POSSIBLE MOBILITY OF RESIDUALS OF AMMONIUM NITRATE FUEL OIL BLASTING AGENTS IN DESERT SOIL AND GROUND WATER

by

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A Thesis Submitted to the Faculty of the

DEPARTMENT OF HYDROLOGY AND WATER RESOURCES

In Partial Fulfillment of the Requirements
For the Degree of

MASTER OF SCIENCE
WITH A MAJOR IN HYDROLOGY

In the Graduate College

THE UNIVERSITY OF ARIZONA

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STATEMENT BY AUTHOR

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ACKNOWLEDGMENTS

The author wishes to express his gratitude, appreciation, and thanks to Dr. L. G. Wilson for his unending help, guidance, and support throughout his graduate career and preparation of this thesis.

Recognition also is extended to Dr. S. N. Davis and Dr. D. D. Evans, members of his graduate committee. Their critical reviews of the manuscript were most helpful in the development of this thesis.

The author wishes to thank J. Meyer, University of California-Riverside, and G. Perry, General Electric-Company—TEMPO, for their help under the uncomfortable climatic and physical conditions encountered in this study. The author wishes to express his gratitude to J. Blackman for the analysis of soil and water samples.

Special thanks to the personnel of the Water Resources Research Center for their contribution to this study and their valuable friendship.

Also recognized are E. Jackson, Army Corps of Engineers Waterways Experiment Station for the sample used in the laboratory study and Dr. D. Marx and members of the Center for Quantitative Studies for their help in the statistical analysis of the laboratory results.

Finally, the author wishes to extend his personal gratitude to his mother and father, and to his fiancee, Theresa, whose help and patience were important factors in completing this thesis.

This study was supported by funds from General Electric Company—TEMPO.

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ABSTRACT

Field and laboratory studies were conducted to assess the effect, if any, of residuals of ammonium nitrate fuel oil (ANFO) detonation on ground-water quality of a portion of Planet Valley, Arizona. Preliminary soil samples were taken near detonation sites to characterize background levels of NO_3-N. Laboratory experiments were performed to determine the mass of NO_3-N removed from soils exposed and unexposed to the products of ANFO detonation. Ground-water samples were taken from eight wells to monitor changes of NO_3-N.

Preliminary soil samples showed that considerable NO_3-N existed at some depths near detonation sites. Laboratory study showed no significant difference between the mass of NO_3-N leached from exposed soil and that eluted from unexposed surface soil. Analyses of water samples from all wells, except one, showed NO_3-N levels to be similar to background values.

The source of NO_3-N which occurred in the one well, either natural soil NO_3-N or NO_3-N introduced as a residual of ANFO detonation, could not be determined. The effect of ANFO residuals on water quality was no more than would occur if natural soil NO_3-N were leached to the ground water.

INTRODUCTION

In recent years, the consumption of ammonium nitrate-fuel oil (ANFO) blasting agents has increased in the United States, primarily because an oxygen-balanced ANFO mixture is the cheapest source of explosive energy. The consumption of ANFO during 1977 increased 15.1 percent over that of 1976, with the coal mining industry consuming 62.5 percent of all ANFO manufactured (U.S. Bureau of Mines, 1978). Table 1 shows the consumption of bulk ammonium nitrate and fuel mixtures in the United States, by state and use, for 1977. Additional uses of ANFO include seismic prospecting, military testing, and improvement of drainage in agricultural soils. Accompanying the increase of ANFO utilization as a blasting agent is the potential for contamination of the soil-air-water continuum by the detonation products.

Various investigators have alluded to the possibility of ground-water contamination by ANFO residuals. Schmidt (1978) stated that nitrate nitrogen (NO_3-N) levels in pit waters within copper mines south of Tucson, Arizona may approach 23 mg/l. In studies on the Decker mine, Montana, Van Voast (1974) indicated that samples of mine effluent contained widely differing, but occasionally very high concentrations of NO_3-N due to fracturing of coal and

Table 1. The consumption of bulk ammonium nitrate and fuel mixtures in the United States, except Hawaii, by state and use, 1977, in thousands of pounds. -- After U.S. Bureau of Mines (1978).

State	Use					
	Coal Mining	Metal Mining	Quarrying and non-metal Mining	Construction Work	Other Purposes	Total
Alabama	212,310	198	29,152	17,947	14,230	273,837
Alaska	---	---	---	972	---	972
Arizona	4,250	115,918	4,900	375	9,877	135,320
Arkansas	2,466	1,356	4,006	2,321	9,144	19,293
California	---	12,035	12,702	3,207	500	28,417
Colorado	5,827	9,166	1,632	2,750	60	19,435
Conneticut	---	---	3,653	2,687	59	6,399
Deleware	---	---	100	---	---	100
Florida	---	---	7,080	406	---	7,486
Georgia	---	---	13,184	1,736	436	15,356
Idaho	---	1,264	690	1,403	---	3,357
Illinois	52,398	---	27,843	792	50	82,100
Indiana	126,388	125	11,410	1,556	186	139,665
Iowa	1,679	---	14,691	1,724	5,209	23,303
Kansas	3,300	---	7,216	268	208	10,992
Kentucky	503,811	126	31,121	22,604	16,955	574,617
Louisiana	---	---	3,825	---	9	3,834
Maine	---	137	320	274	---	704
Maryland	6,775	---	4,920	772	---	12,467
Massachusetts	---	---	2,132	342	22	2,496
Michigan	157	5,040	15,056	2,040	9,386	31,679
Minnesota	---	10,715	2,689	5,409	16,002	34,815
Mississippi	---	---	---	---	206	206
Missouri	12,415	1,774	17,460	9,917	835	42,401

