The second annual report (16) describes a subsequent test in which sterile feed water was percolated through a seeded column of Hanford loam at an hydraulic rate of 1.025 m/day, a BOD loading of 20.4 g/m^2/day, and an ABS loading of 1.02 g/m^2/day. The average breakdown of ABS to sulfate and intermediate products was 25 percent. After 33 days on this regime, a schedule of one hour of percolation and five hours of resting was established. This change produced an improvement of ABS breakdown to 35 percent. More important, the complete degradation to SO_4 increased from 9.3 percent to 31.4 percent. In this later phase of the test the hydraulic loading was only 0.27 m/day, the BOD loading was 5.38 g/m^2/day, and the ABS rate was 0.27 g/m^2/day. Here the hydraulic loading was comparable to the Cincinnati studies, but the organic loadings were much weaker,

Years of experience in the early part of this century with sand filtration of settle sewage proved that this process was capable of producing a well-oxidized stable effluent, but only when the beds were operated intermittently. It was essential that the surface of the beds be exposed to the atmosphere for 75 to 90 percent of the total time in order to allow oxygen to permeate the interstices of the soil and assure aerobic conditions at all times. Under continuous dosing and saturated conditions, the rate of percolation decreased rapidly, and the effluent became putrid. For settled sewage, the maximum rate of hydraulic loading was generally in the range of 0.10 to 0.15 m/day and the BOD loading was in the range of 20 to 40 g/m^2/day. Beds were seeded continuously with the wide variety of biological forms found in municipal wastewaters.

The Cincinnati experiments simulated conventional practice to a very large extent, although loadings were generally quite high. The Berkeley tests, however, included several factors that militated against optimum results. First, a large majority of the tests were run with saturated columns, either sterile or pre-seeded. Hence, there was little or no opportunity for aerobic conditions to obtain. Second, the seeding operations for most tests consisted of one application of highly diluted sewage. These pre-seeding operations did not provide much opportunity for development of rich, widely varied cultures. Third, the columns were dosed with a synthetic sterile feed solution, which afforded no opportunity for continual reseeding and the development of a varied flora and fauna in the beds. Fourth, most tests were of relatively short duration, too short to allow the beds to ripen properly. Fifth, most tests were run at high hydraulic loadings. It is not surprising, therefore, that the Berkeley findings and conclusions do not support the Cincinnati results. The latter were far more realistic and comparable to actual practice in intermittent sand filtration, or to the spreading operations at Whittier Narrows.

The Illinois group (11) found that the retention of ABS by adsorption on relatively coarse and uniform clean Ottawa sand in a column was 3.30 mg of ABS per kg of sand when the ABS in the dosing water was 50 mg/l. This retention was based on the difference between the input and the output, rather than on desorption from the sand. Complete adsorption generally required about one hour of contact. When a biological slime was developed on the sand in a similar column, seven times as much ABS was retained. Tests with sulfur-35 indicated that this retention was due to adsorption and not to biodegradation. The column was not operated as an aerobic intermittent sand filter, and there was evidence of clogging and anaerobiosis. However, this test did serve to prove that the presence of biological growths greatly enhances the ability of sand to adsorb ABS.

Another study at Berkeley by Klein and McGauhey (22) compared the behavior of three syndets, polypropyl ABS, straight-chain ABS or the so-called LAS, and alcohol sulfate in a septic-tank percolation field. This was a model study lasting about 115 days. which used a scaled-down septic tank ($\frac{1}{12}$ of prototype width) and percolation field (having about $\frac{1}{20}$ of the recommended surface area for the particular sand used). Radioactive tracers were added to the influent detergent concentrations which were 25 mg/l. The percolation fields were given rest periods to insure that the soil remained aerobic, generally operations were suspended for about two days every 2 to 3 weeks.

The alcohol sulfate was the most degradable with 20.9 percent degraded and 60.8 percent removed in the septic tank, and for the entire septic-tank percolation field system, 54.3 percent of it was degraded and 99.6 percent removed. Removal efficiency in the percolation field only was 98.8 percent. Straight-chain ABS (LAS) was nearly as readily removed as the alcohol sulfate. The LAS was 96.9 percent removed in the septic-tank percolation field. About 70 percent of the removal could be accounted for by inorganic sulfate. Removal efficiency for the percolation field was 97.1 percent. By comparison, the polypropyl ABS in the septic-tank percolation field system showed a removal of 78.2 percent with only 29.9 percent degradation to inorganic sulfate. The percolation field alone showed a removal efficiency of 74.1 percent for the ABS.

A field and laboratory study by Renn et al. (21) showed that in an extended-aeration activated-sludge system servicing a mobile home park with a population of more than 350 people, LAS proved to be much more readily removed than ABS. By supplying the entire community with LAS products, and adding overloads to the treatment system (up to four times the normal municipal sewage concentrations) removal efficiencies still approximated the overall BOD removal, and a non-foaming effluent with an average methylene-blue assay concentration of 0.5 mg/l was obtainable.

Another field test of LAS was reported by Hanna et al. (20) on the removal efficiency for LAS of a small activated-sludge system in an Ohio community. Removals of LAS reached a maximum of 93.5 percent compared to about 52.2 percent removals for ABS under nearly similar operating conditions.

It is generally concluded that good aerobic treatment systems can expect removals of LAS comparable to the overall BOD removals. Foaming problems should be greatly reduced. However, improperly operated treatment systems without a good aerobic environment existing somewhere in the system may show no improvement of detergent removal with LAS over what was found with ABS.

7.03 *Removal of ABS at the Whittier Narrows and Rio Hondo Test Basins.*

The data showing the monthly averages of the concentration of ABS in the surface waters and percolate for both test basins are presented in tabular form in Chapter 4. Figure 4-12, Chapter 4, shows the concentration as function of depth for the various operating periods. These data are also presented in Tables 7-1 and 7-2 of this chapter. The phenomenal performance of the Whittier Narrows Test Basin is readily seen from the data in Table 7-1. During the initial period of this study, the concentration of ABS was reduced by 24 percent after two feet of percolation, while during the last nine months of this study, the ABS concentration dropped by nearly 87 percent after two feet of percolation.

The results (Table 7-2) from the Rio Hondo Test Basin (the basin which received diluted effluent most of the time) do not show as good removal as for the Whittier Narrows site. The most significant difference is seen under Period III during which time the Whittier Narrows site was loaded three times per week with loads gradually increasing from 0.6 feet per dose to 1.4 feet per dose, while the Rio Hondo site was flooded continuously for about six days with the water depth held nominally at 2 feet and then the basin was allowed to drain and dry for about two weeks. As seen in Table 7-3, the average surface loading of ABS at each basin was about the same, namely 0.22 and 0.24 gm ABS per m^2 per day, during Period III. The average surface loading of ABS for each period was computed by taking the product of the average concentration of ABS in the applied water and the total volume of water applied divided by the product of the bottom area of the basin and the total number of calendar days in the period. The superiority of the intermittent spreading operation relative to the more prolonged flooding for ABS can be judged by the fact that 83 percent of the applied load was removed in two feet of percolation at Whittier Narrows, while only 41 percent of the applied load was removed at the Rio Hondo Test Basin. Nevertheless, after eight feet of percolation the Rio Hondo site showed about 90 percent removal although Whittier Narrows was still better with a 92 percent removal at the same depth. These estimates are somewhat conservative because the concentrations of ABS in the percolate at both basins are generally quite small, namely less than 0.5 mg/l, and the methylene-blue assay on these waters can easily result in an error of ± 0.1 mg/l when the concentration is less than 0.5 mg/l.

7.04 *Soil Extractions for ABS at the Test Basins.*

In order to estimate the amount of removal attributable to adsorption on the soil and not to biodegradation, several core samples were taken from the test basins and extracted for ABS. The results of these extractions are reported in Tables 7-6 and 7-7. To make a material balance for ABS at both basins the product of the total volume of water applied to a basin in a month and the average concentration in the percolate was computed and presented in Tables 7-4 and 7-5.

The extraction process employed for the ABS assay of the soil samples was developed by Jesse M. Cohen of the USPHS at the R. A. Taft Sanitary Engineering Center in Cincinnati. Dried soil samples were extracted continuously for eight hours in a Soxhlet extractor with a solvent mixture of 1 + 1 benzene and methanol containing 0.04N ammonium hydroxide. At

Table 7-1—Summary of ABS concentrations in percolate at Whittier Narrows Test Basin

Period	Dates	Average concentration in milligrams per liter				
		Surface	2 ft.	4 ft.	6 ft.	8 ft.
I	January–April 1963	3.23	2.38	1.72	0.84	0.76
	Percent remaining	100.0	73.6	53.2	26.0	23.5
II	May 1963–January 1964	1.96	1.08	0.59	0.57	0.45
	Percent remaining	100.0	55.1	30.1	29.1	23.0
III	February–June 1964	2.07	0.35	0.24	0.18	0.16
	Percent remaining	100.0	16.9	11.6	8.7	7.7
IV	July 1964–March 1965	2.54	0.34	0.33	0.35	0.29
	Percent remaining	100.0	13.4	13.0	13.8	11.4

Table 7-2—Summary of ABS concentrations in percolate at Rio Hondo Test Basin

Period	Dates	Average concentration in milligrams per liter				
		Surface	2 ft.	4 ft.	6 ft.	8 ft.
I	January–April 1963	1.04	0.46	0.36	0.28	0.29
	Percent remaining	100.0	44.2	34.6	26.9	27.9
II	May 1963–January 1964	1.08	0.15	0.08	0.08	0.09
	Percent remaining	100.0	13.9	7.4	7.4	8.3
III	February–December 1964	0.98	0.58	0.19	0.11	0.10
	Percent remaining	100.0	59.1	19.4	11.2	10.2
IV	January–March 1965	0.33	0.12	0.08	0.08	--
	Percent remaining	100.0	36.4	24.2	24.2	--

Table 7-3—Average surface loading of ABS

Period	Days	Whittier Narrows Test Basin	
		gm. ABS/ m^2/day	lb. ABS/ acre/day
I	125	0.53	4.7
II	276	0.12	1.1
III	151	0.24	2.1
IV	274	0.79	7.0
Weighted mean		0.43	3.7

Period	Days	Rio Hondo Test Basin	
		gm. ABS/ m^2/day	lb. ABS/ acre/day
I	99	0.15	1.3
II	276	0.080	0.71
III	335	0.22	1.9
IV	90	0.028	0.25
Weighted mean		0.14	1.2

the end of the extraction time, the solvent was evaporated, and the residue was dissolved in hot distilled water. Analyses for ABS were performed on this solution by the methylene-blue procedure.

In order to estimate the background of the natural soil at the test basins, that is, the extractables which react with methylene blue to give an apparent ABS concentration, some samples of soil taken with an auger device prior to the installation of the test basins were assayed. These data can be compared with the results of assays of samples taken from test sites on 19 June 1963, 18 October 1963, 5 January 1965, and 30 March 1965, as presented in Tables 7-6 and 7-7. It is recognized that a proper soil extraction study would necessitate the assaying of many soil columns from a network of locations in each test basin. However, the extractions from various depths on each of the 2-inch soil cores gives some idea of the amount of adsorbed ABS. For each of the columns of extraction data, a simple arithmetic mean value was computed. Comparison of all the data taken over nearly a three-year interval shows the mean value of adsorbed ABS to increase apparently during the first year of operation and then decrease to less than the original background data taken before the spreading of waters at

Table 7-4—ABS material balance at the Whittier Narrows Test Basin

Month	Weight of ABS in kilograms at				
	Surface	2 ft.	4 ft.	6 ft.	8 ft.
January 1963*	11.74	8.36	6.14	3.58	3.76
February	7.62	3.54	4.42	1.16	1.02
March	2.91	2.17	0.69	0.79	0.39
April	1.23	1.77	1.16	0.47	0.47[a]
May	1.03	1.54	1.31	1.18	0.77
June	0.78	0.61	0.43	0.30	0.30[a]
July	1.86	0.96	0.34	0.34[a]	0.34[a]
August	1.53	0.81	0.42	0.36	0.36[a]
September	0.94	0.47	0.18	0.19	0.19[a]
October	1.09	0.46	0.11	0.18	0.18[a]
November	1.06	0.34	0.10	0.15	0.11
December	1.35	0.44	0.17	0.22	0.14
January 1964	1.44	0.34	0.16	0.14	0.12
February	1.22	0.27	0.10	0.12	0.09
March	1.96	0.33	0.18	0.15	0.13
April	2.09	0.32	0.20	0.18	0.16
May	2.54	0.47	0.27	0.24	0.24
June	3.98	0.52	0.35	0.35	0.35
July	6.18	0.77	0.50	0.62	0.75
August	6.80	1.11	0.83	0.55	0.69
September	7.10	0.82	0.82	0.69	0.69
October	7.81	1.19	0.89	1.19	1.04
November	6.71	1.05	0.94	1.14	1.00
December	7.90	1.12	1.02	1.33	0.93
January 1965	7.94	0.95	1.11	1.11	0.79
February	9.45	1.21	1.39	1.45	0.87
March	9.34	1.15	1.65	1.92	1.15
Totals	115.60	33.09	25.88	20.10	17.03
Percent remaining	100.0	28.6	22.4	17.4	14.7

[a] No data available. Value extrapolated from data at lesser depth assuming no removal.
* Includes data since 27 December 1962.

Table 7-5—ABS material balance at Rio Hondo Test Basin

Month	Weight of ABS in kilograms at				
	Surface	2 ft.	4 ft.	6 ft.	8 ft.
January 1963*	1.09	0.50	0.42	0.42	0.50
February	1.74	0.97	0.80	0.48	0.44
March	1.19	0.43	0.27	0.20	0.16
April	0.73	0.20	0.10	0.12	0.12
May	1.15	0.17	0.07	0.08	0.08
June	1.11	0.09	0.00	0.00	b
July	1.71	0.20	0.08	0.06	b
August	1.72	0.23	0.08	0.06	b
September	0.54	0.06	0.04	0.04	0.04
October	0.23	0.06	0.06	0.06	b
November	0.31	0.07	0.00	0.02	0.05
December	0.21	0.08	0.08	0.08	0.08
January 1964	0.31	0.10	0.10	0.08	0.08
February	0.53	0.38	0.09	0.19	0.19
March	2.47	1.57	0.22	0.90	0.45
April	‡	‡	‡	‡	‡
May	6.18	4.88	1.63	0.33	b
June	0.36	0.18	0.09	0.09[a]	b
July	0.68	0.23	0.23	0.23	b
August	1.90	0.95	0.28	0.09	b
September	4.71	2.71	1.12	0.37	b
October	1.94	0.91	0.34	0.11	b
November	0.74	0.55	0.18	0.18	b
December	0.34	0.21	0.04	0.04	b
January 1965	0.07	0.02	0.01	0.01	b
February	0.22	0.08	0.06	0.06	b
March	0.47	0.17	0.17	0.17	b
Totals	32.65	16.00	6.56	4.47	†
Percent remaining	100.0	49.0	20.1	13.7	--

[a] No data available. Value extrapolated from lesser depth assuming no removal.
[b] No data available.
* Starting 22 January 1963.
† Total not calculated because there are too few data.
‡ No data available this month.

Table 7-6—Results of extractions for adsorbed ABS from soil cores at the Whittier Narrows Test Basin

Sample depth (feet)	Adsorbed ABS in milligrams per kilogram of dry soil				
	December 1962	19 June 1963	18 October 1963	5 January 1965	30 March 1965
Surface	--	3.8	5.4	5.0	2.6
0.5	--	2.3	--	--	--
1.0	4.7	--	--	--	1.0
1.5	--	--	--	2.8	1.6
2.0	1.9	1.6	2.8	--	--
2.5	--	--	--	1.0	1.2
3.0	--	--	--	--	--
3.5	1.8	--	--	1.5	1.6
4.0	--	5.8	3.7	--	--
4.5	1.0	--	--	1.1	1.4
5.0	--	--	--	--	--
5.5	--	--	--	3.3	1.2
6.0	2.4	5.0	5.9	--	--
6.5	--	--	--	1.1	2.8
7.0	--	--	--	--	--
7.5	--	--	--	1.0	1.1
8.0	1.2	6.8	9.8	--	--
8.5	--	--	--	0.5	1.5
Total	13.0	25.3	27.6	17.3	16.0
Mean	2.17	4.22	5.52	1.92	1.60

Table 7-7—Results of extractions for adsorbed ABS from soil cores at the Rio Hondo Test Basin

Sample depth (feet)	Adsorbed ABS in milligrams per kilogram of dry soil				
	December 1962	19 June 1963	18 October 1963	5 January 1965	30 March 1965
Surface	5.1	5.8	3.6	3.8	1.8
0.5	--	2.6	--	--	--
1.0	--	--	--	--	1.1
1.5	--	--	--	2.0	1.0
2.0	--	0.6	0.6	--	--
2.5	--	--	--	--	0.5
3.0	--	--	--	--	--
3.5	1.2	--	--	0.7	0.4
4.0	--	0.6	0.6	--	--
4.5	--	--	--	0.9	0.6
5.0	--	--	--	--	--
5.5	--	--	--	0.9	--
6.0	--	0.4	1.0	--	--
6.5	0.4	--	--	--	--
7.0	--	--	--	--	--
7.5	--	--	--	1.1	--
8.0	--	0.4	0.5	--	--
8.5	--	--	--	--	--
10	0.6	--	--	--	--
Total	7.3	10.4	6.3	11.3	5.4
Mean	1.82	1.73	1.26	1.88	0.90

the test sites. This phenomenon may be partly an indication of the inhomogeneity of the soil at the basin sites and may also be an indication that during the first years of operation adsorption played a significant part in the overall removal. As the basin ripened and organisms capable of degrading ABS developed, adsorption became only the initial step in the biodegradation process.

The removal data presented for the various periods in Tables 7-1 and 7-2 showed that the percentage removal improved as the basin aged even though the applied loads increased. This change is not characteristic of an adsorption process. A simple estimate of the adsorptive capacity of the test basins can be made assuming an average porosity of 40 percent and a specific gravity of about 2.7. The total weight of soil contained in a volume about 323 m^2 by 2.42 m (8 ft.) deep is about 1280 x 10^3 kilograms. With no background correction, assuming the adsorbed ABS at 2 mg ABS per kg dry soil, then one could expect about 2.56 kilograms of ABS to be adsorbed on the basin soil. From Table 7-4, it is shown that at the Whittier Narrows site over the 27 months of operation about 116 kg of ABS was applied to the basin and the percolate at eight feet carried away about 17 kg, leaving about 99 kg behind. Of the 99 kg left in the basin only about 3 percent can be conservatively accounted for by adsorption to the soil. Consequently, about 97 percent was degraded biochemically.

7.05 *Reliability of Extraction Procedure.*

The efficiency of the extraction procedure developed by Cohen was found to average about 97.5 percent for several quartz sands (personal communication). To test the efficiency of recovery in our laboratory, a simple adsorption study was conducted. Four samples of soil from the surface at Whittier Narrows were processed through the extraction procedure and found to have an average of 5.0 mg ABS per kg of dry soil. These samples after extraction were dried and placed in Erlenmeyer flasks in 250 ml of distilled water to which was added 1 mg of ABS to make an ABS concentration of 4 mg/l. The flasks were allowed to sit at room temperature and were manually agitated about twice each day. Samples of the supernatant were assayed periodically for ABS concentration by the methylene-blue procedure. After about a three-month interval, the soil samples were separated from the supernatant and each was assayed for ABS. The supernatants averaged about 1.5 mg/l. Extractions from the soil samples averaged about 14 mg ABS per kg of dry soil. A material balance accounting for losses of ABS from the main solution with each aliquot of sample removed for analysis resulted in an average ABS recovery for three samples of about 70 percent. The reproducibility of the data was very good. One sample was inadvertently lost and the other three gave values of 70.2, 69.2, and 71.4 percent recovery.

