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Nitrate-Nitrogen in Effluent from Agricultural Tile Drains in California

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Subsurface drainage water of agricultural lands collected in tile lines may contain nitrate-nitrogen which serves to pollute surface waters. The objective of the research was to determine quantities and concentrations of nitrate-nitrogen in tile drainage and their relationship to soil, fertilizer input, crop, and time with respect to water applicaiton. The first research phase consisted of extensive monitoring of tile drainage effluents collected in important agricultural areas in California. More intensive investigation of nitrate-nitrogen within a few soil profiles was done during the second phase. Nitrate-nitrogen concentrations were not well-correlated with any measured parameter. The amount of nitrate-nitrogen contained in tile effluent was quite well-correlated with total water discharge and fertilizer nitrogen application. Higher amounts of nitrate-nitrogen were removed from coarse-textured soil profiles as compared to profiles containing a layer of fine-textured material. Some alluvial soils on the west side of the San Joaquin Valley apparently contain significant native nitrogen which contributes both to high nitrate-nitrogen concentrations and total mass emissions in tile effluent. Implications of the research findings to nitrogen and water management to achieve a desired water quality standard are discussed. It appears that the most positive approach to controlling nitrate-nitrogen in irrigated agriculture is through control of leaching.

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INTRODUCTION

THERE ARE APPROXIMATELY 52 million ha of artificially drained cropland in the United States (U. S. Department of Agriculture, 1973). The primary purpose of drainage systems in irrigated agriculture in semi-arid zones is to facilitate leaching salts from the soil profile. In high precipitation areas, drainage systems are often required to lower the water table and allow adequate soil aeration for crop production.

Since nitrate is soluble in water and mobile in soil profiles, it can be carried through the soil and discharged with tile effluent. The tile line must, however, be in a saturated zone before drainage occurs. It is conceivable that denitrification, which requires anaerobic conditions and an energy source for organisms, occurs as water moves through the saturated zone to the tile line.

Meek, Grass, and Mackenzie (1969) monitored tile effluent from land planted to cotton in Imperial County, California. The nitrate-nitrogen concentrations ranged from 0.48 to 2.06 ppm. The amount of nitrogen removed as nitrate by the tile system represented approximately 1.5 percent of the nitrogen added to the field. Johnston *et al.* (1965) measured nitrate-nitrogen concentrations of tile effluent under various crops in the San Joaquin Valley of California. Individual sample concentrations ranged from 2 to 440 ppm. Carter, Bondurant, and Robbins (1971) found nitrate-nitrogen concentrations of 3.01 to 3.79 ppm in tile-well complexes in Idaho. Bolton, Aylesworth, and Hore (1970) measured nutrient losses through tile drains under three cropping systems and two fertility levels on a Brookston clay soil in Canada. The highest nitrate-nitrogen concentration was 14.0 ppm measured in a fertilized corn plot, and the lowest was 1.1 ppm on a fertilized bluegrass sod plot. Erickson and Ellis (1971) analyzed tile effluent from drainage systems in Michigan. Nitrate-nitrogen concentrations ranged from 0.2 to 11.1 ppm. The highest measured loss was 20 percent of the nitrogen applied as fertilizer during 1 year. Nitrate-nitrogen movement into drainage lines under different soil management systems in Florida was investigated by Calvert and Phung (1971) and Calvert (1975). The nitrate-nitrogen concentrations were highly variable, ranging from 1 to 152 ppm. Highest concentrations were found in the late fall, when nitrogen fertilizer was applied in the fall, and there was sufficient precipitation to cause tile discharge. The nitrate-nitrogen concentrations varied greatly from year to year, depending upon fertilizer application.

A diversity of nitrate-nitrogen concentrations in tile effluents has been reported by the above-mentioned investi-

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gators. This is not surprising, because the systems investigated included a variety of soils, crops, fertilizers, and drainage designs. In some instances, water was added almost entirely by irrigation; in others, the water was from precipitation.

The amount of nitrogen fertilizer applied is considered to be a factor which would influence the nitrate-nitrogen concentration in tile effluent. Some reports have indicated that increased nitrogen fertilizer application is related to increased concentration of nitrate in tile drainage effluent (Erickson and Ellis, 1971; Glandon and Beck, 1971; Johnston *et al.*, 1965; Jones and Zwerman, 1972; Meixner, 1965). Research by Jones (1972) indicated that the quantity of nitrogen applied, and the tile discharge volume, accounted for 82 percent of the variability in loss of nitrates from tile drainage. However, investigations by Glandon and Beck (1969) in the central area of the San Joaquin Valley of California indicate that in this area there is no significant correlation between amount of nitrogen fertilizer applied and nitrate level in the tile drainage. A significant factor is that, in certain areas of the central San Joaquin Valley, nitrogen concentrations in the soil are exceedingly high to a depth of about 15 m. Mackenzie and Viets (1974) cautioned against jumping to the conclusion that nitrate-nitrogen concentrations in drainage waters are completely related to fertilizer. They stated "A frequent error in the literature is to relate the losses to percentages of fertilizer applied. Although fertilization may have increased the loss, the conclusion cannot be drawn that the nitrogen lost came from the fertilizer, or that stopping fertilizer use would stop loss."

Soil profile characteristics can influence nitrate-nitrogen loss through drainage water. Factors which influence soil aeration, pH, redox potential, organic matter content, and temperature affect nitrogen transformation processes in the soil. Kolenbrander (1969) reported higher losses of nitrogen in sandy soils, as compared to the heavier soils, at the same drainage water production. Gambrell, Gilliam, and Weed (1975*a*, 1975*b*) reported evidence that denitrification was occurring in poorly drained soils in the North Carolina coastal plain. They reported a higher amount of nitrate-nitrogen moving into drainage waters from a well-drained soil profile as compared to a poorly drained profile.

Meek *et al.* (1969) found that nitrate-nitrogen concentrations in tile effluent from a field in the Imperial Valley of California were extremely low (1.2 ppm), even though 240 kg/ha of nitrogen were applied. They suggested that the low nitrate concentrations were due to denitrification taking place in the soil solution near the water table. They further reported that decreases in nitrate concentration were associated with increases in soil depth near the water table. Research by Mackenzie and Viets (1974) on nitrate distribution in soil profiles, also in the Imperial Valley, indicated that sharp decreases in the nitrate concentrations were associated with the drop in redox potential at depths greater than 107 cm. The decreases in concentration were not due to plant uptake, because plant roots were restricted to the top 107 cm of soil. Thus, denitrification was implied.

Glandon and Beck (1969, 1971) found that some alluvial soils in the central area of the San Joaquin Valley of California were extremely high in residual nitrates. They also stated that, in general, the finer-textured alluvial profiles in irrigated areas contain higher concentrations of nitrogen in the soil than do coarser-textured soils. They implied that the quantity of drainage water from a given area depends on the interrelationships of geographic position, soil stratigraphy, and texture, as well as irrigation. Nitrate-nitrogen concentrations in the tile drainage effluent were more dependent on the particular soil series and geographic position of the tile systems, than on agricultural practices.

Irrigation practices, particularly as related to the amount of water leached through the soil, are important in determining nitrate losses with drainage water. Bolton, Aylesworth, and Hore (1970) believe that the quantity of water that flows through soil is the main determinant of nutrient loss. In comparing losses from fertilized corn to losses from fertilized bluegrass sod, they found a difference in total annual effluent flows of 15.6 and 6.5 cm. Since drainage flow varied more than nutrient concentration throughout the year, large nutrient losses resulted in seasons when large amounts of drainage flow occurred.

This paper presents the results of research conducted as part of the project "Nitrates in Effluents from Irrigated Lands" by scientists from the Berkeley, Davis, and Riverside campuses of the University of California. The "Tile Drainage" subproject is reported in its entirety in this publication.

The objectives of the tile drainage subproject were to determine quantities and concentrations of nitrate-nitrogen in tile drainage and their relationship to soil, crop, fertilizer inputs, and time with respect to water applications. The research was done in two phases. The first consisted of extensive monitoring of tile drainage effluents collected in important agricultural areas in California. The second phase consisted of choosing fewer sites and making more intensive studies on nitrate-nitrogen within the soil profile. The remainder of this report is divided into three main sections. The first two contain the Phase 1 and Phase 2 research, respectively. The third considers implications of the research findings on nitrogen and water management to achieve a desired water quality standard.

SURVEY OF NITRATE-NITROGEN IN TILE EFFLUENTS

Experimental Procedures

Tile drainage effluents were collected from commercial farming operations in the Imperial, Coachella, Ventura-Oxnard, San Joaquin, and Salinas valleys of California. Most of the tile systems on irrigated farm lands in California lie within these areas. Criteria used for selecting sites to be monitored were 1) the effluent sample drained from a relatively well-defined area, 2) the drainage area contained only one or similar soil series, 3) reasonably easy access to the collection site and tile line, and 4) owner permission to monitor the site. Within these criteria, 19 sites were selected in the San Joaquin Valley, 7 in the Salinas Valley, 11 in the Coachella Valley, 10 in the Imperial Valley and 8 in the Ventura-Oxnard area.

Questionnaires and personal interviews were used to obtain land-use history for the previous 5 years. The history was not obtained in some cases because of land ownership transfer, poor recordkeeping by the owner, or both. A questionnaire was sent out monthly requesting information concerning fertilizer applications and irrigations during the monitoring period.

Tile effluent samples were usually collected from each tile outflow weekly for a period of 1 year. Nitrate-nitrogen concentrations were measured with the Orion Specific Ion Meter (Model 401). To reduce the interference of complexing anions, each sample was diluted on a 1:1 ratio with 0.05 M aluminum sulfate containing a background of 20 ppm nitrate-nitrogen. Samples collected in the southern part of the state were placed in an ice chest and transferred to the laboratory for measurements.

Samples collected in the central part of the state were measured within 2 to 3 hours after sampling. Preliminary investigations indicated that no significant change in nitrate-nitrogen concentration was occurring from the time of sample collection to measurement.

Values obtained with the Orion meter and the Technicon Auto-Analyzer II were compared half way through the year. No significant difference in values was noted except for some samples from the Imperial Valley. In these samples, the Orion meter indicated a greater concentration of nitrate-nitrogen than did the Technicon Auto-Analyzer. This was particularly true for samples in which the nitrate-nitrogen concentration was less than about 20 ppm; the chloride concentration was greater than about 50 meq/liter; and the EC was as high as 25 mmho/cm. All samples from the Imperial Valley were analyzed by the Technicon Auto-Analyzer for the last half of the year. Results measured by the Orion meter during the first half year were adjusted by multiplying them by the ratio of the Technicon Auto-Analyzer to Orion meter results measured on samples taken from the same site.

The tile flow rate was measured, when possible, by collecting water in a bucket for a given period of time and measuring the collected volume. Measurement was not possible at all sites because of inaccessibility of the tile line, extremely large flow rates, or both.

A soil profile, representative of the soils within each drainage site, was sampled and characterized to a depth of 3 m. After air-drying and crushing the soils to pass a 2-mm sieve, the samples were analyzed for particle size distribution by the pipette method, and for NH_4^+–N and NO_3^-–N by steam distillation.

Tile effluent from sites located in the Coachella, Imperial, and Ventura-Oxnard areas was analyzed for soluble organic carbon content on a monthly basis. The samples were analyzed with a Beckman Total Carbon Analyzer.

Results

Nitrate-nitrogen concentrations

Each monitoring site was identified by a symbol which consisted of a letter, or group of letters, and a number to provide owner anonymity. The letters represent the general site location area, and the numbers identify different sites within the area. The Coachella Valley was symbolized by C, the Imperial Valley by I, the Salinas Valley by SAL, and the Ventura-Oxnard area by V. The sites located in the San Joaquin Valley were subdivided based on county, except for the Tulare Lake Basin, which was symbolized by TLB. Sites located in Fresno County were symbolized by F, in Merced County by MER, in Stanislaus County by ST, and in San Joaquin County by SJ.

