WWTPs were also quite low, ranging from 0.01 to 0.06 day^{-1}. These low rates reflect the secondary treatment of wastewater prevalent in the Tualatin River Basin.

The highest rates of CBOD decay were generally observed during a storm in early February. The increase in deoxygenation rates in the main stem and tributaries during the period of peak flow is consistent with the hypothesis that material entering. the river during high flows is less oxidized and therefore more susceptible to bacterial decay Similarly, the observed increase in rates for the WWTP effluent from Rock Creek and Durham early in the storm suggests that treatment efficiency was decreased. Even these slightly elevated rates remained low, however, and indicate that treatment was not seriously compromised during this storm. Decay rates during the base-flow

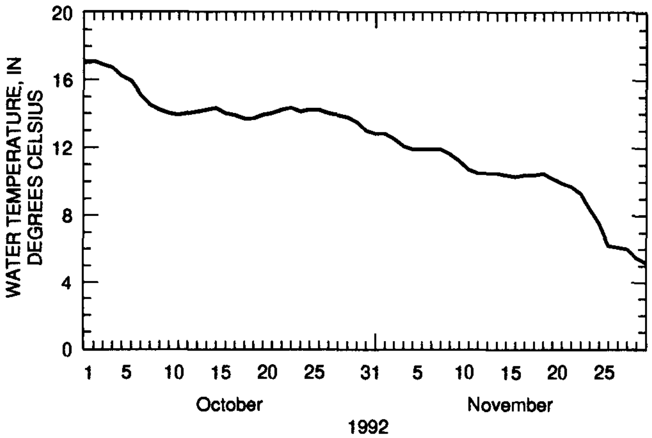


Figure 20. Daily mean water temperature of the Tualatin River, Oregon, at river mile 3.4, October to November 1992.

period were similar to rates observed during the low-flow condition and averaged about 0.02 day^{-1}.

CBOD concentrations in most of the major tributaries, except for Fanno Creek and at selected sites in the main-stem Tualatin River, were found to decrease slightly with increasing streamflow (table 14). This dependence is consistent with a dilution effect. The pattern was similar for the WWTPs during peak streamflow conditions. Mean concentrations of CBOD were less than those observed during winter low flow for all the WWTPs. During the base-flow period, however, the effect of dilution was apparently not a factor, despite reductions in effluent volume. Mean concentrations of CBOD at Rock Creek and Durham, in particular, were considerably less than those associated with winter low-flow conditions. This reduction in CBOD was probably a consequence of the WWTP management process; treatment was more consistent during base-flow conditions than during the wide range of streamflow under winter low-flow conditions.

Factors Affecting Dissolved Oxygen during Winter Low-Flow Conditions

The pattern of DO concentrations at RM 3.4 (14207200) during the 1992 water year (fig. 21) illustrates the extreme fluctuations during the summer, caused by the alternating bloom and die-off cycle of phytoplankton, and the relatively stable concentration during winter, resulting from insignificant algal productivity. DO concentrations during the winter sampling season remained above 8 mg/L for most of the season, except for November 1991 and part of April 1992. Those two periods of low DO concentrations coincided with periods of lower flow and were especially pronounced in November (fig. 11). During most of that winter season, the percent saturation concentrations for DO remained in the 80 to 90 percent range,

Table 13. Laboratory decay rates of carbonaceous biochemical oxygen demand (CBOD) observed in samples from the Tualatin River Oregon, major tributaries, and wastewater treatment plants during November 1991 to April 1992 and November 1992

[Streamflow in cubic feet per second; N, number of sites; WWTP, wastewater treatment plant; --, no data available; *, samples seeded with Tualatin River water instead of WWTP seed]

		Decay rates of CBOD in base 10 at 20 degrees Celsius								
		Tributaries					Wastewater treatment plants			
Date	Streamflow in Tualatin River at West Linn	Tualatin River N=5	Gales Creek	Dairy Creek	Rock Creek	Fanno Creek	Forest Grove WWTP	Hillsboro WWTP	Rock Creek WWTP	Durham WWTP
12-4-91	565	0.02	0.03	0.03	0.04	0.01	0.04	0.03	0.02	0.01
1-8-92	1,530	.02	.04	.03	.03	.04	.04	.03	.02	.01
1-30-92	4,170	--	--	--	--	--	.03	.03	.06	.05
2-3-92	5,060	--	--	--	--	--	--	.03	.04	.01
2-5-92	4,630	.04	--	.06	.02	.05	.04	.02	.05	.01
3-4-92	1,600	.02	.01	.01	.01	.02	.04	.02	.03	.01
4-1-92	444	.04	.03	.04	.05	.05	.04	.04	.03	.01
11-3-92	511	.02	.02	.02	.02	.02	.01*	.02*	.03*	.02*
11-17-92	162	.01	.03	.01	.01	.04	.01*	.02*	.02*	.02*

Table 14. Mean concentrations of ultimate carbonaceous biochemical oxygen demand (CBOD), categorized by selected ranges of streamflow in the Tualatin River at West Linn, Oregon (U.S. Geological Survey streamflow-gaging station 14207500) at selected sites in the main-stem river, major tributaries, and wastewater treatment plants during November 1991 to April 1992 and November 1992)

[Concentrations calculated from measurements of 5-day CBOD on the basis of decay rates determined monthly; N, number of observations; WWTP, wastewater treatment plant; ft^3/s, cubic feet per second]

Site	River mile	Mean ultimate CBOD concentrations, in milligrams per liter					
		Base flow[1]		Winter low flow[2]		Peak flow[3]	
		N	CBOD	N	CBOD	N	CBOD
Tualatin River near Dilley	58.8	4	2.7	18	2.3	7	1.7
Gales Creek	56.7	4	2.2	18	1.6	8	.8
Forest Grove WWTP	55.3	13	13	58	11	5	10
Dairy Creek	44.8	4	4.4	18	2.7	8	1.2
Hillsboro WWTP	42.8	13	15	87	17	5	11
Rock Creek	38.1	3	4.7	18	2.7	8	1.9
Rock Creek WWTP	38.1	13	12	83	23	8	10
Fanno Creek	9.4	3	1.8	18	5.1	8	1.6
Durham WWTP	9.4	12	12	72	21	7	15
Tualatin River at Weiss Bridge	.2	4	4.3	18	3.0	8	1.7

[1] less than 300 ft^3/s, November 6–20, 1992.
[2] 500–2,000 ft^3/s, November 1991 to April 1992.
[3] greater than 2,000 ft^3/s, January 29 to February 5, 1992.

indicating that some oxygen demand persisted despite increased streamflow and decreased temperature. In addition, diel variation of DO was minimal, indicating that photosynthesis was insignificant except for a period of about 2 weeks in late March and early April (fig. 22).

Mean DO concentrations throughout the length of the study reach during winter low-flow conditions were in the range of approximately 9 to 11 mg/L (fig. 23), representing saturation concentrations of about 80 to 90 percent. As the water moved downstream, mean DO concentrations decreased steadily by 1.7 mg/L over the entire reach; the greatest mean DO loss (65 percent of the total mean loss) occurred during passage through the slow-moving backwater reach, below RM 38.5. In this part of the river, the rate of reaeration is very low due to the increased depth and reduced velocities. DO concentrations consistently increased from RM 3.4 to RM 0.2 due to increased velocities and reaeration below the LOC diversion dam.

Wastewater Treatment Plant and Tributary Loads

The relative contributions of mean CBOD, ammonia, and TSS loads from the major WWTP and tributary sources varied significantly during the winter low-flow conditions for the period November 1991 to April 1992 (table 15). Considering the uncertainty inherent to CBOD analysis, the total mean loading of CBOD was essentially the same for the WWTP and tributary sources, with 42 percent from WWTPs and 58 percent from the tributaries. Ammonia loading was almost exclusively from WWTPs, which contributed 95 percent of the total mean load. In contrast, loading of TSS was dominated by the tributary sources, which accounted for 99 percent of the total mean load.

Because most of the major WWTP and tributary sources enter the main stem upstream from RM 38, a significant proportion of the total mean CBOD load must travel through the entire backwater reach. Nearly 75 percent of the total mean CBOD load had entered the river above RM 38, and that was probably a factor in the loss of DO observed downstream from that point. The tributaries contributed a slightly higher mean proportion of this upper river load; the Durham WWTP, which contributed a significant proportion of the total mean WWTP CBOD load (nearly 40 percent), discharges to the main stem at RM 9.3. The mean CBOD load from Fanno Creek, the only major tributary in the lower river, constituted about 20 per-

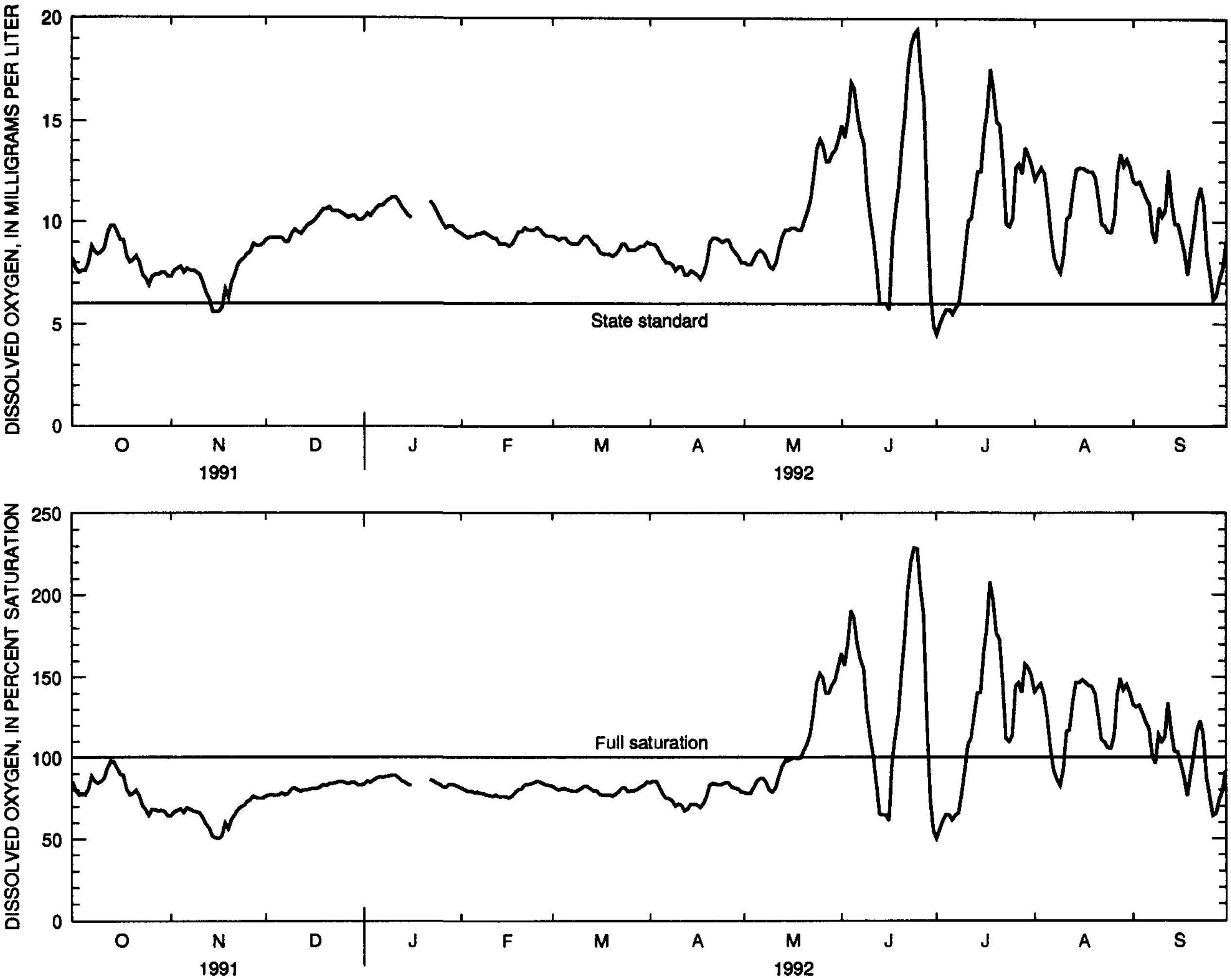


Figure 21. Daily mean dissolved oxygen in the Tualatin River, Oregon, at river mile 3.4 during 1992 water year.

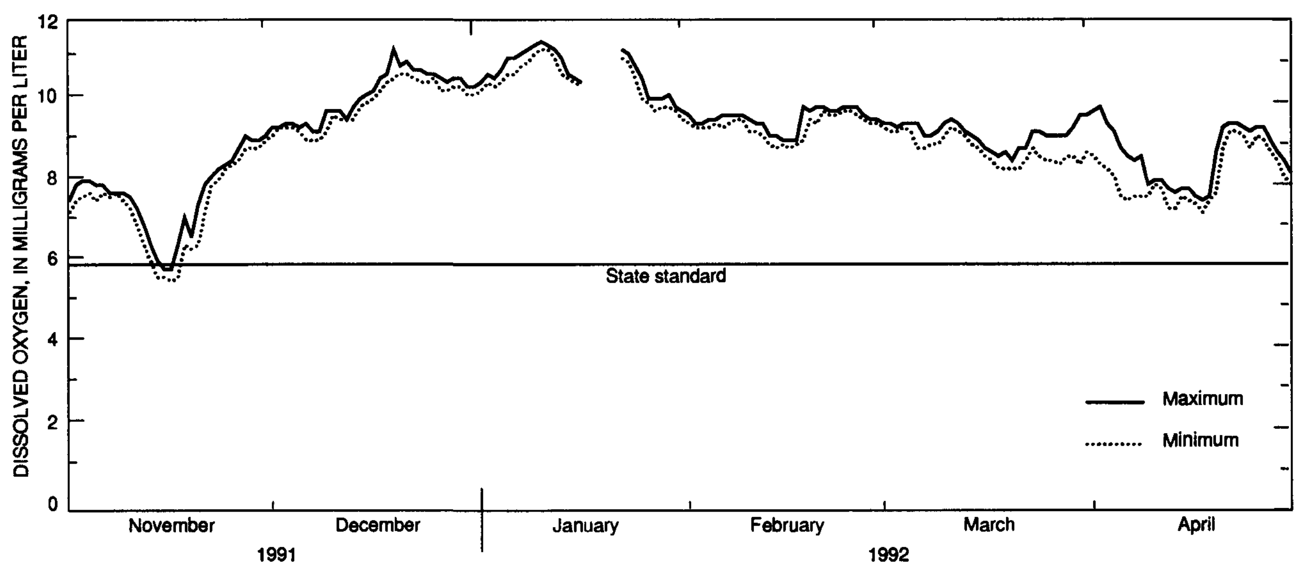


Figure 22. Maximum and minimum daily dissolved oxygen concentrations in the Tualatin River, Oregon, at river mile 3.4, November 1991 to April 1992. Data missing from January 17–21, 1992.

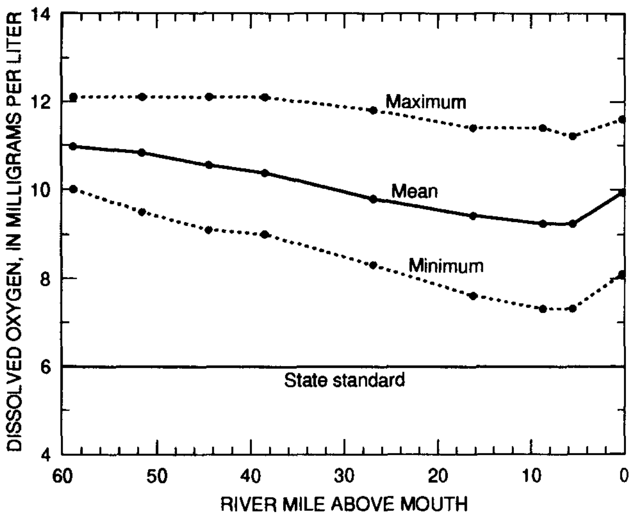


Figure 23. Maximum, mean, and minimum dissolved oxygen concentrations at selected river miles in the Tualatin River, Oregon, during winter low-flow conditions, November 1991 to April 1992 (streamflow in the Tualatin River at West Linn, Oregon, equals 500–2,000 cubic feet per second).

cent of the total tributary load. At the high velocities observed in the river during the winter season, the travel time from Fanno Creek and the Durham WWTP to the mouth of the Tualatin River was typically less than 1 day; consequently, the loads from these sources should have had significantly less of an effect on the system than loads from farther upstream.

In-River Decay of Carbonaceous Biochemical Oxygen Demand

Concentrations of CBOD in the river were moderate, averaging between 1.8 and 2.4 mg/L, and typically did not exceed 4 mg/L. Under winter low-flow conditions, concentrations in the upper river and the tributaries (with the exception of Rock and Dairy Creeks) increased slightly as streamflow increased, possibly because of the incorporation of vegetative organic debris. In contrast, concentrations in the WWTP effluent did not show any relation to river flow, presumably as a result of WWTP treatment procedures. Note that the fraction of particulate CBOD in the river was probably less significant than the dissolved fraction, because the mean concentrations of TSS throughout the river were low, on the order of 10 to 20 mg/L.

From a management perspective, it is useful to evaluate the relative significance of loading to the upper and lower river, particularly in context with the level of streamflow in the river. For the purpose of this analysis, the river was divided into three reaches: RMs 58.8–38.5, 38.5–9.3, and 9.3 to the mouth. The assumption was made that CBOD loss in the river was due entirely to biochemical decay with no settling. The mean decay rate observed during the winter season (k=0.02 day^{-1}, base 10, at 20°C), corrected for the mean in-river temperature (9°C), was used to estimate the consumption of DO due to the known sources of CBOD (fig. 24). These theoretical estimates were calculated for the two streamflows that defined winter low-flow conditions (500 and 2,000 ft^3/s) using the time of travel estimates shown in figure 6.

This analysis, while essentially qualitative in nature, indicates that the effect of CBOD loading was mainly controlled by the streamflow in the river. In each of the selected reaches, the theoretical DO consumption at 500 ft^3/s was approximately 5 to 10 times the consumption at 2,000 ft^3/s. In addition, the analysis indicates the extent to which CBOD loading to the upper river causes a greater reduction in DO, because of the increased in-river residence time, than does the loading to the lower river. The calculated consumption of DO in the two upper river reaches combined was approximately 4 to 5 times the DO consumed in the lowest reach. This effect was most significant at the lower flow level because the DO loss was much greater. A proportionally greater fraction of the upper river load is attributable to tributary sources under winter low flow conditions because of the distribution of WWTPs and tributaries within the river basin. The implication for management, therefore, is that further reduction of WWTP CBOD loads during the winter season will have only a limited effect on DO consumption in the river.

Other Microbial Processes Affecting Dissolved Oxygen

The low temperatures observed in the river during the winter, averaging less than 10°C, are considered inhibitory to the growth of nitrifying bacteria (Thomann and Mueller, 1987). Consequently, it is doubtful that a significant population of nitrifiers would be present during the winter, even in the upper river, where the habitat is more suitable. In addition, the reduced travel times during the winter make it unlikely that nitrifying bacteria could have established a significant population in the lower river before being flushed from the system. Therefore, in spite of increased ammonia loading, nitrification was not considered to be a significant factor in the loss of DO during winter low-flow conditions.