CHAPTER 8

BEHAVIOR OF CARBONACEOUS AND NITROGENOUS SUBSTANCES IN THE FIELD SPREADING BASINS

8.01 *The Concept of Total Oxygen Demand.*

The introduction of wastewaters and reclaimed waters into confined systems, such as ground-water basins where the detention times will be in the order of years, has caused the water-quality-control engineer to re-evaluate the conventional criteria for describing the biological stability of a water. When wastewaters are discharged into surface waters, one of the most critical effects is deoxygenation, for which the rate and magnitude can be approximated by the test for biochemical oxygen demand (BOD). It is conventional to determine the 5-day BOD at 20°C, for within this period of time a flowing stream may have reached a larger body of water such as the ocean, or it has been modified by subsequent discharges or use. In surface waters, moreover, the rate of oxygen utilization is generally more significant than the ultimate demand.

When wastewater is percolated into the ground and reaches the ground-water table, it will probably flow laterally at velocities in the general magnitude of 0.1 foot per day. Thus, the time between its recharge and subsequent withdrawal may easily be thousands of times longer than the detention times associated with surface flows. For this reason, it is important to examine the biochemical behavior of wastewater when it is held in a confined system for long periods of time.

In this study, the source of water for recharge was a well-treated secondary effluent from an activated-sludge treatment plant. The oxygen demand as expressed by the 5-day BOD was normally below 10 mg/l and consequently this effluent was more than satisfactory for discharge to a surface-water system. A long-term measurement of the biochemical oxygen demand of this effluent, however, shows that the total oxygen demand is in the order of 100 to 200 mg/l. This is the value that is important in assessing the oxygen relationships in a ground-water system.

8.02 *A Rapid Method for Predicting the Total Oxygen Demand of a Wastewater.*

The substances that exert an oxygen demand in a well-oxidized wastewater effluent can be classified into two main groups, *viz,* the carbonaceous and nitrogenous materials. Among the carbonaceous substances in the effluent will be found such refractory compounds as the synthetic detergents and other exotic organic materials that do not break down readily during conventional treatment. The nitrogenous substances in the effluent will exist principally in the state of minus-three oxidation number, i.e., primarily ammonia-nitrogen and organic nitrogen.

At the suggestion of Dr. William Samples, Assistant Professor of Civil Engineering at Caltech, a calculation of the long-term biochemical oxygen demand was made using two analyses of a strictly chemical nature. He maintained that the carbonaceous demand of the waste could be found by running the test for chemical oxygen demand (COD) and that the oxygen demand of the nitrogen could be estimated from the measured amount of minus-three oxidation number nitrogen (ie., ammonia-nitrogen plus organic nitrogen) and a caclulation of the amount necessary to convert this nitrogen to the nitrate form. The total oxygen demand is a sum of the carbonaceous and nitrogenous oxygen demands.

An example of such a calculation and its comparison with a conventional test for biochemical oxygen demand run over a period of 100 days is shown in Figure 8-1. The predicted value of the long-term BOD was arrived at by measuring the COD, which turned out to be 40 mg/l, and measuring the ammonia and inorganic nitrogen. Two mols of oxygen (O_2) per mol of nitrogen (N) are required to oxidize the nitrogen nitrate. Thus, 25.2 mg N/l requires 115 mg O_2/l to satisfy its oxygen demand. The predicted long-term BOD is then estimated as 40 plus 115 or 155 mg/l. This value compares very favorably with the measured values shown in Figure 8-1.

Several other features of the oxygen demand exerted by the treatment plant effluent can be seen in Figure 8-1. The 5-day BOD is small, namely 2.7 mg/l or less than two percent of the total oxygen demand. The manner in which the oxygen demand is exerted will be quite different from the textbook examples for raw or settled sewage. There is no evidence of a time separation in the carbonaceous and nitrogenous oxygen demands. In raw sewage the carbonaceous substances are often the readily degradable substances, while in a secondary effluent the remaining carbonaceous compounds are considerably more resistant. Also, in a suitable environment such as an aerobic soil system, it is apparent that the nitrogenous demand will be exerted more rapidly and more completely than the carbonaceous demand. It is also significant to note that the nitrogenous oxygen demand is more than four times the carbonaceous demand.

In the intermittent percolation of wastewater through an aerobic soil system, the nitrogenous oxygen demand and part of the carbonaceous demand are exerted rapidly, even in a few feet of vertical drainage. The comparable oxygen demand in a BOD bottle may require 100 days or longer. Consequently, the computation of total oxygen demand from COD and minus-three oxidation number nitrogen is not only much faster than the determination of the 100-day BOD; it is also more realistic for soil systems.

8.03 *Chemical Oxygen Demand in the Test Basins.*

The chemical oxygen demand (COD) is representative of the unstabilized carbonaceous material in a water or soil system. The averages of the weekly analyses of COD performed on surface and percolated

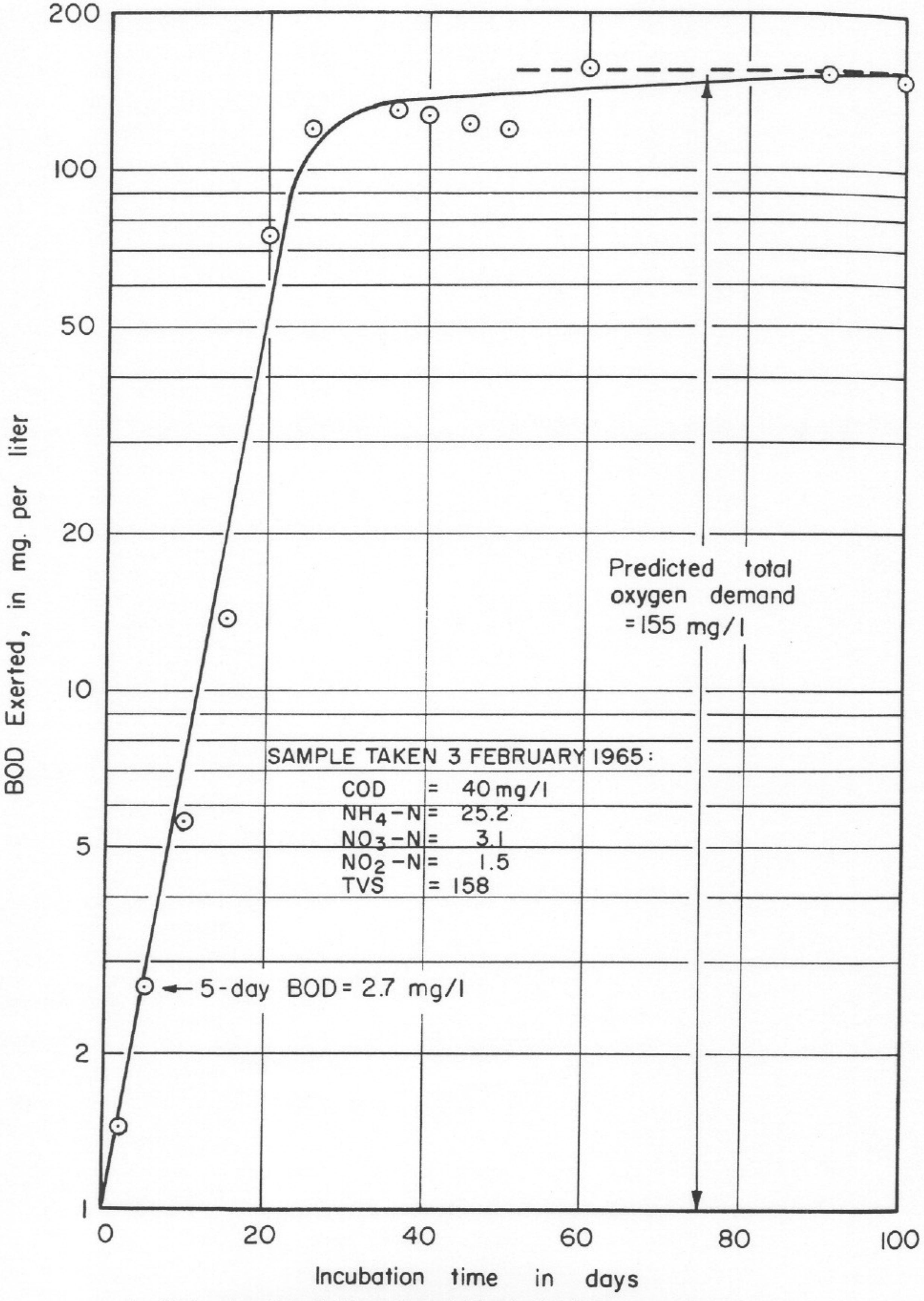

Fig. 8-1—Long-Term Biochemical Oxygen Demand of Sample of Final Effluent from Whittier Narrows Water Reclamation Plant

waters from each of the test basins are presented in Tables 4-9 and 4-19 in Chapter 4. Summaries of the COD data averaged over the periods of significance defined by various hydraulic operations are shown in Tables 4-27 and 4-28, and also in Figure 4-13. For the purpose of discussion in this section, summaries of the COD data presented in Chapter 4 are shown again in Tables 8-1 and 8-2.

In addition to the average concentrations of COD at various depths for the periods of interest, a material balance of COD is also shown in Tables 8-1 and 8-2. This material balance is calculated by taking the average COD concentration during a period and multiplying it by the total volume of water applied to each basin during the period, so that the product yields a weight of COD (expressed as kilograms of O_2) at each level in each test basin. For each period considered, an average surface loading was calculated for both test basins, based on the total number of calendar days in the period and the area of the bottom of each basin.

At Whittier Narrows Test Basin, the COD of the wastewater effluent at the surface was nearly constant over the entire study. At the Rio Hondo site, however, the surface water was subject to a large monthly variation. This fluctuation is to be expected since the sources of water for the Rio Hondo Basin range from storm run-off to nearly undiluted plant effluent. Interestingly, the values of COD at Rio Hondo are not diminished in proportion to the diluting water as are other constituents of the plant effluent; consequently, there is a possible difference in the nature of compounds constituting the COD at each of the basins. At the Rio Hondo site, operating personnel have reported noticeable growths of algae in the water, while at Whittier Narrows fewer algal concentrations have been observed. Apparently the diluted plant effluent contains adequate nutrients to stimulate these algal growths which in turn represent a substantial COD in the surface water at Rio Hondo.

At both of the test sites, there is a significant decrease in the COD of the percolated waters down to the four-foot level. Generally each basin has improved with age in its ability to remove COD from the percolating waters. The best results were obtained at the Whittier Narrows site, during the latter part of the study (Period IV) when the average surface loading was about 12 gm COD per m^2 per day (about 109 lb COD per acre per day), showing approximately a 75-percent removal of COD after four feet of percolation with a resulting COD concentration of about 10 mg/l.

Table 8-1—Summary of COD data for Whittier Narrows Test Basin

Period	Dates	Average concentration of chemical oxygen demand in milligrams O_2 per liter at Surface	2 ft.	4 ft.	6 ft.	8 ft.
I	Jan.–Apr. 1963	47.6	41.1	44.0	52.2	30.3
	Percent remaining	100.0	88.1	94.4	112.0	65.0
II	May 1963–Jan. 1964	33.7	24.5	21.4	79.4	52.9
	Percent remaining	100.0	72.7	63.5	236.0	157.0
III	Feb. 1964–June 1964	36.7	8.8	9.1	12.6	38.8
	Percent remaining	100.0	24.0	24.8	34.3	105.9
IV	July 1964–Mar. 1965	39.3	10.4	9.7	17.0	14.6
	Percent remaining	100.0	26.4	24.7	43.3	37.2
Weighted average	Jan. 1963–Mar. 1965	37.8	19.4	17.8	40.8	29.9
	Percent remaining	100.0	51.3	47.1	108.0	79.0

Period		Material balance of chemical oxygen demand in kilograms* of O_2 at Surface	2 ft.	4 ft.	6 ft.	8 ft.
I		304.6	263.0	281.6	334.0	193.9
II		189.0	137.4	120.0	445.2	296.6
III		204.4	49.0	50.7	70.2	216.1
IV		1081.0	286.0	266.8	467.6	401.6
Total		1779.0	735.4	719.1	1317.0	1108.2

* Equals volume of water spread in period × average COD concentration.

Period	Calendar days	Average surface loading of chemical oxygen demand† gm. O_2/m^2/day	lb. O_2/acre/day
I	125	7.5	67
II	276	2.1	19
III	151	4.2	37
IV	274	12.2	109
Weighted average	826	6.6	59

† Equals weight of COD divided by number of calendar days divided by basin bottom area.

Table 8-2—Summaries of COD data for Rio Hondo Test Basin

Period	Dates	Average concentration of chemical oxygen demand in milligrams O_2 per liter at Surface	2 ft.	4 ft.	6 ft.	8 ft.
I	Jan.–Apr. 1963	27.4	25.4	18.4	15.8	17.0
	Percent remaining	100.0	92.6	67.1	57.6	62.0
II	May 1963–Jan. 1964	34.5	13.5	18.1	12.4	45.5
	Percent remaining	100.0	39.1	52.4	35.9	132.0
III	Feb.–Dec. 1964	35.0	16.0	8.9	7.0	7.4
	Percent remaining	100.0	45.7	25.4	20.0	21.2
IV	Jan.–Mar. 1965	12.5	7.5	6.6	7.6	--
	Percent remaining	100.0	60.0	52.8	60.8	--
Weighted average	Jan. 1963–Mar. 1965	31.4	15.6	13.3	10.7	8.8
	Percent remaining	100.0	49.6	42.3	34.0	28.0

Period		Material balance of chemical oxygen demand in kilograms* of O_2 at Surface	2 ft.	4 ft.	6 ft.	8 ft.
I		128.3	119.0	86.2	74.0	79.6
II		226.1	88.5	118.6	81.2	298.2
III		862.5	394.3	219.3	172.5	182.4
IV		31.0	18.6	16.4	18.8	--
Total		1,247.9	620.4	440.5	346.5	--

* Equals volume of water spread in period × average COD concentration.

Period	Calendar days	Average surface loading of chemical oxygen demand† gm. O_2/m^2/day	lb. O_2/acre/day
I	99	4.0	36
II	276	2.5	22
III	335	8.0	71
IV	90	1.0	8.9
Weighted average	800	4.8	44

† Equals weight of COD divided by number of calendar days divided by basin area.

Both test basins exhibit a peculiar pattern of COD concentrations in the percolates below the four-foot level. At greater depths, the concentration of COD substances in the percolate was normally found to increase. An explanation of this phenomenon leads to some interesting hypotheses. Tables 4-31 and 4-32 show that there are times when the dissolved-oxygen concentration is zero at all sampling points below ground. All oxygen that gets down to the soil organisms is transported by some mechanism from the ground surface. It is reasonable to expect that at a level below four feet, anaerobic conditions can exist for periods of time during which inorganic carbon can be changed to organic carbon. Total carbon measurements indicate that the COD substances in the percolate are truly carbon and not some other interfering substance. The inorganic carbon sources are believed to be principally carbon dioxide in the soil air and bicarbonate and carbonate in the soil water. Although the particular organic substances constituting the high COD in the percolates below four feet are not known, it is certainly reasonable to believe that they are not typical of the man-made exotic organics found in wastewater effluents.

The peculiar nature of the COD in the percolate may also be attributed to some sort of leaching phenomena. It is difficult, however, to explain why the high values ,of COD show a slight seasonal pattern and a tendency to increase and then decrease again after deeper percolation. Tables 8-3 and 8-4 present some physical data taken from soil cores at each of the test basins in March 1965. An arithmetic average for the COD of the soil samples at Whittier Narrows is 13 gm COD per kg dry soil (Table 8-3). Using this value, assuming a soil porosity of 50 percent, a solids density of 2.65 gm per cubic centimeter, the total basin area of 324 square meters, and a soil depth of 2 meters, the total COD contained in this soil volume of 324 x 2 cubic meters is calculated to be about 11,000 kg. Comparing this figure of 11,000 kg of COD with the value given in Table 8-1 of a total of 1779 kg of COD applied to the basin over the period of 27 months of the study, it is evident that the total organic load of the soil as expressed by the COD is more than five times the amount of COD added to the basin by the recharge operation. A similar calculation over a two-meter depth at the Rio Hondo site, using from Table 8-4 an average COD value of 3 gm per kg dry soil, shows the total COD contained in this soil volume to be about 2600 kg. This weight exceeds the total COD of 1248 kg applied to the basin over the 27-month period.

In both cases the COD of the soil appears to be higher than the total amount added by the recharge operations. It is evident, therefore, that organic carbon is being synthesized in the soil system and that a portion of this organic carbon (probably in the form of bacterial bodies) is leached downward to appear in the percolates at lower sampling pans.

8.04 *Nitrification in the Test Basins.*

Nitrification is the enzymic oxidation of nitrogen from ammonium ions to nitrite and nitrate ions. This reaction is common to most aerobic soil systems. The responsible organisms are autotrophic bacteria, i.e., organisms that obtain their energy from the oxidation of inorganic compounds and are capable of utilizing carbon dioxide as a source of carbon.

Nitrification takes place in two steps with *Nitrosomonas* species being responsible for the conversion of ammonium ion to nitrite, and *Nitrobacter* species handling the conversion of nitrite to nitrate. Normally, the second reaction follows the first one rather rapidly, but large amounts of ammonium ion in an alkaline soil can suppress the conversion of nitrite to nitrate because of the apparent toxicity of ammonium hydroxide to *Nitrobacter*.

Certain soil conditions favor nitrification. Since this is an oxidation process, a well-aerated soil encourages nitrification. The most favorable temperature range is between 80 and 90°F. with nitrification almost ceasing at 125°F. Freezing temperatures stop nitrification, but it will slowly commence at 35 to 40°F. A very high or very low moisture content in the soil can retard nitrification; however, the process will continue at moisture contents below the wilting coefficient for higher plants. A low pH has little influence on nitrification if exchangeable base ions are

Table 8-3—Physical data from soil core taken at Whittier Narrows Test Basin

Depth (feet)	d_{50}[1] (mm)	σ_g[2]	Moisture content[3]	Volatile solids[4]	COD[5]	Organic N + NH_3 − N[6]
0	0.011	25	0.40	65	33	1.4
1.0	0.050	13	----	50	21	0.8
1.5	0.145	4.0	0.21	21	9	1.0
2.5	0.105	1.7	----	14	6	0.4
3.5	0.058	2.3	0.31	22	10	0.9
4.5	0.054	1.8	----	18	7	0.3
5.5	0.010	32	0.39	24	12	0.5
6.5	0.062	1.9	----	25	18	0.3
7.5	0.066	1.7	0.33	17	7	0.2
8.5	0.165	1.5	----	6	3	0.1

[1] Mean size from sieve analysis.
[2] Geometric standard deviation, $\sigma_g = d_{84.1}/d_{50} = d_{50}/d_{15.9}$
[3] Weight loss at 105°C, gm water per gm dry soil.
[4] Weight loss at 600°C, gm volatile solids per kg dry soil.
[5] Chemical oxygen demand, gm COD per kg dry soil.
[6] Organic nitrogen + ammonia nitrogen, gm N per kg dry soil.