The nitrate-nitrogen concentrations were plotted as a function of time for each site. Data from several sites are presented in Figs. 1 through 4. The times and amounts of nitrogen application are also indicated in each figure.

The results from the Coachella and Imperial valleys are represented by the data in Fig. 1. The nitrate-nitrogen concentrations remained relatively constant for different sampling times for most of the sites in the Coachella Valley. These results are typified by the data from sites C-4 and C-8, illustrated in Fig. 1. Data from site C-11 were illustrated in Fig. 1 to show the greatest change in nitrate-nitrogen concentration measured at different times. Results from sites C-4 and C-8 closely represent the typical results observed on other sites except for C-9.

A summary of the results for each of the sites in the Coachella Valley is

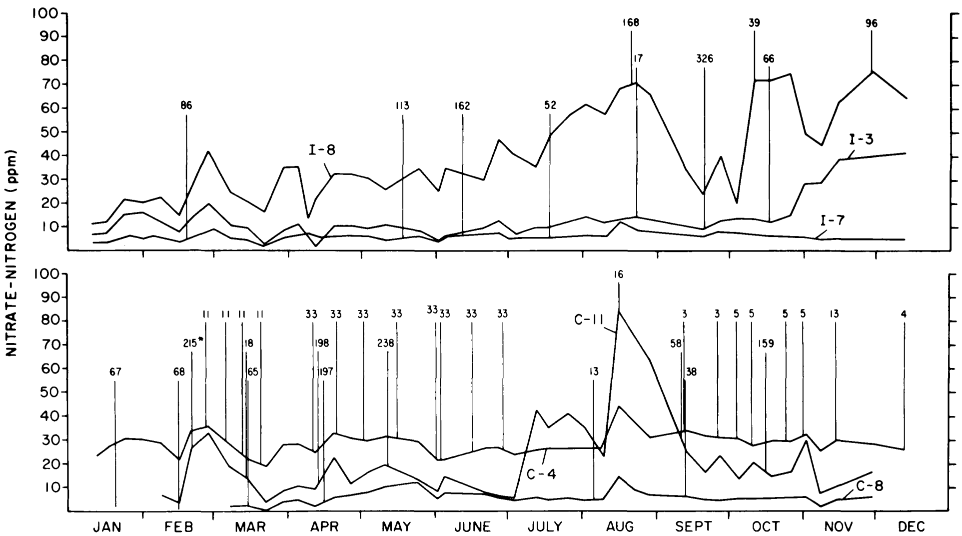

Fig. 1. Nitrate-nitrogen concentration in tile effluent as a function of time for typical sites in the Coachella and Imperial valleys. Numbers indicate the amount (kg/ha) of fertilizer nitrogen applied at each site.

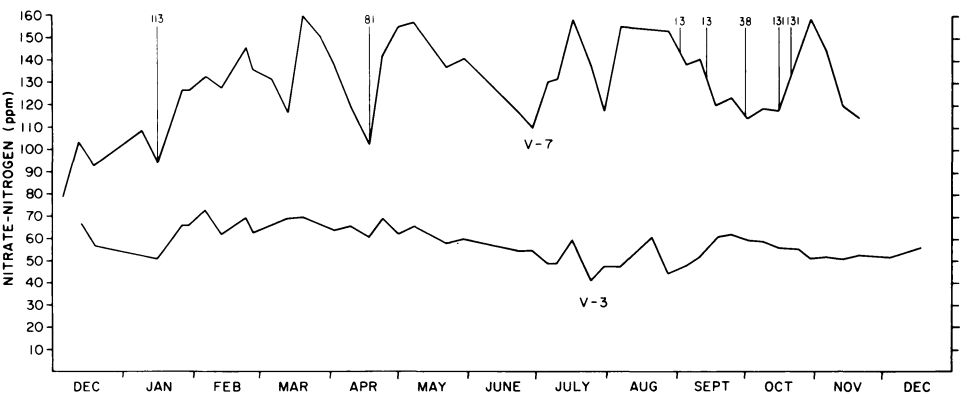

Fig. 2. Nitrate-nitrogen concentration in tile effluent as a function of time for typical sites in the Ventura-Oxnard area. Numbers indicate amount (kg/ha) of fertilizer nitrogen applied at each site. No data were provided on fertilizer input on site V-2.

presented in Table 1. The average nitrate-nitrogen represents the numerical average of all samples collected for the given site. The nitrate-nitrogen range lists the lowest and the highest values measured during the year. The highest nitrate-nitrogen concentration measured in the Coachella Valley was 84 ppm at both C-9 and C-11. The highest average concentration was 56 ppm measured in a date orchard. Other data in Table 1 will be discussed later.

Data from sites I-3, I-7, and I-8 are illustrated in Fig. 1 to indicate results measured in the Imperial Valley. Data from site I-3 represent one of two cases in which a change in nitrate-nitrogen in the tile effluent could be directly associated with the timing of fertilizer application. Alfalfa was the crop at site I-3 during most of the year, and the average nitrate-nitrogen concentration was about 5 ppm during that part of the year. Alfalfa had been grown for at

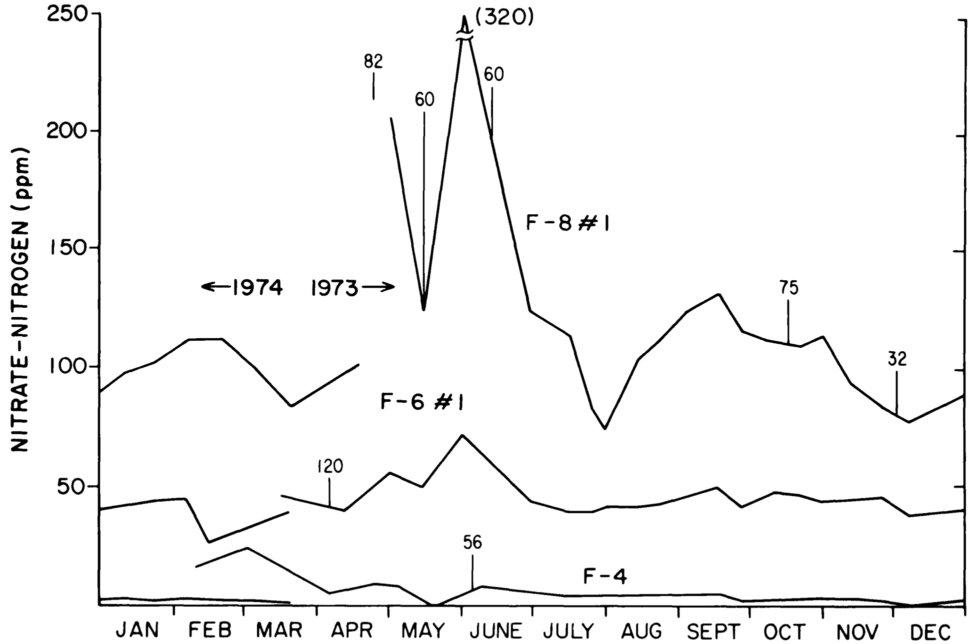

Fig. 3. Nitrate-nitrogen concentration in tile effluent as a function of time for typical sites in Fresno County. Numbers indicate amount (kg/ha) of fertilizer nitrogen applied at each site.

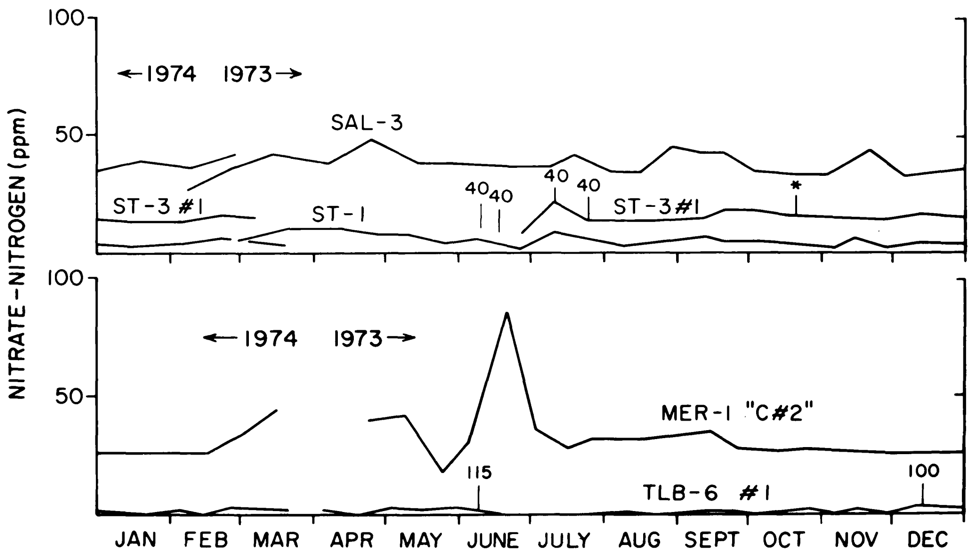

Fig. 4. Nitrate-nitrogen concentration in tile effluent as a function of time for typical sites in the Salinas and San Joaquin valleys. Numbers indicate amount (kg/ha) of fertilizer nitrogen applied at each site. No fertilizer input information was supplied on sites SAL-3 and MER-1. The asterisk on site ST-3 indicates that 90 tons of cow manure/ha were added to one section of the field, and 45 tons of chicken manure/ha were added to another part of that field during October and November.

TABLE 1

NITROGEN AND EFFLUENT DATA FOR SITES IN COACHELLA VALLEY

Site	Soil series	Crop	Avg NO_3^--N	NO_3^--N range	Total N applied	N applied minus manure N	Total NO_3^--N in effluent	Total effluent discharge
			ppm	*ppm*	*kg/ha*	*kg/ha*	*kg/ha/yr*	*ha cm/ha*
C-1	Coachella	Grapefruit	9	2–24	318	31	—	—
C-2	Coachella	Grapefruit	5	Tr.–9	0	0	50	92
C-3	Coachella	Grapes	15	5–28	200	50	—	—
C-4	Indio	Vegetables	28	19–44	350	346	151	53
C-7	Indio	Vegetables	22	13–33	234	234	120	50
C-8	Coachella	Lemons	3	2–6	448	448	—	—
C-9	Coachella-Gilman	Dates	56	14–84	241	241	—	—
C-10	Indio	Rye grass	14	5–29	632	632	—	—
C-11	Gilman	Vegetables	19	3–84	900	685	152	80
C-12	Gilman	Tangerines	7	Tr.–13	327	327	—	—

least 3 years with no nitrogen fertilization prior to our sampling. The soil contained less than 5 percent clay throughout the profile. During August and September of the monitoring year, the field was plowed and planted to lettuce. Thirty-three tons of steer manure per ha were applied in August. Amounts of 17, 326, and 66 kg of commercial fertilizer nitrogen per ha were applied in August, September, and October, respectively. Nitrate-nitrogen concentration rose to 25 ppm on the sampling date following the first irrigation after fertilization, and continued to rise to a high value of 52 ppm on subsequent samplings. The significant increase in concentration in the tile effluent can be directly related to the change in crop and addition of nitrogen fertilizer.

Alfalfa was the crop on site I-8 until the beginning of August, when the crop was plowed. Sugar beets were planted in October. Unfortunately, because of change in property ownership, no information was available on cropping and fertilizer history prior to our sampling dates in January. Nitrogen fertilizer was applied in August, October, and December. The nitrate-nitrogen concentrations varied considerably between sampling dates. The concentrations decreased significantly in the latter part of September and again in the early part of November. The decreases were associated with a measured increase in effluent flow rate, and the increased flow was associated with irrigation, which may have caused dilution of the tile effluent.