Table 15. Mean loads of ultimate carbonaceous biochemical oxygen demand (CBOD), total suspended solids (TSS), and ammonia as nitrogen to the Tualatin River, Oregon, during winter low-flow conditions, November 1991 to April 1992

[Streamflow in the Tualatin River at West Linn, 500–2,000 cubic feet per second; WWTP, wastewater treatment plant; --, not applicable]

Site	River mile	Mean loads, in thousand pounds per day					
		CBOD		TSS		Ammonia	
		WWTP	Tributaries	WWTP	Tributaries	WWTP	Tributaries
Tualatin River near Dilley	58.8	--	2.9	--	26	--	0.03
Gales Creek	56.7	--	1.7	--	20	--	.02
Forest Grove WWTP	55.3	0.37	--	0.30	--	0.01	--
Dairy Creek	44.8	--	3.9	--	31	--	.08
Hillsboro WWTP	43.9	.61	--	.16	--	.42	--
Rock Creek	38.1	--	1.4	--	9.5	--	.01
Rock Creek WWTP	38.1	4.2	--	.51	--	2.0	--
Fanno Creek	9.3	--	2.4	--	17	--	.02
Durham WWTP	9.3	3.4	--	.49	--	.90	--
Subtotal		8.6	12	1.5	104	3.3	.16
Total		20.6		106		3.46	
Percent		42	58	1	99	95	5

Values for SOD in the lower river ranged from 0.5 to 2.6 g O_2/m^2 d (grams of oxygen per square meter per day) during October 1992 (table 16). The temperatures associated with these measurements were in the range 14.5 to 16°C, with water depths ranging from 9 to 20 ft. For these temperature and flow conditions (generally less than 500 ft^3/s), the loss in DO from SOD could be extensive. During the winter sampling season, however, temperatures generally remained less than 10°C, with an extended period of temperatures close to 5°C; under these conditions the effect and extent of SOD processes would be greatly attenuated (Thomann and Mueller, 1987). In addition, because of the inverse relation between SOD and the water depth in determining the loss of DO in the water column, the effect is decreased with increased water volume, or reservoir for oxygen, in the system. Higher flows were coupled with reduced temperatures during winter low-flow conditions; therefore, the effect of SOD was not considered likely to be significant.

Table 16. Mean rates of sediment oxygen demand (SOD) for selected river miles on the Tualatin River, Oregon, during October 1992

[Number of observations, 2; SOD rate in grams oxygen per square meter per day]

River mile	SOD	Depth (feet)	Temperature (degrees Celsius)	Type of bottom material
20.3	1.9	15.3	14.5	Silt, soft clay, organic material
16.7	1.3	15.8	15.0	Mucky, organic material
8.6	.5	9.3	15.0	Coarse sand
5.4	2.6	20.4	16.0	Mucky, organic material

Tributary Input of Dissolved Oxygen

In general, DO concentrations in the major tributaries were close to saturation (90 percent) under winter low-flow conditions, averaging 10 to 11 mg/L. DO concentrations in Gales Creek were similar to those in the main stem at the point where the two joined, causing little change in the DO concentrations of the river. DO concentrations in Dairy Creek and Rock Creek, however, were generally lower than DO concentrations in the upper main stem, causing a reduction in DO as the waters mixed. This effect was particularly significant for Dairy Creek, because its flow effectively doubles the streamflow of the river during the winter. Other inputs to the main stem, such as inputs from ground-water and tile drains, may have contributed water with reduced DO concentrations relative to the main stem. Many of these inputs were sampled during the summer season and, while a

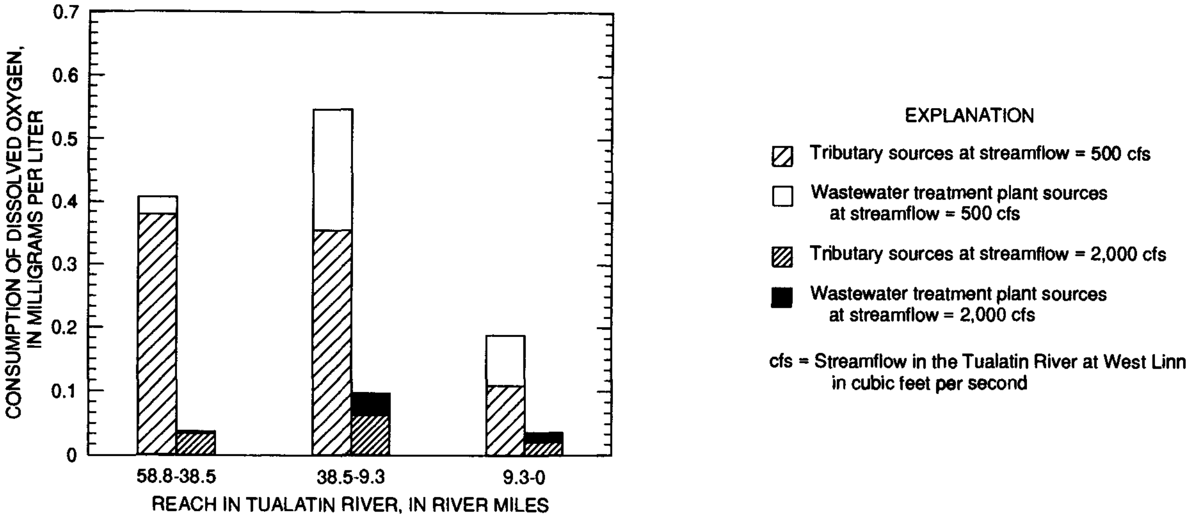


Figure 24. Theoretical consumption of dissolved oxygen in selected reaches of the Tualatin River, Oregon, due to observed mean loads of carbonaceous biochemical oxygen demand from tributary and wastewater treatment plant sources under the upper and lower bounds of winter low-flow conditions (500 and 2,000 cubic feet per second, respectively). An in-river decay rate of 0.02 per day is assumed (base 10 at 20 degrees Celsius), adjusted for mean water temperature of 9 degrees Celsius.

range of DO concentrations were observed, some inputs were significantly depleted in DO.

Using flow-weighted mean concentrations, the mean decrease in DO in the river resulting from combined inflow from Dairy and Rock Creeks was equivalent to about 0.4 mg/L, or approximately 25 percent of the total mean DO loss of 1.7 mg/L. The analysis should be considered qualitative in nature; however, it indicates that depleted DO concentrations in these major tributaries during winter low-flow conditions contributed to decreased concentrations of DO in the main stem. Fanno Creek, in contrast, flowed into the lower river at a point where DO concentrations had been further depleted. As a result, this tributary had a relatively slightly higher mean DO concentration at that point, and slightly increased the DO concentration in the main stem.

Factors Affecting Dissolved Oxygen during Peak-Flow Conditions

During the peak-flow conditions sampled January 29 to February 5, 1992, the DO concentration at RM 3.4 (14207200) exhibited a consistent but slight decrease over the rising limb of the hydrograph as temperatures increased (fig. 25).A part of this decline in the DO concentration was attributable to the decreased saturation potential of the water at higher temperatures. Over the course of the peak-flow period, the loss in DO observed at this site, excluding the DO decrease that had occurred prior to the storm,

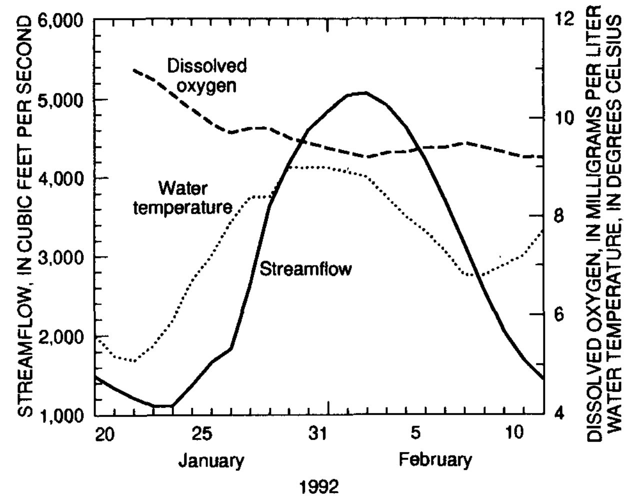


Figure 25. Daily mean streamflow in the Tualatin River at West Linn, Oregon, and daily mean dissolved oxygen and water temperature at river mile 3.4 during peak-flow conditions, January 29 to February 5, 1992 (streamflow in the Tualatin River at West Linn, Oregon, greater than 2,000 cubic feet per second).

amounted to approximately 0.5 mg/L, or about 7 percent saturation. DO concentrations remained greater than 9 mg/L at this site throughout the storm, within the range of approximately 78 to 83 percent saturation.

The water entering the study reach at RM 58.8 during peak flow contained a mean concentration of about 1 mg/L less DO (a loss of approximately 7 percent saturation) than the mean observed for the winter low-flow conditions (fig. 22). Similarly, mean DO concentrations in the largest tributary, Dairy Creek, as well as in Rock Creek, were reduced relative to winter low-flow mean concentrations, although DO concentrations in the other tributaries were increased. As the water flowed downstream, mean DO concentrations were only slightly reduced, by about 0.8 mg/L or 5 percent saturation; the region of DO loss was limited to the upper river (fig. 26). Mean DO concentrations dropped between RM 51.5 and RM 38.5, whereas between RM 38.5 and RM 5.5 the concentrations remained approximately constant. This pattern of slight oxygen depletion resembled that observed historically for peak-flow conditions (see fig. 8).

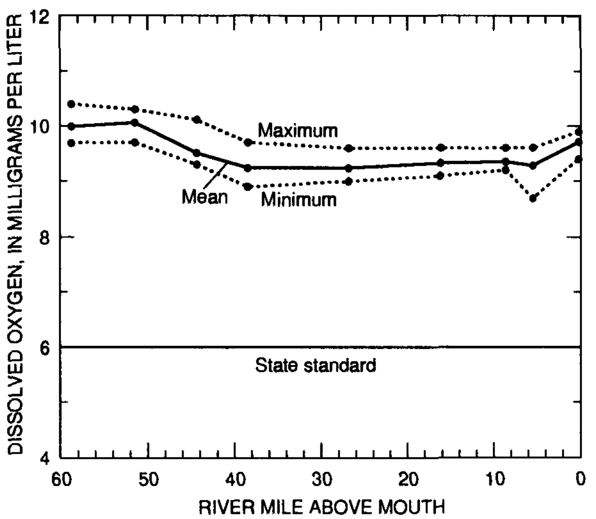


Figure 26. Maximum, mean, and minimum dissolved oxygen concentration at selected river miles in the Tualatin River, Oregon, during peak-flow conditions, January 29 to February 5, 1992 (streamflow in the Tualatin River at West Linn, Oregon, greater than 2,000 cubic feet per second).

Wastewater Treatment Plant and Tributary Loads

During peak-flow conditions, the daily mean load of CBOD from the combined WWTP and tributary sources was about twice the load observed during winter low-flow, increasing from approximately 21,000 lb/d to 41,000 lb/d (table 17). In spite of the increased loads, however, concentrations of CBOD throughout the main stem remained low, with a mean concentration of 2 mg/L. The increase in loading was mainly due to the greatly increased flows, because mean concentrations in WWTP and tributary sources were lower during the peak-flow period than during base-flow conditions (table 15).

The mean CBOD loads from the tributaries during the storm were larger than mean loads from WWTPs. Tributary loads of CBOD increased nearly threefold from their mean loads during winter low-flow and amounted to more than 80 percent of the total mean load during the storm. Nearly 50 percent of the total mean CBOD load during peak flow was from two major tributaries, Dairy Creek and Rock Creek; 20 percent of the total entered the upper reach at Dilley. It is possible that numerous minor tributary sources, which were not sampled during this study, also contributed potentially significant loads of CBOD during this period; as a consequence, the total tributary source loading during the storm may have been underestimated. The total mean load from WWTPs decreased slightly from over 8,000 lb/d during base-flow conditions to 7,000 lb/d during the peak-flow period. Combining both WWTP and tributary sources, 88 percent of the total mean CBOD load entered the river above RM 38.1.

The mean load of ammonia from both WWTP and tributary sources during the storm was slightly higher than during winter low-flow conditions, increasing from about 3,500 lb/d to 4,800 lb/d. Over the course of the rising hydrograph, the combined WWTP loads changed very little, whereas the tributary source loads increased significantly (fig. 27). WWTP loads continued to be primary, however, contributing 83 percent of the total mean ammonia load. The two large WWTPs, at Rock Creek and Durham, accounted for more than 70 percent of the total mean load. The increase in storm loads of ammonia, like those of CBOD, was mainly a consequence of increased flow; mean ammonia concentrations in the tributaries and the WWTP effluent did not significantly differ from concentrations observed during low-flow conditions. Ammonia concentrations in the main stem remained low during the storm (0.2 mg/L or less).

Total mean TSS loads during peak flow were nearly 10 times larger than the mean low-flow load, largely due to the ninefold increase in tributary loads. In contrast, total mean loads of TSS from WWTPs increased by less than a factor of two. The increased tributary load was due to an increase in both concentration and flow. Mean concentrations in the tributaries increased significantly over the course of the storm, most significantly in Gales Creek and Fanno Creek. In the main stem, mean concentrations of TSS increased in a similar fashion. Concentrations in the WWTP effluent, in contrast, were generally unchanged. The peak loads of TSS from combined WWTP and

Table 17. Mean loads of ultimate carbonaceous biochemical oxygen demand (CBOD), total suspended solids (TSS), and ammonia as nitrogen to the Tualatin River, Oregon, during peak-flow conditions, January 29 to February 5,1992
[Streamflow in the Tualatin River at West Linn greater than 2,000 cubic feet per second; WWTP, wastewater treatment plant; --, not applicable]

Site	River mile	Mean loads, in thousand pounds per day					
		CBOD		TSS		Ammonia	
		WWTP	Tributaries	WWTP	Tributaries	WWTP	Tributaries
Tualatin River near Dilley	58.8	--	9.0	--	200	--	0.32
Gales Creek	56.7	--	3.5	--	350	--	.14
Forest Grove WWTP	55.3	0.55	--	0.43	--	0.10	--
Dairy Creek	44.8	--	10	--	240	--	.17
Hillsboro WWTP	43.9	.53	--	.19	--	.36	--
Rock Creek	38.1	--	9.9	--	61	--	.17
Rock Creek WWTP	38.1	2.6	--	.82	--	2.1	--
Fanno Creek	9.3	--	1.2	--	66	--	.044
Durham WWTP	9.3	3.3	--	1.1	--	1.4	--
Subtotal		7.0	34	2.54	917	3.96	.844
Total		41		920		4.80	
Percent		17	83	.3	99.7	82	18
Tualatin River at Weiss Bridge	.2	44		1,200		4.8	

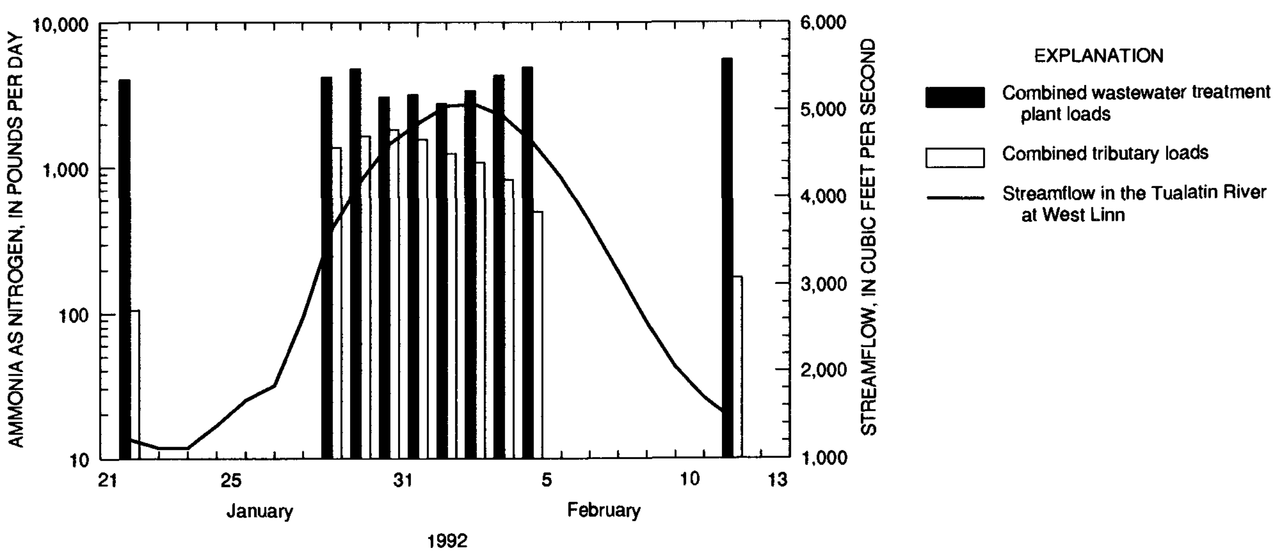


Figure 27. Combined loads of ammonia as nitrogen to the Tualatin River, Oregon, from wastewater treatment plants and major tributaries during peak-flow conditions, January 29 to February 5, 1992 (streamflow in the Tualatin River at West Linn, Oregon, greater than 2,000 cubic feet per second).

tributary sources occurred during the rising stage of the hydrograph (fig. 28) and may have been underestimated due to incomplete sampling at the onset of the storm.

In-River Decay of Carbonaceous Biochemical Oxygen Demand

In spite of the doubling of CBOD loads to the river during the storm, DO concentrations were unchanged throughout the length of the backwater reach, where the effect of deoxygenation due to CBOD would likely be greatest. During peak-flow conditions, the time of travel for the length of the study reach was significantly reduced from winter low-flow conditions and was estimated to 1 day or less. The potential for reaeration and dilution was increased due to increased velocities and low winter temperatures. These factors were probably the most important in determining the effect of CBOD in the river during peak flow.

The theoretical oxygen consumption (using mean WWTP and tributary source loads within selected river reaches) is presented in figure 29. The analysis was similar to that performed for winter low-flow conditions (theoretical DO loss based on observed decay rates [main-stem rate constant, base 10 at 20°C = 0.04 day^{-}], corrected for a mean in-river temperature of 9°C) For the purpose of this analysis, time-of-travel estimates were based on equations derived from dye studies, which assumed streamflow in the lower river to be 4,000 ft^3/s. Residence times may have been overestimated, consequently, as mean flow in the Tualatin River at West Linn during this period was 4,610 ft^3/s. To simulate the potential effect of reduced WWTP treatment capacity associated with peak streamflow, a threefold increase in total WWTP loads was also included in this analysis. Because of the decrease in travel time and the increase in the volume of water, and despite the increase in CBOD loading and decay rate, the theoretical loss in DO attributable to mean loads from all sources was very low (less than 0.15 mg/L). The theoretical consumption of oxygen by CBOD loads from tributary sources was a significant fraction of the total, about 86 percent. Increasing WWTP loads by a factor of three had the greatest effect in the lowest river reach, below RM 8.7; the total consumption of oxygen for the entire study reach, however, remained less than 0.20 mg/L. Approximately 67 percent was still attributable to tributary sources. This analysis suggests that DO loss in the main stem from CBOD loading, particularly from WWTPs, is generally not significant during peak-flow conditions. In addition, if a significant inflow and infiltration occurs during the course of a storm, causing a greatly increased load of CBOD from one or several of the WWTPs, the concentrations of DO in the river probably would not be seriously effected.