Table 8-4—Physical data from soil core taken at Rio Hondo Test Basin

Depth (feet)	d_{50}[1] (mm)	σ_g[2]	Moisture content[3]	Volatile solids[4]	COD[5]	Organic N + NH_3 − N[6]
0	0.60	1.9	0.21	18	10	0.4
1.0	0.21	2.5	0.17	19	5	0.2
1.5	0.091	2.1	0.06	9	2	0.07
2.5	0.24	1.8	0.09	6	0.5	0.04
3.5	0.32	1.4	0.07	8	1	0.1
4.5	0.32	1.6	0.05	7	1	0.1
5.5	0.33	1.9				
6.5	0.29	1.8				
7.5	0.43	1.6				
8.5	0.33	1.7				
10	0.28	1.7				
14.5	0.31	1.8				
16	0.29	1.7				

[1] Mean size from sieve analysis.
[2] Geometric standard deviation, $\sigma_g = d_{84.1}/d_{50} = d_{50}/d_{15.9}$
[3] Weight loss at 105°C, gm water per gm dry soil.
[4] Weight loss at 600°C, gm volatile solids per kg dry soil.
[5] Chemical oxygen demand, gm COD per kg dry soil.
[6] Organic nitrogen + ammonia nitrogen, gm N per kg dry soil.

present. A high pH associated with large concentrations of ammonium ion, as previously mentioned, can inhibit the oxidation of nitrite to nitrate.

The averages of the weekly analyses for the nitrogen components performed on surface and percolated waters at each of the test basins are presented in Tables 4-12 to 4-15 and 4-22 to 4-25 in Chapter 4. Summaries of the nitrate and total nitrogen data averaged over the periods of significance defined by various hydraulic considerations are given in Tables 4-27 and 4-28. Also in Chapter 4, a graphical representation for the nitrate and total nitrogen data is contained in Figure 4-14. For the purpose of this discussion, summaries of some of the nitrogen data presented in Chapter 4 are shown again in Tables 8-5 and 8-6.

From an inspection of the above-mentioned tables and figure in Chapter 4, it is evident that more than 90 percent of the nitrification at both basins takes place in the first two feet of percolation. With regard to the data for total nitrogen at the Whittier Narrows Basin, there is an apparent small decrease in the concentration of total nitrogen in the percolate over the first six feet of percolation. Between the six- and eight-foot depths, the total nitrogen appears to show a slight increase. It is well to remember that after two feet of percolation, changes in total nitrogen reflect principally changes in nitrate-nitrogen. The sharpest decrease in the nitrate-nitrogen takes place at the six-foot depth, the same level at which the COD data discussed previously showed a sharp increase.

An inspection of the data plotted in Figure 4-14 shows nitrogen concentrations in the surface water at Rio Hondo that are considerably lower (about one-fifth) than concentrations at Whittier Narrows, as a result of dilution. Although nitrification at the Rio Hondo Basin was observed to be nearly complete after two feet of percolation, the changes with depth are not as dramatic as at Whittier Narrows.

The data shown in Tables 8-5 and 8-6 indicate that the Whittier Narrows Basin showed better percentage decreases of ammonia and organic nitrogen than did the Rio Hondo site, and also that the Whittier Narrows Basin improved with age. At the highest surface loading (Period IV), the Whittier Narrows Basin provided more than 95-percent conversion of ammonia and organic nitrogen (after two feet of percolation) from a loading of 6.8 gm N per m^2 per day (about 61 lb N per acre per day). The Rio Hondo site at its best showed only a 75-percent conversion

Table 8-5—Summary of Nitrogen data for Whittier Narrows Test Basin

Period	Dates	Average concentration of ammonia plus organic nitrogen in milligrams N per liter at				
		Surface	2 ft.	4 ft.	6 ft.	8 ft.
I	Jan.–April 1963	21.1	4.2	1.9	0.8	0.6
	Percent remaining	100.0	19.9	9.0	3.8	2.8
II	May 1963–Jan. 1964	16.4	2.6	2.1	1.7	0.5
	Percent remaining	100.0	15.8	12.8	10.4	3.0
III	Feb.–Dec. 1964	20.3	0.8	0.7	0.6	0.7
	Percent remaining	100.0	3.9	3.4	3.0	3.4
IV	Jan.–Mar. 1964	21.8	0.9	0.8	0.9	1.0
	Percent remaining	100.0	4.1	3.7	4.1	4.6
Weighted average	Jan. 1963–Mar. 1965	19.6	1.8	1.4	1.1	0.8
	Percent remaining	100.0	9.2	7.1	5.6	4.1

Period		Material balance of ammonia plus organic nitrogen in kilograms* of N at				
		Surface	2 ft.	4 ft.	6 ft.	8 ft.
I		135.0	26.9	12.2	5.1	3.8
II		92.0	14.6	11.8	9.5	2.8
III		113.0	4.4	3.9	3.3	3.9
IV		599.6	24.7	22.0	24.7	27.5
Total		939.6	70.6	49.9	42.6	38.0

* Equals volume of water spread in period × average concentration.

Period	Calendar days	Average surface loading of ammonia plus organic nitrogen†	
		gm N/m²/day	lb N/acre/day
I	125	3.3	29
II	276	1.0	8.9
III	151	2.3	20
IV	274	6.8	61
Weighted average	826	3.5	31

† Equals weight of N divided by number of days divided by basin bottom area.

Table 8-6—Summary of nitrogen data for Rio Hondo Test Basin

Period	Dates	Average concentration of ammonia plus organic nitrogen in milligrams N per liter at				
		Surface	2 ft.	4 ft.	6 ft.	8 ft.
I	Jan.–April 1963	7.6	2.0	1.4	1.2	1.2
	Percent remaining	100.0	25.3	18.4	15.8	15.8
II	May 1963–Jan. 1964	4.7	1.9	2.1	2.0	1.3
	Percent remaining	100.0	40.4	44.7	42.5	27.6
III	Feb.–Dec. 1964	3.1	1.7	0.6	0.5	2.0
	Percent remaining	100.0	54.8	19.3	16.1	64.5
IV	Jan.–Mar. 1965	1.9	0.8	0.4	0.6	--
	Percent remaining	100.0	42.1	21.0	31.6	--
Weighted average	Jan. 1963–Mar. 1965	3.7	1.7	1.3	1.2	1.4
	Percent remaining	100.0	46.0	35.1	32.4	37.8

Period		Material balance of ammonia plus organic nitrogen in kilograms* of N at				
		Surface	2 ft.	4 ft.	6 ft.	8 ft.
I		35.6	9.4	6.6	5.6	5.6
II		30.8	12.5	13.8	13.1	8.5
III		76.4	41.9	14.8	12.3	49.3
IV		4.7	2.0	1.0	1.4	--
Total		147.5	65.8	36.2	32.4	--

* Equals volume of water spread in period × average concentration.

Period	Calendar days	Average surface loading of ammonia plus organic nitrogen†	
		gm N/m²/day	lb N/acre/day
I	99	1.1	9.8
II	276	0.35	3.1
III	335	0.70	6.2
IV	90	0.16	1.4
Weighted average	800	0.57	5.1

† Equals weight of N divided by number of calendar days divided by basin bottom area.

(Period I) at a surface loading of 1.1 gm N per m^2 per day (about 10 lb N per acre per day).

A further discussion of nitrification and its relation to the reduction of COD is presented in the following section.

8.05 *Total Oxygen Demand in the Test Basins.*

It is evident from the discussions in the previous sections that the behavior of carbonaceous and nitrogenous substances in a soil system does not follow the conventional pattern typical of trickling filters. Perhaps the reason for the observed patterns of change is a limitation of available oxygen. Atmospheric oxygen gets down into the pores of the soil only when these pores are not filled with water. While there is some oxygen available to the soil organisms from the dissolved oxygen in the water applied to the basin, this source is small compared with the total oxygen demand of the COD and the ammonia. The principal method of aerating the soil by the intermittent spreading operation comes from the piston action associated with the final drainage of the ponded water into the soil. This hydrodynamic aeration is stimulated by loading a basin with small hydraulic loads at frequent intervals. When the surface is free of ponded water, oxygen may diffuse from the atmosphere into the ground and carbon dioxide may diffuse from the ground to the atmosphere above. The depth of penetration of the oxygen depends on the porosity and permeability of the soil (especially near the surface), the moisture content of the soil, the partial pressure of the oxygen in the soil, and the rates of oxygen utilization by the soil organisms. In the final chapter of this report the relative magnitudes and importance of the oxygen flux associated with each process are discussed.

Using the average data for the amount of ammonia-nitrogen and organic nitrogen in the surface waters and percolate in both basins as presented in Tables 8-5 and 8-6, and using the relationship that two mols of oxygen are needed to convert one mol of ammonia-nitrogen to nitrate, a tabulation showing a material balance for the oxygen demand of the nitrogen is presented in Tables 8-7 and 8-8.

Comparing the data in Table 8-7 with the data for the COD in Table 8-1 for the Whittier Narrows Basin, it is seen that the oxygen demand for the ammonia is about 2.5 times the COD. Hence, the principal burden in terms of oxygen demand is exerted by the oxidation of ammonia. At the Rio Hondo site, a different situation exists. Here the oxidation of ammonia requires only about three-fifths of the oxygen necessary to satisfy the COD. The difference is primarily attributable to dilution of the plant effluent and the high COD of the surface water resulting from algal activity.

Tables 8-9 and 8-10 show summaries of the oxygen demands given in Tables 8-1, 8-2, 8-7, and 8-8. These summaries are essential for an understanding of the behavior of the total oxygen demand within each basin.

The Whittier Narrows Test Basin shows a much better satisfaction of the total oxygen demand than does the Rio Hondo site, even though the surface organic loadings at Whittier Narrows were two to three times heavier than at Rio Hondo. It is believed that this result is attributable primarily to the pattern of frequent floodings of the test basin and the very gradual increase in the volume of water applied to the basin. It may also be due to the apparent heavier biological growth within the Whittier Narrows Basin. The performance of the Whittier Narrows Basin improved with age, such that the best overall removal of total oxygen demand took place during Period IV. Most of the removal was found to occur in the first two feet of percolation (perhaps even

Table 8-7—Oxygen demand of ammonia plus organic nitrogen for Whittier Narrows Test Basin

Period		Material balance of ammonia plus organic nitrogen as kilograms of O_2* at				
		Surface	2 ft.	4 ft.	6 ft.	8 ft.
I		617.0	122.9	55.8	23.3	17.4
II		420.4	66.7	53.9	43.4	12.8
III		516.4	20.1	17.8	15.1	17.8
IV		2,740.2	112.9	100.5	112.9	125.7
Total		4,294.0	322.6	228.0	194.7	173.7

* Equals weight of O_2 necessary to oxidize ammonia plus organic nitrogen to nitrate, i.e., weight as N × 64/14 = weight as O_2.

Period	Calendar days	Average surface loading of ammonia plus organic nitrogen	
		gm O_2/m^2/day	lb O_2/acre/day
I	125	15.2	135
II	276	4.7	42
III	151	10.5	94
IV	274	30.8	274
Weighted average	826	16.0	142

Table 8-8—Oxygen demand of ammonia plus organic nitrogen for Rio Hondo Test Basin

Period		Material balance of ammonia plus organic nitrogen as kilograms of O_2* at				
		Surface	2 ft.	4 ft.	6 ft.	8 ft.
I		162.7	43.0	30.2	25.6	25.6
II		140.8	57.1	63.1	59.9	38.8
III		349.1	191.5	67.6	56.2	225.3
IV		21.5	9.1	4.6	6.4	--
Total		674.1	300.7	165.5	148.1	--

* Equals weight of O_2 necessary to oxidize ammonia plus organic nitrogen to nitrate, i.e., weight as N × 64/14 = weight as O_2.

Period	Calendar days	Average surface loading of ammonia plus organic nitrogen	
		gm O_2/m^2/day	lb O_2/acre/day
I	99	5.1	45
II	276	1.6	14
III	335	3.2	28
V	90	0.74	6.5
Weighted average	800	2.6	23

after a shorter depth, but this was the first sampling point below ground) and with little change thereafter to a depth of eight feet.

The Rio Hondo site did not perform as effectively as the Whittier Narrows Basin during the Periods I and II when hydraulic operations were identical. This difference may be attributed to the weaker organic loads and the fact that the biological growth within the Rio Hondo Basin was not as profuse. In Period III, when this basin had aged somewhat, the removal improved over the early performance, but compared to the results at Whittier Narrows it was far inferior. During Period III, the Whittier Narrows Basin was flooded three times per week on Monday, Wednesday, and Friday with a gradual increase in the depth of flooding, whereas the Rio Hondo Basin was flooded continuously for about one week and allowed to dry out for two weeks. During this period Rio Hondo received its highest surface loading of total oxygen demand. Even though the average hydraulic load (see Table 4-6, Chapter 4) for Rio Hondo was twice that at Whittier Narrows, the load of total oxygen demand was less. Because of the failure of the Rio Hondo Basin to maintain its hydraulic acceptance towards the latter part of Period III and because of the excellent performance of the Whittier Narrows Basin in Period III, the operation of the Rio Hondo site was converted back to smaller hydraulic loads at more frequent intervals in Period IV. The loading at Whittier Narrows was greatly increased in Period IV to nearly three times its loading in Period III and still the basin showed nearly 90 percent removal of total oxygen demand after two feet of percolation.

Table 8-9—Total oxygen demand for Whittier Narrows Test Basin

Period		Material balance of COD plus ammonia and organic nitrogen in kilograms of O_2 at				
		Surface	2 ft.	4 ft.	6 ft.	8 ft.
I		921.6	385.9	337.4	357.3	211.3
		100.0	41.8	36.6	38.7	22.9
II		609.4	204.1	173.9	488.6	309.4
		100.0	33.5	28.6	80.4	50.8
III		720.8	69.1	68.5	85.3	233.9
		100.0	9.6	9.5	11.8	32.4
IV		3821.2	398.9	367.3	580.5	527.3
		100.0	10.4	9.6	15.2	13.8
Total		6073.0	1058.0	947.1	1511.7	1281.9
		100.0	17.8	15.6	24.9	21.1

Period	Calendar Days	Average surface loading of COD plus ammonia and organic nitrogen	
		gm $O_2/m^2/day$	lb O_2/acre/day
I	125	22.7	202
II	276	6.8	61
III	151	14.7	131
IV	274	43.0	383
Weighted average	826	22.6	201

Table 8-10—Total oxygen demand for Rio Hondo Test Basin

Period		Material balance of COD plus ammonia and organic nitrogen in kilograms of O_2 at				
		Surface	2 ft.	4 ft.	6 ft.	8 ft.
I		291.0	162.0	116.4	99.6	105.2
		100.0	55.6	40.0	34.2	36.2
II		366.9	145.6	181.7	141.1	337.0
		100.0	39.8	49.6	38.4	91.9
III		1211.6	585.8	286.9	228.7	407.7
		100.0	48.3	23.7	18.7	33.6
IV		52.5	27.7	21.0	25.2	--
		100.0	52.7	40.0	48.6	--
Total		1922.0	921.1	606.0	494.6	--
		100.0	47.9	31.6	25.8	--

Period	Calendar Days	Average surface loading of COD plus ammonia and organic nitrogen	
		gm $O_2/m^2/day$	lb O_2/acre/day
I	99	9.1	81
II	276	4.1	36
III	335	11.2	100
IV	90	1.7	15
Weighted average	800	7.4	66

CHAPTER 9

BACTERIAL CONSIDERATIONS

9.01 *Coliform Organisms in the Test Basins*

The results of bacteriological tests for coliform organisms in the percolate at both test sites are presented in tabular and graphical form in this chapter. At the beginning of the project only the presumptive coliform test was run routinely. As the study progressed, however, positive results from the lauryl tryptose broth were confirmed by means of either brilliant-green-lactose-bile broth or eosin-methylene-blue agar. Some tests were performed to assay the so-called fecal coliforms, which are defined by the procedure of Geldrich, Clark, and Huff (17). The fecal coliform test is conducted at an elevated temperature, with a loop of broth from a positive presumptive lactose tube being transferred to an EC broth tube. The EC tube is incubated for 24 ± 2 hours at 44.5 to 45.0°C. Gas production shows the presence of fecal coliforms while no gas production is considered to indicate that the coliforms are of non-fecal origin.

The general procedure for bacteriological sampling at the test basins was for LACFCD personnel to collect a sample of the percolate in a sterilized bottle containing sodium thiosulfate, just prior to delivering the samples to the Caltech laboratory. Since the basins were flooded in the late afternoon and the bacteriological samples were taken the following morning, there were numerous occasions when the sampling pans had stopped running by morning, and consequently no samples could be obtained. The samples received for bacteriological analysis were delivered to Caltech within an hour of the time they were collected. At Caltech they were refrigerated and normally analyzed that same day. Five 10-ml portions, five 1-ml portions, and five 0.1-ml portions of the samples or diluted samples were inoculated into lauryl tryptose broth and incubated at 35°C.

Inasmuch as the bacteriological samples from the pans were taken in the morning when the basins were normally free of ponded water, no samples of the applied water were assayed routinely. However, personnel of LACSD conducted bacterial analyses of the plant effluent. Table 9-1 and Figure 9-1 show the distribution of confirmed coliforms in the final effluent from the Whittier Narrows plant, as reported to Caltech by LACSD. Samples of chlorinated final effluent were taken from the end of the chlorine contact chamber, normally in the morning, allowed to sit in the laboratory at room temperature for about 30 minutes, dechlorinated, and then the presumptive test for coliforms was set up. Two 10-ml portions, two 1-ml portions, and two 0.1-ml portions of the samples were inoculated into lactose broth. The reason for letting the samples sit for about 30 minutes after collection was an attempt to stimulate the quality of the plant effluent at the time of its discharge from the plant effluent pipe into the open ditch near the Whittier Narrows Test Basin. Of 180 samples assayed in 1964, 50 percent had an MPN less than or equal to 190 per 100 ml.

Data for the coliform analyses of the percolates at both test basins are presented in Tables 9-2 to 9-12. These tables show the results for presumptive coliforms, confirmed coliforms, and fecal coliforms for the 27 months of the study. Figures 9-2 to 9-5 are graphs of the coliform data on log-probability paper. The data suggest a log-normal distribution, but they do not show a good fit. A similar plot for the final effluent samples (Figure 9-1) shows a better fit, but to be truly logarithmically normal the data must plot as a straight line on log-probability graph paper.

The presumptive coliform data from Whittier Narrows (Figure 9-2) indicate a median MPN value of about 1.5×10^4 per 100 ml at the 2-ft depth, a median MPN value of about 10^2 per 100 ml at the 4-ft depth, a median MPN value of about 10^3 per 100 ml at the 6-ft depth, and a median MPN value of above 2×10^4 per 100 ml at the 8-ft depth. For the confirmed coliform data (Figure 9-4), the median MPN value for the 2-ft depth is about 10^4 per 100 ml, while for the 6- and 8-ft depths, the median MPN appears to be about 5×10^4 per 100 ml.

Covering 27 months of the study, these data indicate that the median MPN values of the percolates at Whittier Narrows are much higher than the median MPN values of the applied water (Figure 9-1). Although there is an apparent decrease in the coliform density between the 2- and 4-ft pans, the concentration rises with depth such that the median MPN concentrations at the 2- and 8-ft depths are about the same.