The data represented in Fig. 1 for site I-7 are typical of the changes in nitrate-nitrogen concentrations among different sampling periods for other sites in the Imperial Valley. On all sites other than those previously discussed, there was no observed change in nitrate-nitrogen concentration which could be directly associated with timing of nitrogen fertilization.

TABLE 2

NITROGEN AND EFFLUENT DATA FOR SITES IN IMPERIAL VALLEY

Site	Soil series	Crop	Avg NO_3-N	NO_3^--N range	Total N applied	N applied minus manure N	Total NO_3-N in effluent	Total effluent discharge
			ppm	*ppm*	*kg/ha*	*kg/ha*	*kg/ha/yr*	*ha cm/ha*
I-2	Imperial	Alfalfa	9	5–12	44	44	4	5
I-3	Coachella	Alfalfa, lettuce	23	5–44	410	402	122	82
I-4	Gilman	Alfalfa	4	3.5–6.6	0	0	30	57
I-5	Holtville	Wheat, alfalfa	13	4–27	149	149	—	—
I-6	Imperial	Wheat, alfalfa	17	15–20	292	292	28	11
I-7	Imperial	Cotton, wheat	7	3–19	654	654	6	5
I-8	Imperial	Alfalfa, sugar beets	45	28–59	302	302	18	2
I-9	Meloland	Sugar beets	28	26–36	112	112	—	—
I-11	Niland	Wheat	18	11–30	203	203	26	8

The average and range of nitrate-nitrogen concentrations measured in the Imperial Valley are presented in Table 2, along with other data which will be discussed later. The highest concentration was 59 ppm, and the highest average nitrate-nitrogen concentration was 45 ppm, both measured on site I-8 where alfalfa was converted to sugar beets.

Results from the Ventura-Oxnard area are illustrated in Fig. 2. Sites V-3 and V-7 were chosen to represent the extremes in variability of nitrate-nitrogen concentrations with sampling dates. (Fertilizer application information was not provided for site V-3). The concentration in site V-3 remained relatively constant, while there were relatively large fluctuations at site V-7. Fluctuations at other sites were intermediate between these two. A variety of vegetables was grown on all sites in this area. In no case was there a significant trend in nitrate-nitrogen concentration with respect to time. This could be expected because the fertilizer applications tend to be applied at consistent intervals throughout the year. The average and range of nitrate-nitrogen concentrations in the Ventura-Oxnard area are presented in Table 3. The highest, individual concentration was 187 ppm, and the highest average concentration was 161 ppm. In general, the concentrations in the Ventura-Oxnard area were considerably higher than those in the Coachella and Imperial valleys.

Examples of data collected in Fresno County are presented in Fig. 3. Data from site F-8 No. 1 represent the greatest fluctuations in nitrate-nitrogen concentrations measured. They also represent some of the highest values observed. Data from site F-6 No. 1 are presented to illustrate the least variability in nitrate-nitrogen concentrations measured at various times in Fresno County. Most of the systems had fluctuations in concentrations at different times intermediate between those of sites F-6 and F-8.

The nitrate-nitrogen concentrations at site F-4, illustrated in Fig. 3, gradually decreased from the first monitoring period in February 1973, to the last monitoring date in March 1974. The crop was converted from barley to alfalfa in the spring of 1973. Although there was some nitrogen applied soon after the alfalfa was seeded, no further nitrogen was applied to the field. The gradual decline in concentration in the tile effluent may have resulted from lack of fertilizer input and removal of nitrogen by the alfalfa crop.

Summary data on all sites in Fresno County are presented in Table 4. Nitrate-nitrogen concentrations ranged from trace amounts to as high as 320 ppm, and average values ranged from 2 to 196 ppm.

TABLE 3

NITROGEN AND EFFLUENT DATA FOR SITES IN VENTURA-OXNARD AREA

Site	Soil series	Crop	Avg NO_3^-N	NO_3^--N range	Total N applied	N applied minus manure N
			ppm	*ppm*	*kg/ha*	*kg/ha*
V-1	Hueneme	Vegetables	149	46–180	394	287
V-2	Hueneme	Vegetables	73	52–91	310	310
V-3	Pico	Vegetables	59	41–73	—	—
V-4	Hueneme	Vegetables	39	25–104	395	288
V-5	Hueneme	Vegetables	93	61–114	657	421
V-6	Pico	Vegetables	63	11–84	589	376
V-7	Anacapa	Vegetables	126	78–159	519	519
			161	131–187		

Data from other areas of California are illustrated in Fig. 4. The results presented for SAL-3 are typical of those observed in the Salinas Valley. (Data on fertilizer input were not provided for SAL-3, but were probably similar to fertilization in other areas in the Salinas Valley, where about 500 kg nitrogen/ha are applied in several applications during the year.) The concentrations remained constant at all sampling dates throughout the monitoring year. Note that the concentrations measured in February 1974 were very close to the values obtained in February 1973. Likewise, the nitrate-nitrogen concentrations in Stanislaus County did not fluctuate greatly with the differing monitoring times. Only one site was monitored in San Joaquin County, and data from that site are not illustrated in the figure. However, the concentrations remained quite constant, ranging between 10 and 22 ppm, as noted from the data presented in Table 5.

Except for SAL-2, which had an average nitrate-nitrogen concentration of 65 ppm, all sites in the Salinas Valley had average concentrations in the 30–50 ppm range. These concentrations were, on the average, lower than those measured in the Ventura-Oxnard area, which is also a vegetable-growing area. The average nitrate-nitrogen concentrations measured in the few sites in Stanislaus and San Joaquin counties were relatively low, with the highest average value being 15 ppm. The highest individual sample was 22 ppm.

Nitrate-nitrogen concentrations in the Tulare Lake Basin were generally very low, as indicated in Table 6. The highest average concentration was 13 ppm, and the highest individual sample concentration was 36 ppm. Since all values were low, the fluctuation in concentration with time was not great. Data for TLB-6, presented in Fig. 4, illustrate typical results for the Tulare Lake Basin area.

Table 4

NITROGEN AND EFFLUENT DATA FOR SITES IN FRESNO COUNTY

Site	Soil series	Crop	Avg NO_3^--N	NO_3^--N range	Total N applied	N applied minus manure N	Total NO_3^--N in effluent	Total effluent discharge
			ppm	*ppm*	*kg/ha*	*kg/ha*	*kg/ha/yr*	*ha cm/ha*
F-1	Oxalis	Alfalfa, tomato	73	40–100	118	118	118	16
F-2	Oxalis							
	Panoche	Cotton, alfalfa	85	58–132	—	—	—	—
F-3	Oxalis	Alfalfa	2	<1–9	0	0	10	36
F-4	Oxalis	Barley, alfalfa	5	<1–24	63	63	11	12
F-6#1	Kettleman	Cotton	45	27–72	151	151	350	77
F-6#2	Kettleman	Safflower, cotton	101	62–126	—	—	100	10
F-7	Oxalis	Tomato, alfalfa, cotton	59	38–94	560	560	—	—
F-8#1	Oxalis	Cotton	114	75–320	150	150	146	12
F8-#2	Oxalis	Melons	196	104–234	—	—	172	9
F-9	Oxalis	Cotton, sugar beets	31	20–60	132	132	46	14

Table 5

NITROGEN AND EFFLUENT DATA FOR SITES IN SALINAS VALLEY, STANISLAUS COUNTY, AND SAN JOAQUIN COUNTY

Site	Soil series	Crop	Avg NO_3^--N	NO_3^--N range	Total N applied	N applied minus manure N	Total NO_3^--N in effluent	Total discharge
			ppm	*ppm*	*kg/ha*	*kg/ha*	*kg/ha/yr*	*ha cm/ha*
SAL-1	Pacheco	Lettuce, celery	30	24–44	637	637	383	134
SAL-2	Cropley-Salinas	Lettuce, celery	65	56–84	480	480	930	106
SAL-3	Clear Lake-Salinas	Lettuce, rye grass	38	27–48	—	—	—	—
SAL-4	Clear Lake	Celery	47	44–56	728	728	277	59
SAL-5	Clear Lake	Lettuce	43	16–60	—	—	—	—
SAL-6A	Clear Lake	Lettuce	42	30–56	580	580	138	32
SAL-6B	Clear Lake	Lettuce	40	26–54	580	580	103	25
ST-1	Myers	Alfalfa	5	2–10	0	0	38	66
ST-2	Delhi	Peaches	10	6–15	296	296	—	—
ST-3#1	Columbia	Corn, Sudan grass	15	9–22	717	160	336	221
ST-3#2	Columbia	Corn, Sudan grass	15	11–21	717	717		
SJ-1	Delhi	Almonds	15	10–22	633	633	—	—

TABLE 6

NITROGEN AND EFFLUENT DATA FOR SITES IN MERCED COUNTY AND TULARE LAKE BASIN

Site	Soil series	Crop	Avg NO_3-N	NO_3-N range	Total N applied	N applied minus manure N	Total NO_3-N in effluent	Total effluent discharge
			ppm	*ppm*	*kg/ha*	*kg/ha*	*kg/ha/yr*	*ha cm/ha*
MER-1A	Panoche	None	44	22–56	0	0	214*	49*
MER-1B	Panoche	Cotton	28	22–46	168	168	90*	33*
MER-1C#1	Panoche	Cotton	31	20–65	168	168	46*	15*
MER-1C#2	Panoche	Cotton	34	18–86	168	168	46*	16*
MER-1D	Panoche	Alfalfa	83	40–108	0	0	67	15
MER-1E	Panoche	Cotton	67	14–84	168	168	—	—
TLB-1	Tulare	Wheat	11	2–36	134	134	7	6
TLB-2	Metz	Cotton	13	3–24	280	280	17	20
TLB-3	Tulare	Wheat, cotton	5	1–17	168	168	—	—
TLB-5	Hacienda	None	4	1–8	0	0	—	—
TLB-6#1	Pacheco	Cotton	1	<1–3	145	145	4	27
TLB-6#3	Pacheco	Cotton	1	<1–7	292	292	3	17

* Minimum values because of excessive flow rates.

Several different tile lines were monitored on the same farm in Merced County. The range and average nitrate-nitrogen concentrations are presented in Table 6 for all sites, and the nitrate-nitrogen concentrations as functions of time for MER-1 C No. 2 are illustrated in Fig. 4. This curve represents typical results measured on the different tile systems in Merced County. (Fertilizer application information was not provided.)

In summary, a variety of conditions were observed. There were cases where the nitrate-nitrogen concentrations remained quite constant at all sampling periods. In other systems, there was considerable variability from date to date. Except for two cases, there was no obvious change in nitrate-nitrogen concentration at any given time which could be related to the time of fertilizer application. In general, the concentration measured on the last sampling date in 1974 was similar to that of the first sample collected and measured in the spring of 1973. Unless there are drastic changes in cropping or fertilizer practices, the values reported are probably typical of those that would be expected in the different areas.

Amount of nitrate-nitrogen loss in tile effluent

The amount of nitrate-nitrogen removed with the tile effluent over the 1-year monitoring period is presented, for the various sites, in Tables 1 through 6. The amount of nitrate-nitrogen lost could only be calculated for those systems in which the rate of effluent discharge could be measured.

Because both the concentration and effluent discharge rate varied with monitoring dates, the following procedure was used to calculate total mass emission. The measured concentration and the discharge rate at a given date were assumed to have remained constant for a period halfway to the previous and following monitoring dates. The calculated amounts for each monitoring period were then totaled to cover 1 year.

The results for Coachella Valley are presented in Table 1. Effluent discharge could not be measured in several of the systems. The range of nitrate-nitrogen loss was 50 to 150 kg/ha.