Tributary Input of Dissolved Oxygen

Early in the storm, tributaries with flashy flow characteristics provided a significant proportion of the total flow in the upper river (fig. 16). These tributaries.

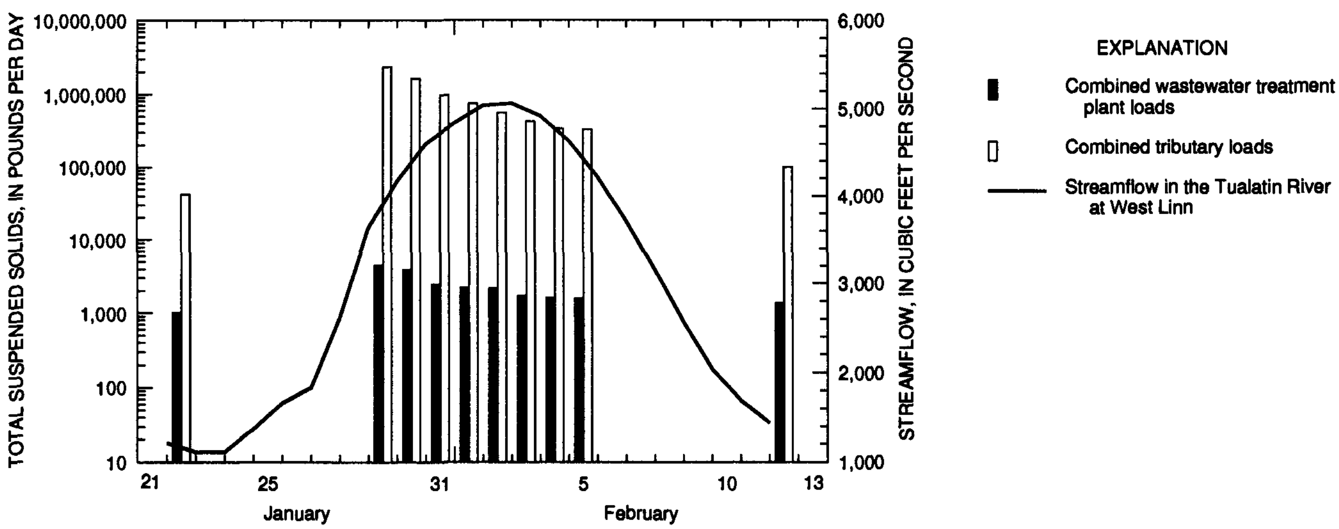


Figure 28. Combined loads of total suspended solids to the Tualatin River, Oregon, from wastewater treatment plants and major tributaries before, during, and after peak-flow conditions, January 29 to February 5, 1992 (streamflow in the Tualatin River at West Linn, Oregon, greater than 2,000 cubic feet per second).

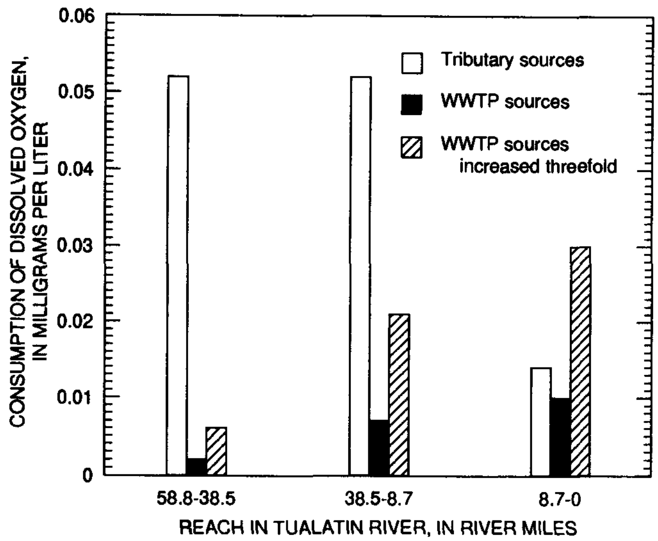


Figure 29. Theoretical consumption of dissolved oxygen in selected reaches of the Tualatin River, Oregon, due to observed mean loads of carbonaceous biochemical oxygen demand from tributary and wastewater treatment plant (WWTP) sources, and mean loads from WWTP sources increased threefold, under peak-flow conditions (mean streamflow equals 4,000 cubic feet per second). An in-river decay rate of 0.04 per day is assumed (base 10 at 20 degrees Celsius), adjusted for mean water temperature of 9 degrees Celsius.

also exhibited relatively high concentrations of DO; consequently, the concentrations of DO in the main stem were higher in the early stages of the peak-flow period. These tributaries included Gales Creek and the Tualatin River near Dilley (RM 58.8). As the peak-flow period progressed, however, input from these sources decreased, and those tributaries with a more sustained peak flow (Dairy and Rock Creeks) provided a major part of the total flow in the river. Dairy Creek was especially significant due to its large volume of flow. These tributaries were characterized by lower concentrations of DO during the sustained period of peak flow; consequently, the concentrations in the main stem were reduced. In contrast, the streamflow from Fanno Creek, emptying into the main stem at RM 9.3, was a relatively minor component of main-stem flow during the peak-flow period. The effect of the reduced DO observed during the mid-peak-flow period in Fanno Creek was reduced, therefore, by the large reservoir of oxygen being transported in the lower river.

It is probable that other input sources to the main stem, such as tile drains, ground water, and interflow under the soil surface, may have contributed water during the storm that was depleted in oxygen. These additional sources would probably not have been confined to the upper river but could have been distributed along the entire length of the study reach. Considering that reaeration was likely to have been an important factor at these elevated flows, especially in the upper river, these sources may have been more significant than suggested by this analysis.

Factors Affecting Dissolved Oxygen during Base-Flow Conditions

The 14-day period from November 6–20, 1992 was characterized by base flow averaging less than 300 ft^3/s (fig. 30). Estimated time of travel from the top of the study reach to the river mouth was about 16 days, given the mean flow of 219 ft^3/s in the Tualatin River at West Linn (14203500), equivalent to 256 ft^3/s if flow diverted by the Lake Oswego Canal is included. Temperatures throughout the main stem decreased during the base-flow period, ranging from approximately 8 to 13°C, but remained above 10°C in the lower river for most of the period. These temperatures would be expected to support bacterial activity, especially when combined with the increased residence time associated with base flow.

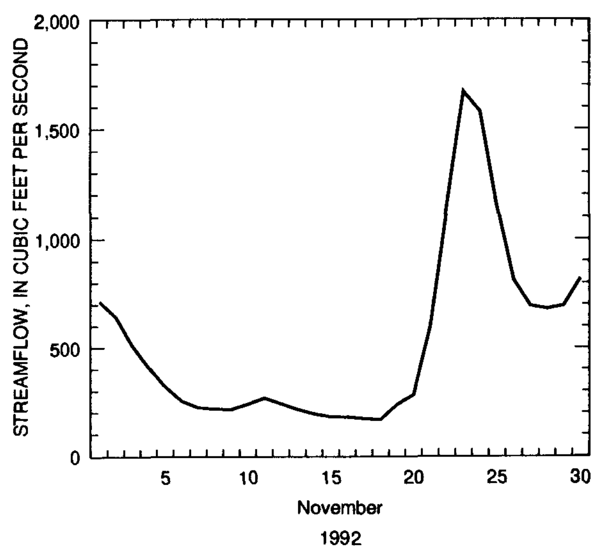


Figure 30. Daily mean streamflow in the Tualatin River at West Linn, Oregon, November 1992.

During the first 2 weeks of November, DO concentrations at RM 3.4 (14207200) were approximately 6 mg/L, the minimum State standard concentration; during the third week of November, concentrations fell below 5.5 mg/L (fig. 31). The short-term decrease in DO on November 19, 1992, was indirectly caused by a fire at a combusting facility on November 13, 1992. Runoff associated with water used to extinguish

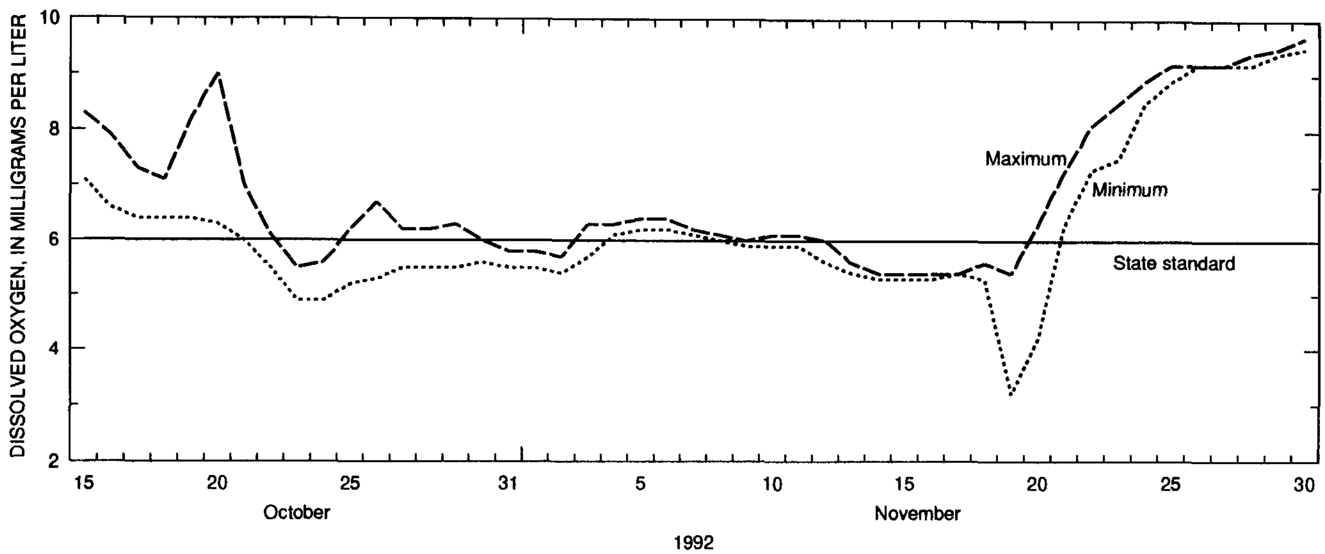


Figure 31. Daily maximum and minimum dissolved oxygen concentrations in the Tualatin River, Oregon, at river mile 3.4, October and November 1992.

the fire entered the river from Rock Creek (south), at RM 15.2, and exerted a significant oxygen demand as it moved through the system. A slow rate of reaeration contributed to the dramatic drop in DO.

Prior to the base-flow period, during October 1992, DO concentrations reflected the diel variation characteristic of significant levels of algal productivity. Diel concentrations sometimes varied by more than 2 mg/L and ranged from 5 to 10 mg/L over the course of several days. After the storm at the end of October, which increased streamflow to around 800 ft^3/s at the mouth (14207500) and presumably flushed the system of planktonic algal communities, diel variation in DO was reduced. Hourly measurements of DO from the minimonitor for the period of low flow (November 6–20, 1992) reflect the absence of photosynthetic activity (fig. 32). In addition, rates of primary production were observed on November 5 to have significantly decreased in the lower river compared to those observed on October 15 (table 18). Thus, it is assumed that photosynthesis did not contribute significantly to DO concentrations in the river during November 1992. Mean DO concentrations over the length of the study reach during the period of low flow decreased by more than 4 mg/L from RM 58.5 to 5.5 (fig. 33); the drop in oxygen was most pronounced between RMs 44.4 and 5.5. This pattern of DO depletion was similar to, although not as pronounced as, the historical pattern observed for base-flow conditions (see fig. 8). In addition, mean DO concentrations in the tributaries were significantly reduced from their mean winter low-flow levels, especially in Dairy and Rock Creeks.

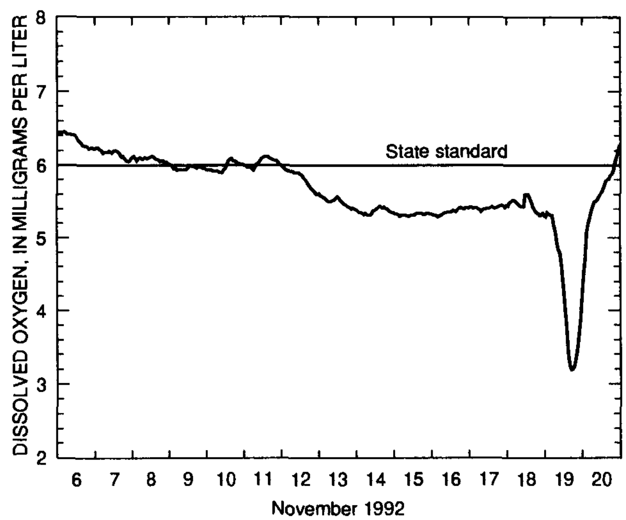


Figure 32. Hourly instantaneous dissolved oxygen concentrations in the Tualatin River, Oregon, at river mile 3.4, November 6–20, 1992.

Wastewater Treatment Plant and Tributary Loads

The total daily mean CBOD load to the main stem during base-flow conditions (6,700 lb/d, table 19) was about one-third the total mean load observed during winter low-flow conditions (table 15), and less than 20 percent of the mean load during peak-flow conditions (table 17). The comparatively small load was a direct consequence of the greatly decreased flow, however, as mean concentrations of CBOD were relatively high, especially in Dairy and Rock Creeks. Mean loads from WWTPs comprised 58 percent of the total daily mean load during this period; given the uncertainty surrounding the laboratory measurement

Table 18. Primary productivity data for selected river miles in the Tualatin River, Oregon, on October 15, 1992 and November 5, 1992
[Primary productivity is expressed as milligrams carbon per cubic meter per hour; --, no data]

River mile	Primary productivity	
	October 15	November 5
58.8	2.0	10.1
51.5	5.8	2.2
44.4	--	3.9
26.9	18	5.7
16.2	40	5.5
11.6	90	--
8.7	--	8.0
5.5	73	10

of CBOD, however, the contribution of WWTP and tributary sources of CBOD were probably essentially equivalent. Approximately 75 percent of the total CBOD load entered the main stem above RM 38.

TSS loads were greatly reduced from the loads observed during other flow conditions, amounting to about 5 percent of the mean load for winter low-flow conditions and less than 1 percent of the mean peak-flow load. This load reduction was a consequence of smaller tributary loads; the tributaries remained the most significant contributor of TSS, accounting for nearly 80 percent of the total. The total mean load from WWTPs during base-flow conditions was similar to that observed during winter low-flow conditions, and differed from the peak-flow load only by a factor of two. In contrast, the total mean TSS load from tributary sources varied from base to peak flow by approximately two hundredfold. This high variability in loads emphasizes the major role of flow conditions in governing the magnitude of tributary loads of TSS; loading from WWTPs is tightly controlled and is relatively independent of flow in the river.

Ammonia loads during the base-flow sampling period were dominated by WWTP sources, with tributary sources accounting for 2 percent of the total load. The total mean WWTP load of ammonia in November represented an increase of nearly seventeenfold over the mean load in October. The load increase was due to the discontinuation of nitrification as part of the effluent treatment process in the two large WWTPs, as well as the initiation of effluent discharge into the river

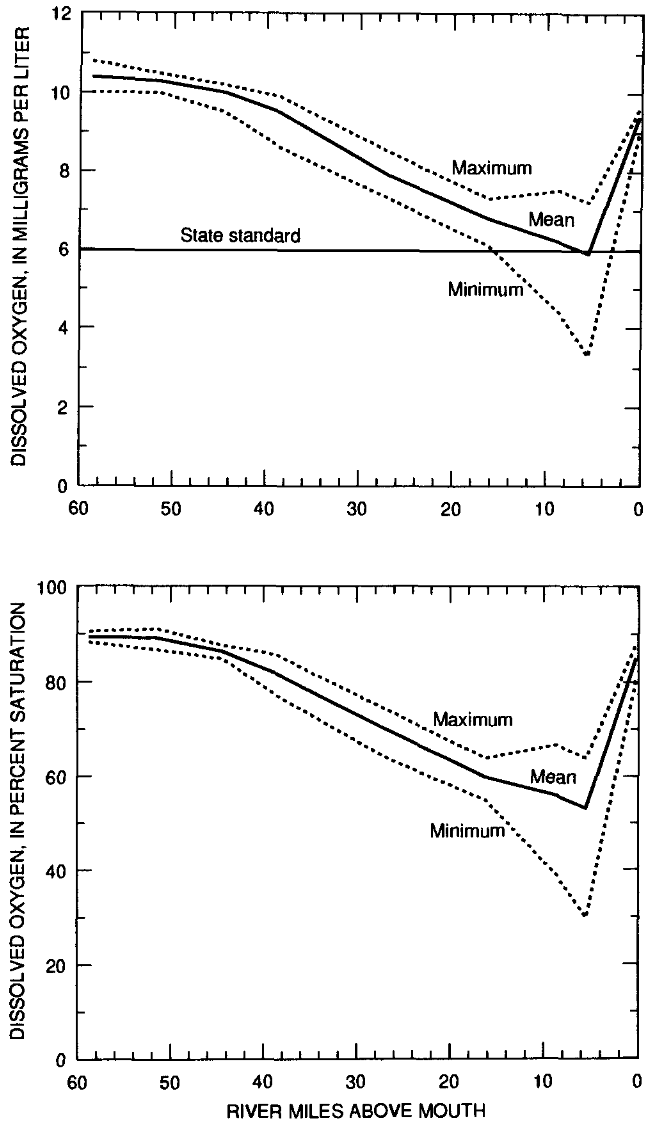


Figure 33. Maximum, mean, and minimum dissolved oxygen concentrations at selected river miles in the Tualatin River, Oregon, during base-flow conditions, November 6–20, 1992 (streamflow in the Tualatin River at West Linn, Oregon, less than 300 cubic feet per second).

from the two smaller WWTPs. Because the flow in the main stem increased by only a factor of two during this period, ammonia concentrations in the river increased significantly. In addition, about 95 percent of the total ammonia load was discharged into the main stem above the backwater reach; 60 percent of that load came from the Rock Creek WWTP.