The presumptive and confirmed coliform densities in the percolates at the Rio Hondo Test Basin (Figures 9-3 and 9-5) show lower median concentrations than the Whittier Narrows site. The presumptive median MPN values at 2-ft and 4-ft depths are about 3×10^2 per 100 ml and less than 50 per 100 ml at the 6-ft and 8-ft depths. These data tend to show a decrease in the coliform density with increased depth.

Table 9-1—Confirmed coliforms in the final effluent from the Whittier Narrows Treatment Plant *

MPN per 100 ml	Number of samples having MPN less than or equal to stated value†
700	152 (93.6)
240	112 (62.2)
130	66 (37.6)
62	60 (33.3)
50	31 (17.2)
23	29 (16.1)
13	13 (7.1)
6	11 (6.1)
4.5	7 (3.9)

* From data supplied by Los Angeles County Sanitation Districts (January-December 1964).

† Bracketed number is percent of the samples having an MPN less than or equal to the stated value. This percent figure is based on Thomas plotting position criteria such that $\% = m/(n + 1)$ where m is the rank of the data point and n is the total number of data points. For this array, $n = 180$. Twenty-eight samples were found to have an $MPN > 700$ per 100 ml.

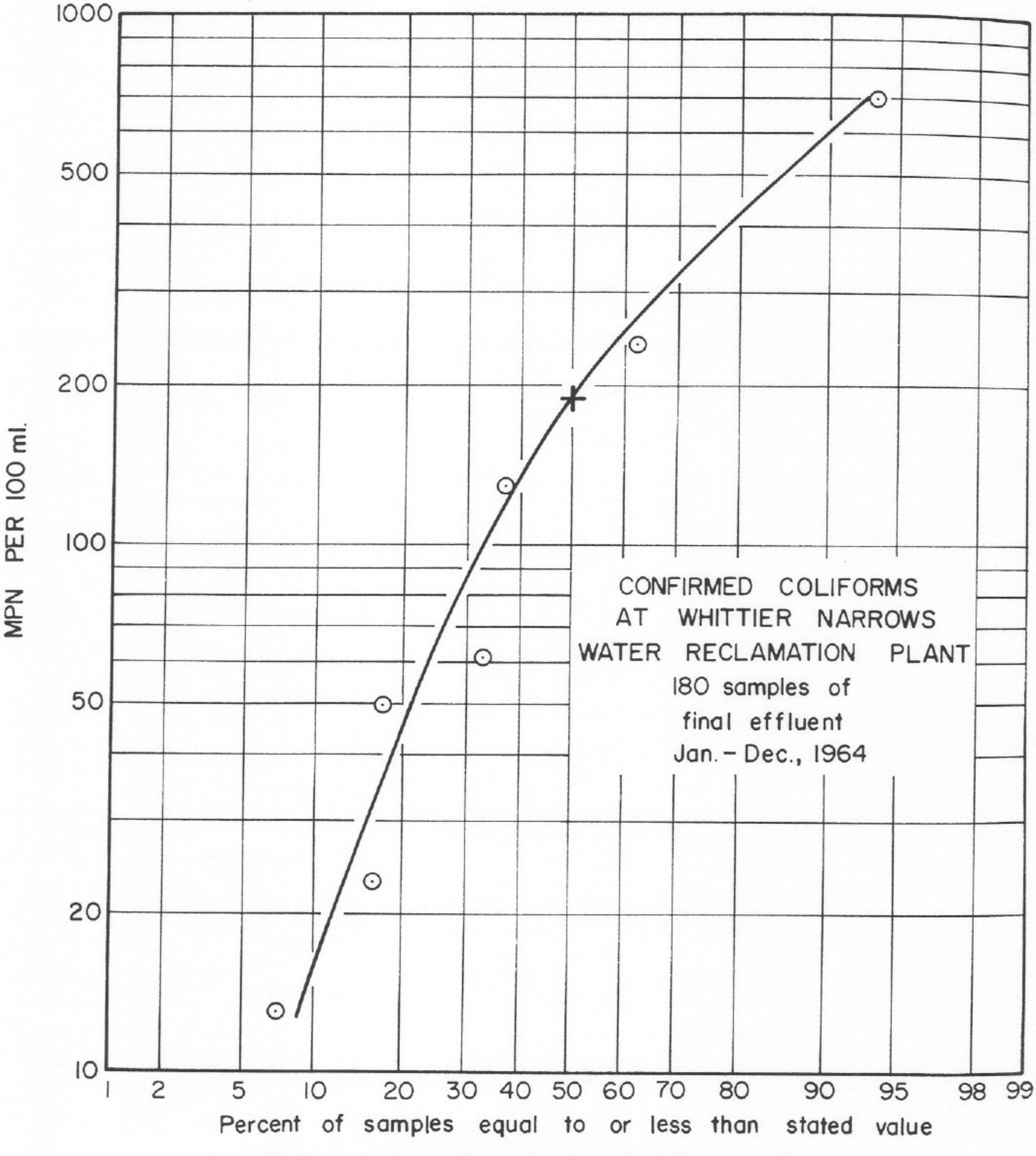

Fig. 9-1—Distribution of Confirmed Coliforms in Final Effluent from Whittier Narrows Water Reclamation Plant

The confirmed coliform densities show little pattern, and all depths have median MPN values about 10^3 per 100 ml.

The multiplication of coliform bacteria in percolating wastewaters runs contrary to the experience reported by others. According to Baars (28), the percolate from polluted river water applied to sand dunes in the Netherlands was free of coliforms. One or two feet of sand filtration removed approximately 96.5 percent of the coliforms in English experiments (29). Stone and Garber (31) found the wastewater at Azusa and Whittier to be free of coliforms after seven feet of percolation. In the early Hyperion tests, coliform concentrations were reduced by 98.0 to 99.9 percent in seven feet of percolation (34, 35). At Lodi, a bacteriologically safe water was produced from settled sewage after percolation vertically through only four feet of soil (4). Yet the Whittier Narrows Test Basin showed the following median concentrations of coliforms:

surface water ----	190
2-foot pan --------	15,000
4-foot pan --------	100
6-foot pan --------	1,000
8-foot pan --------	20,000

Table 9-2—Presumptive coliforms at the Whittier Narrows Test Basin (January–December 1963)

MPN per 100 ml	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
10^{7}	26 (96.3)*	--	--	--
5×10^{6}	25 (92.5)	--	--	--
10^{6}	23 (85.2)	--	--	--
5×10^{5}	21 (77.8)	--	18 (94.7)	--
10^{5}	17 (63.0)	--	17 (89.5)	--
5×10^{4}	15 (55.5)	--	16 (84.2)	--
10^{4}	11 (40.7)	--	14 (73.7)	--
5×10^{3}	11 (40.7)	--	14 (73.7)	6 (85.6)
10^{3}	11 (40.7)	--	14 (73.7)	5 (71.5)
5×10^{2}	11 (40.7)	6 (85.7)	14 (73.7)	5 (71.5)
10^{2}	9 (33.3)	5 (71.5)	9 (47.4)	3 (42.8)
5×10	8 (29.6)	5 (71.5)	8 (42.1)	2 (28.6)
10	5 (18.5)	3 (42.8)	6 (31.6)	2 (28.6)
2	0	3 (42.8)	3 (15.8)	0

* Bracketed number is the percent of the samples having an MPN less than or equal to stated value. This percent figure is based on the Thomas plotting position criteria such that $\% = m/(n+1)$ where m is the rank of the data point and n is the total number of data points.

Table 9-3—Presumptive coliforms at the Whittier Narrows Test Basin (January–December 1964)

MPN per 100 ml	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
5×10^{6}	--	--	25 (96.1)	--
10^{6}	28 (96.5)	--	22 (84.6)	17 (94.5)
5×10^{5}	27 (93.0)	--	21 (80.7)	16 (89.0)
10^{5}	22 (75.8)	--	15 (57.6)	11 (61.1)
5×10^{4}	20 (69.0)	--	15 (57.6)	8 (44.4)
10^{4}	10 (34.5)	--	10 (38.4)	4 (22.2)
5×10^{3}	9 (31.0)	--	10 (38.4)	4 (22.2)
10^{3}	6 (20.7)	2 (66.7)	9 (34.6)	4 (22.2)
5×10^{2}	6 (20.7)	1 (33.3)	9 (34.6)	4 (22.2)
10^{2}	0	1 (33.3)	6 (23.1)	1 (5.6)
5×10	--	0	6 (23.1)	0
10	--	--	6 (23.1)	--
2	--	--	6 (23.1)	--

Table 9-4—Presumptive coliforms at the Whittier Narrows Test Basin (January–June 1965)

MPN per 100 ml	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
5×10^{5}	--	--	--	9 (90.0)
10^{5}	--	--	10 (91.0)	7 (70.0)
5×10^{4}	10 (91.0)	2 (66.7)	9 (81.8)	6 (60.0)
10^{4}	6 (54.5)	1 (33.3)	7 (63.6)	3 (30.0)
5×10^{3}	6 (54.5)	0	6 (54.5)	2 (20.0)
10^{3}	4 (36.4)	--	6 (54.5)	0
5×10^{2}	2 (18.2)	--	5 (45.5)	--
10^{2}	0	--	0	--
5×10	--	--	--	--
10	--	--	--	--
2	--	--	--	--

Table 9-5—Presumptive coliforms at the Whittier Narrows Test Basin (January 1963–June 1965)

MPN per 100 ml	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
10^{7}	64 (98.5)	--	--	--
5×10^{6}	63 (97.0)	--	53 (98.1)	--
10^{6}	61 (93.8)	--	50 (92.5)	32 (96.9)
5×10^{5}	58 (89.2)	--	49 (90.7)	31 (93.9)
10^{5}	49 (75.4)	--	42 (77.7)	24 (72.7)
5×10^{4}	45 (69.2)	10 (91.0)	40 (74.0)	20 (60.6)
10^{4}	27 (41.5)	9 (81.8)	31 (57.4)	13 (39.4)
5×10^{3}	26 (40.0)	8 (72.7)	30 (55.5)	12 (36.4)
10^{3}	21 (32.3)	8 (72.7)	29 (53.6)	9 (27.2)
5×10^{2}	19 (29.2)	7 (63.6)	28 (51.8)	9 (27.2)
10^{2}	9 (13.8)	6 (54.5)	15 (27.8)	4 (12.1)
5×10	8 (12.3)	5 (45.5)	14 (25.9)	2 (6.1)
10	5 (7.7)	3 (27.3)	12 (22.2)	2 (6.1)
2	0	3 (27.3)	9 (16.7)	0

Table 9-6—Presumptive coliforms at the Rio Hondo Test Basin (January–December 1963)

MPN per 100 ml	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
5×10^{4}	22 (95.6)	--	--	--
10^{4}	19 (82.6)	18 (94.7)	--	--
5×10^{3}	18 (78.2)	17 (89.4)	--	--
10^{3}	18 (78.2)	16 (84.1)	--	--
5×10^{2}	18 (78.2)	16 (84.1)	35 (97.2)	8 (88.8)
10^{2}	13 (56.5)	10 (52.6)	33 (91.6)	7 (77.7)
5×10	13 (56.5)	9 (47.4)	32 (88.9)	7 (77.7)
10	6 (26.1)	7 (36.8)	26 (72.3)	5 (55.5)
2	2 (8.7)	3 (15.8)	23 (63.9)	3 (33.3)

Table 9-7—Presumptive coliforms at the Rio Hondo Test Basin (January–December 1964)

MPN per 100 ml	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
10^{5}	--	--	14 (93.4)	--
5×10^{4}	17 (94.5)	18 (94.7)	12 (80.0)	--
10^{4}	15 (83.3)	13 (68.5)	9 (60.0)	--
5×10^{3}	13 (72.2)	13 (68.5)	9 (60.0)	--
10^{3}	6 (33.3)	12 (63.1)	9 (60.0)	--
5×10^{2}	3 (16.7)	11 (57.9)	9 (60.0)	4 (80.0)
10^{2}	1 (5.6)	7 (36.8)	7 (46.7)	2 (40.0)
5×10	1 (5.6)	7 (36.8)	7 (46.7)	2 (40.0)
10	0	1 (5.3)	7 (46.7)	1 (20.0)
2	--	1 (5.3)	6 (40.0)	1 (20.0)

These unusual bacterial results led to the hypothesis that fecal organisms were truly being removed by filtration, but that non-fecal coliforms found the environment of the soil much to their liking. The intermittent daily wetting and aeration, and the availability of organic nutrients from the wastewater should favor the growth of non-fecal coliforms such as *Aerobacter aerogenes.*

Tests for the confirmation of the coliform group on eosin-methylene-blue agar indicated mainly atypical colonies with only an occasional colony showing a metallic sheen. These data supported the hypothesis that the predominant member of the coliform group in the percolate was not *Escherichia coli* but more likely *Aerobacter aerogenes.*

Further analyses of the positive lauryl tryptose tubes for fecal coliforms, employing EC media and incubation at 44.5°C, showed that positive presumptive results were due almost entirely to non-fecal coliforms, i.e., the EC tubes were almost always negative. By the nature of the multiple-dilution technique, the results reported in Tables 9-11 and 9-12 show median MPN values of the same order as the presumptive and confirmed coliform results at the test sites. Normally, the EC tubes for the fecal coliform test were negative so that the results in Tables 9-11 and 9-12 really represent more strongly data having values less than the stated MPN value rather than less than or equal to.

To ascertain further that non-fecal coliform organisms were multiplying in the soil, especially at the Whittier Narrows Test Basin, several intensive samplings were performed. In each of these studies, the samples were taken and inoculated immediately into the fermentation tubes at the test basin site. After the final sample was taken, the entire set of fermentation tubes were brought back to Caltech and incubated in the laboratory.

On 9 July 1964 the Whittier Narrows Test Basin was flooded at 4:00 PM. Samples were taken from the surface and from the pans at 2, 4, 6, and 8 feet during half-hour intervals starting at 5:00 PM. Coliform concentrations were determined by lactose fermentation (5 tubes each of three dilutions) followed by confirmation on EMB. The results, presented in Table 9-13, demonstrate again that the applied effluent is almost free of coliform organisms, but that the percolate contains sizeable concentrations of these bacteria.

On 18 August 1964 another intensive on-site test was conducted at the Whittier Narrows Test Basin. The basin was flooded at 4:00 PM and samples were

Table 9-8—Presumptive coliforms at the Rio Hondo Test Basin (January 1963–December 1964)

MPN per 100 ml	Number of samples having MPN less than or equa lto stated value at			
	2 feet	4 feet	6 feet	8 feet
10^5	--	--	49 (98.0)	--
5×10^4	39 (97.5)	36 (97.4)	47 (94.0)	--
10^4	34 (85.0)	31 (83.8)	44 (88.0)	--
5×10^3	31 (77.5)	30 (81.1)	44 (88.0)	--
10^3	24 (60.0)	28 (75.7)	44 (88.0)	--
5×10^2	21 (52.5)	27 (73.0)	44 (88.0)	12 (92.4)
10^2	14 (35.0)	17 (46.0)	40 (80.0)	9 (69.2)
5×10	14 (35.0)	16 (43.3)	39 (78.0)	9 (69.2)
10	6 (15.0)	8 (21.6)	33 (66.0)	6 (46.1)
2	2 (5.0)	4 (10.8)	29 (58.0)	4 (30.8)

Table 9-9—Confirmed coliforms at the Whittier Narrows Test Basin (January 1963–June 1965)

MPN per 100 m	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
5×10^6	--	--	23 (95.9)	--
10^6	--	--	21 (87.5)	22 (95.6)
5×10^5	30 (96.8)	--	21 (87.5)	21 (91.3)
10^5	26 (83.9)	--	15 (62.5)	17 (73.9)
5×10^4	24 (77.5)	--	11 (45.8)	14 (60.9)
10^4	16 (51.6)	--	8 (33.3)	6 (26.1)
5×10^3	14 (45.1)	--	8 (33.3)	5 (21.8)
10^3	10 (32.3)	1 (50.0)	6 (25.0)	5 (21.8)
5×10^2	8 (25.8)	0	6 (25.0)	5 (21.8)
10^2	2 (6.5)	--	5 (20.8)	2 (8.7)
5×10	2 (6.5)	--	4 (16.7)	2 (8.7)
10	1 (3.2)	--	2 (8.3)	2 (8.7)
2	1 (3.2)	--	0	0

Table 9-10—Confirmed coliforms at the Rio Hondo Test Basin (January 1963–December 1964)

MPN per 100 m	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
10^5	--	--	9 (90.0)	--
5×10^4	14 (93.4)	15 (93.8)	7 (70.0)	--
10^4	12 (80.0)	9 (56.3)	6 (60.0)	--
5×10^3	10 (66.7)	9 (56.3)	6 (60.0)	--
10^3	4 (26.7)	8 (50.0)	6 (60.0)	--
5×10^2	3 (20.0)	8 (50.0)	4 (40.0)	--
10^2	2 (13.3)	8 (50.0)	3 (30.0)	--
5×10	2 (13.3)	8 (50.0)	3 (30.0)	2 (66.7)
10	0	3 (18.8)	2 (20.0)	1 (33.3)
2	--	1 (6.2)	1 (10.0)	0

Table 9-11—Fecal coliforms at the Whittier Narrows Test Basin (October 1964–June 1965)

MPN per 100 m	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
5×10^4	10 (91.0)	--	9 (90.0)	12 (92.2)
10^4	7 (63.6)	--	2 (20.0)	3 (23.1)
5×10^3	7 (63.6)	--	2 (20.0)	3 (23.1)
10^3	6 (54.5)	--	2 (20.0)	3 (23.1)
5×10^2	5 (45.5)	1 (50.0)	2 (20.0)	3 (23.1)
10^2	1 (9.1)	0	1 (10.0)	0
5×10	1 (9.1)	--	1 (10.0)	--
10	1 (9.1)	--	1 (10.0)	--
2	1 (9.1)	--	1 (10.0)	--

Table 9-12—Fecal coliforms at the Rio Hondo Test Basin (October 1964–March 1965)

MPN per 100 m	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
5×10^2	4 (80.0)	1 (50.0)	2 (66.7)	No data
10^2	1 (20.0)	0	0	--
5×10	1 (20.0)	--	--	--
10	1 (20.0)	--	--	--
2	1 (20.0)	--	--	--