The results for Imperial Valley are presented in Table 2. The amount of nitrate-nitrogen lost varied from 4 to 122 kg/ha, with only one site greater than 30 kg/ha. The highest nitrogen loss was at site I-3, where the crop was changed from alfalfa to lettuce during the monitoring year. Extrapolation of the data collected when alfalfa was the crop for a full year would have resulted in a loss of nitrate-nitrogen of 38 kg/ha per year. Extrapolation of the data collected with lettuce as the crop for a full year period resulted in a calculated 281 kg/ha per year nitrate-nitrogen loss.

No data on nitrate-nitrogen loss through tile effluent were available for the Ventura-Oxnard area, because the land area drained for a given monitored tile line could not be accurately determined. The results from Fresno County are presented in Table 4. Ten to 350 kg/ha of nitrate-nitrogen were lost through the tile effluent at the various sites in the county. Except for site F-4 where the crop was changed from barley to alfalfa, the effluent from sites which have more than one crop listed was from tile lines located under land areas which had more than one crop.

Data from the Salinas Valley, Stanislaus County, and San Joaquin County are presented in Table 5. The nitrate-nitrogen loss in the Salinas Valley ranged from 103 to 930 kg/ha (the calculated value of 930 is suspicious, since it represents twice the amount of fertilizer applied. No explanation for this extraordinarily high value is available.) Mass emission data are available for two sites in Stanislaus County. On a site where alfalfa was the crop, the loss was 38 kg/ha. In contrast, 336 kg/ha were lost in tile effluent coming from land which was cropped to corn and Sudan grass.

The data for the Tulare Lake Basin and Merced County are presented in Table 6. Very low mass emission (less than 17 kg/ha) was observed in the Tulare Lake Basin. Accurate values for the sites monitored in Merced County were not possible because of excessive flow rates. The numbers reported in Table 6 represent minimum values, because we know the flow rate was greater than the values reported in the table.

In summary, there was a wide variation in the amount of nitrate-nitrogen lost through tile effluents from various systems monitored. Factors which may have affected the amount of loss will be discussed later. In comparing the results from the various areas in California, one notes the extremely low mass emissions in the Tulare Lake Basin. In general, the highest mass emissions were in the Salinas Valley, where the crop consisted of vegetables in each case. Comparatively low mass emissions were observed in the Imperial Valley.

Factors affecting nitrate-nitrogen in tile effluent

One of the factors expected to affect the level of nitrate-nitrogen in tile effluent would be nitrogen application. The average concentrations in tile effluent for a given site are plotted in Fig. 5 as a function of the amount of nitrogen applied for the 1-year monitoring period. Classification of the soil series listed in Fig. 5 is presented in Table 7. Data from sites which received manure during the year are not included in Fig. 5, nor in other analyses to be reported later. The reason for excluding such sites was the uncertainty concerning the amount of mineralization of nitrogen which occurred during the year from the manure. Study of the data in Fig. 5 indicates that there was no significant correlation between fertilizer application and nitrate-nitrogen concentration

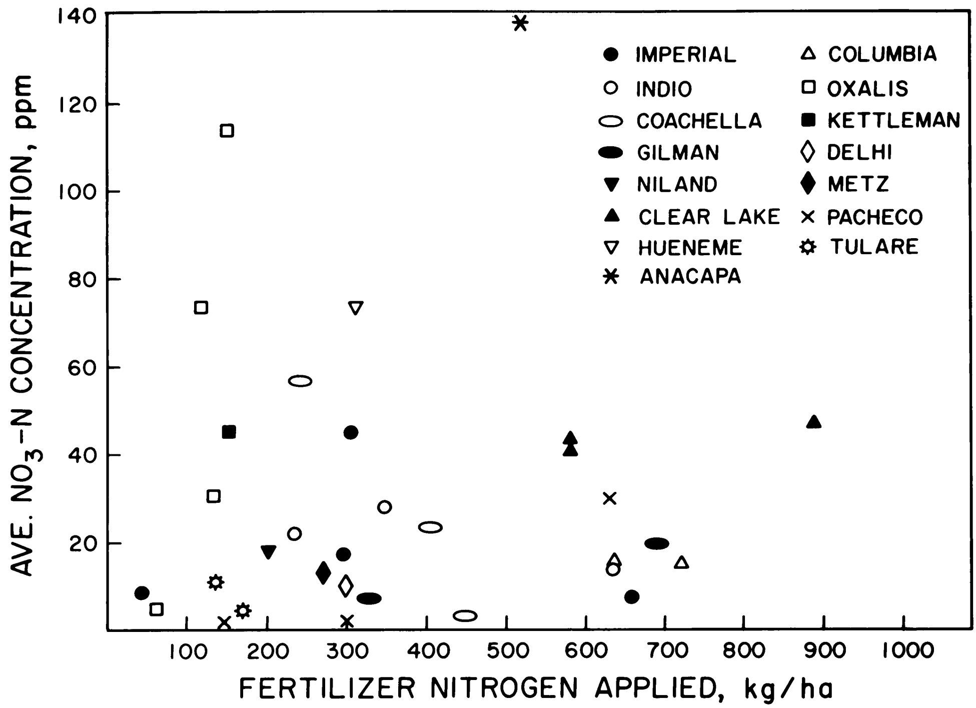

Fig. 5. Average nitrate-nitrogen concentration measured in tile effluent from sites located on various soil series and receiving various amounts of fertilizer nitrogen during the monitored year.

in the tile effluent in the 1 year. Indeed, a linear regression analysis resulted in the equation $C = 29.4 - 0.0007N$, where C is the average nitrate-nitrogen concentration and N is the applied fertilizer nitrogen in kg/ha per year. Neither $F = 0.001$ nor $r = 0.006$ was statistically significant. The concentration would be expected to be dependent on such factors as leaching fraction, denitrification, crop removal, native nitrogen in the profile, previous nitrogen applications, and nitrogen applications during the year. The data in Fig. 5 indicate that

TABLE 7

CLASSIFICATION OF SOIL SERIES DESCRIBED AT THE STUDY AREAS

Soil series	Family	Subgroup
Anacapa	coarse-loamy, mixed, thermic	Calcic Pachic Haploxerolls
Clear Lake	fine, montmorillonitic, thermic	Typic Pelloxererts
Coachella	sandy, mixed, hyperthermic	Typic Torrifluvents
Columbia	coarse-loamy, mixed, nonacid, thermic	Aquic Xerofluvents
Delhi	mixed thermic	Typic Xeropsamments
Gilman	coarse-loamy, mixed (calcareous), hyperthermic	Typic Torrifluvents
Hueneme	coarse-loamy, mixed (calcareous), thermic	Aquic Xerofluvents
Imperial	fine, montmorillonitic (calcareous), hyperthermic	Vertic Torrifluvents
Indio	coarse-silty, mixed (calcareous), hyperthermic	Typic Torrifluvents
Kettleman	fine-loamy, mixed (calcareous), thermic	Typic Torriorthents
Metz	sandy, mixed, thermic	Typic Xerofluvents
Niland	sandy over clayey, mixed (calcareous), hyperthermic	Typic Torrifluvents
Oxalis	fine, montmorillonitic, thermic	Vertic Xerochrepts
Pacheco	fine-loamy, mixed, thermic	Fluvaquentic Haploxerolls
Pico	coarse-loamy, mixed, thermic	Calcic Entic Haploxerolls
Tulare	fine, montmorillonitic (calcareous), thermic	Typic Haplaquents

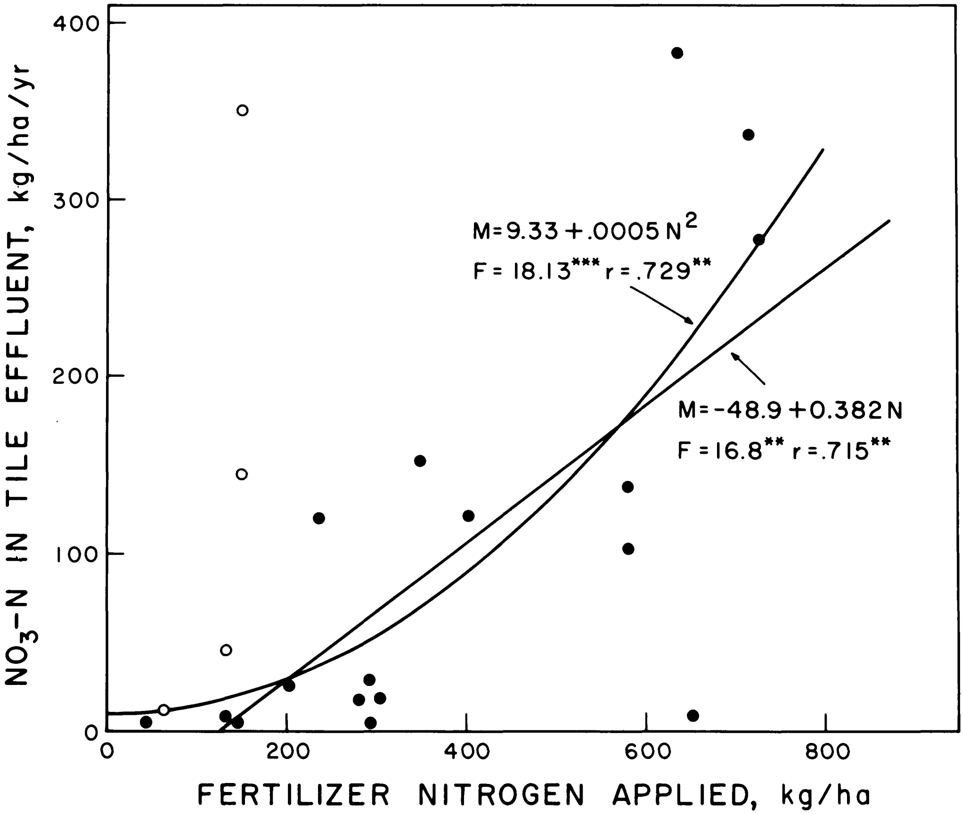

Fig. 6. The amount of nitrate-nitrogen removed with tile effluent as a function of fertilizer nitrogen applied during the monitored year. Open circles represent sites on the west side of the San Joaquin Valley suspected of containing large amounts of "native" nitrogen.

one or more of these other factors has a significant effect on the nitrate-nitrogen concentration.

The sites located on Oxalis and Kettleman soil series were on the west side of the San Joaquin Valley. Glandon and Beck (1969) reported that these soils apparently contain considerable amounts of native nitrogen in the soil profile. Note on Fig. 5 that, with one exception, the concentration for a given fertilizer application on these soil series is considerably greater than on other soil series. Thus, our data support the previously reported assertion of Glandon and Beck that considerable native nitrogen apparently exists in the alluvial soils on the west side of the San Joaquin Valley. Assuming that the native nitrogen in these soil profiles significantly affected the nitrate-nitrogen concentration, a linear regression analysis was conducted on the data, neglecting data points from Oxalis and Kettleman soil series. The resultant equation was $C = 7.92 + 0.036N$, with $F = 5.93^*$ and $r = 0.532^*$ (*, **, and *** will be used to denote significance at the 5, 1, and 0.1 percent level, respectively). Elimination of those sites suspected of having native nitrogen in the soil profile resulted in a better relationship between effluent concentration and applied fertilizer. The linear relation, however, still was not highly significant.