Simulation of Flow and Water Quality

Flow and water quality within the lower part of the Tualatin River were simulated during the November base-flow period using CE-QUAL-W2,

Table 19. Mean loads of ultimate carbonaceous biochemical oxygen demand (CBOD), total suspended solids (TSS), and ammonia as nitrogen to the Tualatin River, Oregon, during base-flow conditions, November 6–20, 1992
[Streamflow in the Tualatin River at West Linn less than 300 cubic feet per second; WWTP, wastewater treatment plant; --, not applicable]

		Mean loads, in thousand pounds per day					
		CBOD		TSS		Ammonia	
Site	River mile	WWTP	Tributaries	WWTP	Tributaries	WWTP	Tributaries
Tualatin River near Dilley	58.8	--	1.3	--	2.0	--	0.015
Gales Creek	56.7	--	.23	--	.16	--	.0027
Forest Grove WWTP	55.3	0.28	--	0.15	--	0.068	--
Dairy Creek	44.8	--	.93	--	1.5	--	.0082
Hillsboro WWTP	42.8	.40	--	.11	--	.50	--
Rock Creek	38.1	--	.32	--	.64	--	.0028
Rock Creek WWTP	38.1	1.6	--	.77	--	1.3	--
Fanno Creek	9.4	--	.055	--	.22	--	.0022
Durham WWTP	9.4	1.6	--	.17	--	.14	--
Subtotal		3.9	2.8	1.20	4.52	2.01	.031
Total		6.7		5.72		2.04	
Percent		58	42	21	79	98	2

a two-dimensional model originally developed by the U.S. Army Corps of Engineers (Cole and Buchak, 1995). This model is well-suited to simulate the reservoir-like nature of the lower Tualatin River. The hydrodynamic component of the model calculates water velocity, surface elevation, and heat flow; the water-quality component can simulate 22 different constituents or measurements, of which only 5 were modeled during this study. These included a tracer (chloride), ammonia, nitrite-plus-nitrate nitrogen, CBOD, and DO. Because of the negligible photosynthetic activity during this period, the algal dynamics were omitted. The concentration of DO during this period was considered to depend only on oxygen input at the model boundaries and the processes of reaeration, nitrification, SOD, and CBOD decay.

Several important modifications were made to the model code that make it different from the documented version. These modifications are (1) inclusion of an algorithm to calculate the flow over the LOC dam, using weir characteristics defined by the user (new code provided by Scott Wells of the Department of Civil Engineering, Portland State University), (2) utilization of one-dimensional segments for shallow reaches (revised in part by Scott Wells, and in part by USGS personnel), (3) modification of the algorithm which calculates reaeration, now a function of water velocity and depth, rather than solely a function of wind speed, (4) modifications of the temperature dependence of SOD and nitrification, according to the equations previously described, and (5) use of two nitrification rates in two characteristically different reaches of the river.

The simulated reach extended for approximately 35 miles from RM 38.5 to the LOC diversion dam at RM 3.4, and was discretized using a grid composed of 155 longitudinal segments of variable length and 16 vertical layers of variable height. Channel bathymetry was determined by midchannel longitudinal depth sounding measure- ments, extending from the LOC dam upriver to RM 37, as well as by 52 cross-sectional traverses made at longitudinal intervals of approximately 0.5 to 1.0 mile during base-flow in October 1990. Midchannel depth data were used to estimate mean cross-sectional depths based on a regression between measured midchannel and cross-sectional depths. Four major inflows to the model grid included: Rock Creek, Rock Creek WWTP, Fanno Creek, and Durham WWTP. One withdrawal was specified at the Lake Oswego Canal. Flow over the dam at the downstream boundary was calculated within the model.

Input files used by the model are available from the U.S. Geological Survey, WRD, Portland, Oregon; values for the rate constants are given in table 20. Model simulations began on October 15, 1992, about 2 weeks prior to the base-flow period of interest (November 6–20, 1992).

Table 20. Rate constants used in the model CE-QUAL-W2, calibrated to November 1992 base-flow conditions

[Nitrification and carbonaceous biochemical oxygen demand (CBOD) decay rates as base 10 per day at 20 degrees Celsius; sediment oxygen demand (SOD) rate in grams oxygen per square meter per day at 20 degrees Celsius]

Process	Rate constant
Nitrification—segments 1–32	0.095
Nitrification—segments 33–155	.024
CBOD decay	.020
SOD	1.80

Model Calibration

Streamflow

Streamflow at the upstream boundary, RM 38.5, was taken from continuous records of stage (OWRD), which were converted to daily mean streamflow using a rating curve. The streamflow at the downstream boundary, RM 3.4, was calculated internally and compared to the daily mean streamflow at the Tualatin River at West Linn (14207500) at RM 1.8. The flow in the river was variable throughout the simulated period, ranging from about 100 ft^3/s on October 15, 1992, when the simulation began, to a peak of approximately 700 ft^3/s on November 1; the base-flow period of interest extended from November 6–20, 1992, when the flow was less variable—150 to 250 ft^3/s.

Rock Creek tributary inflows were obtained from gage-height readings made concurrent with water-quality sampling that were converted to streamflow on the basis of rating curves maintained by the OWRD. As a consequence of intermittent sampling, daily streamflow data were not available for Rock Creek; in particular, no streamflow data were available during the peak of the storm of October 29 to November 2, 1992. Daily mean streamflow from Fanno Creek was available from a USGS gaging station (14206950). Daily discharges of effluent from the Rock Creek and Durham WWTP were obtained from USA records. No data were available for minor tributary inputs.

The calculated flow at the downstream boundary was very sensitive to the weir characteristics of the LOC diversion dam. The width and height of the LOC dam are easily modified (within certain limits) by the placement or removal of a series of flashboards on the crest of the dam. The number and height of the flashboards are managed in order to maintain a river elevation (at RM 6.7) sufficient to provide a gravity-feed diversion of roughly 60 ft^3/s through the Lake Oswego canal and into Lake Oswego. It was necessary to calibrate the water surface elevation at the canal in order to correctly model the residence time within the backwater reach of the river. Daily mean values for elevation at the canal were obtained from records of stage maintained by the OWRD; these data were considered preliminary by OWRD but were assumed to be sufficiently accurate.

Figure 34 illustrates the simulated and observed streamflow at the downstream boundary during the period from October 29 to November 20, 1992. During the peak storm period, the simulated streamflow was significantly less than the observed streamflow; this discrepancy was apparently a consequence of insufficient input data during the high-flow period, including minor tributary inflow. The simulated and observed streamflows matched well beginning on day 308 (November 3, 1992), however, and continued to be in excellent agreement throughout the base-flow

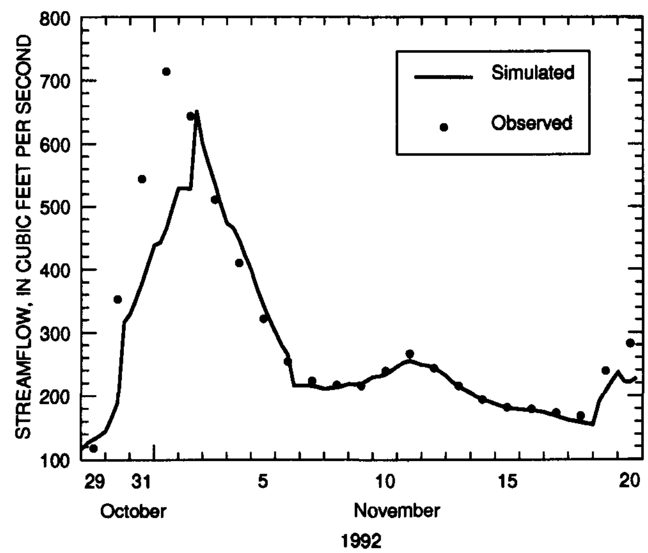


Figure 34. Observed streamflow in the Tualatin River, Oregon, at river mile 1.8 and simulated streamflow at river mile 3.4 for model simulation of October 29 to November 20, 1992.

period. The observed and simulated water surface elevation at the Lake Oswego canal for this period were also considered to be in close agreement (fig. 35).

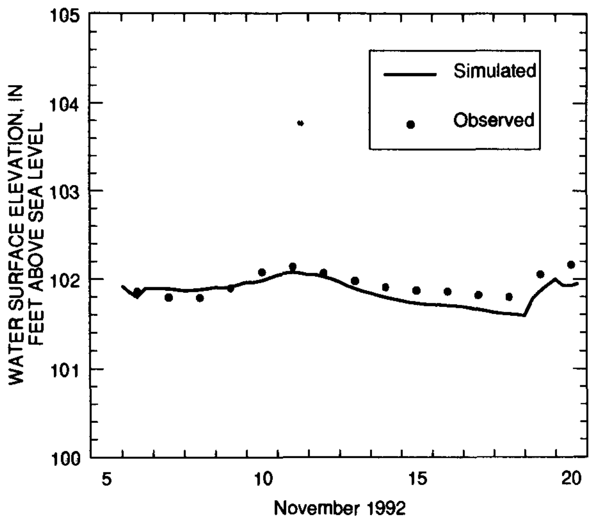


Figure 35. Observed and simulated water-surface elevatior in the Tualatin River, Oregon, at the Lake Oswego Canal, river mile 6.7, during base-flow conditions, November 6–20, 1992 (streamflow in the Tualatin River at West Linn, Oregon less than 300 cubic feet per second).

Temperature

Heat flow in the model was simulated using meteorological input data that included wind speed and direction, air and dewpoint temperatures, and insolation. Daily wind and temperature data were provided by Tualatin Valley Irrigation District (TVID) from monitors that were located near Forest Grove. Hourly values for temperature were estimated prior to being input to the model from daily maximum and minimum values using a sinusoidal function. Insolation data were obtained from a solar monitor located at the Durham WWTP, which measured only photosynthetically active radiation (PAR). These data were used to estimate total insolation by using a correlation between PAR and total insolation developed from measurements made in the South Umpqua River near Brockway, Oregon (14312002) and at the WWTP, Winston-Green, Oregon (430812123240101) during the summer 1991 and 1992.

The simulated and observed water temperature at RM 3.4 for the base-flow period are illustrated in figure 36. Simulated temperatures were consistently less than the observed values, which suggests that actual heat input may have been greater than in the

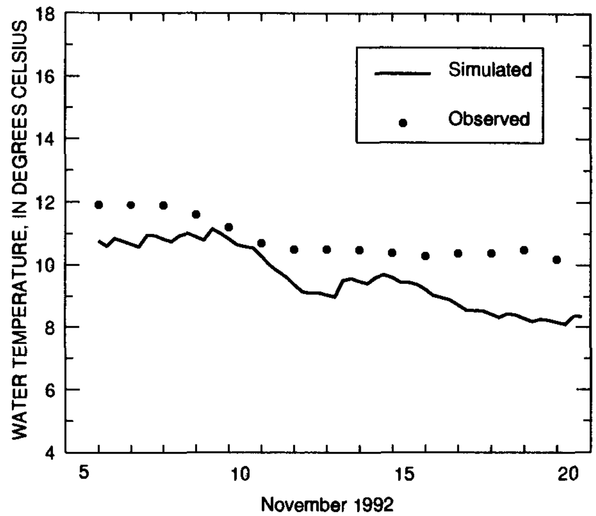


Figure 36. Observed and simulated water temperature in the Tualatin River, Oregon, at river mile 3.4 during base-flow conditions, November 6–20, 1992 (streamflow in the Tualatin River at West Linn, Oregon, less than 300 cubic feet per second).

simulation; the discrepancy ranged from approximately 0.2 to 2.0°C. Several factors may have been involved in these errors: heat input may have been underestimated by the use of a correlation conversion between PAR and total insolation observed during summer conditions, which tended to be clear and sunny, with the sun's angle near perpendicular. In contrast, skies were cloudy in the Tualatin River Basin for much of November; as a result, the quantity of infrared radiation (>700 nm) may have been greatly underestimated. In addition, simulated heat losses caused by evaporative cooling may have been overestimated. During fall mornings, localized foggy conditions are not uncommon along the Tualatin River; evaporative cooling would be expected to be small during foggy conditions. The model does not simulate fog; therefore, evaporation may have been overestimated.

The sensitivity of other processes in the model to the discrepancy between observed and simulated temperatures was evaluated by analyzing the effect of the temperature difference on various rates of deoxygenation used by the model. Using the temperature compensation equations previously described, the transformed rates of CBOD ($k = 0.02$ day^{-1} at 20°C), nitrification ($k = 0.095$ day^{-1} at 20°C), and SOD (1.8 g O_2/m^2 day) were calculated over the temperature range 8 to 12°C. In all cases, the change in rate over this temperature range was very slight, that is, about 10 to 15 percent for the maximum discrepancy of 2°C

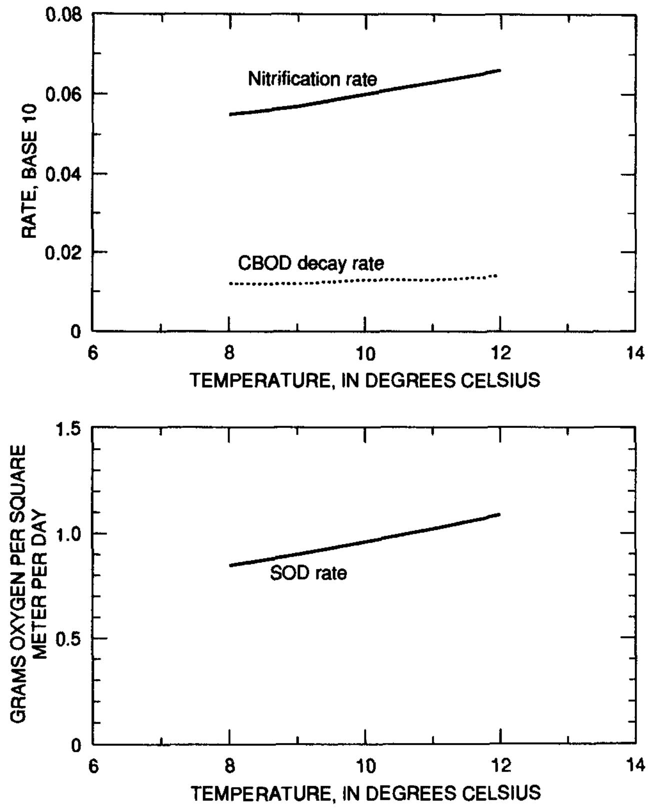


Figure 37. Effect of temperature on the rate of carbonaceous biochemical oxygen demand (CBOD) (k=0.02 per day, base 10 at 20 degrees Celsius), the nitrification rate in the upper river (k=0.095 per day, base 10 at 20 degrees Celsius), and the rate of sediment oxygen demand (1.8 grams oxygen per square meter per day at 20 degrees Celsius) during simulation of base-flow conditions, November 6–20, 1992 (streamflow in the Tualatin River at West Linn, Oregon, less than 300 cubic feet per second).

(fig. 37). This level of error is well within the uncertainty inherent in the initial determination of these rates. As a consequence, the calibration of the model with respect to temperature was considered to be adequate for the purpose of this study.

In-River Decay of Carbonaceous Biochemical Oxygen Demand

The model requires ultimate CBOD as input for the determination of CBOD in-river decay. Considerable uncertainty is inherent in the simulation of this characteristic, because direct measurement of ultimate CBOD was made only twice during November, 1992. Analysis of 5-day CBOD was made more frequently, usually two to three times per week for tributary and main-stem samples, and four to five times per week for WWTP effluent. This frequency of CBOD analysis was considered adequate to determine the typically low rates of decay characteristic of the basin, as well as the range of CBOD loading and concentrations. It was inadequate, however, to thoroughly and quantitatively describe the loading of CBOD to the river over the variable flow conditions that occurred during the model simulation, especially during the storm immediately prior to the baseflow period of November 6–20, 1992. In addition, the uncertainty in the CBOD loads was compounded by the relative significance of tributary sources for CBOD, the incomplete streamflow and loading data for Rock Creek, and the lack of consideration of minor tributary sources during the model simulation.

Several important assumptions were made in the determination and use of the input CBOD data and the in-river decay rate of CBOD during the simulation of base-flow conditions. First, estimates of ultimate CBOD from 5-day values were made on the basis of rates observed for each site, assuming that no change in rate occurred with changing flow conditions. Second, missing data were interpolated as the mean between two observed or estimated values, because analysis for CBOD was not always performed when other measurements were made. Third, the model assumed that the constituent concentrations provided at the boundaries were constant until new values were supplied, generally for intervals of 1 to 2 days and occasionally up to 4 to 5 days. Finally, the in-river decay rate was held constant throughout the simulation, in spite of storm conditions.

As a consequence of the uncertainties inherent in the loading data for CBOD, calibration of the model was performed only at the downstream boundary (RM 3.4). Laboratory decay rates were measured for samples taken throughout the main stem during November 1992 and were found to be in the range 0.01–0.03 day^{-1}; the most prevalent rate was 0.02 day^{-1}. Several rates were tested during calibration of the model, and k = 0.02 day^{-1} was chosen on the basis of a visual comparison between simulated and observed CBODs during the base-flow period in November (fig. 38). Note that only one of the observed values for CBOD during the base-flow period was directly measured as ultimate CBOD, on day 322 (November 17, 1992); the other observed values were estimated from 5-day CBOD values using k = 0.02 day^{-1}. Considering the assumptions and consequent uncertainties in the simulation of this constituent, the fit between observed and simulated values was adequate.

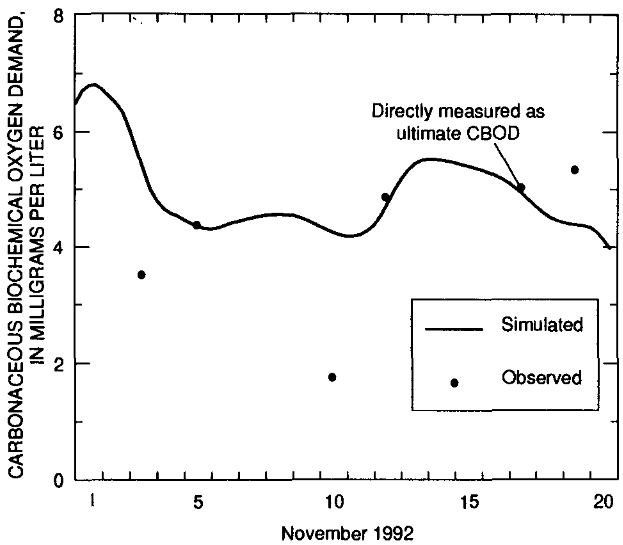


Figure 38. Observed ultimate carbonaceous biochemical oxygen demand (CBOD) in the Tualatin River, Oregon, at river mile 0.2 and simulated values at river mile 3.4, during November 1992. Direct measurement of ultimate CBOD was made on November 17; other values were estimated from 5-day CBOD analyses using $k = 0.02$ per day

Nitrification

The nitrogen cycle in aquatic systems is complex and includes a variety of transformation processes that act upon the many forms of nitrogen in freshwater. During the simulation of the base-flow period in November, 1992, several assumptions were made regarding the sources and fate of the key nitrogen species, ammonia and nitrite-plus-nitrate nitrogen. First, the only sources of nitrogen to the river were considered to be WWTP effluent and tributary input. Other potential sources, such as ammonia released from the sediments resulting from bacterial decay of organic material, were assumed to be negligible. Second, the most significant process governing the quantification of nitrogen during this period was considered to be nitrification, the oxidative decay of ammonia to nitrate. Other processes, such as denitrification (conversion of nitrate to dinitrogen gas [NO_2 and N_2], which escapes to the atmosphere) and biological uptake, were considered relatively insignificant and were not simulated. Third, the two-stage process of nitrification, which involves two separate bacterial groups, was assumed to proceed under a uniform rate with an identical response to temperature changes. Fourth, the extent of ammonia decay was assumed to be related only to the concentration of ammonia in the water and the residence time, dependent on water temperature, with no provisions in the model for simulating the growth of nitrifying bacterial communities as a response to increasing ammonia loads. Finally, two rates of nitrification were applied in different reaches of the river with the distinction between them based on channel geometry or the availability of habitat for bacterial growth. The rate in the upper river, a higher rate, was applied to the shallow reach from the upstream boundary to segment 33 (RM 29.7), where the water depth began to consistently exceed 6 feet; the second rate was applied from that point to the downstream boundary.