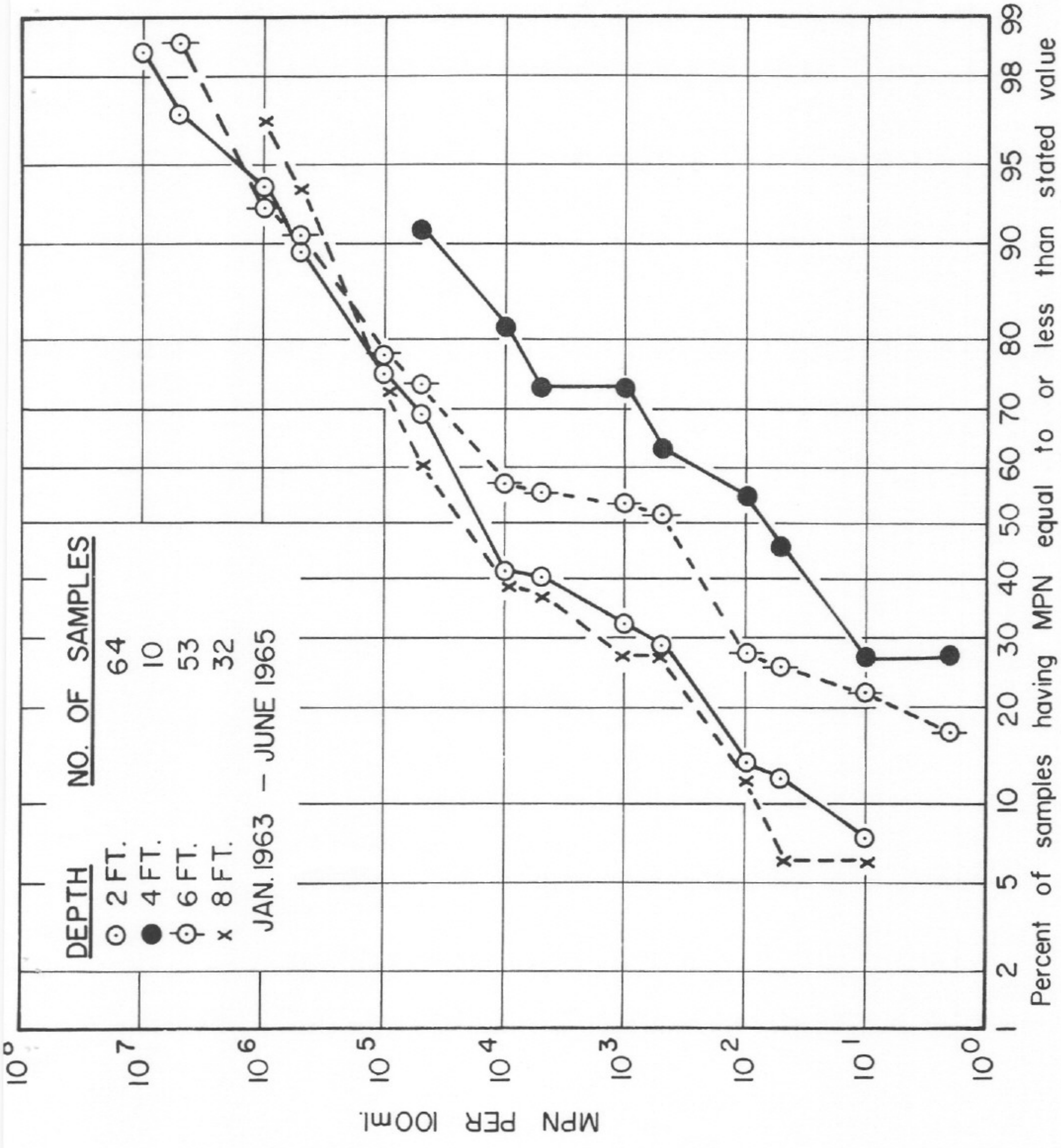

Fig. 9-2—Presumptive Coliforms as a Function of Depth at the Whittier Narrows Test Basin

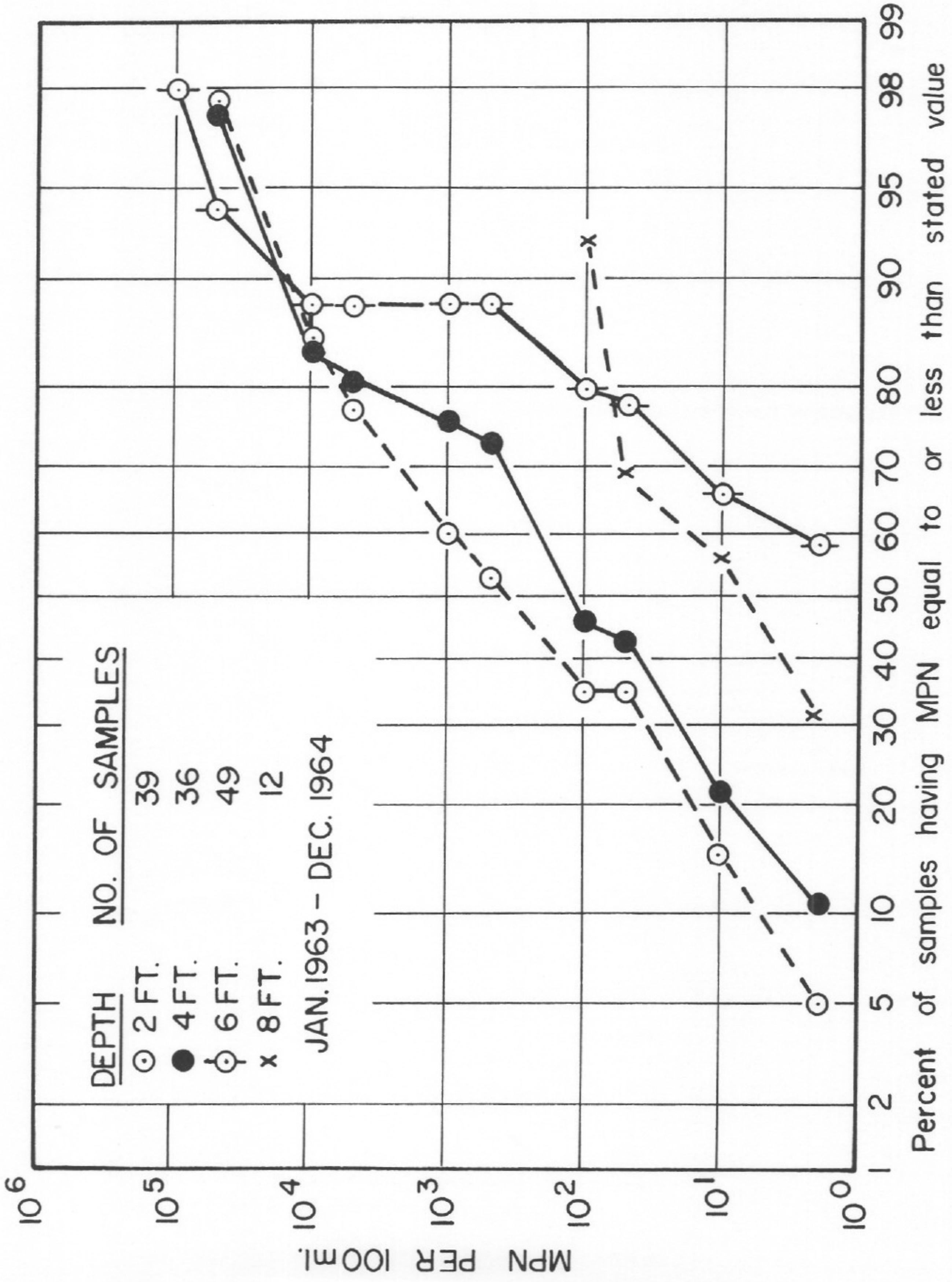

Fig. 9-3—Presumptive Coliforms as a Function of Depth at the Rio Hondo Test Basin

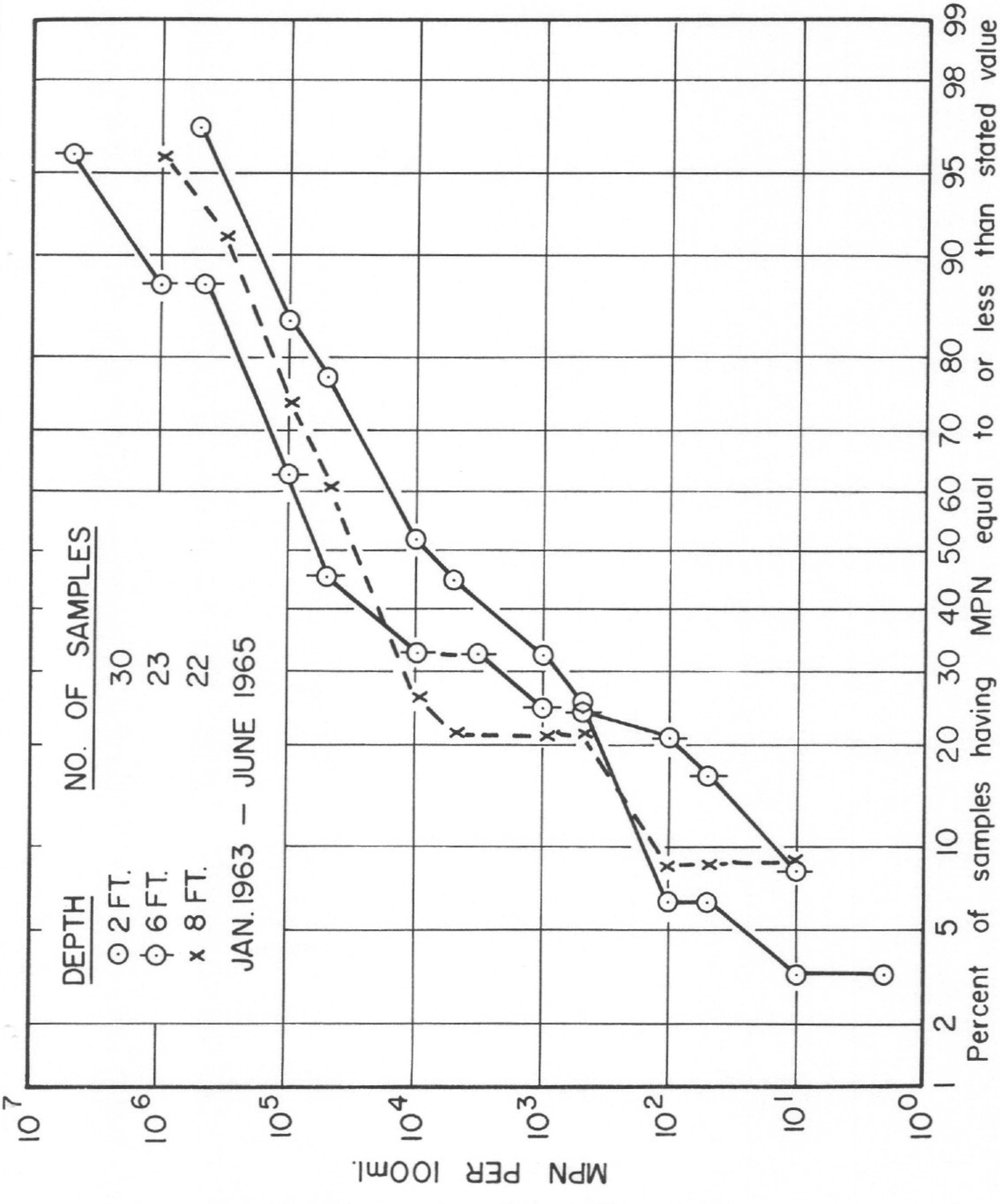

Fig. 9-4—Confirmed Coliforms as a Function of Depth at the Whittier Narrows Test Basin

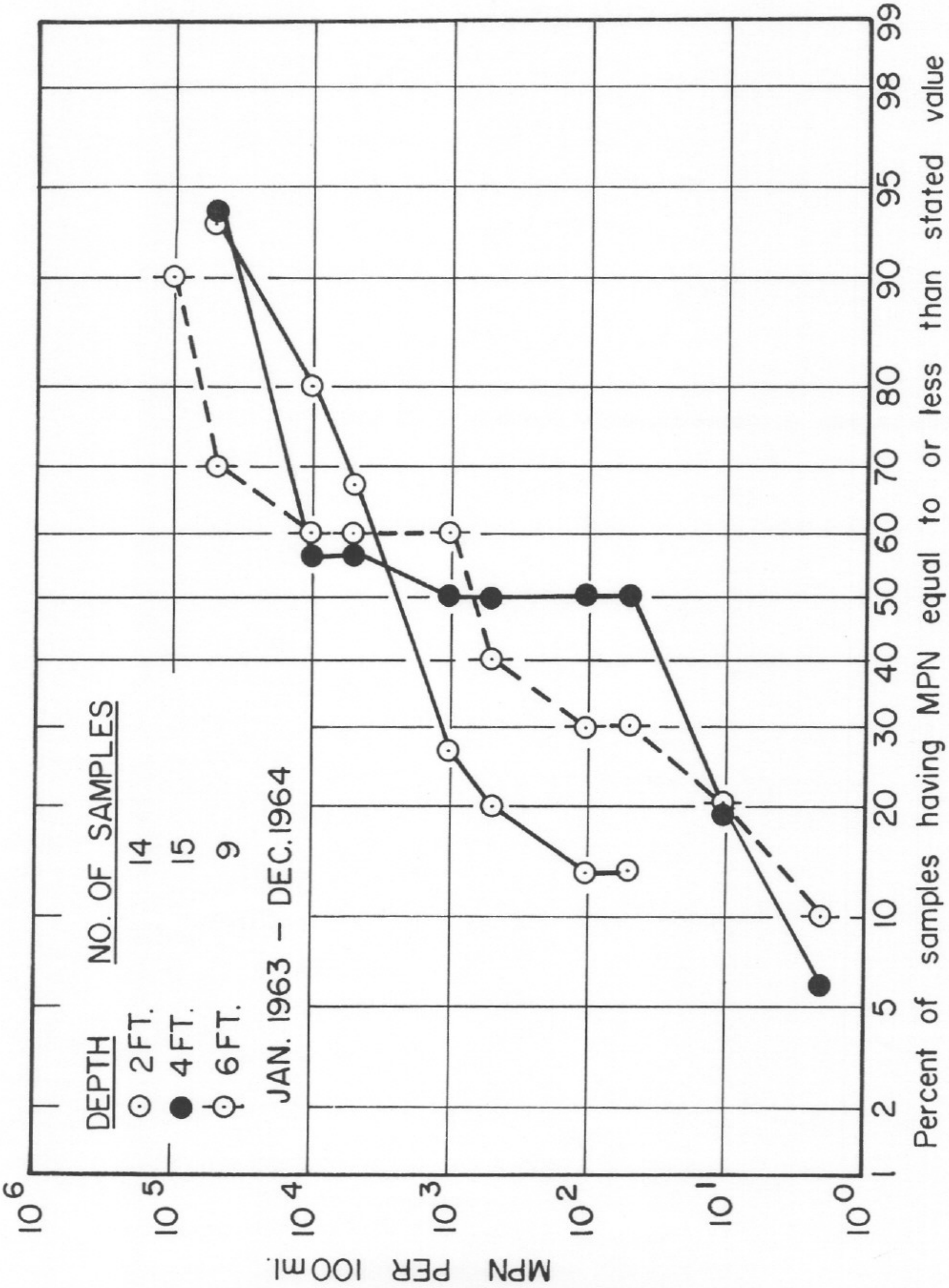

Fig. 9-5—Confirmed Coliforms as a Function of Depth at the Rio Hondo Test Basin

taken from the surface and from the pans at half-hour intervals starting at 5:00 PM. Coliform concentrations were determined by lactose fermentation (5 tubes each of three dilutions) followed by confirmation on EMB. Table 9-14 shows the results of the assays.

From the above data, the pattern suggested by the earlier samplings was repeated, namely, there were few coliform organisms in the applied water but sizeable concentrations in the percolate.

There has been some evidence that coliform organisms exposed to monochloramine (which forms when ammonia-containing effluent is chlorinated) are not completely killed but merely inactivated, and that they can be revived by removal of the chloramine and exposure to optimum conditions for growth. In order to see if the high concentrations of coliforms in the percolate could be attributed to a resuscitation of the organisms in the applied water, a special test of the Whittier Narrows effluent was performed. A sample of dechlorinated effluent was allowed to sit in the laboratory. It was assayed for coliforms on each of five successive days. Coliform concentrations were determined by lactose fermentation and confirmed on EMB. The presumptive tubes were incubated and read at 24-hour intervals for five days. If resuscitation occurs, one would expect that the incubated lactose tubes would show a significant increase in positive samples with time. The results, presented in Table 9-15, do not show this pattern. Hence, the high concentrations in the percolates are not attributable to a revival of the dead coliforms in the applied water.

A third intensive on-site sampling was conducted at the Whittier Narrows Test Basin on 16 December 1964. In addition to coliform sampling, assays were run for fecal streptococci using a multiple-tube fermentation (5 tubes of 3 dilutions each) with azide-dextrose broth. Positive tubes were confirmed with ethyl-violet-azide broth. The basin was flooded at 4:00 PM and samples were taken from the surface and pans at one-hour intervals beginning at 5:30 PM. The results, presented in Table 9-16, show high coliform densities in the percolate. Unfortunately, the proper dilutions of the samples were not always run, so that most of these data are somewhat inconclusive. The tubes that were positive, indicating presumptive coliforms, were assayed for fecal coliforms using the high temperature incubation in EC medium. All samples transferred to EC medium gave negative results. Hence, it may be concluded that none of the coliforms in the percolate were of fecal origin.

Table 9-13—Concentrations of confirmed coliforms in the percolate at Whittier Narrows Test Basin

(Test of 9 July 1964; basin dosed at 4 p.m.)

Sample location	MPN per 100 ml at					
	5:00 PM	5:30 PM	6:00 PM	6:30 PM	7:00 PM	Mean†
Surface	5	2	0*	5	0*	2.4
2 feet	>542	212	84	141	175	>231
4 feet	>542	>542	94	56	426	>330
6 feet	345	0*	>542	212	>542	>328
8 feet	130	23	5	23	109	58

* Strictly speaking less than 2, for all tubes were negative.
† Arithmetic mean of indicated numbers.
> Signifies "greater than," i.e., all tubes were positive.

Table 9-14—Concentrations of confirmed coliforms in the percolate at Whittier Narrows Test Basin

(Test of 18 August 1964; basin dosed at 4 p.m.)

Sample location	MPN per 100 ml at				
	5:00 PM	5:30 PM	6:00 PM	6:30 PM	7:00 PM
Surface	11	5	2	17	33
2 feet	5×10^4	2×10^4	172×10^4	11×10^4	13×10^4
4 feet	79×10^4	109×10^4	$<2 \times 10^4$	33×10^4	79×10^4
6 feet	49×10^4	33×10^4	27×10^4	$<2 \times 10^4$	$<2 \times 10^4$
8 feet	49×10^4*	33×10^4*	33×10^4*	33×10^4*	2×10^4*

< Signifies "less than," i.e., all tubes were negative.
* The pan at the 8-ft. depth did not start to flow until 7 PM so that the times for these samples are really 7:15 PM, 7:40 PM, 7:50 PM, 8:05 PM, and 8:15 PM.

Table 9-15—Presumptive coliforms in surface water from Whittier Narrows Test Basin (1 September 1964)

Incubation period (hours)	MPN per 100 ml from samples taken on				
	Day 1	Day 2	Day 3	Day 4	Day 5
24	2	2	2	<2	<2
48	8	8	13	<2	2
72	8	13	13	<2	2
96	8	13	13	<2	2
120	11	13	13	<2	5
Confirmed	8	8	2	--	2

< Signifies "less than," i.e., all tubes were negative.

Table 9-16—Bacterial sampling of the percolate at the Whittier Narrows Test Basin

(11 December 1964; basin dosed at 4 p.m.)

Sample location	MPN per 100 ml at		
	5:30 PM	6:30 PM	7:30 PM
	Presumptive coliforms		
Surface	$<2 \times 10^4$	$<2 \times 10^4$	$<2 \times 10^4$
2 feet	2×10^4	2×10^4	$<2 \times 10^4$
4 feet	2×10^4	$<2 \times 10^4$	$<2 \times 10^4$
6 feet	$<2 \times 10^4$	$<2 \times 10^4$	$<2 \times 10^4$
8 feet	$<2 \times 10^4$	5×10^4	2×10^4
	Confirmed coliforms		
Surface	----	----	----
2 feet	2×10^4	2×10^4	----
4 feet	2×10^4	----	----
6 feet	----	----	----
8 feet	----	5×10^4	2×10^4
	Presumptive fecal streptococci		
Surface	33	240	79
2 feet	33	33	31
4 feet	8	33	8
6 feet	13	31	23
8 feet	27	23	49
	Confirmed fecal streptococci		
Surface	2	5	<2
2 feet	8	<2	<2
4 feet	<2	<2	<2
6 feet	2	2	<2
8 ffet	<2	2	8

< Signifies "less than," i.e., all tubes were negative.