The relationship between nitrogen applied during the year and mass emissions of nitrate-nitrogen was investi-

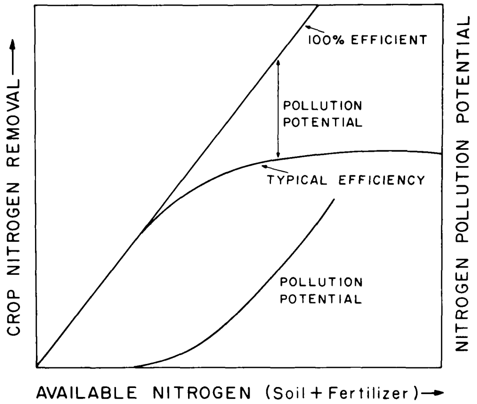

Fig. 7. Theoretical relationship between amount of nitrogen removed with the crop and available nitrogen in the soil. The difference between a 100% efficient system and typical efficiency represents the nitrogen pollution potential, which is plotted as a function of available nitrogen in the soil. (Curves depict general and not quantitative relationships.)

gated. The data are plotted in Fig. 6. The curves were calculated omitting the data from sites suspected of having native nitrogen in the soil profile. Using the general relationship $M = a + bN^c$ where $c = 1,2,3$, or 4, the best fit equation was $M = 9.33 + 0.0005N^2$, where M is mass emission of nitrate-nitrogen in kg/ha per year. The linear regression curve is also presented in Fig. 6 for comparison. Note that the F and r values indicate a much better relationship between mass emission and fertilizer application than was found for nitrate-nitrogen concentration and fertilizer application. When data points from the sites containing native nitrogen were included in the analysis, the best fit equation was $M = 66.1 + 4 \times 10^{-8}$ N, with $F = 5.22$ * and $r = 0.46$*.

The fact that a curvilinear regression described the data better than a linear regression is not surprising. Fig. 7 illustrates the theoretical relationship between nitrogen in the soil and nitrogen removal by crop. The difference between input and removal is available for leaching, and thus for movement into tile drainage. With low applications of nitrogen (depending upon the initial soil nitrogen content) most of the fertilizer is taken up by the crop. As the amount of nitrogen increases, a point is reached where further uptake by the plant decreases, and thus the potential for groundwater pollution increases. Shape of the theoretical relationship between nitrogen input and groundwater pollution approximates the best fit curvilinear curve presented in Fig. 6.

The amount of water leached through the profile and discharged in a tile line could influence both concentration and amount of nitrate-nitrogen in the tile effluent. The relationship between the average nitrate-nitrogen concentration and effluent volume is presented in Fig. 8. Those sites suspected of containing native nitrate-nitrogen are identified separately from other sites. Those sites that received no fertilizer and the crop was alfalfa, are also identified in Fig. 8. There is no apparent significant correlation between the average nitrate-nitrogen concentration and effluent volume. A linear regression analysis between the concentration and effluent volume, using data from all sites, resulted in $C = 44.9 - 0.182W$, with $F = 0.42$ and $r = 0.14$, where W refers to amount of effluent in ha cm/ha. A linear

Fig. 8. Average nitrate-nitrogen concentration in the tile effluent for a given site as a function of the amount of tile effluent for the monitoring year. Open circles represent data from sites in the San Joaquin Valley suspected of having high native nitrogen, and triangles represent sites which had alfalfa and no fertilizer application.

regression analysis using only data points from sites other than those receiving no nitrogen, or coming from sites containing native nitrogen, resulted in $C = 17.6 + 0.119W$, with $F = 1.38$ and $r = 0.29$. In neither case was the F value significant, and the correlation coefficient was extremely low in both instances. One ordinarily expects the average nitrate-nitrogen concentration to decrease with increased effluent volume, because of the dilution effect. The data in Fig. 8 indicate that the expected behavior may have been masked by factors other than effluent volume, which affect nitrate-nitrogen concentration.

Sites from the west side of the San Joaquin Valley had higher nitrate-nitrogen concentrations for a given amount of effluent than did other sites, providing added evidence for the existence of native nitrogen in the profiles. Significantly, as would be expected, those sites which had alfalfa with no fertilizer application had low nitrate-nitrogen concentrations, compared to other sites.

The amount of nitrate-nitrogen discharged during the given year is plotted in Fig. 9 as a function of amount of water discharged. Again, sites suspected of being unusual because of native nitrogen, lack of nitrogen input, or both, are identified separately from the other data points. Linear regression analysis using data exclusive of those from the suspected sites, resulted in an equation $M = -4.52 + 2.66W$, with $F = 55.0^{***}$ and $r = 0.89^{**}$. Inclusion of the data points in the analysis from the sites expected to have native nitrogen resulted in the linear regression equation $M = 22.0 + 2.58W$, with $F = 37.8^{***}$ and $r = 0.80^{**}$.

The amount of nitrate-nitrogen removed from a field is definitely related to the amount of tile effluent. Again, the data in Fig. 9 illustrate that the sites on the west side of the San Joaquin

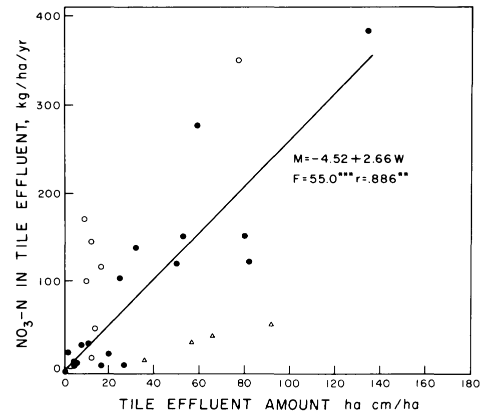

Fig. 9. Amount of nitrate-nitrogen removed with the tile effluent as a function of the amount of tile effluent during the year monitored. Open circles represent data from sites in the San Joaquin Valley suspected of having high native nitrogen. Triangles are for sites with alfalfa and no fertilizer application. Linear regression analysis is for data represented by the solid circles only.

Valley had much higher nitrate discharge, as compared to the other sites, when compared on an equal water discharge basis. Furthermore, those sites which received no nitrogen fertilizer also had lower loss of nitrate-nitrogen for a given effluent discharge.

The amount and concentration of nitrate-nitrogen discharge would be expected to be related to both the amount of nitrogen fertilizer applied and the amount of effluent discharge. The computer was programmed to select the equation which would best fit the data points from the basic equation:

$$C = a + bW + cN + dW^2 + eN^2 + fW^2N + gN^2W + hN^2W^2 + iNW$$

where symbols are as previously defined. The same equation was used for mass emission of nitrate-nitrogen, M. The analysis was conducted by both including and excluding data points from the west side of the San Joaquin Valley, suspected of having high native nitrogen. With all data points used, the best fit equation for concentration was $C = 28.1 + 0.30\ W$, with $F = 0.03$ and $r =$

0.039. When the data points from the west side of the San Joaquin Valley were eliminated, the best fit equation was $C = 7.92 + 0.36\ N$, with $F = 5.93^{*}$ and $r = 0.532^{*}$, as discussed previously.

When all data points were used, the best fit equation for mass emission was $M = 7.72 + 2.76\ W$, with $F = 47.8^{***}$ and $r = 0.840^{***}$. Elimination of the data points from the west side of the San Joaquin Valley resulted in the best fit equation $M = 16.4 + 0.0042\ NW$, with $F = 82.9^{***}$ and $r = 0.920^{***}$. As was found previously, there was a good relationship between the amount of nitrate-nitrogen removed through tile effluent and the drainage volume. When the sites with native nitrogen were eliminated from the analysis, the best overall relationship was with the product of the amount of water drained and the amount of fertilizer applied.

Previous studies in California have shown that textural changes in profiles can have significant effects on nitrate-nitrogen loss below root zones (Lund, Adriano, and Pratt, 1974; Pratt, Jones, and Hunsaker, 1972). High clay-containing layers could reduce the leaching fraction from an area, and thus reduce nitrate-nitrogen loss by leaching. Reduced water flow under a given fertilizer input could cause either an increase or a decrease in the concentration of nitrate-nitrogen in the tile effluent. If nitrate-nitrogen were a conservative element, as is chloride, a decrease in leaching fraction would result in an increased nitrate-nitrogen concentration in the effluent. However, since nitrate-nitrogen is subject to denitrification, the concentration of nitrate-nitrogen in the effluent may be reduced by restricted water flow. Denitrification takes place under anaerobic conditions, and profile characteristics which inhibit water transmission can also cause anaerobic conditions.

In Fig. 10, the amount of nitrate-nitrogen lost per year for a number of sites is plotted as a function of the highest percent of clay found in a horizon above the tile line (excluding the 0 to 25 cm layer). Data from the Oxalis and Kettleman soil series are not included in Fig. 10 because of the postulated high native nitrogen in the soil profile, as previously discussed. Also, data from the Salinas Valley, where vegetable crops were exclusively grown, are not included in Fig. 10. The amount of nitrate-nitrogen lost from these sites at a given clay content was higher than the curve shown in Fig. 10. The higher loss from the Salinas Valley could be accounted for by a prolonged history of high fertilizer application, and the relatively high water application used in growing vegetables.

Excluding the above discrepancies, the data in Fig. 10 indicate that the amount of nitrate-nitrogen removed in tile effluent from field crops is closely associated with the highest amount of clay found in the profile. It is not possible to determine whether the clay horizon primarily restricts water flow or causes denitrification. However, based on a previous study (Lund *et al.*, 1974) both mechanisms are likely involved.

Significantly, all available data from sites monitored in the Coachella and Imperial valleys, Tulare Lake Basin, and Stanislaus areas are included in Fig. 10. These sites represent a variety of crops and fertilizer applications. Soil profile characteristics obviously have a significant effect on the amount of nitrate-nitrogen which may be lost through the tile system. Soils containing high-clay layers would be expected to have less nitrogen loss by leaching, as compared to soil profiles without these layers.

It has mistakenly been assumed on occasion that tile effluent which has a low concentration of nitrate-nitrogen results in low mass emissions of nitrate-nitrogen, and vice versa. This assumption can not necessarily be made, if one considers a few specific instances from

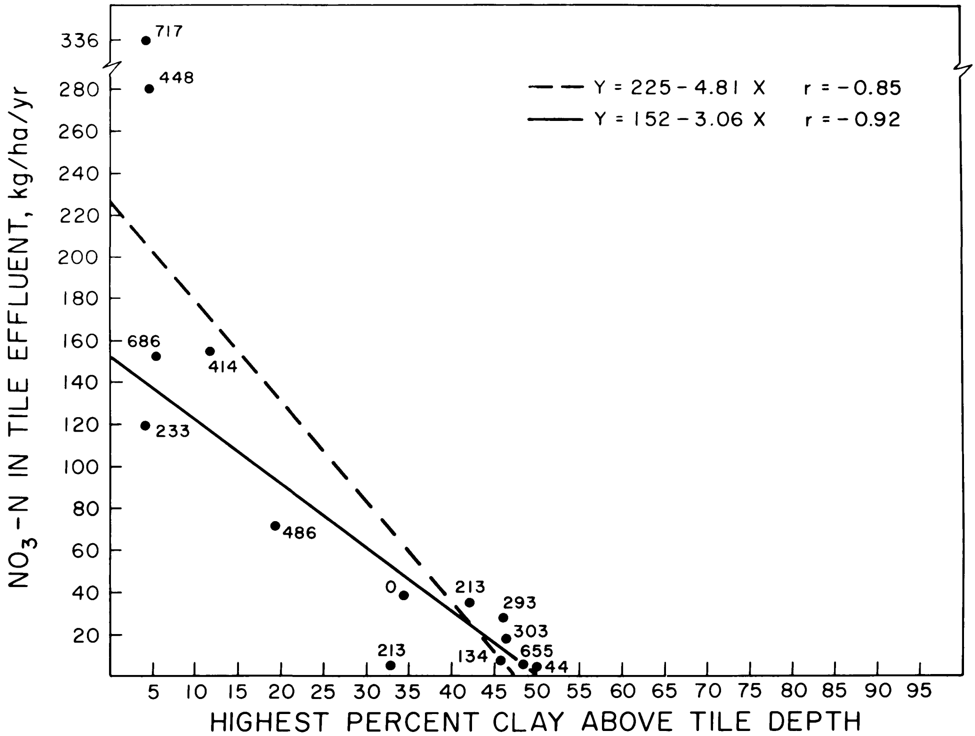

Fig. 10. The amount of nitrate-nitrogen removed through tile systems placed in soil profiles containing various amounts of clay. The dashed and solid lines were calculated using all data points and neglecting two highest amounts of loss, respectively. Numbers refer to the amount of fertilizer nitrogen applied (kg/ha) during the monitoring year.