Generally, the course of nitrification in a stream is followed by observing the disappearance of ammonia and the concomitant appearance of nitrate as the water travels downstream from a major input of ammonia. Calibration of this process during November base-flow conditions was initially done by comparing observed ammonia concentrations at selected RMs with simulated values. The use of a single nitrification rate throughout the model reach did not provide a good fit with the measured concentrations; when two rates were used, the correspondence between the simulated and observed data was considered to be very good (fig. 39).

The simulated nitrification rates produced nitrate concentrations that matched the observed data fairly well at RM 26.9 and RM 16.2 (fig. 40). At RM 5.5, however, the comparison of observed and simulated values revealed nitrate to be overestimated by approximately 20 to 30 percent. The simulation of nitrate concentrations at this site was likely complicated by the nonuniform loading of nitrate from the Durham WWTP, located about 4 miles upstream at RM 9.3. Nitrate loads from the Durham plant were a large fraction of the total nitrate loading to the river during this period, with the daily mean nitrate load from Durham amounting to nearly 2,000 lb, or more than 80 percent of the total mean nitrate load at RM 5.5. In addition, loading from the Durham plant is characterized by a distinct diel variation, as demonstrated by a 24-hour sampling survey conducted in July 1991 (fig. 41). The composited daily mean flow varied considerably during this survey and effluent flows during the middle of the night were less than daytime flows. As a consequence, nitrogen loading after 3 a.m. was up to 30 to 40 percent less than the composited daily mean load. Because the nitrate load from the Durham plant constituted a significant fraction of the total load, this diel load signature would be

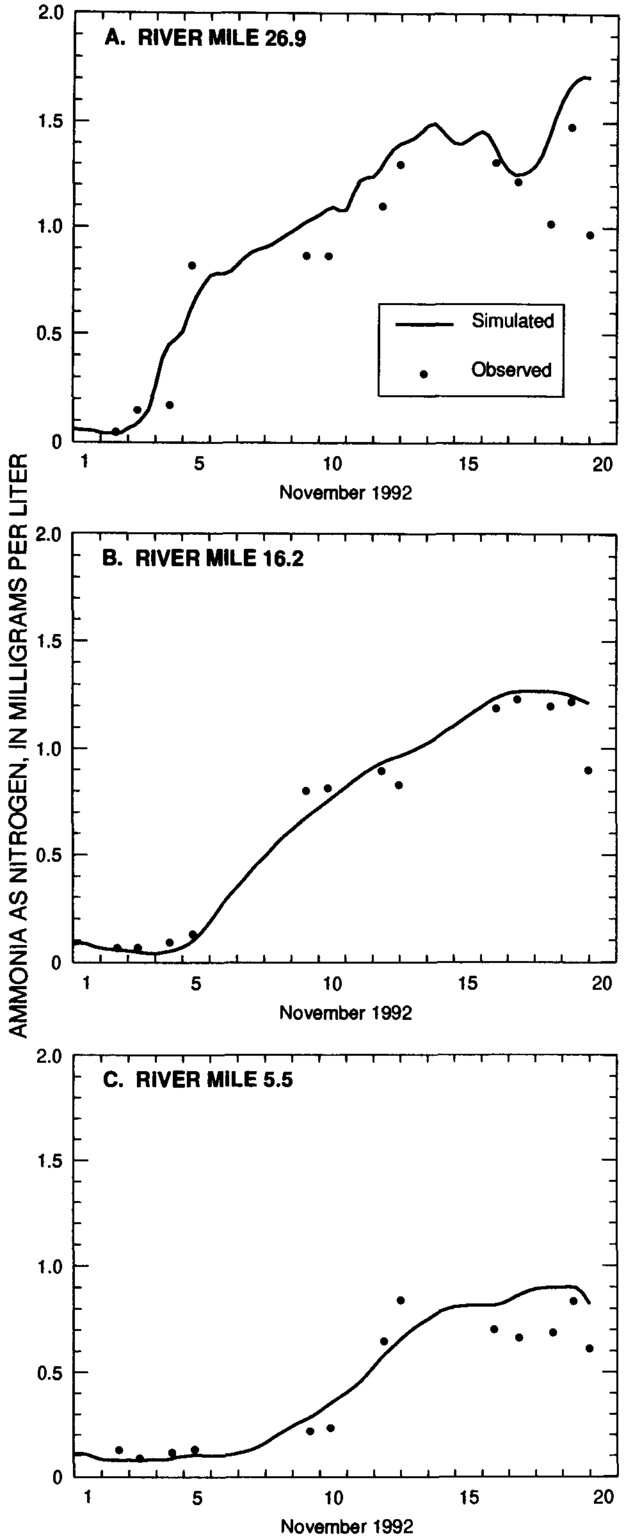


Figure 39. Observed and simulated concentrations of ammonia as nitrogen at selected river miles in the Tualatin River, Oregon, before and during base-flow conditions (November 6–20), 1992 (streamflow in the Tualatin River at West Linn, Oregon, less than 300 cubic feet per second).

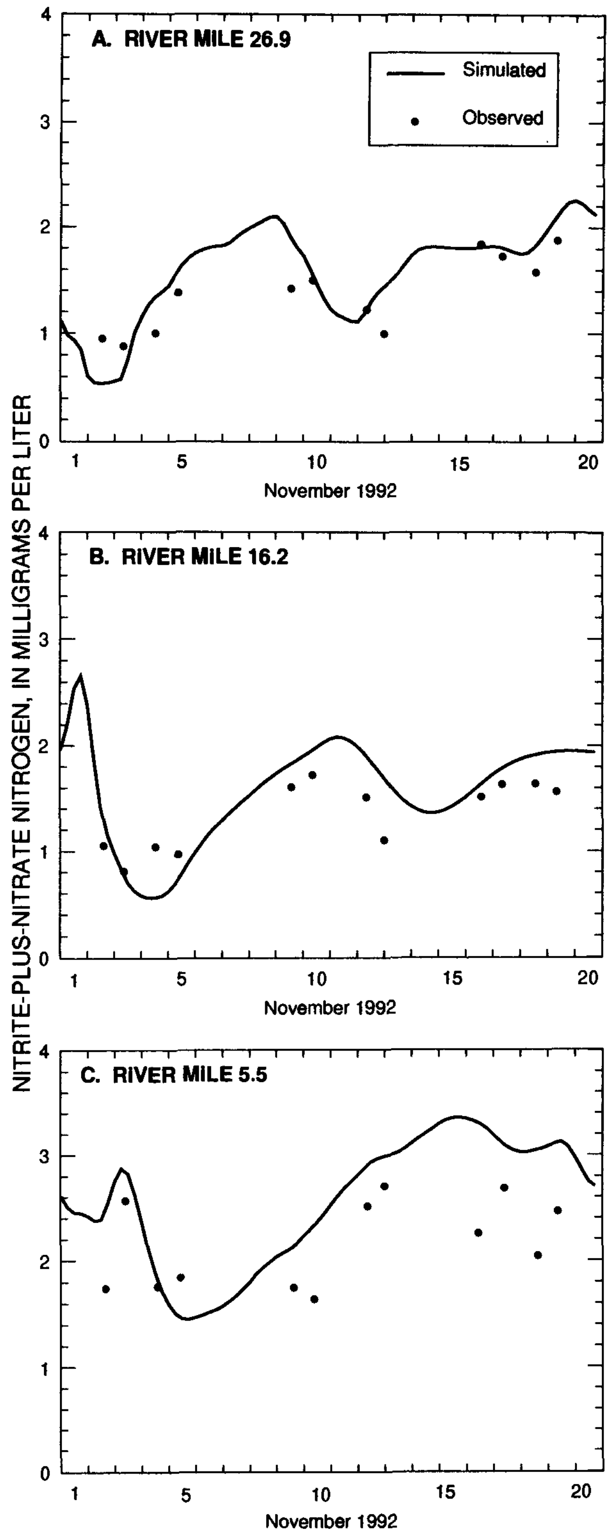


Figure 40. Observed and simulated concentrations of nitrite-plus-nitrate nitrogen at selected river miles in the Tualatin River, Oregon, before and during base-flow conditions (November 6–20), 1992 (streamflow in the Tualatin River at West Linn, Oregon, less than 300 cubic feet per second).

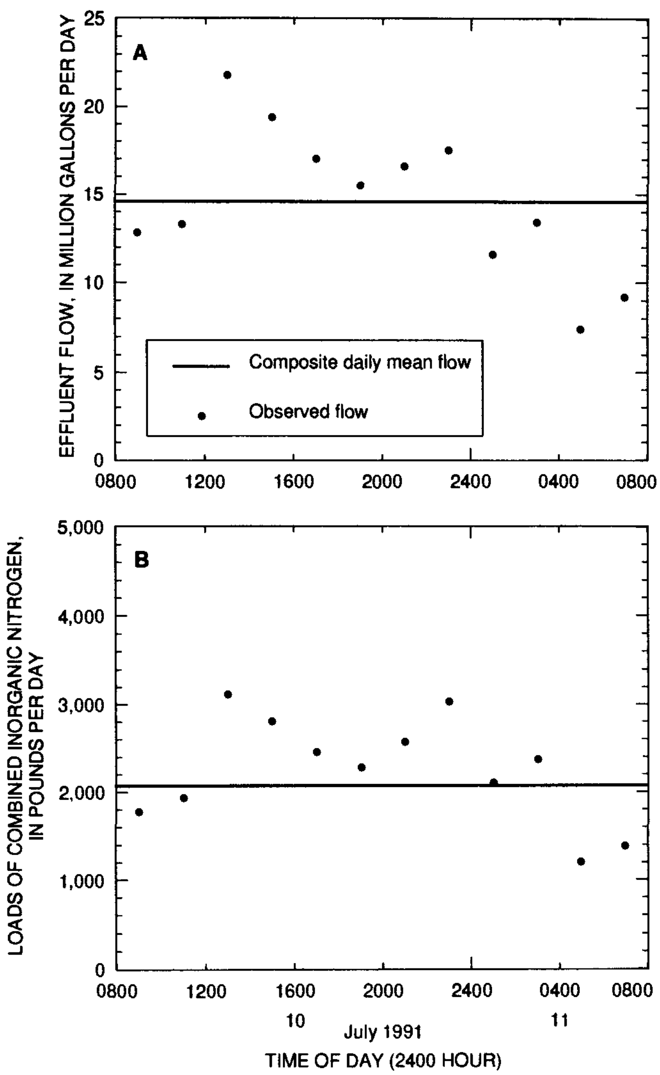


Figure 41. Hourly sampling of effluent flow and combined inorganic nitrogen (CIN) load and the composited daily mean effluent flow and CIN load from the Durham wastewater treatment plant to the Tualatin River, Oregon, during a 24-hour survey conducted on July 10–11, 1991. (A) Effluent discharge and (B) loads of combined inorganic nitrogen.

expected to be measurable as the load moved downstream.

Several factors combine to help explain the nitrate discrepancy observed in the model simulation. Data input to the model for the Durham plant was in the form of composited daily mean values, not hourly samples. In addition, the estimated time of travel during the base-flow period between the Durham outflow and RM 5.5 was approximately 32 hours. Midday sampling at that site, therefore, would reflect a load from Durham that was discharged between 2:00 a.m. and 6:00 a.m. These factors indicate that composited daily mean loads of nitrate input to the model from Durham WWTP were high relative to the actual loads associated with the observed values. Simulated nitrate values based on the composited mean loads, therefore, would be high when compared to observed values, which reflect the lower nighttime loading.

Additional corroborating evidence of the nitrification process is provided by the appearance of nitrite (as nitrogen), the intermediate product of nitrification, as the water flows downstream from an ammonia source. Nitrite is highly unstable and is typically observed in the water column at low concentrations in the absence of nitrification. Background concentrations of nitrite in the upper reach of the Tualatin River during November were low, as shown in figure 42. Downstream from the Hillsboro WWTP at RM 43.8 and the Rock Creek WWTP at RM 38.1, the major sources of ammonia during this period, nitrite concentrations increased considerably. The highest concentrations of nitrite were observed at RM 26.9, an increase of threefold to fourfold over the background concentration. This increase reflects the higher nitrification rate potential associated with the relatively shallow and fast-moving region above RM 30, mentioned previously. The persistence of nitrification throughout the lower river, in spite of the greater depth of the water column and consequent lower rate, was indicated by nitrite concentrations that were two to three times the background concentration at RM 16.2 and RM 5.5.

Dissolved Oxygen

After the calibration of CBOD and nitrification was completed, and assuming that the physical process of reaeration needed no calibration, the only remaining parameter to be defined in these simulations was SOD. Calibration of DO was evaluated by comparing simulated DO concentrations at the downstream boundary with daily mean measurements from the minimonitor located at RM 3.4 (14207200). In addition, field data from sampling sites throughout the model reach were compared with simulated values. An excellent fit to the observed data was obtained by a value for SOD equal to 1.8 g O_2/m^2 day at 20°C (figs. 43 and 44). This value was assumed to be valid for the entire model reach. This SOD rate lies well within the range of values observed during October 1992 (table 16).

Although the calibrated rates were not verified by application to a separate set of observed data, the

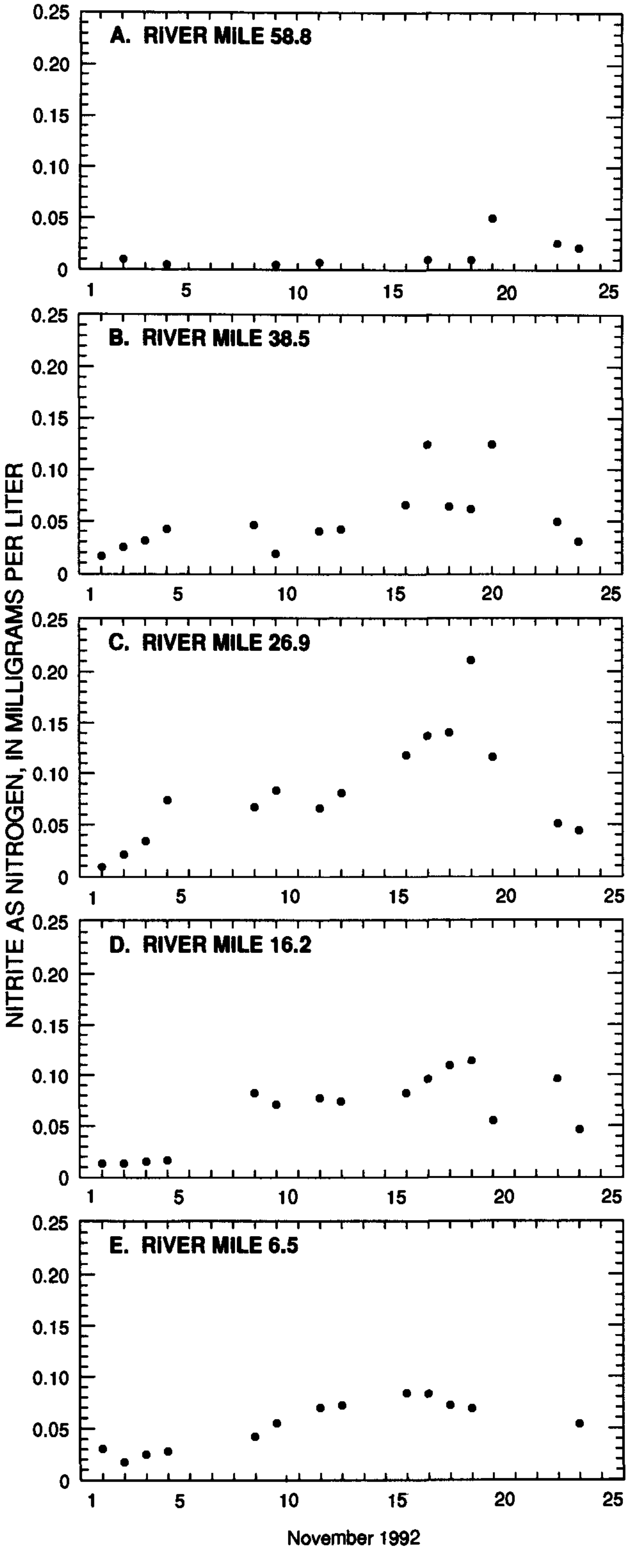


Figure 42. Observed concentrations of nitrite as nitrogen from selected river miles on the Tualatin River, Oregon, before, during, and after base-flow conditions (November 6–20), 1992 (streamflow in the Tualatin River at West Linn, Oregon, less than 300 cubic feet per second).

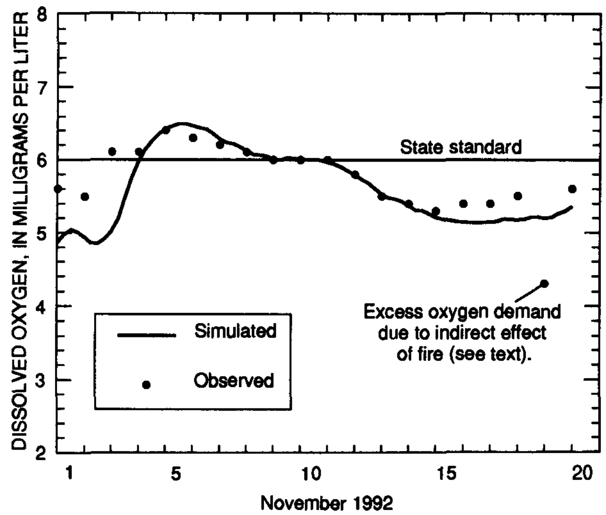


Figure 43. Observed and simulated dissolved oxygen at river mile 3.4 in the Tualatin River, Oregon, before and during base-flow conditions (November 6–20), 1992 (streamflow in the Tualatin River at West Linn, Oregon, less than 300 cubic feet per second).

good agreement between the model simulation and the observed data in November 1992 indicates that the CE-QUAL-W2 model as calibrated can be considered to be a useful tool for predicting water-quality conditions, subject to the following limitations:

(1) The applicable reach extends only as far upriver as RM 38.5 and does not apply below the LOC dam at RM 3.4.

(2) Whereas the model is dynamic, and therefore not subject to steady-state hydrologic conditions, the nitrification rate used in the upper river was the result of a relatively stable ammonia loading to the upper river (on the order of 2,000 lb/d) over the entire period of base-flow conditions. Under conditions of variable ammonia loading, as is often the case during this transition period, when the WWTPs can no longer support in-plant nitrification, this relatively high rate would not apply.

(3) A key element in the November base-flow conditions during 1992 was the absence of algal activity, despite the low flows and relatively warm temperature conditions. The absence of algae was caused by the flushing effect of the storm at the end of October. If significant algal activity is occurring in the river, primary production would be an oxygen source that these simulations do not model, and the predicted values for DO would probably underestimate actual oxygen concentrations.