Table 1. Continued

State	Use					
	Coal Mining	Metal Mining	Quarrying and non-metal Mining	Construction Work	Other Purposes	Total
Montana	7,908	86	282	67	47	8,390
Nebraska	---	---	770	259	---	1,029
Nevada	704	13,904	4,215	---	---	18,823
New Hampshire	---	4	244	308	---	556
New Jersey	51	213	887	326	10	1,487
New Mexico	32,596	48,758	10,807	2,881	---	95,042
New York	431	4,238	8,300	2,932	315	16,216
North Carolina	40	80	12,702	6,217	---	19,039
North Dakota	---	---	33	---	---	33
Ohio	147,948	---	21,155	7,723	9,494	206,320
Oklahoma	32,188	---	5,946	1,451	277	39,862
Oregon	---	---	592	7,265	---	7,857
Pennsylvania						
Anthracite	129,408					
Bituminous	262,535	66	22,239	21,984	8,384	444,616
Rhode Island	---	---	46	87	---	133
South Carolina	---	---	2,826	471	---	3,297
South Dakota	---	---	---	---	---	28
Tennesse	36,384	7,665	23,441	41,857	18,364	127,711
Texas	33,700	96	14,581	21,785	212	70,474
Utah	152	25,440	---	---	96	25,688
Vermont	17	---	24	763	13	817
Virginia	123,244	368	18,413	17,915	935	160,875
Washington	---	770	---	9,389	---	10,159
West Virginia	138,448	2	5,337	29,306	11,591	184,684
Wisconsin	---	194	7,199	222	933	8,548
Wyoming	72,306	6,465	7,723	1,125	418	88,137

overburden by ANFO. Similarly, Dettman, Olsen, and Vinikour (1976) examined mine discharge from a coal strip mine near Sheridan, Wyoming. They correlated high NO_3-N and ammonia nitrogen (NH_3-H) concentrations in pit discharge to the use of ANFO during coal extraction. Occasionally, farmers in the San Joaquin Valley, California, have observed an improvement in tree growth and production following ANFO use to remove impervious soil layers (Meyer, 1979).

Investigations to characterize the effect of ANFO residuals on ground-water quality appear to be rare or even nonexistent. During the summer of 1978 the Department of Defense (DOD) conducted explosive tests at Planet Valley, Arizona, to evaluate conditions for installing the proposed MX missile; ANFO was used as a high energy source to simulate the cratering and ground shock phenomena resulting from a nuclear explosion. The test program involved the detonation of a single 109-metric-ton charge of ANFO on the land surface on June 28, 1978, and an array of six, 109-metric-ton charges on August 30, 1978. The DOD military testing provided a source of ANFO residuals to study and perhaps better understand the hydrologic consequences of detonating ANFO in arid and semi-arid regions.

Specific objectives of this study were: (1) to monitor spatial and temporal changes in the chemical composition of soil water and ground water in the vicinity of

the detonations, particularly with respect to nitrogen species, and (2) to conduct laboratory leaching experiments to determine the mass and pattern of removal of NO_3-N and NH_3-N from soil affected by ANFO detonation and from surface and subsurface materials unaffected by ANFO detonation within the study area.

CHARACTERISTICS OF BLASTING AGENTS

Definition and Types

The main distinction between explosives and blasting agents is related to their safety in handling (U.S. Bureau of Mines, 1978). Blasting agents are not as sensitive to initiation as explosives and require a primer for detonation. Also, explosives vary widely in composition, whereas blasting agents are essentially ammonium nitrate (AN) plus a fuel.

Blasting agents may be classified as dry or slurry. Dry blasting agents are basically a mixture of granular AN and a fuel oil (FO) or other carbonaceous material (Dick, 1968). Experimentation has proved that the most practical and efficient formulation, in terms of field performance and explosive properties, is a 94% AN, 6% FO mixture known as ANFO. Advantages of ANFO are its safety in transportation, storage, and handling, as well as its low price compared with that of other explosives. However, because ANFO is primarily composed of highly soluble AN, it may deteriorate on exposure to water.

Various methods have been developed for field formulation and use of ANFO mixtures. Most commonly, mixed ANFO is loaded, either by hand or mechanically, into boreholes that have been drilled and spaced to insure adequate

fragmentation of the rock medium. A suitable primer is added to achieve detonation. In surface or unconfined blasting, AN is stacked in bags and FO is poured over the AN, which is then detonated with a primer.

The ANFO mixtures are suitable when a moderate degree of blasting is required, as in coal strip mining, open-pit mining, and blasting of stratified limestone, sandstone, and shale (Manon, 1977).

Slurry blasting agents are the most recent development in blasting agent formulations. They were originally designed to overcome the problems of ANFO and to produce a denser explosive (Reed, 1974). A slurry is essentially a mixture of an oxidizer and a fuel sensitizer in a liquid medium, thickened with a gum, and gelled with a cross-linking agent (U.S. Bureau of Mines, 1978). In most slurries, AN is the primary oxidizer, although some sodium nitrate is often added if more available oxygen and a higher density are needed (Dick, 1972). Fuel sensitizers are either explosive or nonexplosive. The thickening and gelling of the slurry are required to prevent leaching of soluble nitrate salts used as the oxidizers (Dick, 1972).

Generally, slurries equal or exceed the properties of ANFO in the energy available in a given weight of explosive. For this reason their use has become widespread, especially under harder rock conditions (Reed, 1974).

Chemistry and Products of ANFO Detonation

If properly initiated, ANFO undergoes detonation. Dick (1972) has defined detonation as a self-propagating, exothermic decomposition, at a rate supersonic to the unreacted material, which produces more stable products, usually gases. Chaiken, Cook, and Ruhe (1974) have shown the detonation of a stoichiometric ANFO mixture (AN-FO mass ratio = 94.5 to 5.5) to be represented in gross terms by the following overall reaction:

$$NH_4NO_3 + 0.047C_{7.17}H_{14} \rightarrow N_2 + 2.33H_2O + 0.336CO_2 + \text{Energy}$$

Nonstoichiometric amounts of fuel oil will produce a negative oxygen balance and change the amount and type of products and the amount of energy release.

Chaiken et al. (1974) determined the detonation products of ANFO mixtures containing varying amounts of fuel oil. Table 2 depicts the theoretical detonation properties of an ANFO composition of 6% FO (most common for commercial use) and the predicted amount of detonation products from a 109-metric-ton ANFO charge.