9.02 *Fecal Streptococci in the Test Basins.*

In October 1964 a routine assaying of the bacteriological samples from the test basins for fecal streptococci was started. This procedure was part of an attempt to ascertain the nature of the biological flora in the test basins. A multiple-fermentation-tube assay procedure was employed (5 tubes each of three dilutions) using azide-dextrose broth for the presumptive test and ethyl-violet-azive broth for the confirmed test.

The results, presented in Tables 9-17 to 9-20, for both test basins indicate that the concentrations of confirmed fecal streptococci are very low in both percolates. Presumptive fecal streptococci densities are at least a factor of 10 to 100 times lower than the concentrations of presumptive coliforms in the same waters.

9.03 *Summary of the Bacterial Results.*

The data presented in the previous sections of this chapter show that the percolates at the two test basins contain sizeable concentrations of coliform organisms, an almost complete absence of fecal coliforms, and measurable but low concentrations of fecal streptococci.

Previous studies (4) at Lodi, California, sponsored by the California State Water Quality Control Board, concluded that a bacteriologically safe water can be produced from settled sewage or final effluent if the water is percolated through at least four feet of soil. The term "bacteriologically safe" means capable of meeting the U. S. Public Health Service drinking water standards for the coliform group. Analyses were also made for fecal streptococci, but the concentration of these organisms in the final effluent used for the Lodi study was considered too low to be of statistical value.

Table 9-17—Presumptive fecal streptococci at the Whittier Narrows Test Basin (October 1964–June 1965)

MPN per 100 ml	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
10^3	--	--	--	11 (91.6)
5×10^2	13 (93.0)	2 (66.7)	15 (93.8)	10 (83.3)
10^2	10 (71.5)	1 (33.3)	10 (62.5)	5 (41.6)
5×10	9 (64.3)	1 (33.3)	10 (62.5)	4 (33.3)
10	4 (28.6)	0	3 (18.8)	1 (8.3)
2	3 (21.4)	--	2 (12.5)	1 (8.3)

Table 9-18—Presumptive fecal streptococci at the Rio Hondo Test Basin (October 1964–March 1965)

MPN per 100 ml	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
10^3	--	3 (75.0)	--	No data
5×10^2	--	2 (50.0)	--	
10^2	--	2 (50.0)	--	
5×10	5 (83.3)	2 (50.0)	7 (87.5)	
10	2 (33.3)	0	3 (37.5)	
2	0	--	2 (25.0)	

While it is generally accepted that filtration through fine-grained media tends to remove bacteria, it seems reasonable that with proper care a soil environment can be made suitable for the culture of coliform bacteria. The test basins have been operated in a manner to encourage the development of a highly active biological flora in the unsaturated zone between the ground surface and the ground-water table. Chemical analyses from the test basins demonstrate the successful cultivation of organisms necessary for nitrification, volatilization, ABS degradation, and the removal and production of organic compounds represented by the COD. In such an active biological environment, which is at times aerobic and at times anaerobic, it is not surprising to find high concentrations of coliform organisms. The work of stabilizing the water takes place in this near-surface region. Here also, the food supplied by the nutrients in the applied water is the most plentiful and available. It is therefore most likely that coliform organisms traveling with the percolating waters into the main ground-water system will find at greater depths and in saturated soil an environment less amply supplied with nutrients. As a result, they will soon be greatly reduced in concentration.

Table 9-19—Confirmed fecal streptococci at the Whittier Narrows Test Basin (October 1964–June 1965)

MPN per 100 ml	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
5×10	--	--	10	8
10	8	--	8	7
2	6	2	3	6

Table 9-20—Confirmed fecal streptococci at the Rio Hondo Test Basin (October 1964–March 1965)

MPN per 100 ml	Number of samples having MPN less than or equal to stated value at			
	2 feet	4 feet	6 feet	8 feet
5×10	5	--	--	No data
10	4	2	6	
2	3	0	4	

CHAPTER 10

VIROLOGICAL ANALYSES

10.01 *Preliminary Virus Study at Whittier Narrows.*

In January and February 1963, an unusual opportunity was afforded by the Sabin polio vaccine program to monitor part of the Whittier Narrows water reclamation system for massive doses of enteric viruses. The field sampling was undertaken by personnel of the LACSD and LACFCD with preliminary sample preparation such as pH adjustment and freezing being conducted at Caltech. Final virus assays were performed at Children's Hospital of Los Angeles under the direction of Robert McAllister, M.D. and Clyde Goodheart, M.D.

The samples assayed by Doctors McAllister and Goodheart were divided chronologically into three groups. Each portion was a one-liter grab sample. The first group, taken on 31 January and 1 February 1963 for background data, consisted of a liter sample each of primary effluent (i.e., settled sewage), chlorinated activated-sludge effluent, and a sample collected after percolation through two feet of soil at the Whittier Narrows Test Basin. No enteric viruses were detected in 800 ml portions of any of these samples. As shown later, enteric viruses are seldom detected in normal primary effluent by this method of assay, except during summer months.

The second group of samples was taken after a massive community feeding of the Sabin poliovirus vaccines. The grab samples of settled sewage taken on 13 February 1963 contained 82 plaque-forming units (PFU) per 800 ml (about 102 PFU per liter). Both poliovirus type III and type I were identified. The grab sample of activated-sludge effluent taken on 13 February 1963 assayed at 202 PFU per 800 ml (about 252 PFU per liter). Those plaques picked for further identification were all determined to be polio type III. Again, however, no enteric viruses were de-

Table 10-1—Assay Procedures for Enteric Viruses

A. *Preparation of Sewage Samples for Virus Assay.*

1. Composite sewage sample obtained.
2. Add 25 ml of sterile 2.4% Gelatin to each liter of sewage.
3. Raise pH to 8.0 with NaOH.
4. Sample can be stored at —60° F at this point.
5. Samples clarified by lowspeed centrifugation (approximately 3000 rpm for 20 min.)
6. Centrifuge above supernate at 100,000 g for two hours.
7. Sediment in each tube is dissolved by adding a small quantity of La Ye medium.
8. Treat with penicillin and streptomycin (100 units and 100 μ/ml) at room temperature for 1 hour.
9. Inoculate thioglycollate medium (Difco's B430) to test for bacterial contamination.
10. If contamination is present obtain antibiotic sensitivities and with appropriate drug (Kanamycin 100 μg/ml often useful).

B. *Assay Procedure for Enteroviruses, Including Sabin Type III Poliovirus.*

Host Cell

Primary Rhesus Monkey Kidney Cells with SV_5 antiserum purchased in 2-oz. plastic flasks.

Assay

1. Serial dilutions are performed in Hanks' Balanced Salt solution (Hanks' BSS) to obtain proper dilution for plating.
2. Medium is removed from cells by suction and 2 ml of Hanks' BSS added and removed by suction to wash the calf serum from the cells. The wash process is repeated three times.
3. 0.1 ml diluted virus sample is placed in each 2-oz bottle of cells.
4. Cells are then incubated for ½ hour at 37° C. Bottles are tilted in a slow rotating motion every 10 minutes to aid adsorption of virus to cell.
5. Meanwhile, white agar is melted and cooled to 43–44°C in a hot water bath. Note:—agar must not be hotter than 44°C.
6. The 2x La Ye medium is heated to 44°C.
7. Equal parts of the agar and La Ye medium are mixed.
8. After ½ hour of incubation the cells are washed with 2 ml of Hanks' BSS to remove unadsorbed virus.
9. To each 2-oz flask, 5 ml of the agar La Ye medium is added and allowed to cool.
10. Cells are incubated at 37°C for 3 days.
11. After 3 days of incubation the cells are stained.
12. At 43–44°C equal parts of red agar and 2x La Ye medium are mixed.
13. To the cells 3 ml of the red agar La Ye medium is added and allowed to cool.
14. The cells are placed in the incubator at 37°C and observed for plaques on the fourth and fifth day.

C. *Media.*

Hanks' Balanced Salt Solution (Hanks' BSS)

Chemical Compound	*1 liter contains*
NaCl	8.00 gm
KCl	0.40 gm
$MgSO_4.7H_2O$	0.20 gm
KH_2PO_4	0.06 gm
$Na_2HPO_4.7H_2O$	0.09 gm
$CaCl_2$	0.14 gm
Glucose	1.00 gm
$NaHCO_3$	0.05 gm
Phenol Red	0.02 gm

White agar
  1.3 gm of Difco Nobel Agar
  55.0 ml of deionized water
Red agar
  2.6 gm of Difco Nobel agar
  11.0 ml of 10x Neutral Red solution
  99.0 ml of deionized water
    Note: 10x Neutral Red solution contains 0.3% Neutral Red by weight.
La Ye (Lactalbumin hydrolysate—yeast extract)
  5 gm of Lactalbumin
  1 gm Yeast Extract (Difco)
  1000 ml Hanks' BSS and bicarbonate
  20 ml Calf serum
  200 units per ml penicillin
  200 mcg per ml streptomycin
  100 mcg per ml Kanamycin
  50 mcg per ml Fungizone

tected in 800 ml of the sample of water that had percolated through two feet of soil at the Whittier Narrows Test Basin. Samples taken from this same depth on the same date showed 48-hour presumptive coliforms having an MPN of 130 per 100 ml.

The third set of samples was taken on 14 March 1963 about a month after the Sabin vaccines had been used extensively. This set consisted of only two samples, one of settled sewage and the other of activated sludge effluent. The settled sewage assayed at five PFU per 800 ml (about 6.3 PFU per liter) with all five plaques being identified as polio type III. The effluent had one PFU in 800 ml (1.25 PFU per liter), which proved to be polio type III.

10.02 *Enteric Virus Sampling and Analysis.*

Prior to this investigation, none of the personnel directly connected with the project had had any training in assay procedures for viruses. Fortunately, however, a Caltech graduate student in Environmental Health Engineering, Mr. John Cookson, was performing doctoral research on the adsorption of viruses and consequently had developed considerable experience in virological techniques. With the continued interest and assistance of Dr. Robert McAllister of the Children's Hospital, Mr. Cookson developed and supervised the assay procedures used in this investigation. Most of the laboratory work was performed by Mrs. Jean Edens.

The assay procedure developed by Mr. Cookson and Dr. McAllister for enteroviruses is presented in Table 10-1. No attempt was made in this investigation to identify the plaques other than to obtain a gross count of enteric viruses.

Samples taken from the field study were normally one to four liters in volume. The samples from the Whittier Narrows Treatment Plant were 24-hour composite samples using the proportional sampling units owned and operated by the LACSD. Surface samples from the Whittier Narrows Test Basin were one-gallon grab samples, while the percolate was sampled by placing a one-gallon bottle at each of the various sampling taps within the central well, allowing it to fill up and overflow overnight.

From limited virological data in the literature, it is evident that enteric viruses occur in sewage during warm summer months, but that they are difficult to detect in winter sewage except by the swab technique. This procedure involves the suspension of an adsorbent pad in the sewage stream for three days or longer. Viruses are adsorbed partially to the pad and can be squeezed off for assay. The procedure indicates the presence or absence of viruses, but not the concentration per liter. The swab technique was not utilized in the Whittier Narrows investigations. Instead, an attempt was made to get a quantitative evaluation of enteroviruses in the summer and winter.

The results of virological assays on samples taken during August 1964 and February–March 1965 are presented in Table 10-2. Out of 33 samples, detectable enteric viruses were found in only one instance, namely, a sample of settled sewage taken on 4–5 August 1964 which showed a concentration of 93 PFU per liter. None of the winter samples showed any viruses whatsoever in one liter.

10.03 *Supplementary Studies with Sabin Type III Virus.*

Compared to bacterial indicators of pollution, enteric viruses occur in very low concentrations. Normally, the density of viruses in treated sewage effluents is so small that a quantitative study of these concentrations is most difficult. With the laboratory techniques described in Table 10-1, it is apparent that concentrations of enteroviruses lower than 50 to 100 PFU per liter are difficult, if not impossible, to detect. Without doubt, a large percentage of the viruses is lost by adsorption on particulate matter which is removed in the initial low-speed centrifuging. Hence, the assay procedure related only to the viruses that are completely independent of even the finest particulate matter.

In order to measure quantitatively the behavior of enteric viruses in the Whittier Narrows system, certain preliminary experiments were indicated. Two studies of the die-off with time of polio viruses added to samples of Whittier Narrows Treatment Plant effluent were conducted in the laboratory. The added viruses were purchased from Pfizer Laboratories in

Table 10-2—Enteric viruses at Whittier Narrows Treatment Plant and Test Basin

Location	No. of samples	Dates	PFU/liter
A. Whittier Narrows Treatment Plant (all 24-hour composites)			
1. Primary effluent	1	4–5 Aug. 1964	93
	1	19 Aug. 1964	none
	2	11, 25 Feb. 1965	none
	3	4, 10, 18, March 1965	none
2. Activated-sludge effluent	1	4–5 Aug. 1964	none
	1	11 Feb. 1965	none
3. Final effluent (after foam separation and chlorination)	1	20 Aug. 1964	none
	2	11, 25 Feb. 1965	none
	3	4, 10, 18 March 1965	none
B. Whittier Narrows Test Basin (all grab samples)			
1. Surface water	2	9, 23 Feb. 1965	none
	4	3, 9, 16, 17 March 1965	none
2. Two-foot sampling pan	2	10, 23 Feb. 1965	none
	4	3, 9, 17, 24 March 1965	none
3. Four-foot sampling pan	2	10, 23 Feb. 1965	none
	4	3, 10, 17, 24 March 1965	none

the form of live polio vaccine, oral concentrate, Sabin type III. The stock solutions were assayed and found to have concentrations of about 1 to 4×10^4 PFU per milliliter. About 15 ml of the stock solutions were added to 1500 ml of the reclaimed water. Hence the concentration in the reclaimed water should have been about 1 to 4×10^5 PFU per liter. Assays of the virus concentration in the seeded samples were performed over selected time intervals. The results of these assays are presented in Table 10-3 and Figure 10-1. The measured virus concentrations at the beginning of each test were only about 2.5 to 5.0 percent of the concentration expected from the known virus stock concentration and the measured volumetric dilution. A die-off of 60 to 70 percent from the initial measured concentration occurred in 2 to 4 days, and a 98 percent die-off was found after 11 days.

On 15 July 1965 a special virus study was initiated at the Whittier Narrows Test Basin. Dr. J. H. Brown of Wyeth Laboratories, Inc. in Marietta, Pennsylvania kindly supplied at no cost to this project 10 liters of Sabin Type III poliovirus vaccine. The concentrate assayed at about 10^{10} PFU per liter in our laboratory. The concentrated virus vaccine was added to the wastewater effluent spread at the Whittier Narrows Test Basin on 15 July. Enough sodium sulfite was also added to the spread water to eliminate a combined chlorine residual of 2.0 mg/l in 185 cubic meters of water. Samples of the first water coming from the sampling pans were taken as a composite from 4:50 to 6:00 P.M. on 15 July. These samples were placed in a container with dry ice. They were picked up and delivered to Caltech by 7 P.M. and placed in the dry freeze at −60°C. A 5-gallon sample of the surface water was composited from grab samples taken at various locations around the test basin. Subsequently, the test basin was routinely flooded on 16, 17, and 18 July and samples were taken of the percolate from the sampling pans on each of these days at 7 to 8 P.M., in an effort to determine if viruses were being flushed out of the basin. No virus was added to the surface water except on 15 July 1965. The flooding of the test basin required about 45 minutes. The virus concentrate was added slowly to the influent stream over this interval in order to assure a uniform distribution of the virus in the surface water. The sodium sulfite was added similarly to the influent stream.

The total number of added viruses (10^{11} PFU) was diluted in 191.1 cubic meters of wastewater effluent, making a theoretical concentration of 5.23×10^5 PFU per liter in the water being spread. Based on the previous tests of virus in this effluent, the measurable concentration should have been 2.5 to 5.0 percent of the theoretical density, or 13 to 26×10^3 PFU per liter. Actually, a sample of the surface water showed only 3×10^3 PFU per liter, which was the lowest concentration that the tested dilutions would detect.

None of the samples from the 2, 4, 6, and 8-ft. pans at 5–6 PM or 7–8 PM on 15 June, nor at 7–8 PM on 16, 17, and 18 June 1965, showed any enteroviruses (including polio type III) in the dilutions tested. Actually, these results should be reported as less than 1000 or less than 1500 PFU per liter, because these were the lowest concentrations that could be detected in the dilutions used for this test.

It is recognized that the results of these field studies of the fate of enteroviruses in percolating water are very inconclusive. The test should have been repeated, but unfortunately the project had to be terminated.

Table 10-3—Die-off of attenuated polio vaccine in Whittier Narrows reclaimed water

Test I. Sabin Polio Vaccine Type III added to reclaimed water.* Sample allowed to sit on magnetic stirrer in laboratory at room temperature.

Time (hours)	Measured virus concentration (PFU per liter)	Percent remaining
0	11×10^3	100
0.5	10×10^3	91.0
1	8×10^3	72.7
4	5×10^3	45.5
10	6×10^3	54.5
20	3×10^3	37.2
48	2×10^3	18.2
72	2×10^3	18.2

* pH = 7.8.

Test II. Sabin Polio Vaccine Type III added to reclaimed water. Sample placed on magnetic stirrer in constant temperature room at 20°C.