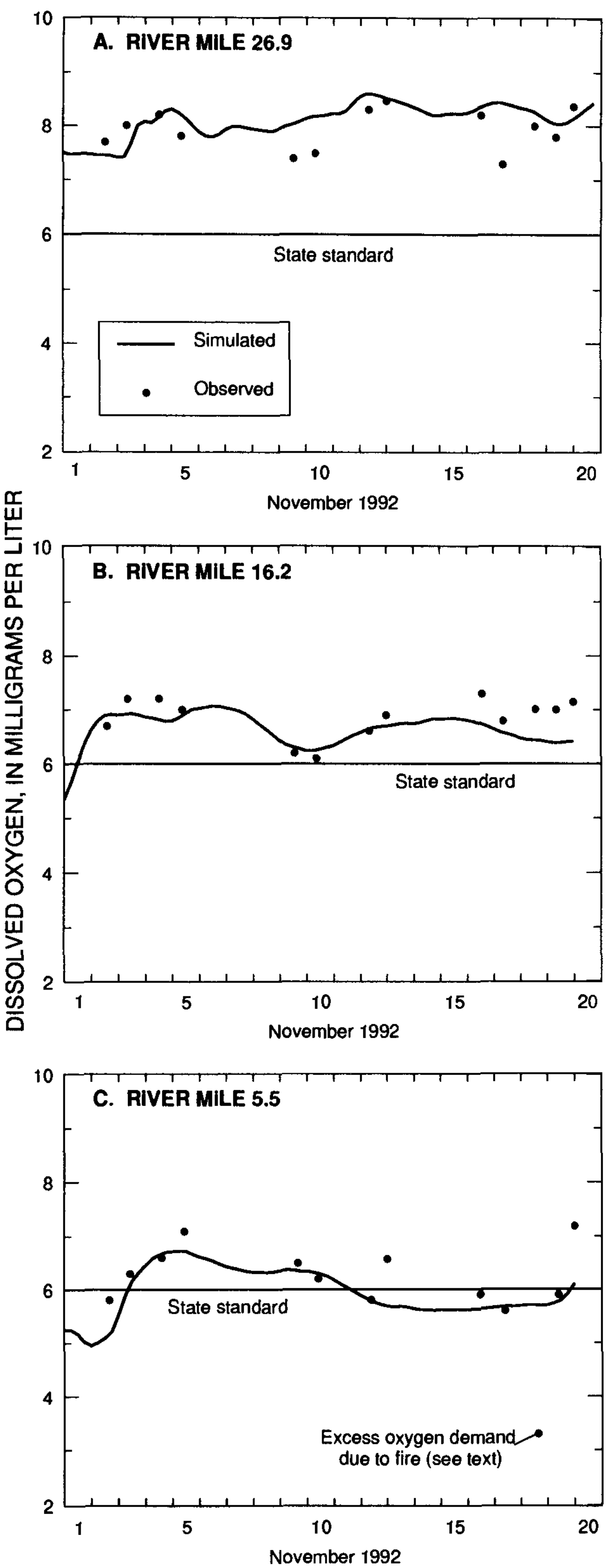


Figure 44. Observed and simulated dissolved oxygen at selected river miles in the Tualatin River, Oregon, before and during base-flow conditions, November 6–20, 1992 (streamflow in the Tualatin River at West Linn, Oregon, less than 300 cubic feet per second).

Relative Significance of Deoxygenation Processes

The relative significance of the various deoxygenation processes in the Tualatin River during the November base-flow period was evaluated using the model in several steps. First, time of travel information for a parcel of water entering the model reach at the upstream boundary on November 10, 1992, was determined; the parcel required approximately 9 to 10 days to move through the model reach, arriving at the downstream boundary just before the end of the base-flow period on November 19, 1992. The model was then modified to track this water parcel according to the time of travel information and output rates of change in DO in each segment for each major process. These rates were then converted to cumulative losses or gains for selected subreaches based on the time-of-travel information. This analysis allowed the loss or gain of DO resulting from each of the processes to be isolated during the time period the parcel was moving through the reach. Because reaeration and SOD are surface-dependent phenomena, and therefore highly dependent upon channel characteristics, these terms of the DO budget were calculated indirectly by difference based on model runs that did not simulate these processes. When calculated in this manner, the DO budget was subject to error (about 1 to 2 percent) due to the nonlinear nature of the reaeration process. A summary of this analysis for each of these processes is given in figure 45 and table 21.

These results demonstrate that SOD was the dominant factor governing deoxygenation throughout the model reach, accounting for 53 percent of the total DO loss. Even though a spatially constant rate was assumed, the effect of SOD was not the same for all the subreaches because it is dependent on the depth of the overlying water and the travel time. In the shallower upper river reach, SOD was exerted on the traveling water parcel at a rate of approximately 0.6 mg/L day, but the net effect was reduced by the relatively short residence time. In the lower river, the rate of oxygen depletion in the water parcel due to SOD was reduced to about 0.2 to 0.4 mg/L day as a result of the increased water depth, whereas the increase in travel time caused the overall DO loss to be intensified. In assessing DO management options in the Tualatin River, this high level of SOD is significant because it represents an "intrinsic" oxygen demand that is not directly amenable to treatment. These data suggest that maintaining a minimum level of river streamflow,

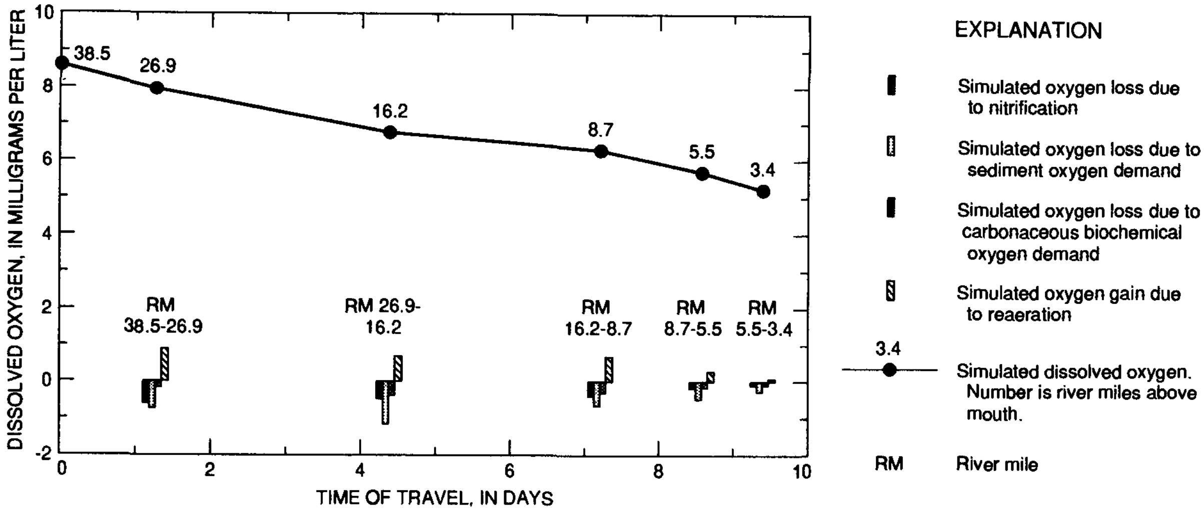


Figure 45. Simulated concentrations of dissolved oxygen in the Tualatin River, Oregon, as a function of time-of-travel, and losses and gains in oxygen concentrations between selected river reaches during November 10–19, 1992.

Table 21. Summary of simulated gains and losses in dissolved oxygen (DO) in the Tualatin River, Oregon, river miles 38.5 to 3.4, November 10–19, 1992
[mg/L, milligrams per liter; SOD, sediment oxygen demand; CBOD, carbonaceous biochemical oxygen demand]

Factor causing change in DO	Change in DO (mg/L)	Percentage of total loss or gain in DO
Nitrification	-1.8	28
SOD	-3.4	53
CBOD	-1.2	19
Total loss	-6.4	100
Reaeration	2.7	90
Tributary input	.30	10
Total gain	3.0	100
Net change	-3.4	

thereby increasing depth and reducing travel time, would limit oxygen loss due to SOD during base-flow periods.

Nearly 30 percent of the total loss of DO during this period was due to nitrification, which was especially pronounced in the upper river. The relatively high nitrification rate of about 0.5 mg/L day was possible because of the abundance of habitat for nitrifying bacteria coupled with elevated concentrations of ammonia; oxygen loss from nitrification accounted for nearly one-half the oxygen lost within this reach. In the lower river, because of the decrease in suitable microbial habitat and the associated reduction in nitrification rate, this process became less important—although it still accounted for approximately 20 to 25 percent of DO loss. The significance of nitrification as a major process affecting oxygen concentrations during November base-flow conditions was the result of ammonia releases from WWTPs; essentially all the ammonia in the river was of WWTP-source origin. Therefore, a reduction in ammonia loads from WWTPs during this transition flow period should have a significant positive effect on DO concentrations, especially in the upper river.

In contrast, the effect of CBOD on DO in the river during this period was small, accounting for less than 20 percent of the DO loss. Decay of CBOD occurred at a rate of about 0.1 mg/L day throughout the length of the model reach. Note that only about 50 percent of the CBOD was of WWTP origin and therefore subject to management control. Considering the low CBOD decay rate observed in the river during this period, these results suggest that further reductions in CBOD loading from WWTPs during the November base-flow period would not have a significant effect on DO concentrations.

The total input of oxygen from reaeration was relatively low, and was insufficient to offset the loss of DO as the water parcel moved through each reach. Even in the upper river, the increase in reaeration caused by the relatively shallow water and swift current velocity did not compensate for the oxygen demand from nitrification and SOD. As the water moved downstream, the reaeration was reduced by the reservoir-like quality of the backwater reach. These data reflect the susceptibility of the lower Tualatin River to oxygen-demanding processes during base-flow periods, particularly when algal productivity is not providing oxygen; to a limited extent, reaeration could be increased by increasing river streamflow.

Evaluation of Hypothetical Management Alternatives during Base-Flow Conditions

Following calibration to the November base-flow period, the model was used in a series of hypothetical simulations to evaluate the effect of various WWTP loading scenarios over a range of base-flow and temperature conditions. In these simulations, hydrologic and loading conditions were kept constant. The WWTPs included in this analysis were the two large plants at Durham and Rock Creek and the smaller plant in Hillsboro. The Forest Grove plant, generally characterized by small loads and located far above the model upstream boundary, was not considered. The effect of a range of ammonia and CBOD-5 loads from each plant was analyzed separately; no other loads of WWTP oxygen demand were added during each scenario. The modeling effort was focused on an evaluation of the various management strategies for achieving required DO concentrations during the transition period from late fall to the early winter, given continued population growth in the basin. Steady-state hydrologic and loading conditions were assumed; therefore, the results of these simulations should only be used in a semiquantitative manner to compare various loading strategies.

The selection of streamflow and temperature conditions for the hypothetical simulations attempted to bracket conditions under which DO problems would likely occur during November. The hypothetical streamflows are referenced to streamflow in the river at RM 33.3 and included:

(1) 150 ft^3/s, the "worst case" scenario, which is typically exceeded about 90 percent of the time at this site during November;

(2) 300 ft^3/s, an "average base-flow condition," which is exceeded approximately 70 percent of the time; and

(3) 500 ft^3/s, which is considered a potential threshold for detrimental DO depletion and is exceeded approximately 50 percent of the time.

Hypothetical temperature conditions included:

(1) 6°C, which represents a minimum value, likely to be exceeded 90 to 95 percent of the time in November;

(2) 12°C, which represents an "average to warm" value, likely to be associated with the "average base-flow" condition during November; and

(3) 18°C, the "worst case" condition, rarely observed during November except at very low flow.

It should be stressed that the assumption of negligible algal activity is unrealistic under the "worst case" scenarios when extended flow conditions of 150 ft^3/s or water temperatures of 18°C exist. The "worst-case" scenarios provide simply a measure of the oxygen demand from the various WWTP loads, therefore, which would probably be counteracted by photosynthetic oxygen input under these conditions.

Baseline Conditions

Initial simulations were performed for the full range of flow and temperature conditions assuming no loading of CBOD or ammonia from any of the three WWTPs. This analysis provided context for the loading scenarios by describing the natural or background oxygen demand in the river that is not subject to direct management control. Boundary conditions were defined by mean data observed during the base-flow period in November 1992 (table 22). Streamflow values for the tributaries were calculated as a fraction of the selected river streamflow, according to the observed ratio of tributary and river streamflows during the base-flow period. WWTP effluent discharges were considered to be more rigorously managed within the range of flow conditions selected and were kept constant for all flow levels. Input DO concentrations were computed for the various temperature conditions on the basis of observed mean percent- saturation data from the base-flow period. Baseline ammonia, CBOD, and DO concentrations at the upstream boundary (RM 38.5) were calculated using data from further upstream at RM 44.4, as an estimate of conditions not affected by oxygen demand from the Hillsboro WWTP at RM 42.8.

Results from the initial baseline simulations reflect the high level of "intrinsic" oxygen demand, which would be expected under baseflow conditions without significant algal activity (fig. 46). Simulated DO concentrations at RM 3.4 are less than the minimum State standard of 6 mg/L under flow conditions approximating 150 ft^3/s unless the water temperature

Table 22. Summary of baseline boundary inputs to the model CE-QUAL-W2 used during hypothetical simulations of November 1992 base-flow conditions
[DO, dissolved oxygen, in percent saturation; ammonia, ammonia as nitrogen, in milligrams per liter; CBOD, carbonaceous biochemical oxygen demand, in milligrams per liter; WWTP, wastewater treatment plant]

River mile	Description	DO	Ammonia	CBOD
38.5	Upstream boundary	86.3	0.057	4.4
38.1	Rock Creek	77.9	.025	4.7
38.1	Rock Creek WWTP	83.0	0	0
9.3	Fanno Creek	81.6	.043	1.8
9.3	Durham WWTP	83.0	0	0

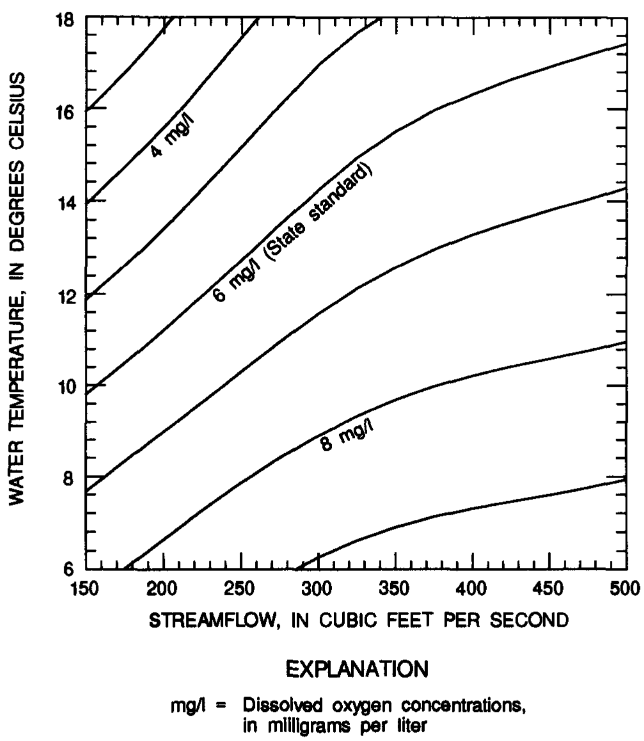


Figure 46. Hypothetical baseline dissolved oxygen concentrations in the Tualatin River, Oregon, at river mile 3.4 as a function of temperature and streamflow at river mile 33.3. Baseline conditions assume zero loads of ammonia and carbonaceous biochemical oxygen demand from the Hillsboro, Rock Creek, and Durham wastewater treatment plants.

is less than 10°C; conversely, simulated DO concentrations at flows of 500 ft^3/s are greater than 6 mg/L unless water temperature is near 18°C. Baseline DO concentrations generally remain above the minimum State standard as temperatures are increased as long as flows are also increased. The implication for management is that the Tualatin River is inherently very vulnerable to oxygen demand under base-flow conditions, especially when water temperatures are warm. This baseline vulnerability is due primarily to the effect of SOD, which decreases as streamflow increases.

Loading Scenarios

The loading scenarios for ammonia and CBOD-5 from the WWTPs were selected to reflect different hypothetical management strategies. Increased ammonia loading from WWTPs during a typical November transition period often approach permitted limits and are a significant factor in oxygen depletion under base-flow conditions. Hypothetical ammonia loads from the two larger plants bracketed "reference" loads, which approximate the permitted loads; loads were set at approximately one-half the reference load, the reference load, and approximately twice the reference load (table 23). The intent was to assess the effect of maximum ammonia loads characteristic of WWTP effluent that is not receiving nitrification treatment. The winter permit for the smaller plant at Hillsboro does not include ammonia; ammonia loads from this plant were designed to bracket the load typically observed during November. Because the Hillsboro plant is located above the model reach, all

Table 23. Hypothetical loads of 5-day carbonaceous biochemical oxygen demand (CBOD-5) and ammonia as nitrogen used in the model CE-QUAL-W2 during simulation of hypothetical November 1992 conditions
[Loads in pounds per day; WWTP, wastewater treatment plant; Mgal/d, million gallons per day; --, not applicable because WWTP located upstream from model boundary]

WWTP	River mile	Discharge (Mgal/d)	Ammonia	CBOD-5
Hillsboro	42.8	--	250	200
		--	500	400
		--	750	600
Rock Creek	38.1	15.6	500	500
			1,000	1,000
			2,000	2,000
Durham	9.3	15.5	200	300
			500	600
			1,000	1,000

loads from this plant were calculated as an increase in the upstream boundary concentrations, weighted to the selected level of flow. Reference and permit level ammonia loads, as well as the mean loads observed during the period 1986–92 for November flow conditions <500 ft^3/s, are listed in table 24.

Table 24. Summary of ammonia permit loads, mean loads of ammonia as nitrogen, and mean loads of 5-day carbonaceous biochemical oxygen demand (CBOD-5) observed during base-flow conditions (streamflow less than 500 cubic feet per second), November 1986–92
[Loads in pounds per day; WWTP, wastewater treatment plant; N, number of observations; --, not applicable because Hillsboro WWTP not subject to ammonia permit; Rock Creek WWTP ammonia permit for streamflow greater than 200 cubic feet per second; Durham WWTP ammonia permit for flow 300–500 cubic feet per second]

WWTP	Ammonia permit	Mean ammonia load	N	Reference ammonia load	Mean CBOD-5 load	N	Reference CBOD-5 load
Hillsboro	--	420	28	500	120	46	200
Rock Creek	1,040	1,100	87	1,000	450	53	500
Durham	390	810	81	500	230	67	300

In contrast to ammonia, WWTP-source loading of CBOD-5 during November base-flow conditions is typically low relative to winter permit levels; those levels are generally approached only during peak-flow conditions. Hypothetical loads of CBOD-5 from the various plants, therefore, were designed to evaluate an increase in mean rather than maximum loading conditions. Loading scenarios included the approximate mean load as the initial reference load, and sequentially increased loads to provide perspective on the effect of increased loading that might be expected as a result of future growth in the basin (tables 23 and 24).

All values for CBOD-5 from the WWTPs were converted to ultimate CBOD before being input to the model. These conversions used observed rates from each plant under the November base-flow conditions when CBOD loading was low. These low loads were typical of base-flow conditions and the concomitant high efficiency of WWTP treatment. The assumption was made that increased loading would occur under similar treatment conditions, although increased loads may be a result of decreased treatment efficiency associated with higher decay rates. Low rates of decay were assumed to be associated with the increased loads, and had corresponding high conversion factors between 5-day and ultimate CBOD; as a consequence, therefore, the effect of increased loading may be overestimated. Loads from the Durham WWTP, in particular, were characterized by extremely low rates of CBOD decay.