Ammonium nitrate-fuel oil mixtures are non-ideal explosives (Reed, 1974) in that other factors besides fuel oil percentage influence their explosive properties and the amount and types of detonation products. The explosive properties and products are affected by density, charge

Table 2. Theoretical detonation properties of ANFO (6% FO) and predicted amount of products from a 109-metric-ton charge. -- After Chaiken et al. (1974).

Detonation Properties (NH_4NO_3--$C_{7.17}H_{14}$)

Temperature--3170°K Pressure--36100 atm

Detonation Product Composition

Chemical	Moles/ kg	Percent by Weight	109-metric-ton Charge (kg)
H_2O	27.4	49.0	54,432
N_2	11.7	32.8	34,927
O_2	0.002	1.0	1,179
H_2	0.253	0.05	54
NO	0.053	1.7	1,860
NO_2	$5x10^{-5}$	0.0322	34
N_2O	$3x10^{-4}$	0.0132	14
NH_3	0.050	0.085	91
O	$2x10^{-4}$	0.0032	4
OH	0.006	0.0289	3
CO_2	3.53	15.5	16,783
CO	0.761	2.1	2,268
HCN	0.002	0.0054	6
CH_4	$2x10^{-5}$	$3.2x10^{-4}$	<0.45

diameter, confinement, priming, water conditions, particle size of the AN, and the degree of reaction (Reed, 1974). However, the amount and types of detonation products as depicted in Table 2 will be assumed to exist for the purpose of this study. Particular constituents that may be introduced or formed in soil and ground-water systems as a result of ANFO detonation include cyanide, nitrate nitrogen, and ammonia nitrogen.

NITROGEN FORMS AND MOBILITY

The wide distribution of nitrogen compounds in soil and ground water and their intimate involvement with life processes make their study complex. Commonly, the relationship between the various compounds and the transformations that can occur are presented schematically in a diagram known as the nitrogen cycle. Figure 1 shows the major aspects of the nitrogen cycle associated with the soil and ground water.

The sources of nitrogen compounds include precipitation and dustfall, waste water or waste-water effluent, plant and animal matter, and artificial fertilizers (U.S. Environmental Protection Agency, 1975). Also, nitrogen gas is fixed by microorganisms living in the root nodules of leguminous plants, by soil organisms, both aerobes and anaerobes, and by some blue-green algae living in soil and water (Feth, 1966). These sources of nitrogen, once incorporated into the soil or ground water, can generally be classified as organic or inorganic.

Organic Nitrogen

In soils, by far the greater total amount of nitrogen (N) occurs as part of the organic matter complex. However, the organic forms of soil N are, for the most

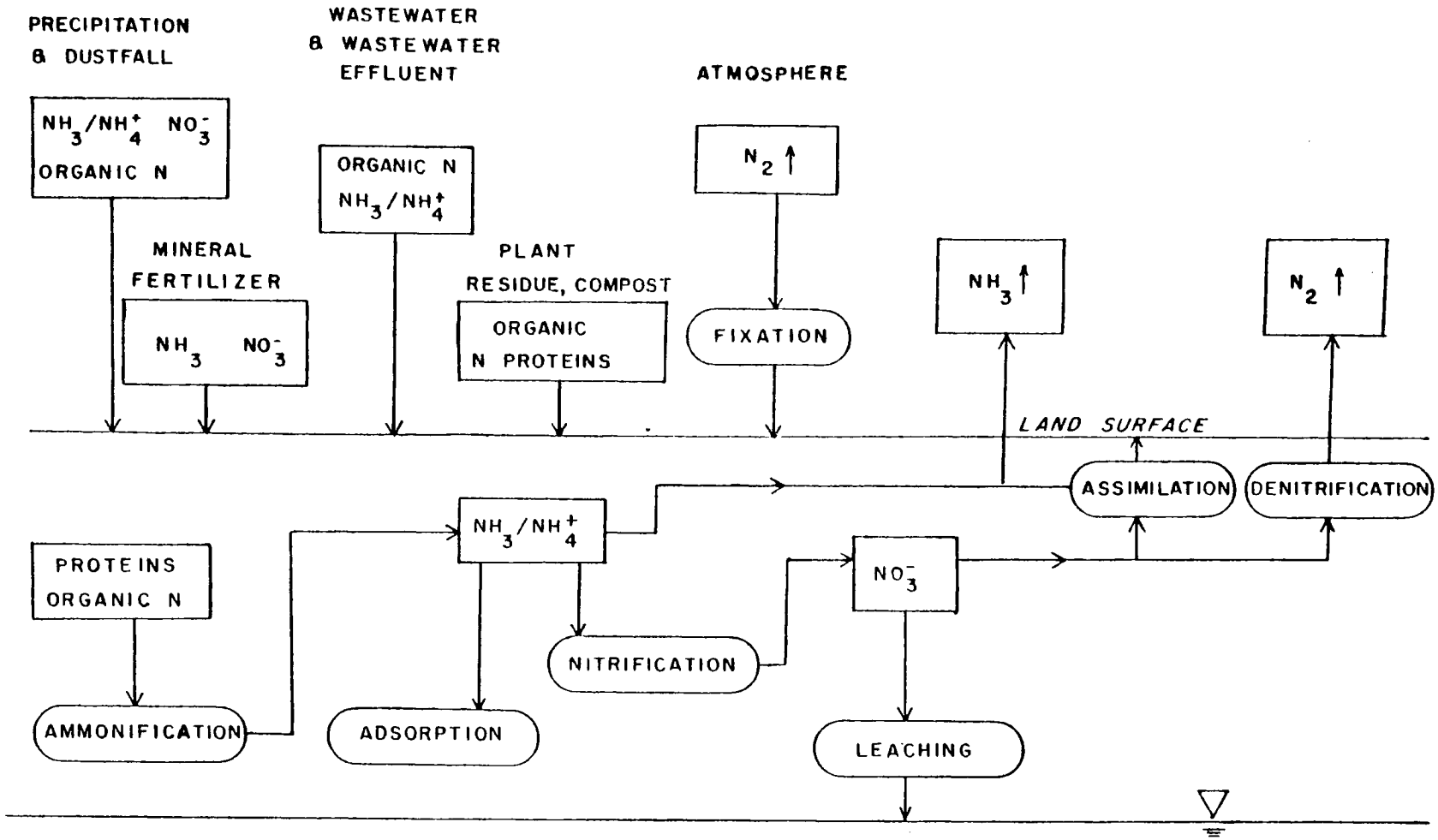

Figure 1. Nitrogen cycle of the soil and ground water. -- After U.S. Environmental Protection Agency (1975).

part, considered immobilized and unavailable for plant growth or leaching (Buckman and Brady, 1969). In arid and semi-arid regions, arable soils are usually characterized by a great deficiency in organic matter, and hence organic N.