our study. The greatest contrast can be observed by comparing site ST-3 with site I-8. In site ST-3, the average nitrate-nitrogen concentration was 15 ppm, and the mass emission was 336 kg/ha per year. On the other hand, the concentration at site I-8 was 45 ppm, with a resultant mass emission of only 18 kg/ha per year. The average concentration on site I-8 was three times greater than that on site ST-3. However, the mass emission from site I-8 was only 5 percent the amount from site ST-3. This difference is a direct result of different amounts of effluent volume from the two sites. Indeed, conversion of concentration to mass emission is directly related to effluent volume. Other relationships between the average nitrate-nitrogen concentration and mass emission in the tile effluents for our sites are illustrated in Fig. 11. Data collected from sites where the soil was predominantly sandy throughout were differentiated from sites where clay or silty clay layers occurred within the profile. Note that for a given concentration, there was always higher mass emission from sandy soils as compared to the clay-layered soils. This result is not too surprising, because the water transmission coefficients of sandy soils generally are higher than for soils containing clay or silty clay layers. However, the tile effluent volume is not only dependent upon transmission properties of soil, but also on water application by the grower. The data in Fig. 11 indicate that soil profile characteristics have a much greater effect

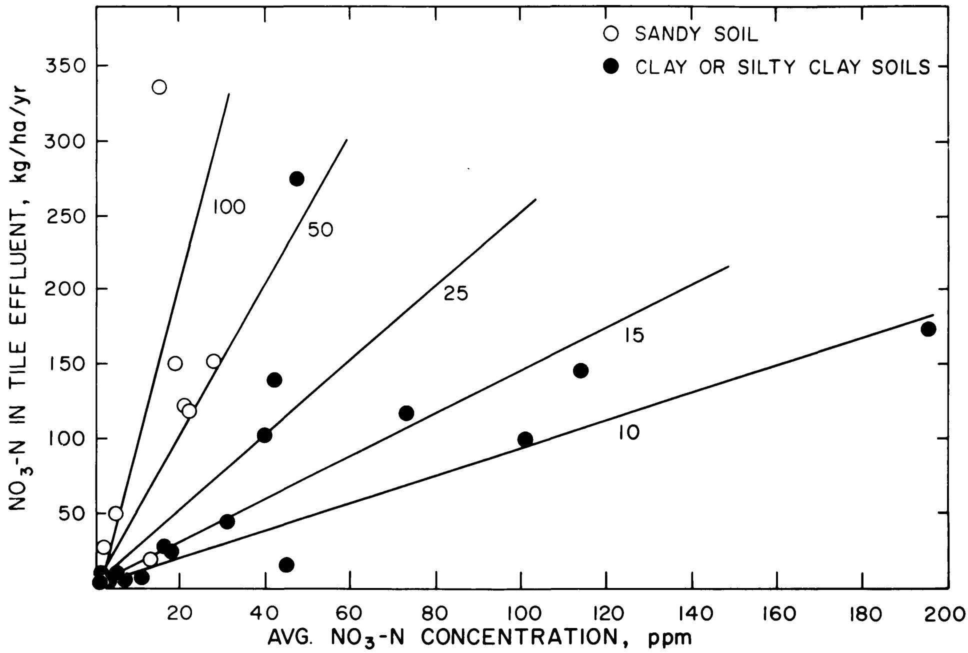

Fig. 11. Relationship between amount of nitrate-nitrogen removed with the tile effluent and average nitrate-nitrogen concentration in the tile effluent. The curves indicate the relationship between these two variables when the amount of tile effluent is equal to the number listed by each curve in ha cm/ha.

on the amount of water flowing through the profile than does the irrigator, under present practices in California. Significantly, these data are from several locations throughout California. The curves in Fig. 11 represent relationships between average concentration and mass emission expected for different amounts of tile effluent, in centimeters of water. In almost every case where the soil was sandy, 50 or more cm of water were leached through the profile. Some of our sites had 10 or fewer cm of water lost through the tile effluent.

We have previously reported the significant relationship between mass emission of nitrate-nitrogen and amount of water discharged from the tile system. We also reported an inverse relationship between mass emission of nitrate-nitrogen and clay content in the soil profile. The data in Fig. 11 indicate that less water flowed through soils which have clay or silty clay layers, as compared to the sandy soils. Clay layers in a soil can reduce mass emission by reducing water flow through the profile. This does not negate the effect of the clay layer in causing denitrification. The data collected during the survey did not allow an analysis of denitrification.

The effect of crop on nitrate-nitrogen loss in tile effluent appears to be most closely associated with the amount of fertilizer applied to the crop. Undoubtedly, factors, such as plant uptake and removal with the harvested product, influence the nitrogen balance in the soil. Our data, however, were not sufficiently refined to detect these differences. Rather consistently where alfalfa was the crop receiving no fertilizer, both the concentration and amount of nitrate-nitrogen removed in the tile were low compared to other crops. On the other hand, vegetables which received relatively high amounts

TABLE 8
SOLUBLE ORGANIC CARBON IN THE TILE EFFLUENT

Site	Avg soluble organic carbon	Soluble organic carbon range
	ppm	*ppm*
Ventura-Oxnard:		
V-1	13.0	4.5–16.9
V-2	17.8	12.9–23.7
V-3 #1	15.0	8.1–24.1
V-3 #2	12.1	2.5–18.3
V-4	23.0	9.0–40.6
V-5	21.1	15.0–35.9
V-6	13.6	3.0–31.1
V-7 #1	17.8	8.4–31.2
V-7 #2	15.9	6.0–28.6
Avg	16.6	
Coachella Valley:		
C-1	14.9	4.2– 31.6
C-2	16.3	8.2– 22.3
C-3	12.8	2.2– 25.2
C-4	10.8	1.9– 16.3
C-7	16.9	Tr. – 37.5
C-8	12.3	Tr. – 21.0
C-9	46.0	Tr. –100.0
C-10	10.6	Tr. – 18.6
C-11	15.9	Tr. – 28.7
C-12	10.4	.9– 17.3
Avg	16.8	
Imperial Valley:		
I-2	28.5	19.2–42.5
I-3	23.3	10.8–38.0
I-4	26.4	8.0–54.7
I-5	18.4	5.6–31.4
I-6	33.6	13.9–56.1
I-7	29.3	15.7–49.7
I-8	39.8	12.8–61.6
I-9	20.3	9.1–31.0
I-11	22.9	8.5–34.6
I-12	27.9	15.2–38.3
Avg	27.1	

of nitrogen fertilizer had tile effluent which was quite high in mass emission of nitrate-nitrogen, and rather consistently high in nitrate concentrations.

Soluble organic carbon in the tile effluent

The soluble organic carbon data are summarized in Table 8. The average soluble organic carbon concentration was highest in the Imperial Valley, at 27 ppm. Both the Coachella Valley and Oxnard-Ventura area averaged about 17 ppm. The highest soluble organic carbon concentrations were found in a coarse-textured soil under a date grove (site C-9). It was not ascertained whether these high concentrations were due to plowing under of date fronds each year, or to some other factor.

Fluctuations in soluble organic carbon concentrations occurred at each of the sites on a monthly basis. The concentrations ranged from trace amounts to a high of over 100 ppm. These changes in concentrations were not related to changes in tile effluent volume but were more closely related to seasonal changes. Soluble organic carbon concentrations were lowest during the summer months of July, August, and September, and highest during the

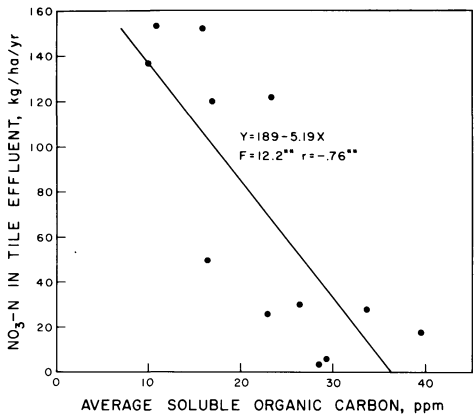

Fig. 12. Amount of nitrate-nitrogen removed with the tile effluent, plotted against the average soluble organic carbon measured on various sites in the Coachella and Imperial valleys.

months of April, May, and June. Thus, changes in biological activity induced by changes in soil temperature appear to be possible factors in influencing measured soluble organic carbon concentrations in the tile effluent.

Because no estimates of denitrification were possible from our data, it was impossible to determine whether there is a relationship between soluble organic carbon and denitrification. In Fig. 12, the average soluble organic carbon concentration is shown to be related to the total amount of nitrate-nitrogen in the tile effluent. These data are from the Imperial and Coachella valleys. There was an inverse relationship between average soluble organic carbon concentration in the tile effluent and the quantity of nitrate-nitrogen removed in the effluent.

Solute travel time to tile drains

Estimates on travel time for a solute to reach the tile from various parts of the field are important in understanding relationships between time of nitrogen application and its appearance in tile effluent. Obviously, travel time for nitrogen placed immediately over a tile line would be considerably less than nitrogen placed midpoint between two tile lines. Jury (1975*a*, 1975*b*) presented a theoretical approach to estimating solute travel time for tile-drained fields, and compared theoretical calculations to measured results. Factors that influence travel time are spacing between tile lines, depth of tile line, depth to an impermeable soil zone, soil porosity, and mean discharge rate for the time of study. Increases in drain spacing, depth of tile, depth to an impermeable zone, and soil porosity all contribute to greater travel times for a solute to the tile drain. Increasing the discharge rate decreases the travel time. For typical tile spacing, depth, and discharge rates in our study, Dr. Jury calculated that travel time for solute placement midway between tile lines would be in terms of years. Thus, any sample collected from the tile line represents nitrogen which may have been applied at the surface at several times in history.

Consideration of solute travel time raises a question concerning comparisons between mass emission and concentration, and fertilizer applied during the year, as was done earlier in this report. The data used in our analysis came only from those sites for which we had a 5-year record, and for which the fertilizer application was not greatly different over this period of time. Systems in which there was a significant departure in fertilizer practice over recent years were excluded from the analysis. Thus, our calculations and conclusions should be approximately correct.

In general, we reported no significant change in nitrate-nitrogen concentration in tile effluent that could be related to a time of fertilizer application to the surface. This observation can readily be explained on the basis of solute travel time. Parts of the pulse of nitrogen input at the surface reached the tile line at different times after application. The one exception was the observation of a significant increase in nitrate-nitrogen concentration in the tile effluent after a crop was changed from alfalfa to lettuce, and considerable nitrogen fertilizer was applied. This increase occurred because nitrogen reached the tile from zones over and adjacent to the tile line. This behavior was predicted by Jury (1975*a*, 1975*b*).

The time sequence between nitrogen application and appearance in tile effluent is expected to be different in the relatively deep, wide-spaced tile systems of the arid, irrigated regions, as compared to the humid regions. Data reported by Bolton *et al.* (1970), Erickson and Ellis (1971), Calvert and Phung (1971), Calvert (1975), and Jones and Zwerman (1972) are from systems with shallower and more closely-spaced tile lines than those in the present report. The above authors reported a closer association between nitrate concentrations measured in the tile and the time of fertilizer or water application or both, than we observed. Transfer of information from tile systems which are shallow and closely spaced to areas where tiles are placed deeper and more widely spaced, and vice versa, may lead to erroneous conclusions.