Results

An initial evaluation of the relative effects of loading from the various WWTPs is provided in figures 47 and 48, which show simulated DO concentrations at selected river miles over the length of the model reach. For the sake of simplicity, only one combination of flow and temperature conditions is depicted in these figures: the "average base-flow/warm" conditions of 300 ft^3/s and 12°C. Baseline concentrations of DO are also shown, providing important context for the loading scenarios. Note that the DO profiles in these figures are plotted as a function of river mile rather than travel time; the slopes of the curves do not reflect the rates of oxygen depletion, as illustrated in figure 45. As an example, the steepness of the DO profile in figure 47B between RM 8.7 and RM 5.5 might suggest a rapid rate of deoxygenation but is actually a result of the residence time within that reach.

Effect of Ammonia Loads

The effect of varying maximum or permit-level ammonia loads from each of the WWTPs on DO concentrations is depicted in figures 47A, B, and C. The uppermost curve in the plots represents the baseline DO under these flow and temperature conditions; these curves indicate the background DO depletion, which is exerted separate from WWTP oxygen demand. The other curves represent DO profiles associated with the three hypothetical loads. Oxygen concentrations are lowest at RM 3.4, just above the LOC dam.

The greatest effect of WWTP ammonia loading is from loads to the upper river, with the greatest extent of DO consumption resulting from the comparatively large loads discharged from the Rock Creek WWTP, located at RM 38.1. Loads from the Durham plant at RM 9.4, in contrast, cause relatively little oxygen loss within this range of ammonia loads. The larger oxygen reduction from loads in the upper river is a result of the relatively high rate of nitrification used in this part of the river, the long

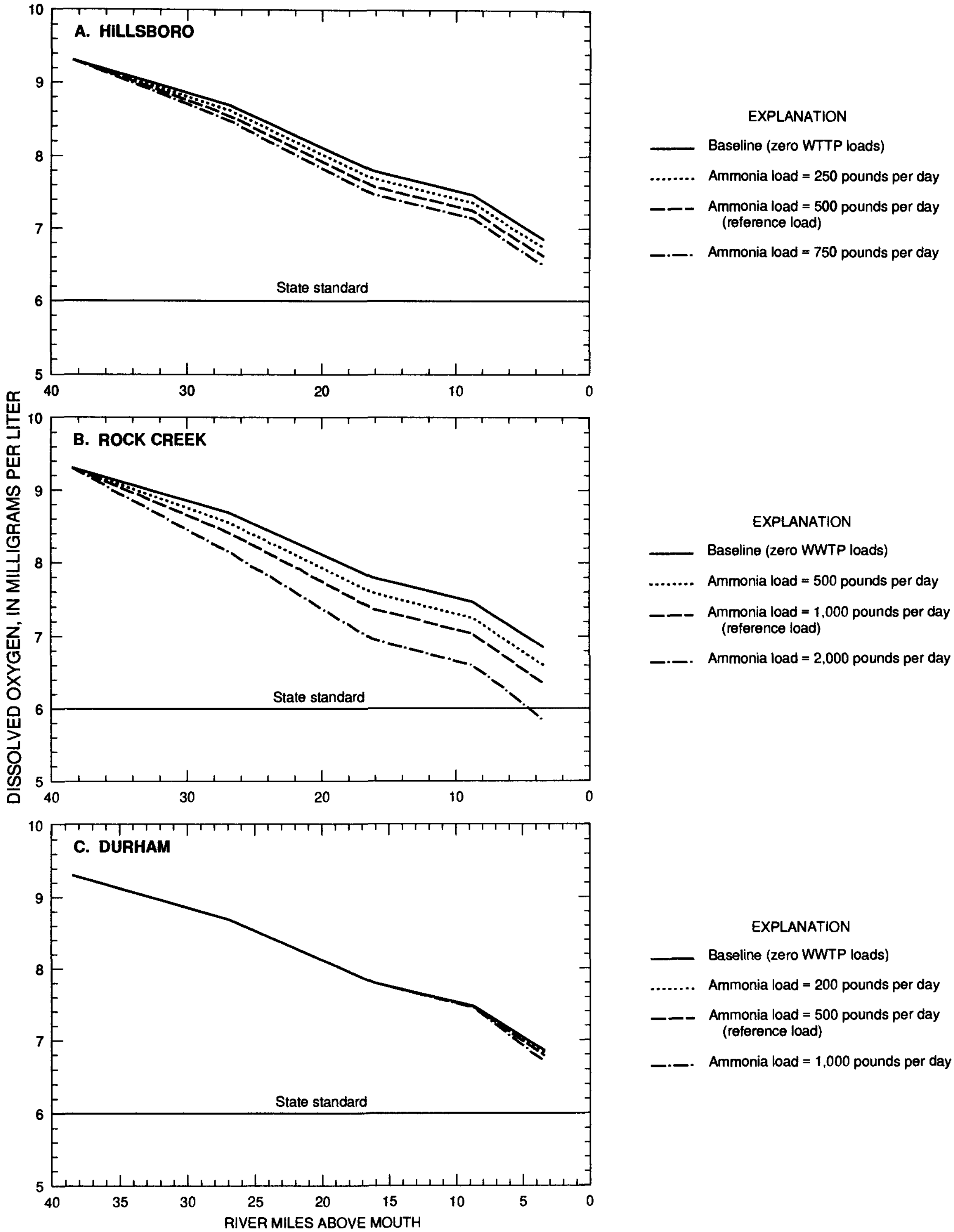


Figure 47. Simulated concentrations of dissolved oxygen at selected sites in the Tualatin River, Oregon, for selected loads of ammonia as nitrogen from the Hillsboro, Rock Creek, and Durham wastewater treatment plants (WWTP), under streamflow and temperature conditions of 300 cubic feet per second and 12 degrees Celsius, respectively. Model predictions for the Hillsboro WWTP start at the model boundary (river mile 38.5) rather than at the plant discharge (river mile 42.8).

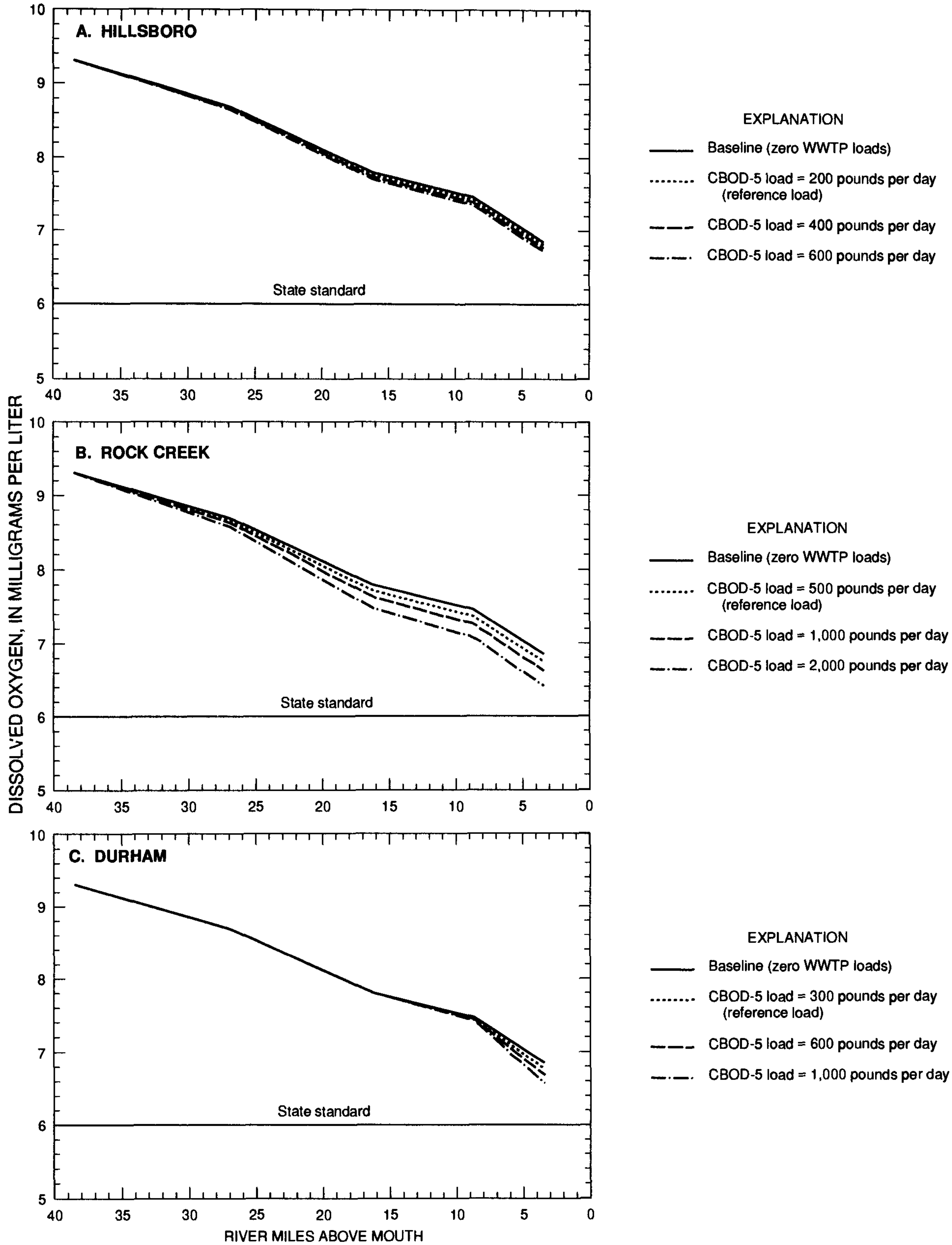


Figure 48. Simulated concentrations of dissolved oxygen at selected sites in the Tualatin River, Oregon, for selected loads of 5-day carbonaceous biochemical oxygen demand from the Hillsboro, Rock Creek, and Durham wastewater treatment plants (WWTP), under streamflow and temperature conditions of 300 cubic feet per second and 12 degrees Celsius, respectively. Model predictions for the hillsboro WWTP start at the model boundary (river mile 38.5) rather than at the plant discharge (river mile 42.8).

potential for reaeration in the backwater reach. Loads to the lower river have a minimal effect because of the low rate of nitrification coupled with the short residence time.

Most of the ammonia load to the upper river is from the Rock Creek WWTP; therefore, the largest benefit to DO concentrations can theoretically be derived from reduced loading from that plant. Under these flow and temperature conditions (300 ft^3/s, 12°C), however, and assuming a reduction in the reference load by 50 percent from both the plants in the upper river basin, the combined benefit at RM 3.4 is slight—an increase in DO concentrations of less than 0.5 mg/L. Furthermore, reducing the combined loads even to near zero (that is, baseline conditions) would result in a DO gain of only about 0.7 mg/L. If the combined reference loads from these two WWTPs are increased, DO concentrations would be reduced by about 0.7 mg/L, a reduction that would result, in theory, in a violation of the State standard. The effect of reference load reduction at the Durham WWTP is essentially negligible; similarly, doubling of the reference load would result in a DO loss of less than 0.1 mg/L under these conditions.

These results imply that any benefit to be gained from reduction of ammonia loads must be placed in the context of baseline DO concentrations that reflect the extensive oxygen demand intrinsic to the lower river. The oxygen depletion that is outside the scope of management control under base-flow and warm-water conditions is high; therefore, assuming that algal activity is negligible, additional oxygen demand from WWTPs must be minimized to avoid violations of the State standard during these river conditions. In spite of the relatively marginal benefits to be gained, therefore, these results suggest that the maintenance of nitrification within the WWTPs in the upper river basin under base-flow conditions would be necessary to maintain DO concentrations in the lower river. Reduction of ammonia loads from the Durham plant within this range of loading provides relatively little benefit.

Effect of Loads of 5-day Carbonaceous Biochemical Oxygen Demand

In general, the effect of mean WWTP loads of CBOD-5 to the upper river is less than the effect of ammonia loading owing to the low rate of in-river decay characteristic of the secondary treatment of WWTP effluent in the basin (fig. 48). The oxygen demand associated with the reference loads from the two plants in the upper river is exerted steadily throughout the length of the river reach, amounting to a combined DO loss at RM 3.4 of less than 0.2 mg/L. With the addition of the demand attributable to the reference load from the Durham plant, the extent of combined oxygen loss remains very low, less than 0.3 mg/L. A doubling of reference loads of CBOD-5 from all three plants results in a combined oxygen demand of approximately 0.5 mg/L, which is significant only because baseline DO concentrations are already excessively depleted. Increases in loads from the two larger plants, Rock Creek and Durham, produce the greatest effect, because their hypothetical loads are higher than those from the Hillsboro plant.

These results indicate that WWTP CBOD loads currently typical of November base-flow conditions are a minor component of oxygen demand in the lower river. If loads are increased on the order of twofold or more, especially from the two larger plants, and if algal activity is assumed to be inconsequential, the resultant oxygen demand becomes significant primarily because the baseline conditions are already depleted. A greater positive benefit in terms of DO concentrations in the lower river is gained, therefore, by reducing the effect of the baseline oxygen demand via increased streamflow compared to further reductions in WWTP loads of CBOD during base-flow conditions.

Interaction of Streamflow and Temperature

Streamflow and water temperature have a dramatic impact on the effects of ammonia and CBOD-5 loading from each of the WWTPs (figs. 49 and 50). Only the DO concentration from RM 3.4 only is presented in these graphs, because the greatest oxygen depletion is expected to occur at this location. These graphs allow the integration of all three factors (streamflow, temperature, and loading), thereby providing a useful tool for the management of water quality under November base-flow conditions. Each graph is composed of three sets of curves, corresponding to the three temperatures used, 6°C, 12°C, and 18°C, which plot the DO concentrations at RM 3.4 as a function of streamflow in the range of 150 to 500 ft^3/s. A splining technique (Paul Turner, Oregon Graduate Institute of Science and Technology, written commun., 1993) was used to estimate DO concentrations between the selected streamflow values; linear interpolation is possible between the

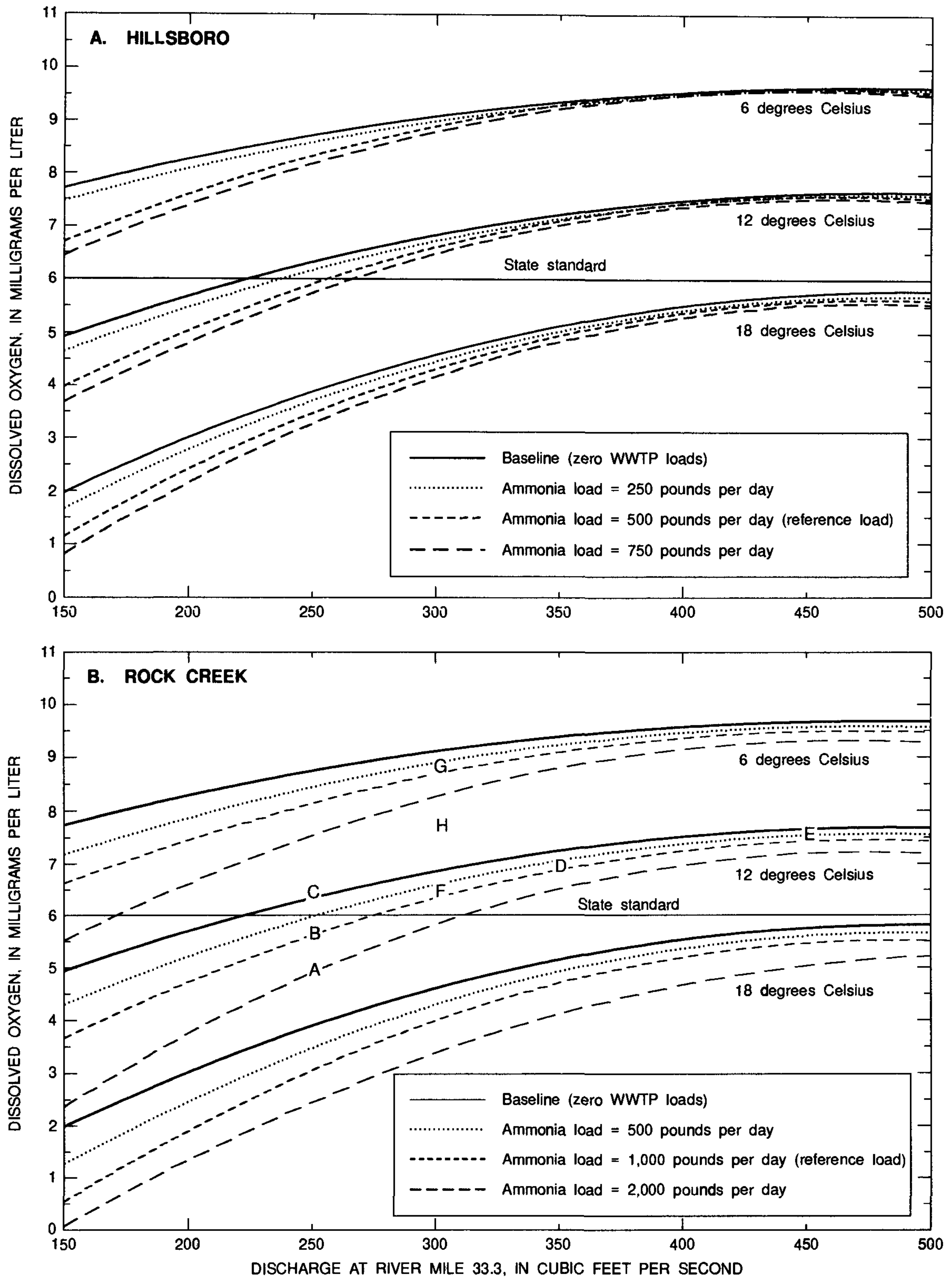


Figure 49. Simulated concentrations of dissolved oxygen in the Tualatin River, Oregon, at river mile 3.4, for selected loads of ammonia as nitrogen from the Hillsboro, Rock Creek, and Durham wastewater treatment plants, under a range of streamflow and temperature conditions. Refer to the text for discussion of lettered points on the graph 49B.