In ground water, organic N is rare unless the water contains organic waste. Some sources of organic waste are municipal waste-water effluents, industrial waste, and feedlot runoff.

Organic matter is capable of being oxidized by heterotrophic organisms. This results in a mineralization of N to ammonia, a process known as ammonification (Figure 1).

Inorganic Nitrogen

Ammonia Nitrogen

Ammonia nitrogen represents the most reduced form of inorganic N commonly found in soil and ground water. NH_3 may be protonated in solution to form NH_4^+, or it may remain nonionic. This conversion is, of course, highly pH dependent. The pH at which the transformation of aqueous NH_3 to NH_4^+ is half complete is about 9.24 (Sillen and Martell, 1964). Hem (1978) therefore suggested that in most natural waters, any NH_3 in solution would have the form NH_4^+. However, in soils and ground water of arid regions, solutions may have considerable base status,

leading to the occurrence of both NH_3 and NH_4^+. In any form, ammonia nitrogen in ground water seldom exceeds a few milligrams per liter (Bouwer, 1978).

The cationic nature of NH_4^+ permits its adsorption and retention by colloidal material, largely clay and organic matter. However, it is necessary that the porous medium have a sufficiently high exchange capacity to retain NH_4^+ or the NH_4^+ will be mobile, as is the water. Sandy mediums with low exchange capacities permit appreciable movement of NH_4^+ into the subsurface (Tisdale and Nelson, 1975). Nonionic NH_3 is also capable of being retained by soils as a consequence of polarity, which produces localized charge within the molecule (Bohn, McNeal, and O'Connor, (1979). This mechanism is most prominent if the organic fraction of the soil is high.

Another possible fate of NH_4^+ is its fixation by clays with an expanding lattice (Buckman and Brady, 1969). Montmorillonite, illite, and vermiculite have internal negative charges, which attract NH_4^+ to surfaces between crystal units and render the NH_4^+ a part of the crystal. This process is reversible, and NH_4^+ may be replaced by cations, which expand the lattice (Ca^{2+}, Mg^{2+}, Na^+, and H^+, for example). However, this process can appreciably modify the rate of movement NH_4^+ in soil solution.

Ammonia nitrogen is capable of being assimilated by plants (Figure 1), or, due to its oxidation state of

minus 3, it is capable of being oxidized by bacteria to other, more stable forms of inorganic N in the process of nitrification (Figure 1). Although nitrite (NO_2^-) is an intermediate product of oxidation, under most natural conditions it is further oxidized to nitrate nitrogen.

Nitrate Nitrogen

Nitrate nitrogen, as the end member of the oxidation chain of N, is the N-bearing ion most commonly occurring in well-aerated soils and ground water.

In porous mediums, in general, NO_3^- is highly mobile and therefore subject to great spatial fluctuation. This mobility is attributed to the electronegative nature of the nitrate ion, which causes it to be repelled by the exchange complex under most circumstances. Under some conditions, that is, low pH and the presence of a mineral whose surface charges are significantly pH dependent, NO_3^- is capable of being adsorbed onto mineral surface sites. However, little NO_3^- adsorption occurs at pH greater than 7, even in soils containing considerable pH-dependent charge (Bohn et al., 1979).

Nitrate nitrogen mobility is affected by textural, climactic, and cultural conditions. Bates and Tisdale (1957) has presented data from column experiments that show that the coarser the texture and greater the large pore space of a soil, the greater will be the mean downward

movement of NO_3^- under the influence of a given quantity of added water. Therefore, NO_3^- can be leached from a clay soil as readily as from a sandy soil, except that the rate of movement will be slower in the finer textured soil.

In humid areas and where irrigation is practiced, NO_3^- can be leached out of the upper soil layers by infiltrating water (Bouwer, 1978). Conversely, in arid and semi-arid regions and where upward, capillary movement of water is possible, NO_3^- may accumulate in the upper horizons of the soil (Tisdale and Nelson, 1975). Under both climatic conditions, if sufficient water exists to leach NO_3^- from the surface layers, NO_3^- may reach the ground water.

Nitrate nitrogen concentrations in ground water are highly variable and subject to seasonal variations. Bouwer (1978) stated that where the natural N-enrichment (nitrogen-fixation) processes have been going on for some time, NO_3-N levels in ground water may be on the order of 1 to 50 mg/l. Kreitler and Jones (1975) found the ground waters of Runnels County, Texas to be highly contaminated with NO_3-N. The average NO_3-N concentration was 57 mg/l with a range of less than 1 mg/l to over 700 mg/l. They concluded the major source of NO_3-N in the ground water to be natural soil NO_3-N formed by oxidation of soil organic N and leached to the ground water.

Feth (1966) discussed seasonal variations of NO_3-N concentrations in ground water. In municipal wells in Kansas, NO_3-N concentrations varied as much as 3.4 mg/l from low values in summer to high values in winter, where the maximum NO_3-N concentration was 7.9 mg/l (Metzler and Stoltenberg, 1950). Seasonal leaching of NO_3^- from the soil was inferred to be responsible for the variations observed.

Nitrate loss from the soil can also occur as a result of assimilation by plants and a few animals to form protein and other nitrogen-containing compounds. In addition, NO_3- is capable of being reduced to N_2 under anoxic conditions by a broad range of bacteria in the process of denitrification (Figure 1). Organic carbon serves as the energy source for this conversion.