Time (days)	Measured virus concentration (PFU per liter)	Percent remaining
0	5×10^3	100
1	3×10^3	60.0
2	1.2×10^3	24.0
3	1.1×10^3	22.0
4	1×10^3	20.0
5	8×10^2	16.0
6	6×10^2	12.0
7	7×10^2	14.0
11	1×10^2	2.0

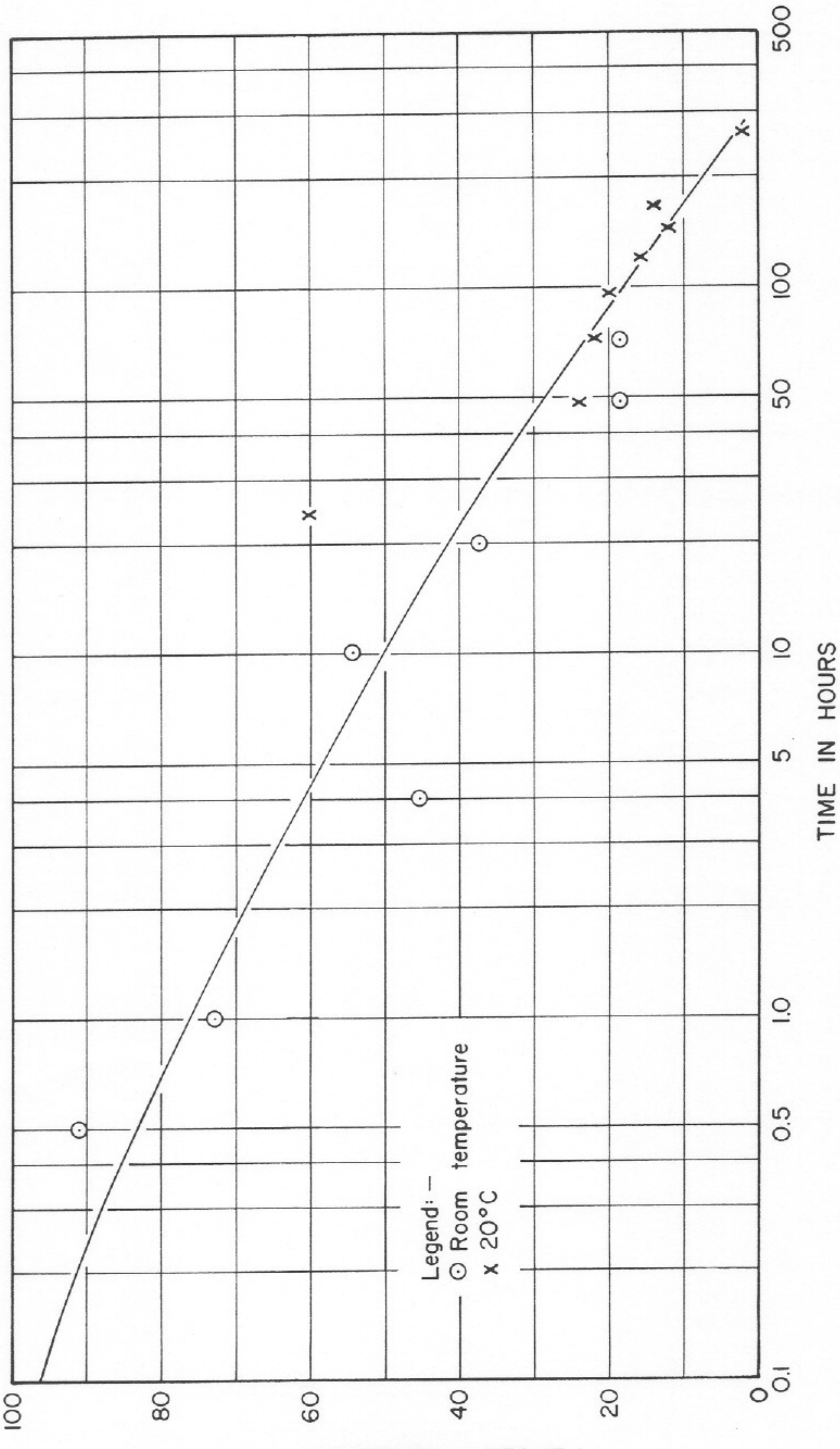

Fig. 10-1—Die-off of Polio Type III Vaccine in Whittier Narrows Reclaimed Water

required for spreading, and

b. to achieve an optimum degree of tertiary treatment so that the percolated water will be suitable for subsequent beneficial uses.

Both of these objectives are related, quite obviously, to the quality of wastewater available for spreading and to the nature of the soil system. For a given wastewater and a given soil system, however, both objectives are a function of the manner in which spreading operations are conducted. Should basins be flooded continuously or should they be dried out occasionally? If intermittent operation is indicated, how frequently should basins be rested, i.e., what is the optimum cycle for dosing and drying? Is the optimum operation for hydraulic acceptance also most advantageous for tertiary treatment and quality control? These are questions that this chapter attempts to evaluate.

11.02 *Relevant Findings.*

From the data presented in foregoing chapters of this report, the following observations are considered to be pertinent to this discussion:

a. The intermittent dosing of undiluted effluent at the Whittier Narrows Test Basin resulted ultimately in much better hydraulic loading and higher rates of infiltration than the continuous flooding of diluted wastewater at the Rio Hondo Test Basin, despite the fact that the soil at Whittier Narrows was initially much less amenable to infiltration and percolation.

b. The rate of hydraulic acceptance at the Whittier Narrows Test Basin decreased during the initial five months of operation but then rose significantly to a phenomenal rate as the bed ripened.

c. Hydraulic acceptances at both test basins were improved dramatically by a surface coating of six inches of pea gravel.

d. With daily dosing and drying at the Whittier Narrows Test Basin, more than 90 percent of the ammonia and organic nitrogen was oxidized to nitrates and approximately half of the residual carbonaceous demand of the effluent (as measured by COD) was removed in the first two feet of percolation. In effect, about 80 percent of the total oxygen demand of the effluent was exerted in the upper two feet.

e. Wastewater that had percolated two feet or more at the Whittier Narrows Test Basin was frequently devoid of dissolved oxygen. Nevertheless, there was

concentrations in the percolates were higher than in the applied water. Fecal coliforms and fecal streptococci, however, were almost completely removed by a few feet of vertical percolation. The high density of coliforms is attributed to a heavy growth of non-fecal soil-type bacteria (such as *Aerobacter aerogenes*) in the beds. This growth may account for the increase in COD after six and eight feet of percolation.

On the basis of these relevant findings, the following rationale for operating spreading basins has been developed.

11.03 *Reaeration Mechanisms in Soil Systems.*

The natural decomposition of organic matter in a reclaimed water will occur most rapidly, and probably most effectively, in an aerobic system. To maintain aerobic conditions in the soil of a spreading basin used for the recharge of reclaimed waters requires planning and a knowledge of the mechanisms for reaerating the soil.

There are three principal methods for reaeration of soil: (a) dissolved air carried into the soil by the percolating water, (b) the hydrodynamic flow of air resulting from the "piston-like" movement of a slug of water through the soil, and (c) diffusion of air into the soil body when there is no standing or ponded water on the ground surface. Strictly speaking, the biochemical soil system is not concerned with air in general but rather with the exchange of oxygen and carbon dioxide, the two principal gases involved in aerobic stabilization.

If a soil is flooded continuously, the only oxygen supplied to the soil system comes from the dissolved oxygen in the applied water. When this applied water has an oxygen demand in excess of its dissolved oxygen, the system will become anaerobic after a short depth of percolation, for there is no mechanism to supply additional oxygen to meet the demand. Even when they are saturated at normal temperatures and pressures, wastewaters contain less than 10 mg/l of dissolved oxygen. In contrast, the total oxygen demand of a highly treated wastewater, most of which can be exerted in the first two feet of percolation, will exceed 100 mg/l. Consequently, dissolved oxygen in the applied water is relatively insignificant.

When a spreading basin is free of ponded water and the pores of the soil are draining under gravity, conditions obtain whereby the soil can be reaerated by the diffusion of oxygen into the ground and the

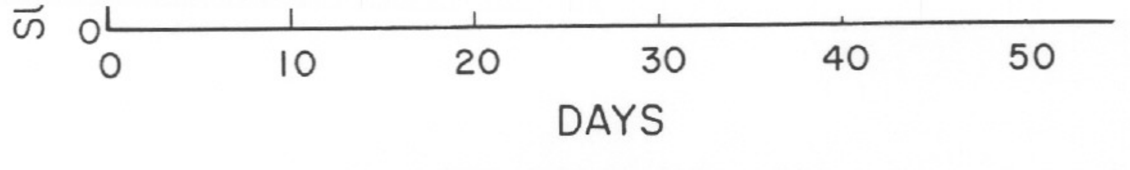

Fig. 6-2—ABS and LAS in Column Effluents as a Function of Time—Experiment I.

diffusion of carbon dioxide out. The thin films of water retained on the grains of soil in the zone of aeration will assume a degree of saturation that is dependent on the soil structure, the soil grain size, and the organic growth within the soil system. When this water is subjected to an oxygen demand, the dissolved oxygen may well be utilized by the microorganisms as fast as the water can be reaerated with oxygen from the soil atmosphere. In this event, diffusion across the air-water interface of the thin film might well be the limiting mechanism, especially if the partial pressure of oxygen in the soil interstices is low. Consequently, the partial pressure of oxygen in the atmosphere within the soil interstices is a significant factor in controlling the rate of aerobic stabilization. Similarly, an excess of carbon dioxide in the soil atmosphere may serve to inhibit aerobic action. The problem of soil reaeration in a system that is diffusion controlled at the ground surface is one that has been considered by soil physicists for many years in relation to root growth (55, 56). This problem is also related to the aeration of buried refuse in sanitary landfill (57).

As a simple example of some of the information obtainable from the diffusion problem, consider first an idealized system. Assume that a one-dimensional soil column has drained to field capacity (i.e., there is no further flow of water under gravity) so that its pores are partially but not wholly filled with water. There is a continuous air film down from the ground surface to the water table. The ground surface is free of ponded water. At the beginning of observation, let the air contained in the soil body be devoid of oxygen while the air above ground has a normal partial pressure of 0.21 atmospheres of oxygen. In order to describe diffusion into the soil quantitatively in a simple manner, it must be assumed further that a known diffusion coefficient can be assigned to the system. This problem is treated as a molecular diffusion problem, and the molecular diffusion coefficient of oxygen in free air is well known. The presence of the soil grains tends to reduce the effective diffusion coefficient, so that for soil this modified diffusion coefficient is determined as the product of the molecular diffusion coefficient in free space times the effective porosity of the medium times a dimensionless constant with the value of about 0.66. This relation is empirical and results from experiments by Penman (57). For gas transfer in soils, the degree of saturation changes the effective porosity, i.e., if the soil is saturated with water the effective porosity is zero.

The transfer of gases by diffusion is governed by Fick's law, which can be written

$$q = -D_p \text{ grad } C \tag{1}$$

where q is the transfer rate per unit area of the medium, D_p is the molecular diffusion coefficient for the soil body, and grad C is the concentration gradient. D_p is related to the diffusion coefficient in free space (D_0)

$$D_p = 0.66 \text{ P } D_0 \tag{2}$$

where P is the effective porosity of the soil. The following values of the diffusivity of oxygen and carbon dioxide are found in the Handbook of Chemistry and Physics (58):

	Diffusivity, D_0, in *sq. ft. per day*	*sq. meters per day*
Oxygen	17.5	1.62
Carbon dioxide	14.0	1.30

Mathematically, the problem of diffusion into a porous medium governed by the conditions stated previously that the above-ground concentration of oxygen is constant for all time, and the below-ground oxygen concentration of oxygen is initially zero at all depths (a semi-infinite medium), has the solution

$$C = C_0\left[1 - \text{erf}\left(\frac{x}{2\sqrt{D_p t}}\right)\right] \tag{3}$$

where C is the concentration of oxygen at any depth x for any time t, C_0 is the surface concentration of oxygen, and D_p is the oxygen diffusion coefficient for the porous medium. The term "erf" is the error function which is a well-known tabulated mathematical expression, whose argument in this problem is the dimensionless group, $x/2(D_p t)^{\frac{1}{2}}$. The total flux of oxygen across the ground surface is found by using equation 3 and Fick's law (equation 1) and integrating over time t, to get the total amount of oxygen (M) moving into the ground from t = 0 to t = T, *viz,*

$$M = 2C_0\left(\frac{D_p T}{\pi}\right)^{1/2} \tag{4}$$

For a typical numerical example, consider a soil that has a porosity of 40 percent and is 50 percent saturated with water. The effective porosity of this soil for gas is therefore 20 percent. The oxygen in the atmosphere above ground is about 21 percent by volume of the air or about 300 mg O_2 per liter of air. The diffusion coefficient for oxygen in air is about 1.62 m^2 per day (17.5 ft^2 per day). Initially, the soil atmosphere is void of oxygen at all depths. The problem is to find how much oxygen will flow into the ground and how the concentration of oxygen in the soil atmosphere will vary as a function of depth and time.

Data:

Porosity (volume voids per total bulk volume of soil) $P = 0.40$
Saturation with water, $S = 0.50$
Diffusion coefficient for oxygen, = 1.62 m^2 per day
C = oxygen concentration
x = depth below surface, x = 0 is the ground surface
t = time

Boundary Condition:

C at x = 0 for all time equals a constant, $C_0 = 300$ mg/l of air

Initial Condition:

C at t = 0 for all x > 0 (below ground surface) equals zero

0.05	0.9436
0.1	0.8875
0.2	0.7773
0.3	0.6714
0.4	0.5716
0.5	0.4795
1.0	0.1573
1.5	0.03389
2.0	0.004678

C_0 = 300 mg O_2 per liter of air = 300 gm/m^3
= surface concentration

D_p = 0.66 (0.5)(0.4)(1.62) m^2/day
= 0.214 m^2/day

Substituting the values of the parameters into equation 3 gives the following typical results. At a depth of 0.5 meter below the surface, it requires one day for the concentration of oxygen in the soil air to reach 54.4 percent of the surface concentration. At a depth of 5 meters, it requires 100 days for the concentration of oxygen in the soil air to reach this value. It should be apparent, therefore, that diffusion is a relatively slow process. The solution to this problem has the property of depending only on the dimensionless parameter, $x/2(D_p t)^{1/2}$, which means that the time required for any point to reach a given concentration is proportional to the square of its distance from the ground surface and varies inversely as the diffusion coefficient.

The second part of the problem was to determine the amount of oxygen that would pass into the ground in a specific amount of time. This information is obtained from equation 4. Let M equal the total amount of oxygen per unit area passing into the ground by diffusion.

Equation 4 shows that the amount of oxygen passing into the ground is directly proportional to the atmospheric concentration at the surface and proportional to the square root of the amount of time passed since the diffusion process began. Thus to get twice as much oxygen into the ground under the conditions of this problem requires four times as long.

By way of comparison of the oxygen demand with the amount that can be supplied by diffusion, a typical undiluted wastewater applied at the Whittier Narrows Test Basin had a total oxygen demand of about 150 mg/l, of which approximately 80 percent, or 120 mg/l, was exerted in the first two feet of percolation.

reclaimed water percolating down to the water table. However, it serves to illustrate the order of magnitude of oxygen transfer into an idealized soil system that has no water ponded on the surface and has a continuous passage for gas flow down into the soil. By the nature of the diffusion process, the soil close to the surface has a considerably greater opportunity to be reaerated than the deeper parts of the ground. The simple model permits a quantitative estimate of how much faster the near-surface soil atmosphere will reach a given oxygen concentration than the soil at greater depths.

The real problem of reaeration of pellicular water in an unsaturated soil is very complicated. The oxygen must enter the soil from above ground, move through the unsaturated pores and pass into the film of water covering the soil grains. The rates at which gases are exchanged across the air-water interface within the soil and the rate at which the oxygen is utilized by organisms during the decomposition or organic matter are not well-known facts. Mathematical equations can be formulated for the soil-water and soil-air systems which include diffusion, oxygen depletion, gas exchange across the interfaces, and transport of gases by the water and air. These equations, however, are complicated and the most simple cases are not easily solved. The major roadblock is a lack of knowledge of the physical parameters, such as rates of organic decomposition and gas exchange coefficients, needed to evaluate the numerical solutions to these models.

Up to this point, molecular diffusion has been the mechanism under consideration. Experience in physical chemistry indicates that if other transfer processes can be invoked, the transport of substances can be in-

$$M = 2C_0\left(\frac{D_p T}{\pi}\right)^{1/2} \qquad (4)$$

For T = ½ day,

C_0 = 300 mg O_2/liter, D_p = 0.214 m^2/day

$$M = 2(300)\left(\frac{0.214(0.5)}{3.14}\right)^{1/2}$$

$$= 110 \text{ gm } O_2/\text{m}^2$$

and for T = 1 day,

$$M = 155 \text{ gm } O_2/\text{m}^2$$

creased ten, one-hundred, or even a thousand times over the molecular process. It is possible, for example, that the hydrodynamic transport of oxygen into the soil may cause considerable reaeration. For example, a cubic meter of atmospheric air contains about 300 gm of oxygen. Consider a case where a slug of water passes down through a soil body as a reasonably compact volume (this is generally not true). Such a water slug would displace the air in front of it (pushing it out laterally from its path, perhaps) and cause air to flow in behind it from above the ground surface. A slug of water could thus conceivably draw into the soil a volume of air equal to the volume of the pores filled and drained by the water during its downward flow through the soil.

By way of example, let the true porosity of a soil be 0.40 and the effective porosity be 0.2 after drainage by gravity. Then a cubic meter of such soil contains 0.2 cubic meters of air, with $0.2 \times 300 = 60$ gm O_2. Hence reaeration by this piston action would supply 60 gm O_2/m^2 in a one-meter depth. This quantity could be partially in addition to that supplied by molecular diffusion.

From what is known of the way water passes into and through soils it is not believed that percolating water moves as a slug completely filling the pores through which it flows. Unless a soil is covered with ponded water for a very long time, the soil pores do not become completely saturated. The rate at which ponded water infiltrates into the ground is dependent in part on the rate at which the water moves down in the soil, i.e., on the permeability of the soil. Frequently, however, percolation through the soil is more rapid than infiltration past the soil surface. The soil permeability for water depends on the degree of water saturation. Permeability is highest when saturation is highest. If the permeability is unity for the saturated soil, it is about one-tenth of unity when the saturation is reduced to 50 percent. By changing the degree of saturation a small amount, the rate of transport of the water can be increased many times.

How fast a volume of air can pass into a soil will depend on the rate of percolation. It is readily apparent that hydrodynamic displacement as a mechanism of soil aeration can be important only if the soils fill and drain rather quickly, or if the zone of aeration is very deep. Diffusion of gases in and out of the soil is probably of equal importance with hydrodynamic phenomena in the near-surface environment. Whatever the mechanism, it is apparent that sufficient oxygen entered the soil at the Whittier Narrows Test Basin to satisfy the oxygen demand in the top two feet. Below this depth, the oxygen requirement of the pellicular water probably exceeded the rate of supply, because there was evidence of anaerobiosis.

11.04 *Oxygen Demand of the Applied Water.*

In the previous section, an approximation of the amount of oxygen that could be transferred from the atmosphere to the soil body by several mechanisms was presented. The question to be discussed herein is the amount of oxygen necessary to stabilize a wastewater. Chapter 8 of this report describes how to estimate the total oxygen demand of a wastewater. By total oxygen demand is meant the biochemical oxygen demand exerted over a very long period of time in a BOD bottle. This demand is approximately equal to the COD plus the oxygen required to convert all ammonia and organic nitrogen to nitrates. Although this demand may extend over 100 days in a BOD bottle, most of it appears to be exerted very rapidly in percolation through two feet of well-ripened soil, provided of course that sufficient oxygen is present in the soil interstices.

If sufficient oxygen is not supplied to the soil, e.g., if a basin is flooded continuously with wastewater so that the only available oxygen is that dissolved in the applied water, the carbonaceous and nitrogenous oxygen demands of a percolating wastewater cannot be satisfied. Such a water will become anaerobic. Some of the organic solids, to be sure, will be removed by adsorption on the soil grains and indeed some of the ammonia and organic nitrogen may be into anaerobic biosynthesis. In all likelihood, sulfates will be converted to hydrogen sulfide, so that the water will be highly odorous when later pumped to the surface. With an aerobic system, the production of nitrates provides a strong safeguard against conversion of sulfates to sulfides inasmuch as oxygen will be stripped from nitrates before sulfates.

11.05 *Some Chemical Changes in Water Quality Resulting from Surface Spreading.*

Many biochemical and physical-chemical phenomena occur under steady-state conditions as a result of surface spreading operations. Phenomena such as adsorption and ion exchange are important initially but a soil body has only a finite capacity for such processes. An operation extended over a long period of time might easily exceed the capacity of the soil body for adsorption and/or ion exchange.