Conclusions

A few general conclusions can be drawn from the data accumulated during Phase One research:

1) Nitrate-nitrogen concentrations in the tile effluent are not well-correlated with any measured parameters such as fertilizer application rate, effluent volume, or soil profile properties.

2) The amount of nitrate-nitrogen lost through tile effluent was quite well-correlated with total water discharge, fertilizer application, and soil profile characteristics. Lowest amounts were associated with soil profiles which had layers of clay or silty clay. The best correlation was found between amount of nitrate-nitrogen loss, and fertilizer application times the amount of tile effluent volume.

3) The alluvial soils on the west side of the San Joaquin Valley contain significant native nitrogen which contributes both to high nitrate-nitrogen concentrations and total mass emissions in the tile effluent. Drainage water from these areas have higher nitrate-nitrogen than other areas receiving comparable nitrogen input and total drainage volume.

4) Quite low nitrate-nitrogen concentrations and mass emissions were observed for systems in which alfalfa was the crop, and no nitrogen fertilizer had been applied.

5) Soils which contain a clay or silty clay layer have lower total emission of nitrate-nitrogen in the tile effluent possibly because of decreased water flow through the profile, or contribution to denitrification. Highest losses of nitrate-nitrogen were on sandy profiles receiving relatively high quantities of nitrogen fertilizer.

6) Measurement of nitrate-nitrogen concentration in the tile effluent does not allow conclusions on the mass emission over a given period of time. Considerable variability in the amount of water flow for different tile systems makes this comparison meaningless. High mass emissions were measured for systems having low average concentrations, and vice versa.

NITRATE-NITROGEN MOVEMENT THROUGH SELECTED SOIL PROFILES

Experimental Procedures

Following the first year's survey of several tile drainage systems, six sites were selected for more intensive study during the second year. The sites selected were located in southern California, and were investigated during a 6-month period (June through October). Some sites selected had soils with differing profile characteristics. Uniform, coarse-textured profiles were represented by sites C-4, C-8, and C-9. A new site (referred to as I-10) was selected during the second year to represent a uniform, fine-textured profile. Sites I-9 and I-11 were selected for further investigation because these sites had soils with textural discontinuities; i.e., they contained sandy soil underlain by horizons high in clay. The average sand and clay contents of the upper part of the profile representative of site I-9 were 87.3 and 2.1 percent, respectively. Depth to the higher clay layer (30 to 40 percent clay) varied in the field, with the depth being 100 cm at the point of instrumentation. The soil in site I-11 had a mean clay content in the upper part of the profile of 1 percent, and a clay content of approximately 51 percent in the lower part. Depth to the contrasting layer was between 70 and 90 cm. At the point of instrumentation, the boundary was at 76 cm.

Porous ceramic cups were installed at the 61-, 91-, 122-, and 183-cm depths, midway between tile lines, to remove soil solution for analyses. Three cups were installed at each depth. Tensiometers were installed at depths of 61, 91, and 122 cm to measure soil suction and to calculate hydraulic gradients. Platinum electrodes, as described by Mann and Stolzy (1972) and Whisler, Lance, and Linberger (1974), were placed in the fields 8 weeks before termination of the experiment to measure redox potentials. Three electrodes were placed at the 91 and 183 cm depths in all sites except I-10, which received no measurements.

Water samples were usually collected on a weekly basis at all six sites. Samples taken from tile lines and ceramic cups were immediately refrigerated and brought to the laboratory for chemical analysis. Solutions were analyzed for nitrate-nitrogen with a Technicon AutoAnalyzer II, for manganese with a Perkin-Elmer atomic absorption spectrophotometer, for electrical conductivity with a standard electrical conductivity bridge, and for chloride with an Aminco chloride titrator.

Information on irrigation and nitrogen fertilizer inputs was received monthly from the growers.

Results

Detailed results of this study have been published by Devitt *et al.* (1976). Less detailed results will be reported in this publication to provide a complete report of the tile drainage subproject and to place the results of the second year's investigation in context with results from the first year's survey.

A summary of the amount of nitrogen applied, the amount of nitrate-nitrogen in the effluent, the average nitrate-nitrogen concentration of the tile effluent, the crop, and salinity of the tile effluent are presented in Table 9.

The results for sites C-4, C-8, and C-9, which represent the coarse-tex-

TABLE 9

CROPS, FERTILIZER NITROGEN APPLIED, AMOUNT OF NO_3^--N, AVERAGE NO_3^--N CONCENTRATION, AVERAGE E.C., AND AVERAGE CHLORIDE IN TILE EFFLUENT

Site	N applied		NO_3^--N in effluent		Average NO_3^--N		Average E.C.		Average Cl		Crop
	1973	1974	1973	1974	1973	1974	1973	1974	1973	1974	
	kg/ha/yr		*kg/ha/yr*		*ppm*		*mmhos/cm*		*meq/liter*		
C-4	345	396	151	155	28	29	3.00	3.28	9	8	corn, carrots
C-8	448	26	—	46*	3	4	2.41	2.30	6	6	lemons
C-9	241	149	—	62*	56	49	10.88	10.77	49	47	dates
I-9	112	492	—	71	28	21	8.78	5.60	47	26	cotton
I-10	—	224	—	119	—	92	—	42.87	—	454	milo
I-11	203	169	26	35	18	16	11.73	8.37	77	52	cotton

* Calculated using ratios of E.C. and water applied with site C-4 to determine volume of the effluent. Values of 42 and 36 for sites C-8 and C-9, respectively, were obtained if chloride concentrations were used instead of E.C.

tured profile throughout, will be discussed first. The redox potential was high at both the 91- and 183-cm depths for all three sites. The lowest average redox potential (393 mv) was measured at the 183-cm depth on site C-8. Meek *et al.* (1969) suggested that a redox potential lower than about 300 mv is required for nitrate reduction. Our redox potential results indicate that reducing conditions may not have existed in any of the coarse-textured profiles to a depth of 183 cm.

Manganese concentrations were measured as an indicator of reducing conditions. Parr (1969) stated that manganese is reduced in the redox potential range of 200 to 400 mv. The manganese concentration in the soil solution was less than 0.1 ppm throughout the profile for sites C-4 and C-9. The manganese concentration was less than 0.1 ppm at all depths except 183 cm, where the average concentration was 0.4 ppm at site C-8. Both the redox potential measurements and the manganese concentrations indicate that reducing conditions contributing to denitrification did not occur in any of the coarse-textured profiles to a depth of at least 183 cm.

The measured hydraulic gradient at site C-4 was always positive downward, indicating that the irrigation practices were continuously causing leaching. The hydraulic head gradients on site C-8 were always positive downward between depths of 91 and 122 cm, and positive downward 70 percent of the time between 61 and 91 cm. Thus, the irrigation practices on site C-8 were such to cause almost continual leaching through the profile. In contrast, the irrigation practices resulted in a measured upward hydraulic gradient during most of the monitoring period on site C-9. Such a gradient would result in minimal leaching during the observation period. The measured hydraulic head gradients are consistent with data on the EC and average chloride concentrations measured in tile effluents from the three sites. Both the chloride concentration and EC were considerably higher on site C-9 as compared to the other sites. These high values indicate a lower leaching fraction on site C-9.

The rate of tile discharge could not be measured on sites C-8 and C-9 because of the extremely high discharge rate on C-8, and the inaccessibility of the tile line on C-9. Calculations of volume outflow were made for both sites by using EC and water-applied ratios with site C-4. The calculated effluent volume was then used to calculate the amount of nitrate-nitrogen in the effluent for sites C-8 and C-9.

Results from C-8 are somewhat difficult to interpret because of the large difference in amount of nitrogen applied during the 2 years and evidence that fertilizer was not uniformly applied to the field. In comparing the results from site C-4 with those of C-9, it is evident that the average nitrate-nitrogen concentration in the tile effluent from site C-9 was approximately twice the concentration from site C-4. In contrast the total amount of nitrate-nitrogen in the effluent in site C-9 was approximately 40 percent of the amount removed from site C-4. These data clearly illustrate the importance of leaching fraction as it affects the concentration of nitrate-nitrogen and the amount of nitrate-nitrogen in the tile effluent. A low-leaching fraction, as on site C-9, results in relatively high concentrations but comparatively low amounts lost in the tile effluent, as compared to the higher-leaching fraction exemplified by site C-4. The very low nitrate-nitrogen concentrations measured at site C-8 can be explained by the very high-leaching fraction and the relatively large amount of water applied to the field. Despite the concentration at site C-8 being less than 10 percent of the concentration at site C-9, the amount of nitrate-nitrogen in the tile effluent was only slightly lower on site C-8 than on C-9. These data are consistent with the results reported earlier that the amount of water flowing through the profile is extremely important in determining how much nitrate-nitrogen is removed in the tile effluent. It is particularly significant in sandy profiles where there is a low denitrification potential and the applied nitrogen can be leached through the soil profile if not taken up by the crop in a reasonably short time after application.

Sites I-9, I-10, and I-11 had high clay contents in certain parts of the profile. Site I-10 was selected to be monitored because it had a soil profile with high clay content throughout. It was anticipated that this profile would represent one with high denitrification potential. Contrary to expectations, very high nitrate-nitrogen concentrations were found throughout the profile, and they remained relatively constant during the monitoring period. The concentrations at the 61-cm depth were approximately 60 ppm. The values at greater depths were in the range of 150 to 200 ppm. The average concentration in the effluent from tiles placed at approximately the 122-cm depth was 92 ppm. This field had just been brought into cultivation and irrigated, and the extremely high nitrate-nitrogen concentrations are believed to be the result of nitrogen stored in the profile, and not the result of fertilizer application. The very high EC and chloride readings reported in Table 9 indicate that very little previous leaching of this profile had occurred. This site illustrates the importance of "native" nitrogen within soil profiles as a source of nitrate in groundwaters. Addition of irrigation water apparently caused considerable mineralization of organic nitrogen, which contributed to the nitrate concentration in the water.

Redox measurements were not made on site I-10. The manganese concentrations at each depth were higher than on any other site at a comparable depth. The concentrations at the 150-cm depth were about 3.5 ppm. The high manganese concentrations are indicative of reducing conditions within the profile, as we had anticipated because of the clay content. A decrease in nitrate-nitrogen concentration due to denitrification, however, was not readily evident from our data. The concentrations at any depth remained relatively constant over the entire monitoring period. Concentration also tended to increase with depth, an indication that the lower concentrations near the surface resulted from downward movement due to leaching.

Site I-9 had the clay layer at a depth of approximately 100 cm. Extremely low nitrate-nitrogen concentrations were measured in samples extracted from the 183-cm depth. Concentrations in samples from the other depths ranged between 10 and 30 ppm. The manganese concentrations were less than 0.1 ppm through most of the monitoring period at the 61-, 91-, and 122-cm depths on this site. However, concentrations were higher (maximum of 0.45 ppm) at the 183-cm depth. The redox potential was approximately 500 mv at the 91-cm depth, and approximately −200 mv at the 183-cm depth, except at the end of the monitoring period when the redox potential increased after irrigation ceased and the profile started to dry. The redox potential, nitrate-nitrogen concentration, and manganese concentration data are consistent, and indicate that conditions for denitrification occurred at some depth between 122 and 183 cm.

Increase in clay content occurred in the profile from site I-11 at approximately 76 cm. Except for a relatively high value initially measured at the 61-cm depth, all nitrate-nitrogen concentrations were very low throughout the site I-11 profile. The manganese concentrations increased with increasing depths. Values at the 61-cm depth were less than 0.1 ppm throughout the monitoring period, and the highest measured value was 0.45 ppm at the 183-cm depth. The redox potentials were approximately −100 mv during most of the monitoring period at both the 91- and 183-cm depths. The water table fluctuated near the sand-clay interface, which could have created conditions for the resultant low nitrate-nitrogen concentrations, low redox potentials, and comparatively high manganese concentrations measured within the profile. Definite gleying was observed below the clay-sand interface.