The maximum permitted concentration of NO_3-N for public water supplies is 10 mg/l. This limit is imposed due to the susceptibility of young infants to methemoglobinemia (Bouwer, 1978).

Cyanide

The cyanide (CN^-) is not a primary constituent of soil or ground water but finds its way into these systems primarily through the activities of man. Because CN^- is toxic (maximum allowable in drinking water is 0.2 mg/l), its behavior when introduced is of considerable concern.

Recently, Fuller (1977) reviewed the chemistry of CN^- in soils. Cyanides were classified into simple and complex forms. Simple forms contained an alkali or alkaline earth metal as the cation, with CN present as CN^-. The complex forms were presented as bearing an alkali metal cation in combination with a transition metal and cyanide anion radical.

Alesii (1976) studied the mobility of simple and complex forms of cyanide in soils. Cyanide as $Fe(CN)_6^{3-}$ and CN^- in water were both found to be very mobile in soils. Soil properties such as low pH, high percentage of free-iron oxide and kaolin, chloride, and gibbsite type clay (high positive charges) tended to increase attenuation of cyanide. High pH, presence of free $CaCO_3$ (high negative charge), low clay content, and montmorillonite clay tended to increase the mobility of the cyanide forms.

The stability of cyanide forms in aqueous solution is unclear; however, microorganisms are known to decompose simple alkali cyanides and many alkali-metallic cyanides to CN^- in aqueous solution. The CN^- is then subject to further decomposition to NH_3 and NO_3^- under aerobic conditions and to N_2 under anaerobic conditions (Fuller, 1977).

DESCRIPTION OF THE STUDY AREA

Location and Physiography

The area of investigation is a portion of Planet Valley, Arizona (Figure 2), in Mohave and Yuma Counties. The explosive test sites are immediately north of the intermittent Bill Williams River in sec. 36, T. 11 N., R. 16 W. of the Gila and Salt River Base and Meridian. Ground-water quality was monitored for this study by sampling wells occurring in sections 25, 35, and 36 of T. 11 N., R. 16 W. Figure 3 illustrates the study area and the location of the ANFO detonation sites.

The study area is in the Western Range and Irrigated Region of the Sonoran Desert section of the Basin and Range province. The actual testing sites lie within a valley at an average elevation of about 207 m MSL, at the approximate depositional confluence of a desert wash (named Misers Wash) and the Bill Williams River. The valley is bordered on the east by Black Mesa (Figure 3) and terrace deposits of ancestral streams that were once tributary to the Bill Williams River. Misers Wash enters the valley approximately 1.2 km northeast of the test sites, through low-lying terrace and alluvial fan deposits (Figure 3). The Bill Williams River and the metamorphic rocks of the Buckskin Mountains lie south of the valley (Figure 3).

Figure 2. Location of Planet Valley, Arizona

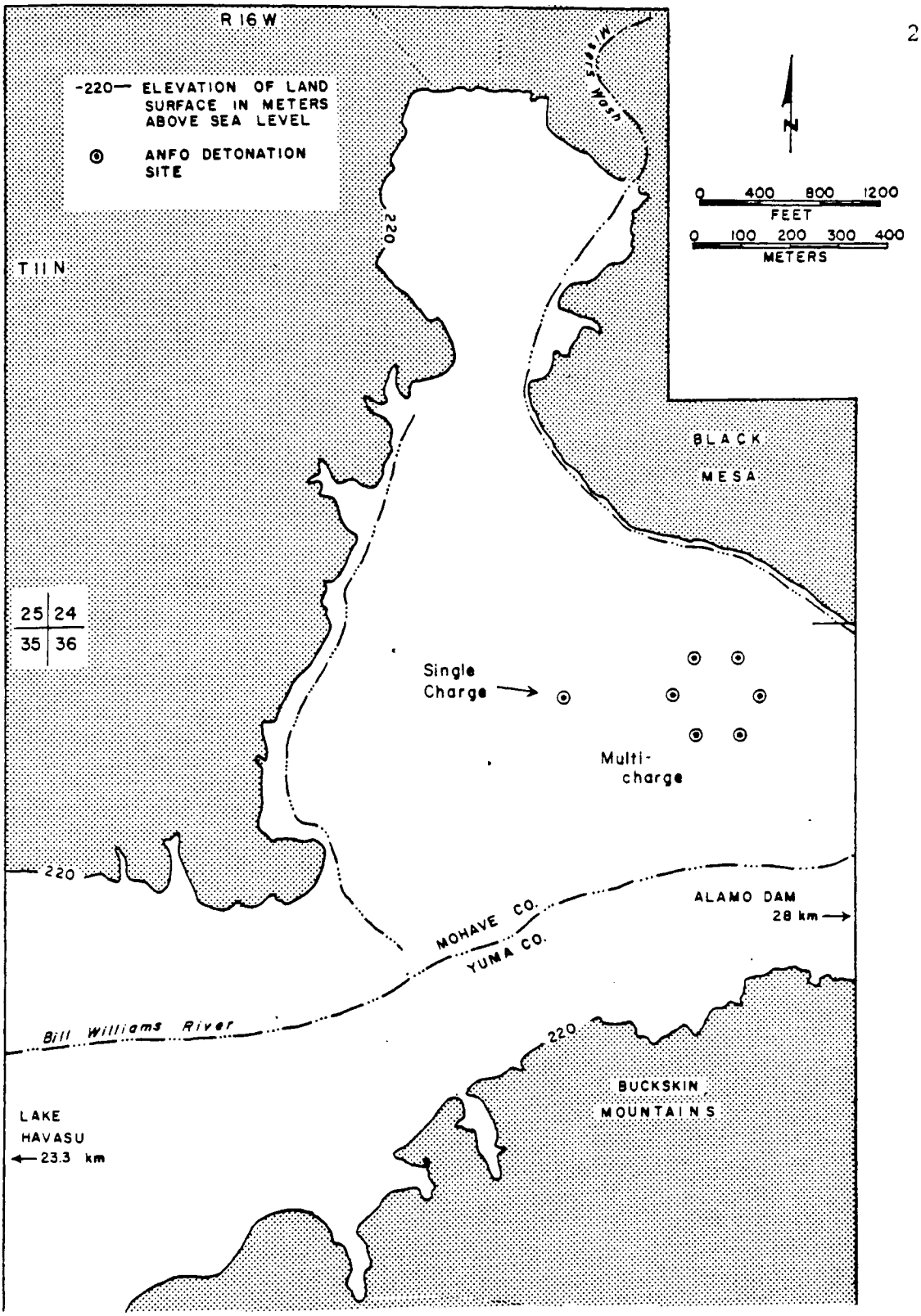

Figure 3. Location of the study area and ANFO detonation sites

West of the valley lie terrace and alluvial fan deposits of both ancestral tributary streams and the Bill Williams River. The Bill Williams River flows intermittently through the study area. About 13.0 km west of the study area it becomes perennial and empties into Lake Havasu 10.3 km west of this location.