The aerobic stabilization of wastewater in treatment plants is normally controlled in such a manner as to prevent the oxidation of ammonia and organic nitrogen. Problems of denitrification causing the bulking of sludge in secondary tanks is often cited as a reason for not nitrifying in the plant. As discussed in Chapter 8 of this report, the oxygen necessary to stabilize the nitrogen in the reclaimed water at Whittier Narrows was generally two to three times the oxygen required for stabilization of the carbonaceous substances. Unlike a secondary treatment plant process, which although aerobic, uses oxygen somewhat inefficiently, the oxygen utilized by a spreading system may be 100 percent of what is supplied. It is necessary, therefore, to decide in a complete system at what point it is most economical and beneficial to stabilize the nitrogenous oxygen demand. Nitrification occurs quickly and easily in the near-surface environment of a spreading basin. However, it requires an enormous proportion of the total oxygen demand. If nitrification could be completed prior to the spreading operation, the oxygen supplied by the natural processes associated with percolation might be available and used for stabilization of other substances.

The presence of chemically combined oxygen such as nitrate may prove beneficial to the water when it gets deeper in the soil where reaeration is almost zero and all dissolved oxygen has been utilized. Many facultative organisms are capable of reducing nitrate to nitrogen gas. Four mols of nitrate-nitrogen (as N) reduced to nitrogen gas is equivalent to 5 mols of oxygen (as O_2) in contrast to the oxidation of the ammonia-nitrogen to nitrate which required 2 mols of oxygen (as O_2) to oxidize one mol of nitrogen to nitrate (as N). Volatilization of nitrate will not give back the equivalent amount of oxygen as oxidation because of the difference in the oxidation states of the end products. Nevertheless, the presence of nitrate may inhibit the reduction of substances like sulfate whose reduced compounds are quite noxious. The reduction of nitrates to nitrogen gas can aid in lessening the total dissolved solids in a water as well as removing the health hazard to young babies relating to methemoglobinemia.

Another phenomenon associated with nitrification in a soil system is the high production of free hydrogen ions. This reaction as observed in laboratory columns is discussed in Chapter 6 of this report. Unbuffered waters could be strongly affected by the nature of the soil matrix through which they are percolated. No attempt will be made to discuss bicarbonate and carbonate equilibria other than to mention that biodegradation of carbonaceous substances also leads to hydrogen ion production. A soil matrix containing calcium carbonate or iron minerals is susceptible to being dissolved by waters of low pH. Such dissolution could then result in an increase in total dissolved solids as well as an increase in water hardness. These phenomena are described to point out that soil systems are very dynamic chemically and the quality of the percolate at any depth can vary appreciably in relation to the quality of the applied water and the nature of the soil through which the percolate flows.

Not only is the quality of the water affected by its soil environment but in turn the rate at which water percolates through a soil can be strongly reduced or increased by the nature and distribution of its dissolved ions. Effects of ionic composition on permeability are of great concern to irrigation practices and are widely discussed in agricultural and soil science literature (59) resulting in classifications of waters for irrigation in terms of the relative amounts and ratios of sodium, calcium, magnesium, and potassium.

11.06 *Physical Behavior of Waters Recharged by Surface Spreading.*

Filtration of public water supplies for the removal of suspended material and microscopic organisms has been relatively common throughout the world in this century. Filtration also occurs during the vertical percolation of wastewater from a spreading basin. Natural soils are normally finer grained than artificial sand filters; consequently it is to be expected that soil will become clogged with material removed by filtration. Unlike an artificial sand filter, a soil system cannot be cleaned by backwashing; hence another mechanism for cleansing must be available. It is common practice to scrape the surface of intermittent sand filters to maintain their hydraulic acceptance, especially after primary effluent has been applied.

The rate at which an unsaturated soil will accept water, known as the infiltration capacity, has been measured by hydrologists and agricultural engineers under many conditions. It depends on the physical characteristics of the soil (e.g., grain size and distribution, porosity, and clay content), the chemical composition of the soil (especially the exchangeable cations), temperature, the chemical quality of the applied water, and the characteristics of suspended and colloidal solids in the applied water.

When an unsaturated natural soil or fine-grained sand is first dosed with water, the initial rate of infiltration is high, frequently in excess of four inches per hour for the first few minutes (36). In effect, the dry soil acts as a sponge. As interstices near the surface of the soil become filled or partially filled with water, however, the infiltration capacity drops off rapidly to half or less of its initial value within 30 minutes or so. Thereafter, the rate of infiltration diminishes slowly as suspended or colloidal solids clog the interstices. If a spreading basin is allowed to dry out and reaerate, it will exhibit another high initial rate of infiltration with subsequent decrease as flooding continues. Hydrologists have found that with frequent resting, the total hydraulic acceptance of a basin will be higher than for a basin flooded continuously.

Even with intermittent operation, it has been the experience of most observers that the infiltration capacity of a spreading basin gradually decreases over a period of months or years. This diminution may be attributed to the formation of a surface film, to deposition of particulate matter in the soil interstices, and/or to the exchange of sodium for calcium in the clay portion of the soil matrix with resultant swelling of the clay. A surface film can be removed by scraping or be broken up by a disk harrower. Sodium adsorption can be reversed by the application of calcium salts. But there is generally no remedy for the clogging of innterstices by suspended and colloidal particles. As a consequence it has been the general experience that the infiltration capacity of spreading basins decreases slowly with time.

The changes in hydraulic acceptance at the Whittier Narrows Test Basin, however, have been contrary to all expectations. After decreasing markedly for five months, as might be predicted, the infiltration capacity stabilized for about half a year and then began a remarkable increase (see Figures 4-7 and 4-9, or Table 4-1). Initially at a rate of 1.45 feet per day, the infiltration capacity dropped off to 0.59 feet per day in the first five months of operation. Then it increased to 5.20 feet per day in August 1964, undoubtedly favored by the effect of high temperatures on viscosity, decreased slightly during the cold winter months, and started to rise again with warmer spring temperatures. The most dramatic increase in filtration capacity occurred in February 1964 after six inches of pea gravel (0.25 to 0.50 inch) had been applied to the surface. Suter and Harmeson (43, 44) reported a similar experience for the test pits used for ground-water replenishment at Peoria, Illinois.

How can the improvement in filtration capacity prior to the application of pea gravel and the continued increases even after the pea-gravel effect be explained? Why did the pea-gravel prove so effective? The following hypotheses have been advanced:

a. The soil at the site of the Whittier Narrows Test Basin had been used for many years to raise crops such as alfalfa, with irrigation from ground water. A typical analysis for such irrigation water would show approximately:

	mg/l	*me/l*
Calcium	40	2.0
Magnesium	12	1.0
Sodium	23	1.0
Dissolved solids	350	--
Conductivity (EC $\times 10^6$)	540	--

The initial percolates at this site, however, contained about 3500 mg/l of dissolved solids, indicating that the soil moisture was about ten times as concentrated as the irrigation water. The sodium adsorption ratio (SAR) for the concentrated soil moisture at the start of percolation is then computed to be 2.6. For the applied wastewater, the SAR averaged about 4.2.

Both of these SAR values are extremely low on the scale described in USDA Handbook 60 (59), so there should be very little effect, if any, from the exchangeable sodium. If anything, the applied water would cause a very slight increase in exchangeable sodium with a consequent decrease in infiltration capacity. Actually, as Table 4-33 reveals, the calcium and magnesium were higher and the sodium somewhat lower in the 8-foot percolate than in the applied water, indicating a slight accumulation of sodium in the soil. This effect would decrease infiltration capacity rather than provide the increase that was observed. It is concluded, therefore, that exchangeable sodium does not account for the improvement in the rate of hydraulic acceptance.

b. A more plausible chemical explanation can be derived from Table 4-33. During its use for agriculture, the soil had been in equilibrium with water having a pH value of about 8.3. The pH of the applied wastewater was generally slightly lower than 8.0 and it was diminished even more by nitrification in the upper two feet of soil. Consequently, it is not surprising that calcium and magnesium were dissolved from their carbonates and hydroxides. Table 4-33 shows a large increase in soluble calcium and a moderate increase in magnesium after eight feet of percolation.

Assume that the data in Table 4-33 are typical of the entire period of operation and that 100 mg/l of calcium were leached from the soil matrix with each passage of percolating water. If the density of calcium carbonate is taken as 2.5 gm/cc, computations show that the volume of calcium carbonate leached from the soil in two years of operation was less than 0.2 percent of the volume of voids in the soil. Consequently this very slight increase in the porosity cannot account for the very large rise in hydraulic acceptance.

c. The third hypothesis relates to the organic growth that developed in the well-ripened test basin, as witnessed by the increase in coliform bacteria in the percolate and the high COD of cores taken from the bed. It is conceivable that the biological growth within the bed forced the individual soil grains farther apart, especially when the growth was wet. When the bed was aerated, the growth dried out partially but the soil grains did not return to their original compact state. The effect of biological ripening, therefore, may have been to cause the bed to swell, with a concommitant increase in porosity and permeability. Unfortunately, no physical measurements were taken to prove or discount this hypothesis.

d. The coating of pea-gravel undoubtedly acted as a flocculating mechanism, converting collodial solids and pin-point floc in the activated-sludge effluent to larger suspended solids which could be removed at the surface of the sand, without moving into and clogging the interstices. Some of the flocculated solids undoubtedly were adsorbed to the pea-gravel itself, there to decompose in the alternate wetting and drying.

The gravel blanket on the Whittier Narrows Test Basin also reduced markedly the need for basin maintenance. Before the gravel was applied, a standing crop of weeds more than one foot high might easily cover the basin in a month's time. There are advantages and disadvantages of allowing weeds to grow in spreading basins. The action of root growth undoubtedly keeps the soil porous, enhancing infiltration rates. On the other hand, increased water loss from evapotranspiration by the plant is not beneficial.

During initial operation at the Whittier Narrows site, weeds were removed and reductions in infiltration rates attributed to soil plugging were relieved by scarifying and disking the basin surface. Such maintenance required attention on a monthly basis. After placing of the gravel, more than a year's operation followed with no need for disking the basin. Furthermore, the infiltration rates increased. Weed removal became a minor problem on the graveled bottom requiring the removal of isolated plants at random intervals. Although the mechanism behind the ability of a gravel pre-filter to enhance the infiltration rates of the soil body is not clearly understood, the reduction in basin maintenance resulting from such a cover merits consideration in any spreading operation.

Recently the process of sand filtration of water has come under more severe scrutiny by the sanitary engineering profession. Some of the ideas on the nature of clogging and the hydraulics of filtration discussed by Camp (60) are applicable to ground-water recharge by surface spreading.

The draining of porous media in the unsaturated state is a very complicated problem. Essentially, drainage is a diffusion problem where the diffusing substance is the water. The problem is made difficult because the diffusivity or diffusion coefficient is generally a very complicated function of the moisture content of the medium. A bibliography and summary of the nature of these problems was prepared by Wang (61).

11.07 *Practical Considerations.*

The requirements of the engineer faced with groundwater recharge by surface spreading of a reclaimed water can be summarized by a need to know several specific things:

1. How much reclaimed water is available for spreading? What is the water quality, i.e., how much pre-treatment has occurred and what is the total oxygen demand of the water to be spread?
2. What is the nature of the spreading area, specifically, the total spreading area available, the depth to the water table, the initial infiltration rates for saturated and unsaturated flow, the soil profile between the ground surface and the water table with regard to layers that may have reduced permeability for saturated or unsaturated flow?
3. What happens to the recharged water after it reaches the main ground-water system?
4. How long will it be in the ground, how far will it travel, and what directions will the water follow until it is pumped out again for re-use?
5. What treatment, if any will the pumped water receive, and to what beneficial use will the water be put?

If definite answers can be given to these questions, a suitable set of spreading basins can be designed rationally.

For a hypothetical operation, somewhat similar to the Whittier Narrows Test Basin, the following general functional design calculations are indicated. If a wastewater reclamation plant can produce and sell 20 million gallons per day (MGD) or 61.4 acre-feet per day, and if field measurements indicate that infiltration rates are 4 acre-feet per acre per day (4 ft per day) when a basin is flooded to a depth of say 2 feet, then under steady-state conditions 30.7 acres are required for recharge from the 20 MGD plant. This calculation assumes no breakdown of spreading facilities and no maintenance requirements.

Since the reclaimed water has an oxygen demand, continuous flooding or saturating the soil pores with water will ultimately result in the reclaimed water becoming anaerobic in the soil system. To prevent this occurrence, the water must be reaerated in the soil. How much reaeration depends on the long-term or total oxygen demand of the reclaimed water. Assume the treatment plant effluent to be well-stabilized, having a COD of about 50 mg/l, and about 22 mg/l of ammonia and organic nitrogen. The total oxygen demand for such a water would be about 150 mg/l. Knowing the oxygen demand of the reclaimed water and the daily discharge of the treatment plant allows one to calculate the total flux in oxygen demand of the water. This is the demand that must be satisfied by the spreading operation.

To be conservative, the rate of spreading will be calculated on the assumption that the oxygen flux into the ground to reaerate the percolating waters comes only from gas diffusion, i.e., dissolved oxygen in the surface water and hydrodynamic air flow induced by the filling and draining of the soil pores will be neglected. A 20-MGD plant effluent with a 150 mg/l total oxygen demand represents a total oxygen demand of 25,000 lb oxygen per day. In section 11.03 it was calculated that, into a soil body with a 20-percent effective porosity and initially void of oxygen in its soil atmosphere, about 110 gm oxygen per m^2 (980 lb oxygen per acre) would diffuse in one-half day from the atmosphere above. To handle the oxygen demand of the reclaimed water by the diffusion process with a one-half day reaeration requires that the water be infiltrated into the ground over sufficient area so that in the steady-state the water takes no more than 12 hours to seep into the ground and spread itself as a thin film throughout the zone of aeration. The remaining 12 hours in each day is the time for soil reaeration. On this basis, the 20 MGD plan with its 25,000 lb of oxygen demand per day requires an area of soil surface of about 25 surface acres in order to have the oxygen supplied in 12-hour intervals at 980 lb oxygen per acre. This calculation assumes that half the plant discharge is spread over a 12.5-acre plot in a 12-hour period and then this plot is free of ponded water for 12 hours when it is reaerated. A second 12.5-acre plot is operated out of phase in the wet-dry cycle to handle the other half of the plant discharge. Hydraulically, this system requires that each 12.5-acre plot accept half the plant flow, i.e., 10 MG or 30.8 acre-feet, in a 12-hour period for an average infiltration rate of about 2.56 feet per day. Again no allowance is considered for basin maintenance. The system can be operated to achieve better aeration if the length of the dry cycle is increased. The hydrodynamic flow of air into the ground is enhanced by frequent draining and drying of the spreading basins, and the division of a spreading area into smaller modules that can be flooded in some alternating sequence should further improve the operation.

Up to this point the functional design was based on knowledge of the total oxygen demand of the water to be spread and managing the spreading operation so that during the time that a basin is free of ponded water the flux of oxygen into the ground by diffusion will match the oxygen demand of the water applied during the flooding cycle. This condition would necessitate that all oxygen diffusing into the ground be mixed with the applied water and that the biological processes proceed at a rate rapid enough to be completed before the percolating waters reached the zone of saturation. Not a great deal is known about the rate of stabilization of wastewaters in soil systems, but by referring back to Chapter 6 of this report it can be seen that the operation of the Whittier Narrows Test Basin satisfies the above design criteria quite well, and yet a complete stabilization of the percolate was not achieved. It is felt that the diffusion model is a conservative one, and therefore the rate of stabilization of the percolate must be limited by the rate of degradation of the unstabilized substances by the soil organisms.

The test basins operated in this study improved with age and permitted more frequent dosing at heavier loads as time progressed. The primary operating criterion was to supply an extensive reaeration period after each flooding. Secondly it was found that a given load received greater stabilization if applied in several small doses intermittently rather than in one continuous flooding. These are the tenets of stabilization of waters by intermittent spreading through the zone of aeration.

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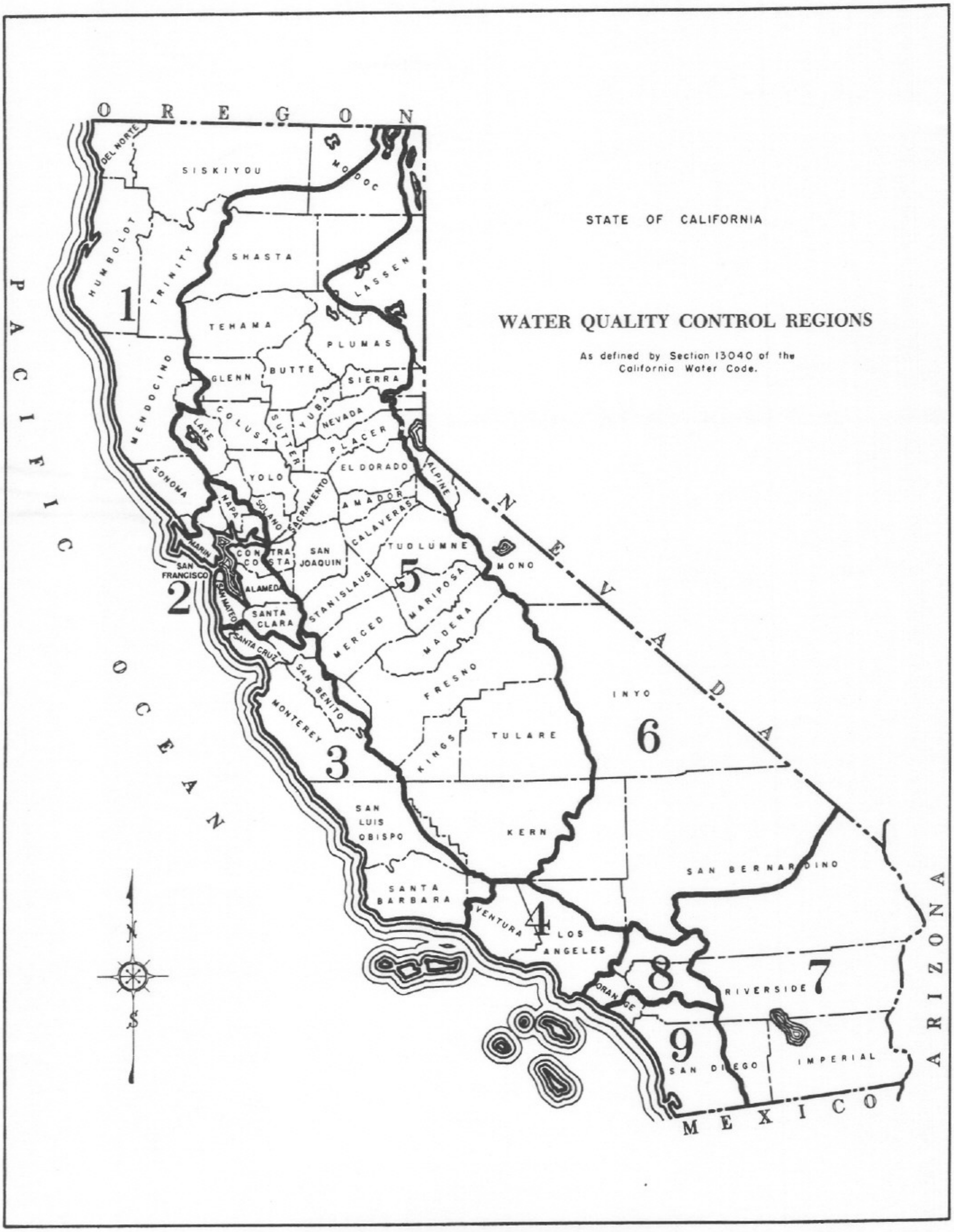

STATE OF CALIFORNIA

WATER QUALITY CONTROL REGIONS

As defined by Section 13040 of the California Water Code.

STATE OF CALIFORNIA

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