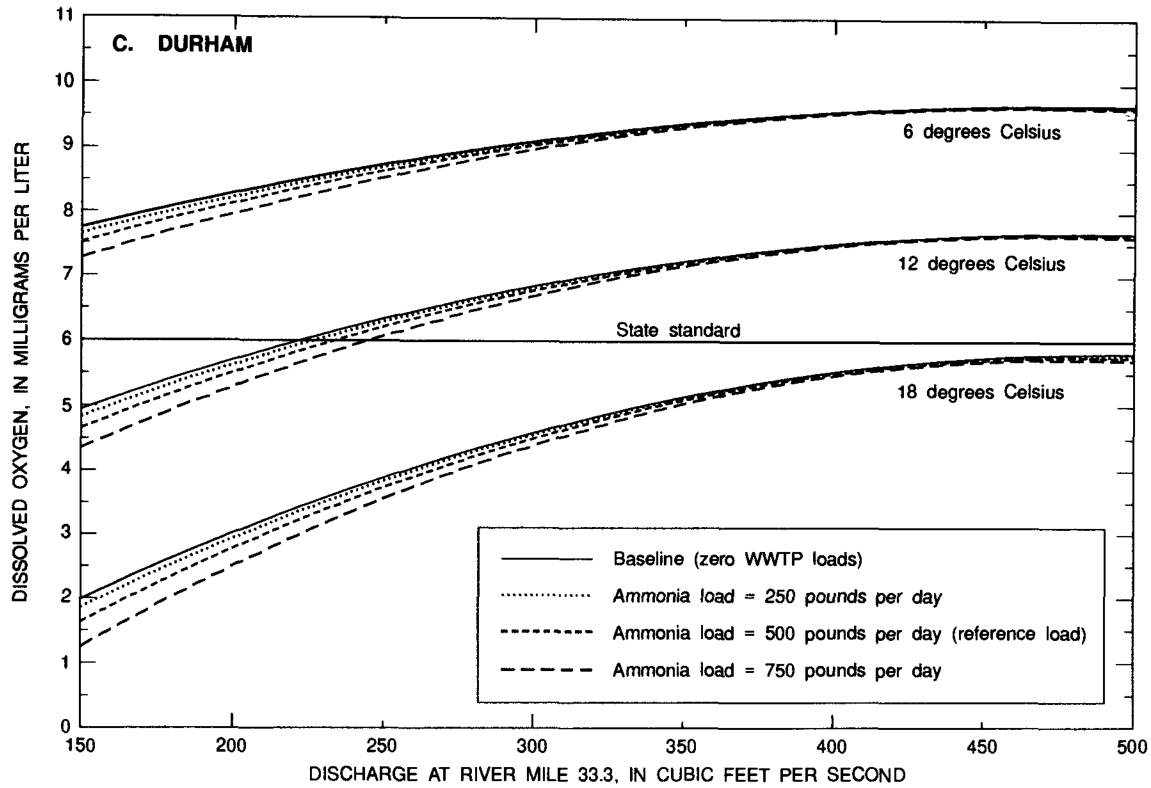


Figure 49. Simulated concentrations of dissolved oxygen in the Tualatin River, Oregon, at river mile 3.4, for selected loads of ammonia as nitrogen from the Hillsboro, Rock Creek, and Durham wastewater treatment plants, under a range of streamflow and temperature conditions. Refer to the text for discussion of lettered points on the graph 49B—Continued.

various temperature curves. Within each set of curves, there are four individual curves that represent the range of loading conditions.

Streamflow and water temperature have a characteristic effect on the basic shape of these curves. The slopes of the curves decrease, and the vertical spread within each set of curves, representing the effect of increased WWTP loads, becomes less pronounced at higher flow levels. A decrease in water temperature also flattens the curve sets and increases the overall DO concentrations. These effects result from the inhibitory effect of lower temperature on biological deoxygenation processes and the increased saturation capacity of colder water for dissolved oxygen.

The potential use of figures 49 and 50 can be illustrated by a detailed examination of figure 49B. Figure 49B was selected because it depicts the effect of ammonia loads from the Rock Creek WWTP, which exert a significant oxygen demand under certain conditions; therefore, figure 49B provides a good demonstration of the interaction of streamflow and temperature in determining the extent of oxygen depletion in the lower river. Three selected combinations of streamflow, temperature, and ammonia load are illustrated by the points A–H.

(1) Points A, B, and C represent changes in ammonia loads (2,000 lb/d, 1,000 lb/d, and baseline, or 0 lb/d, respectively) under unchanging conditions of streamflow (250 ft^3/s) and temperature (12°C). Points A and B both lie below the minimum State standard DO concentration of 6 mg/L; the baseline concentration (C), at approximately 6.3 mg/L, lies scarcely above the minimum concentration. These points clearly demonstrate that, under these flow and temperature conditions, the assimilative capacity of the river for ammonia loads to the upper river is very low. In this case, only by reducing ammonia loading essentially to the baseline level can violations of the State standard be avoided.

(2) Points B, D, and E illustrate the effect of increases in streamflow, at a constant temperature of 12°C, for the reference loading level (1,000 lb/d). The greatest benefit to DO concentrations occurs between points B and D, with an increase in streamflow from 250 ft^3/s to 350 ft^3/s providing a concomitant increase in DO of nearly

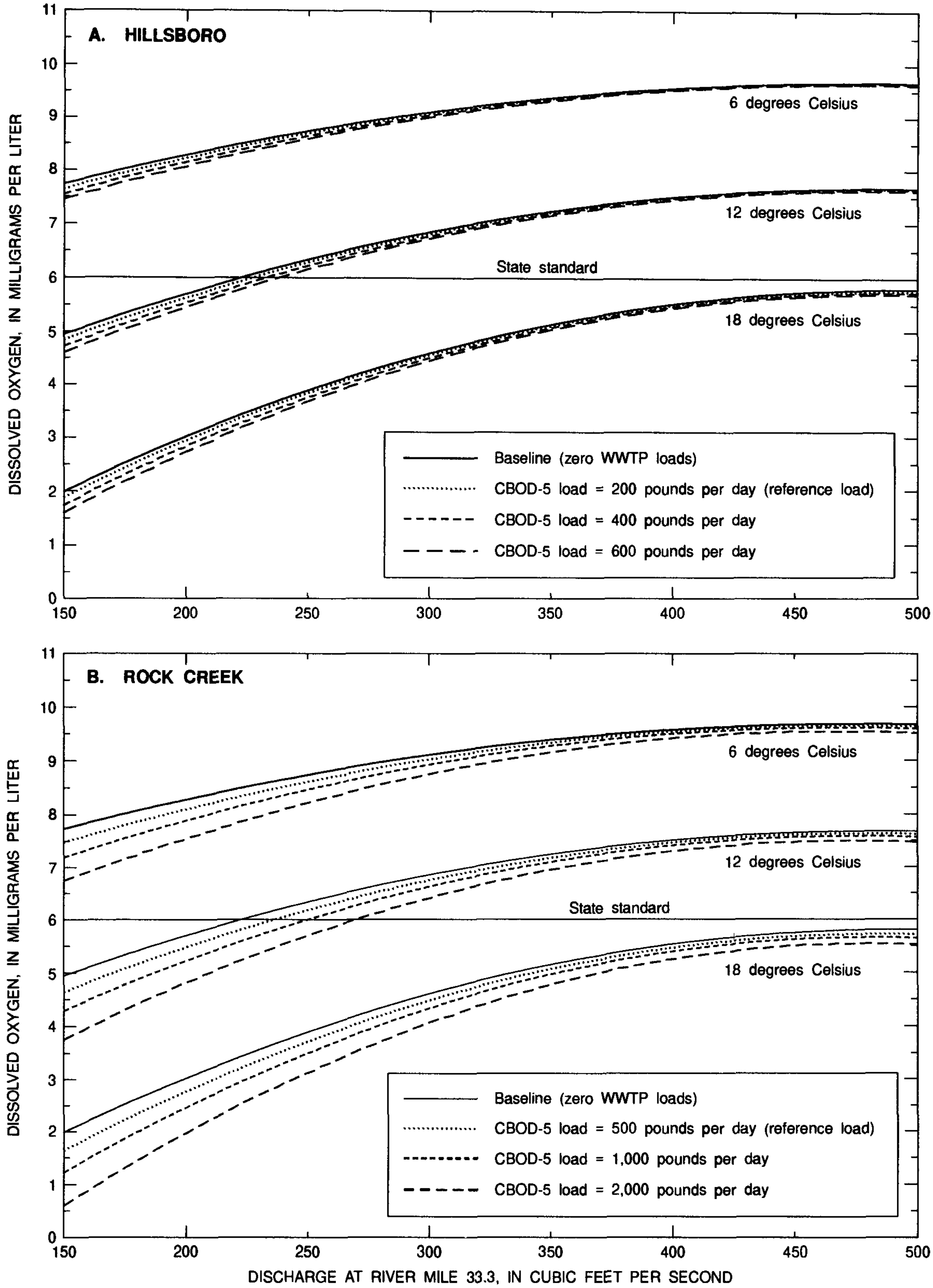


Figure 50. Simulated concentrations of dissolved oxygen in the Tualatin River, Oregon, at river mile 3.4, for selected loads of carbonaceous biochemical oxygen demand from the Hillsboro, Rock Creek, and Durham wastewater treatment plants, under a range of streamflow and temperature conditions.

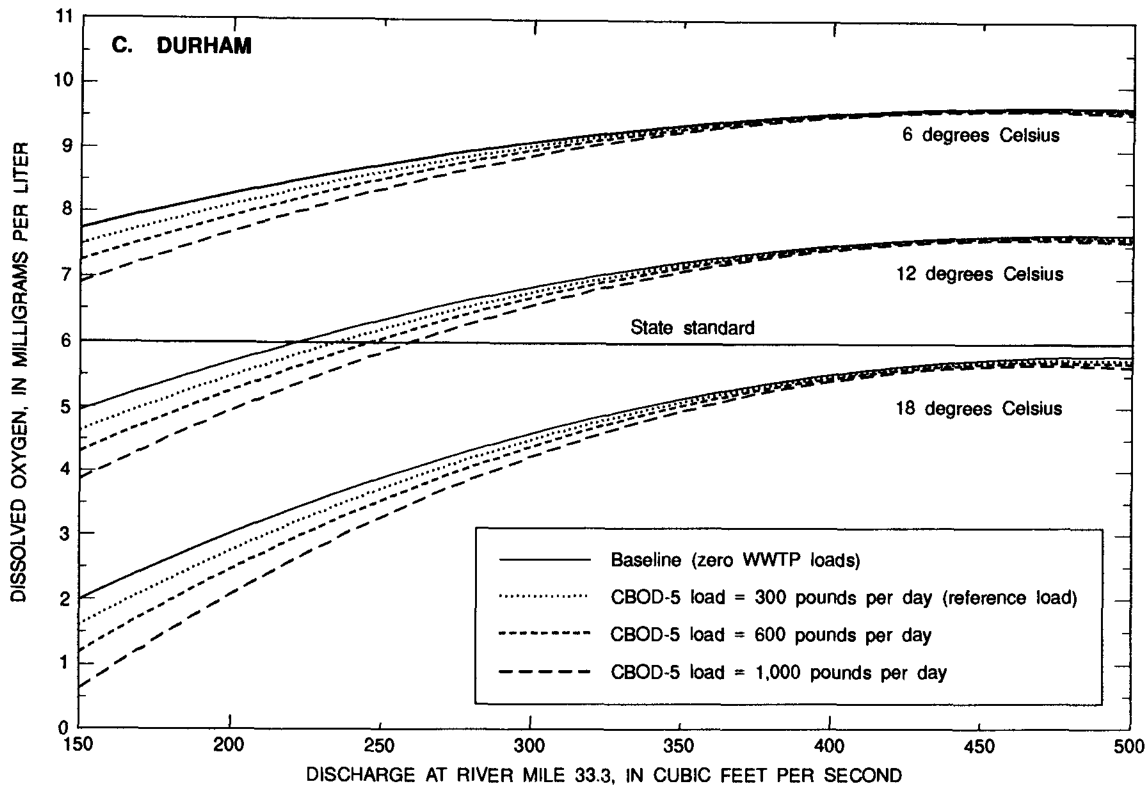


Figure 50. Simulated concentrations of dissolved oxygen in the Tualatin River, Oregon, at river mile 3.4, for selected loads of carbonaceous biochemical oxygen demand from the Hillsboro, Rock Creek, and Durham wastewater treatment plants, under a range of streamflow and temperature conditions—Continued.

1.5 mg/L. As streamflow is increased further, however, the relative benefit decreases gradually; an increase from 350 to 450 ft^3/s results in a DO gain of approximately 0.6 mg/L, from about 6.9 mg/L to 7.5 mg/L. These results suggest that flow augmentation is most effective as a means to deal with increased ammonia loads for flows less than 300 to 350 ft^3/s.

(3) Points F, G, and H show the effect of temperature upon a particular flow and loading condition and illustrate the method of interpolation for temperature values not included in the simulations. Points F and G represent reference loads (1,000 lb/d) at 300 ft^3/s, at 12°C and 6°C, respectively. Point H is calculated as the midpoint on a straight line connecting F and G, and represents the same flow and loading condition at 9°C. In this case, a change in temperature of 3°C alters the expected DO concentration by about 1.2 mg/L. This result reflects the significant effect that temperature has on the capacity of the river to assimilate increased loads, especially at base-flow.

In evaluating figures 49 and 50, it is necessary to remember the assumptions underlying these simulations of November base-flow conditions. The most important of these is the absence of significant algal activity, which is most likely to occur under conditions of extreme base-flow and warm temperature, the "worst-case" scenarios. In addition, the assumption that increased loads of CBOD-5 from the WWTPs would not be associated with a loss in plant treatment efficiency, (that is, an increase in the rate of decay), may not be true for the higher loads simulated. Particularly in the case of the Durham plant, the magnitude of the higher loads may be overestimated. Within the limits of these assumptions, however, figures 49 and 50 can be used to investigate any possible combination of streamflow, temperature, and WWTP load within the ranges simulated by the model. Hopefully, these results will be of practical use for water-quality planning and management purposes.

SUMMARY AND CONCLUSIONS

This report describes the capacity of the Tualatin River to assimilate oxygen-demanding material during winter streamflow conditions, with an emphasis on peak-flow and winter base-flow conditions. The study examined major processes governing concentrations of dissolved oxygen (DO) in the river under different streamflow conditions, as well as the effect of streamflow and temperature on these processes. Water-quality modeling was used to evaluate various wastewater treatment plant (WWTP) loading scenarios during winter base-flow conditions as an aid to management decisions in the basin.

Winter Low-Flow Conditions

Several conclusions can be drawn from the general analysis of winter low-flow conditions. First, in spite of measurable DO depletion in the lower river, DO concentrations throughout the main stem remained well above the State standard of 6 mg/L. The periods of relatively low DO concentrations occurred during periods of lower flow, higher temperatures, and, probably, increased biological activity. In general, however, DO concentrations in the main stem during most of the winter sampling season did not violate minimum State standards.

Second, biological processes during winter low-flow conditions were much less important than during summer base-flow conditions, due mainly to reduced temperatures and travel times. Photosynthesis was reduced essentially to zero for most of the winter sampling season as a consequence of reduced sunlight and decreased residence time. The most important factors governing oxygen concentrations in the main stem were the carbonaceous biochemical oxygen demand (CBOD) decay of loads from both WWTP and tributary sources, and input of tributary waters deficient in DO relative to the main stem. Other microbial processes, including nitrification and sediment oxygen demand (SOD), were probably insignificant.

Third, loads of CBOD from WWTPs and major tributaries were fairly evenly distributed between those sources; WWTPs were the major contributors of ammonia to the river, accounting for 96 percent of the total mean load, and total suspended solids (TSS) loads were predominantly of tributary origin. The relative effect of the loads of CBOD, in particular, depended in large part on the geographic distribution of the sources throughout the river basin, as well as the level of streamflow in the river. The backwater character of the lower river, extending for approximately 30 miles above the Lake Oswego Corporation (LOC) diversion dam, created travel times that were sufficient for CBOD deoxygenation to have a detectable effect in spite of reduced temperatures and higher flows. As a consequence, CBOD loads that were delivered to the main stem above this backwater reach were likely to have exerted a measurable DO demand. Tributary sources were more important than WWTPs in contributing to the mean CBOD load in the upper river. Loads that entered the lower river, however, were less significant in their effect on oxygen consumption, simply as a function of lower residence time.

Peak-Flow Conditions

During peak-flow conditions, oxygen concentrations throughout the main stem remained high, typically greater than 9 mg/L, or about 80 percent saturated. These concentrations were well above the minimum State standard of 6 mg/L. Over the length of the study reach, a mean decline in DO of about 0.8 mg/L was observed in the upper river, between RM 51.5 to RM 38.5. Below RM 38.5, the DO concentrations remained essentially constant.

Loading of CBOD and TSS during peak-flow conditions was dominated by tributary sources, accounting for 83 percent and 99 percent, respectively, of the total daily mean load during the storm. Ammonia loading remained predominantly due to WWTPs. The mean WWTP loads were not significantly larger than those observed during winter low-flow conditions. Tributary loads, especially of TSS, were significantly increased by factors ranging from nearly threefold to tenfold.

Estimated travel time for the length of the study reach during peak-flow conditions was approximately 1 to 2 days. Temperatures were observed throughout the river within the range of approximately 5 to 9°C. These conditions made it unlikely that biological deoxygenation processes, especially nitrification and SOD, would have been significant during the storm. The effect of CBOD was considered to have been minimal in the main stem, largely due to dilution and the greatly reduced residence time. The predominant factor governing the slight reduction in DO concentrations observed in the main stem was the input of tributary water that was relatively depleted in oxygen; other nonpoint sources of oxygen-depleted water may have been involved as well.

Base-Flow Conditions

The winter base-flow period was characterized by a 2-week interval from November 6–20, 1992, with flows averaging less than 300 ft^3/s and water temperatures that generally remained greater than 10°C. These conditions were conducive to significant bacterial activity, although algal activity was minimal. Concentrations of oxygen throughout the main-stem river during this time dropped as the water flowed downstream; mean DO values ranged from 10.0 mg/L (86 percent saturated) at RM 44.4 to 5.9 mg/L (53 percent saturated) at RM 5.5. Daily mean values at RM 3.4 were about 6 mg/L for the first 2 weeks of November in 1992, decreasing to nearly 5 mg/L during the third week, in violation of the minimum State standard.

Mean loads of CBOD and TSS were significantly reduced relative to winter low-flow and peak-flow conditions, by factors from about three to more than two orders of magnitude. The decrease in mean total TSS loading was especially pronounced, reflecting the close relation between streamflow and tributary loading of TSS. CBOD loads from WWTP and tributary sources were essentially equivalent during the base-flow period, whereas TSS loads were dominated by tributary sources. In contrast, ammonia loads were only slightly decreased relative to the higher winter low-flow conditions, and were almost exclusively from WWTP origin. More than 90 percent of these ammonia loads entered the main stem in the upper river, upstream from RM 38.1.

Water-quality modeling of the main-stem river between RM 38.5 and the LOC dam at RM 3.4 was used to determine the relative importance of the major processes depleting DO during the base-flow period. The primary process was SOD, which was especially pronounced in the slow-moving lower river where input of oxygen from reaeration was low; nitrification of ammonia loads to the upper river was also important. Decay of CBOD played a minimal role.

Evaluation of Hypothetical Management Alternatives during Base-Flow Conditions

Hypothetical simulations of a range of flow, temperature, and WWTP loading scenarios demonstrate several key points regarding November base-flow conditions. First, the baseline or "intrinsic" demand operating in the lower river, resulting largely from SOD, is high, particularly at lower flows and warmer temperatures. All loads from WWTPs in the basin must be considered in the context of this intrinsic demand. Second, ammonia loading to the upper river has the most significant effect of the WWTP loads, with most of the ammonia coming from the Rock Creek WWTP. Reduction of WWTP ammonia loading, especially from this one source, would offer more benefit to DO concentrations during November than reduction of loads of CBOD-5, which contribute very little oxygen demand at current levels. Third, the interaction between streamflow and temperature is critical in determining the effect of WWTP loads. For temperatures close to 12°C, maintaining DO concentrations above the minimum State standard require streamflows near 350 ft^3/s.

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