The chloride to nitrate-nitrogen ratios were calculated for all six sites at all monitoring depths. Ratios were based on the average solution concentrations at each respective depth (Fig. 13). If one assumes that chloride and nitrate-nitrogen move with water in a similar fashion, and that neither reacts with the soil to any significant degree, the ratio should be fairly constant with depth beyond the root zone. However, if the chloride to nitrate-nitrogen ratio increases with depth below the zone of plant uptake, then denitrification is a reasonable assumption. The results presented in Fig. 13 indicate that a significant increase in the ratio with depth did occur in sites I-9 and I-11. The increase in ratio for site I-11 occurred at the 91-, 122-, and 183-cm depths. Since the profile was coarse-textured to approximately a depth of 76 cm, the data indicate that denitrification occurred in the profile after the clay content significantly increased. The chloride to nitrate-nitrogen ratio at site I-9 was high only at the 183-cm depth. This profile was coarse-textured to a depth of 100 cm, and gradually increased in clay content at greater depths. Again, the data indicate denitrification occurred in the lower parts of the profile where the clay content became relatively high.

The chloride to nitrate-nitrogen ratios remained relatively constant for all depths for the other sites monitored, indicating no specific zones of stratification in those profiles.

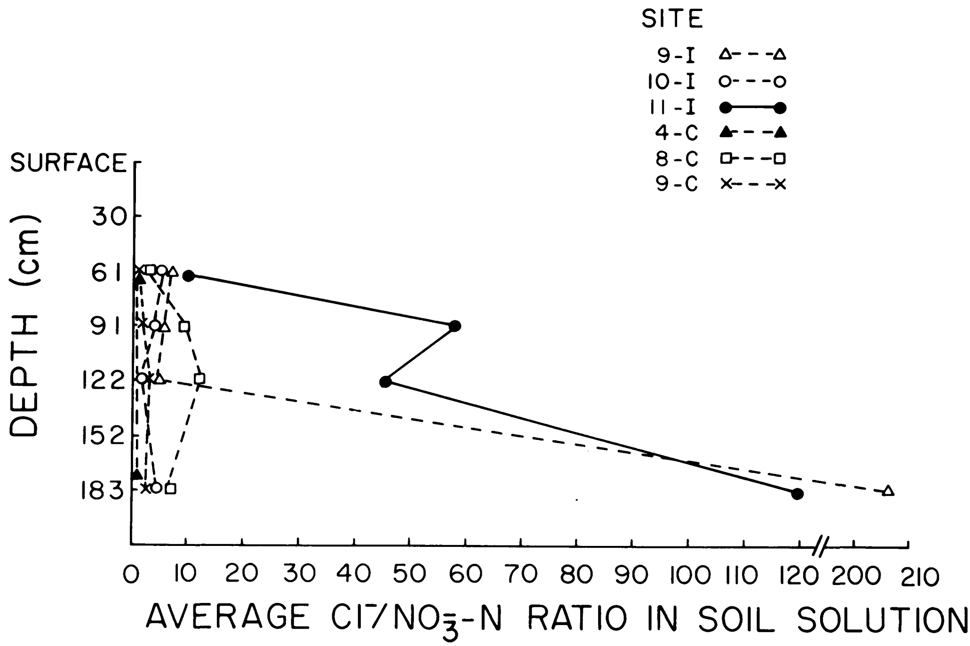

Fig 13. The chloride to nitrate-nitrogen ratio in soil solution as a function of depth for the various sites.

Summary

Our data indicate that soil profile characteristics significantly affect nitrate-nitrogen movement through soil. The coarse-textured profiles have high redox potentials throughout, with apparently very low denitrification potential. Nitrate in these profiles moves with the water as a direct consequence of irrigation practices. Irrigation, which creates downward hydraulic gradients and considerable leaching, will remove considerable amounts of nitrate from the soil. The concentration in the tile effluent is a consequence of the amount of fertilizer applied, fertilizer taken up by the plant, and the leaching fraction. A low-leaching fraction may result in relatively high nitrate-nitrogen concentration in the tile effluent, with a relatively low total loss as compared to higher-leaching fractions. The profiles with layers of high clay content had zones where the redox potential was low, and apparently denitrification occurred. Layers of clay restrict water movement, thus reducing the leaching fraction and promoting anaerobic conditions favorable for denitrification. Significantly, the changes in chloride to nitrate-nitrogen ratio observed in the soil profile were related to the position of the clay layer. Our results are in general agreement with those reported by Gambrell *et al.* (1975*a*, 1975*b*).

CONTROLLING NITRATES IN TILE EFFLUENTS: SOME CONSIDERATIONS

Nitrate can be considered a water pollutant under certain circumstances. Nitrogen in the water can contribute to eutrophication and stimulate vigorous biological activity when the activity is undesirable. High nitrate concentration in water is also considered to be a health hazard when given to infants. The present U.S. Public Health standard is 10 ppm nitrate-nitrogen. Nitrate-nitrogen concentrations in tile effluent quite commonly exceed this level. These and other factors have led to consideration of regulations related to nitrate in water supplies following agricultural use.

One approach would be to legally regulate the concentration of nitrate which can be discharged in the effluent. This approach has commonly been applied to municipalities and industries which must treat their water to lower the concentration of a given constituent to a designated level before discharge. Although this approach is reasonable, and has proved successful when applied to municipalities and industries, it is not a proper approach for regulation or management of subsurface waters derived from agricultural practices. A chief difference between an agricultural system and that of a municipality or industry involves the relationship between concentration and mass emissions of a constituent in water. For municipalities or industries, concentration is directly proportional to mass emission, because the volume of water to be discharged is relatively constant. For a fixed volume of water, decreasing the concentration by 50% results in a 50% decrease in mass emission. Thus, regardless of whether concentration or mass emission is the better criterion for controlling pollution of the environment, the same control strategy and conclusion are drawn for the industries or municipalities. On the other hand, for drainage waters derived from irrigated agricultural land, there is no proportional relationship between concentration and mass emissions, as can be noted from the data plotted on Fig. 11. A large amount of tile effluent with relatively low nitrate-nitrogen concentration may result in a larger mass emission of nitrate into the environment than a small amount of tile effluent with a much higher nitrate-nitrogen concentration. Therefore, with agricultural systems, careful consideration should be given to whether concentration or mass emission is the more valuable criterion for water pollution.

When a potential pollutant is discharged into the environment, its negative effect depends on the assimilative capacity of the environment for that constituent. The environment is negatively "impacted" when the amount discharged exceeds the assimilative capacity. In this respect, mass emission is a better parameter than concentration. Thus, the parameter to be regulated is mass emission, rather than concentration of nitrate-nitrogen, in drainage water.

However, regulation is relatively worthless unless there are means for detecting violations of it—which means measuring mass emissions. It is difficult to determine mass emissions, because both concentration and tile effluent volume must be measured periodically. In our studies we found measurement of effluent volume difficult; in several cases it was impossible. To monitor mass emissions for several tile drainage systems would present nearly insurmountable problems. Furthermore, monitoring is necessarily "after the fact." It is only after the discharge that one can determine whether or not the amount is legal. Thus, control strategy for mass emission would be hard to design.

Concentration is rather easily measured and can be compared to a prescribed value. It is tempting to use concentration as the criterion, but succumbing to this temptation could lead to increased pollution rather than to decreased pollution. This "reverse" effect is explained as follows: In sandy soils where water movement is not restricted, low nitrate-nitrogen concentration can be achieved by applying considerable water, causing a high leaching fraction. The added water dilutes the nitrate-nitrogen and results in a low concentration in the effluent. At the same time large quantities of water are flowing through the profile, and the mass emission is greatly increased. Thus, the strategy a grower would likely use to achieve the concentration limit would result in an increase in mass emission. The increased mass emission would represent a greater assault on the environment than the higher concentration at a lower volume of discharge.

Our data illustrates how low concentration may actually lead to higher mass emission. In investigating sites C-4 and C-9 during the second year of the study, we found that the average nitrate-nitrogen concentration in C-4 was 29 ppm, whereas the average nitrate-nitrogen concentration for site C-9 was 49 ppm. If concentration were the criterion for pollution, one would conclude that the grower on site C-4 more closely approached compliance to a regulation. Mass emission on site C-4 was 155 kg/ha per year, whereas the mass emission on site C-9 was only 62 kg/ha per year. Mass emissions on site C-9 were assaulting the environment to a much lesser extent than they were on site C-4. If a regulation was adopted specifying a low concentration of nitrate-nitrogen, the grower on C-9 could choose to increase irrigation, thus lowering the concentration. However, he also increases the mass emission. Fig. 9 clearly indicates that increasing the amount of tile effluent volume in irrigated soils increases the amount of nitrate-nitrogen which will be discharged to the environment.

Another argument against concentration as the criterion for regulation is that we found poor correlation between it and other measureable variables, such as fertilizer input, soil properties, or amount of water discharged. Our data strongly indicate that significant interaction occurs and that no one factor predominates throughout the state of California. For example, increasing the amount of leaching water would be expected to decrease the concentration because of dilution; yet, we found no such correlation (see Fig. 8). We hypothesize that growers may have learned to compensate for high leaching of nitrate-nitrogen by adding more fertilizer to maintain crop production. This may account for the lack of correlation we found between concentration and the amount of tile effluent.

Any regulation proposal should acknowledge the possibility of high "native" nitrogen in some soil profiles. The grower has no control over this. We found "native" nitrogen in several soils on the west side of the San Joaquin Valley; it was also detected in the Imperial Valley on site I-10 during the second year of study.

Concentrations in tile effluent may show the accumulative effect of several years of nitrate-nitrogen-producing practices. Even when a grower adjusts his management practices to minimize nitrogen leaching beyond the root zone, the effects of his modified management may not be completely detected in the tile effluent for several years. This was clearly illustrated by Jury (1975*a*, 1975*b*) in his discussion on solute travel time to tile drains. Should such a grower be penalized for previous management over which he may have had little or no control?

We conclude that the traditional strategies used for water quality control in municipalities or industries are not practical for agricultural waters. Our data and discussion were restricted to water which has been collected through tile drainage systems. However, most of the points we made are valid for waters that percolate beyond the root zone and continue moving through the profile to the water table. The largest proportion of agricultural land uses the so-called "free drained" system, where artificial drainage systems are not installed.

If traditional control and regulatory

standards are not applicable, are there alternative approaches? Our data indicate that mass emission is strongly correlated to the amount of effluent. When sites containing high native nitrogen were excluded from consideration, mass emission was most strongly correlated to the product of effluent volume and fertilizer nitrogen application. Therefore, mass emissions in irrigated soils can be regulated by controlling the leaching and the amount of applied fertilizer. A decrease in one or both will result in lower mass emissions beyond the root zone. Mass emission seems to be more strongly related to the amount of percolating water than to fertilizer application. We suggest that water management be considered as a control strategy for reducing nitrate pollution. Irrigation should result in a very low leaching fraction, sufficient only to maintain an appropriate salt balance. Fortuitously, irrigation to produce a low leaching fraction is also a control strategy for decreasing the salt burden to the groundwaters. With decreased percolation and its associated leaching of nitrate, less fertilizer would have to be applied to achieve the same crop production.

Reduced leaching is not easily achieved, however. In some cases, different and costly irrigation systems may be required. Considerable instrumentation may also be required to guide irrigation management to achieve low leaching.

In brief, it appears that the most positive approach to controlling nitrate-nitrogen in irrigated agriculture is through the control of leaching. There are many serious problems to be overcome, but we believe that attention should be directed toward this goal.

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