Previous Investigations

Turner (1962) conducted a field investigation in Planet Valley to characterize the geology and water resources of the area, including a water balance. During three summers, 1974, 1975, and 1976, hydrologic studies were conducted in the Planet Valley during a summer field course in the Department of Hydrology and Water Resources of The University of Arizona. These studies resulted in reports by the students on the water balance of the area. Using field data for the three years, Bond (1976) adapted the ground-water flow model of Prickett and Lonnquist to the hydrologic conditions of the area.

An environmental assessment of Planet Valley was done by the U.S. Bureau of Land Management (1977) when the valley was being considered as a potential townsite.

Geotechnical studies done prior to and after the military testing of the summer of 1978 included those by the Air Force Weapons Laboratory (AFWL), Fugro National Incorporated (FUGRO), U.S. Army Corps of Engineers

Waterways Experiment Station (WES), and the University of New Mexico Civil Engineering Research Facility (CERF). Pink (1979) summarized the results of the geotechnical investigations in the vicinity of the test sites.

Climate

The study area, as previously noted, is in the southwestern desert region. It is characterized by extremely hot summers, mild temperate winters, low rainfall, low relative humidity, and a long growing season.

The study area has a mean temperature in July of 41.6°C, with highs ranging from 46.1°C to 51.7°C. Approximately 100 days per year have mean temperatures over 37.8°C (U.S. Bureau of Land Management, 1977).

The average rainfall is about 12.7 cm per year in the study area. Approximately 60 percent of this precipitation results from winter storms, and the remainder from summer cloudbursts. The mean evaporation is approximately 203 cm (U.S. Bureau of Land Management, 1977).

Agriculture and Vegetation

Irrigated agriculture has not been practiced in the valley near the test sites, although at one time it was considered (Cluff, 1979). Farming is done, however, upstream and in portions of Planet Valley west of the study area. Cattle have been allowed to graze on native grasses in the valley after military operations were complete; however,

it is not known if they grazed in the valley prior to ANFO detonation. Vegetation within the valley include creosote bush, cacti, dodder, and mallow (Kuch and Dobson, 1978).

Soils

Soils in the study area are described by Chamberlin and Richardson (1974) as belonging to the Gilman-Vint-Brios association. This association occupies the floodplain of the Bill Williams River and the alluvial fan of Misers Wash. Slopes are dominantly 0 to 2 percent (Chamberlin and Richardson, 1974). The soils are more than 150 cm deep and are formed in recent alluvium from mixed igneous and sedimentary rocks. Because the soils are relatively young, they show little horizon development.

Soils of this association are intermixed with soils of the Agualt, Antho, Maripo, and Carrizo series. In general, soil texture ranges from fine sand to silt loam. Locally, areas of these soils are slightly to strongly saline. All soils included in this association are subject to brief periods of flooding (Chamberlin and Richardson, 1974).

Geology

The geologic formations in the alluvial valley and surrounding area range in age from Precambrian to Holocene and include sedimentary, igneous, and metamorphic rocks of wide variety. Rocks which delineate the alluvial valley

are of primary concern to this study. Figure 4 shows a geologic map of the test valley and surrounding area (FUGRO, 1978).

Directly northeast of the ANFO detonation sites lies Black Mesa, a Quaternary basalt flow that rises above the valley, with local relief of 15 to 30 m. The basalt is moderately vesicular to dense and contains interbedded volcanic agglomerates (FUGRO, 1978). The area to the far north in the test valley is composed of alluvial fan deposits of intermediate age and terrace deposits of Misers Wash (Figure 4).

Northwest of the detonation sites are low-lying hills of young to old alluvial fan deposits deposited by a tributary stream of the Bill Williams River, which does not flow into the study area. Outcrops of igneous rocks and conglomeritic sandstone also occur northwest of the detonation sites (Figure 4).

The area directly west of the detonation sites is composed of terrace deposits of the ancestral Bill Williams River and colluvial deposits derived from the slopes of these terraces (Figure 4).

The area south of the detonation sites is characterized by Precambrian metamorphic rocks of the Buckskin Mountains, which are predominantly gneiss (Figure 4).

The surficial valley fill consists of modern alluvial and alluvial fan deposits from Misers Wash and

Figure 4. Geologic map of the test valley and surrounding area. -- After FUGRO (1978).

Surficial Geologic Units

A1--Modern Alluvial Deposits

A1r--Bill Williams River
A1w--Misers Wash
A1s--Other tributary streams

A2--Older Alluvial Deposits

A2r--Ancestral Bill Williams River alluvium underlying terraces

Relative age, oldest (1), youngest (2), of deposits indicated by subscripts: $A2r_1$, $A2r_2$.

A3--Alluvial Fan Deposits

Relative age indicated by lower case letters:

A3y--Young
A3i--Intermediate
A3o--Old

A4--Colluvial Deposits

Rock Units

I--Igneous extrusive rocks
M--Metamorphic rocks
S--Sandstone

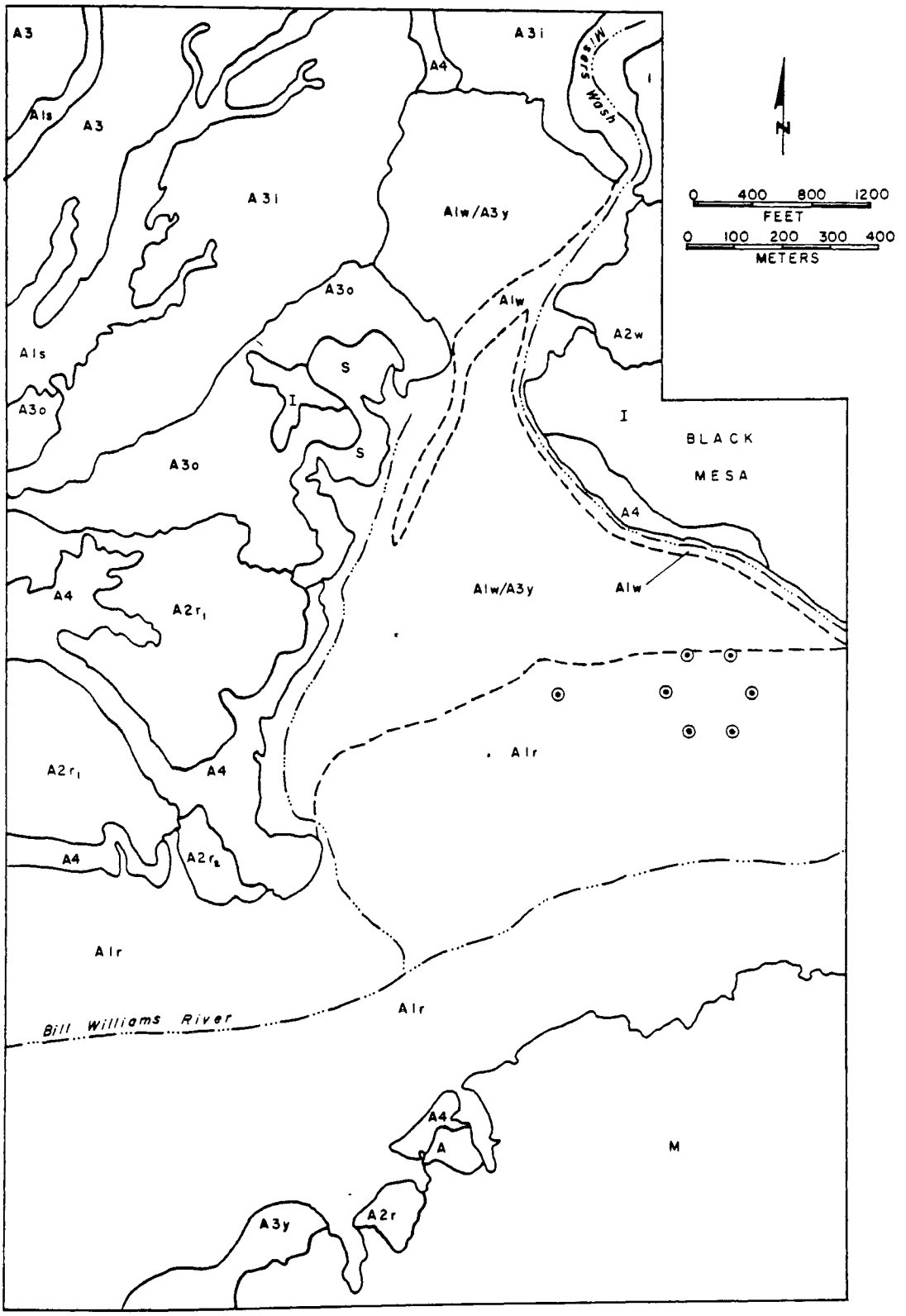

Figure 4. Geologic map of the test valley and surrounding area. -- After FUGRO (1978).

modern alluvial deposits of the Bill Williams River (Figure 4). Modern alluvial deposits consist of sand with silt and gravel-size material occurring locally. Young alluvial fan deposits of Misers Wash are intermixed with modern alluvial deposits. They consist of poorly sorted sand and gravel, with varying amounts of silt, cobble, and boulder-sizee material, and show little to no cementation.

The complex lithology of the basin-fill deposits was studied by AFWL, FUGRO, WES, and CERF; however, their investigations were primarily focused in the vicinity of the ANFO detonation sites (Pink, 1979).

In the vicinity of the single ANFO detonation, a surface layer of silt extends to a depth of 1.4 m. This is underlain by a 7.0 m layer, which is predominantly sand, but includes some gravel (U.S. Army Corps of Engineers WES, 1979). Beneath this is a 5.8 m deposit of silt and clay, which is in turn underlain by a 15.1 m stratum of sand and gravel. A 4.3-m layer of coarse sand and gravel occurs before bedrock is reached at a depth of 33.6 m. Bedrock is composed of fanglomerate with a high proportion of sand and silt (FUGRO, 1978).

In the vicinity of the multiple ANFO detonations, a surficial layer of silt also occurs to a depth of 1.4 m but pinches out to the east and south. The surficial silt layer is underlain by 9.7 m of sand and gravel, which is in turn underlain by 3.3 m of silt and clay. Beneath this

latter layer is a 34.1-m bed of sand with gravel. Gravel content in this bed increases with depth. A 3.4-m stratum of coarse sand and gravel lies beneath this layer, before bedrock is reached at a depth of 51.9 m (U.S. Army Corps of Engineers WES, 1979).

As is evident from the description of the two lithologies, bedrock is deeper in the vicinity of the multiple ANFO detonation site. Other data (Air Force Weapons Laboratory, 1978) suggest that bedrock pinches out to the west of the single detonation site, near the periphery of the study area.

In the study area, the basin fill, as delimited by the surficial and bedrock geology, is the primary medium for water transport, both above and below ground.

Hydrology

Surface Water

The Bill Williams River is an intermittent stream whose present-day flows are controlled by releases from the Alamo Dam, which lies 28 km east of the test site (Figure 3). Alamo Dam was built in 1968 and is operated by the U.S. Army Corps of Engineers as a flood control structure and to reduce sedimentation in Lake Havasu.

Two gaging stations are presently operated on the Bill Williams River below Alamo Dam. First, the Alamo gaging station is 27 km upstream from the test site.

During water year 1977 the mean discharge ranged from 1.525 m^3/s in December 1976 to 4.64 m^3/s in June 1977 (U.S. Geological Survey, n.d.). In water year 1978 Arizona experienced unusually heavy rainfall during late February and early March. Runoff from these rains filled the reservoir behind Alamo Dam to unacceptable levels necessitating large releases from the dam. Rainfall and consequently releases from the dam followed much the same pattern in water year 1979.

The second gaging station is operated 13.0 km downstream from the test site just below where the Bill Williams River returns to the surface. Instantaneous discharges in water year 1977 ranged from 0.025 m^3/s on August 11, 1977 to 0.312 m^3/s on October 4, 1976. Records for water year 1978 show that the instantaneous discharges ranged from 0.043 m^3/s on October 3, 1977 to 6.971 m^3/s on March 6, 1978 (U.S. Geological Survey, n.d.). The latter discharge reflects the large releases from Alamo Dam during February and March of 1978.

The Bill Williams River enters Planet Valley 2.7 km southeast of the test site and maintains a gradient of approximately 0.0061 through the study area. Under normal rainfall conditions, the Bill Williams River does not maintain a flow through the study area. However, workers associated with the military testing reported large flows during February and March of 1978 in response to releases

from Alamo Dam. Conditions in the study area during February and March of 1979 were similar to those of the previous year, although releases from Alamo Dam were begun earlier.

Sheet flow through the valley, derived from flood flows on Misers Wash, has been retarded by a diversion levee built by the land owner at the top of the valley. Cluff (1979) stated that the land owner hoped to trap fine sediment to improve the texture of the soil in the valley. This levee channels water southeastwardly, near the base of Black Mesa (Figures 3 and 4). The levee does not divert all water, and during flash floods, some water flows on the surface near the western boundary of the study area toward the Bill Williams River as evidenced by downed vegetation. The gradient of the valley proper is steeper than the Bill Williams River, with an average gradient of 0.012 maintained through the valley.

Ground Water

Ground water occurs principally in the Holocene sand and gravel along the channel of the Bill Williams River and in the valley proper. The igneous and metamorphic rocks are impermeable except for some slight movement along fault zones (Turner, 1962). The older sediments have been so metamorphosed and silicified that they can be considered relatively impermeable. These igneous,

metamorphic, and well-cemented rocks form the sides and the bottoms of the valley basins and thus form the ground-water reservoirs (Turner, 1962).

The ground water in the Holocene alluvium is recharged principally by intermittent flow in the Bill Williams River and to a much less extent by flood flows along Misers Wash. In fact, as the water levels in the alluvium can be very close to the surface during periods of high discharge from Alamo Dam, it is possible that recharge is being rejected during a small portion of the water year.

Another source of recharge to the alluvium is rainfall directly on the land surface. This is likely to occur when water levels are close to the land surface. Part of this is retained in the soil and discharged by evapotranspiration, and part reaches the water table, however, only in a small portion of the valley.

Ground water is discharged from the study area predominantly by underflow and evapotranspiration. Turner (1962) has estimated the underflow out of Planet Valley at a narrows located 13.0 km downstream of the valley to be 3,700 m^3/day, but noted that it could be as much as 7,400 m^3/day. Fogg (1976) estimated the outflow at the same location to be 4,600 m^3/day.

No experiments have been made to determine direct evaporation of ground water from wetted sands or to

estimate transpiration loss by phreatophytes in the study area. Evapotranspiration is probably greatest where the water table is less than 1.5 m and decreases as the depth to water increases. It is probably very small where the depth to water is greater than 6.0 m. Inasmuch as high water levels are very seasonal and farming is not practiced in the study area, losses by these mechanisms are thought to be localized within the study area.

Water-table fluctuations in the Holocene sand and gravel can be great. Differences of 12.0 m are possible between maximum levels in the late winter and early spring and minimum levels in the late fall and early winter. The degree of variability, of course, depends on releases from Alamo Dam.

Turner (1962) conducted aquifer tests at three locations east of the study area in Planet Valley, which produced hydraulic conductivity values ranging from 262 m/day to 1,640 m/day. These values indicate that the natural rates of ground-water flow are probably very high within the alluvial fill of the study area. Turner (1962) did not compute the storage coefficient during these tests; however, Wolcott, Skibitzke, and Halpenny (1956) reported storage coefficients of Bill Williams River alluvium to be between 0.10 and 0.15.

Turner (1962) found the chemical quality of ground water in Planet Valley to be suitable for domestic or

irrigation usage. However, boron was present in concentrations up to 1.5 mg/l in certain samples. NO_3-N concentrations in Planet Valley ranged from 0.1 to 0.4 mg/l. Wolcott et al. (1956) reported NO_3-N concentrations in Bill Williams River alluvium ranging from 0.1 to 1.8 mg/l in wells 1.6 km to 28.8 km upstream of the test site. They found that 88 percent of the well samples analyzed for NO_3-N had concentrations less than 0.5 mg/l.

EXPERIMENTAL PROCEDURES

Field Study

As previously noted, two ANFO detonations were performed in the summer of 1978. The first event consisted of 109 metric tons of ANFO stacked in 23-kg bags in a cylindrical shape on the land surface. The location of the first test was designated ground zero 1 (GZ1). The second event utilized six 109-metric-ton ANFO charges placed in a hexagonal array on the land surface, with a charge spacing of approximately 90 m. The location of the second test was designated GZ2. Figure 3 shows the location of the first event and the configuration of the second event, as well as the proximity of the detonation sites to the Bill Williams River.

Field studies were undertaken to assess the effect, if any, of these ANFO detonations on soil and ground water.

Soil Sampling and Chemical Analyses

Preliminary Soil and Subsurface Sediment. Soil and subsurface sediment samples were obtained on March 20, 1978 prior to ANFO detonations in the vicinity of GZ1 and GZ2 (Figure 5). The objective of this sampling program was to characterize background levels of soil constituents from

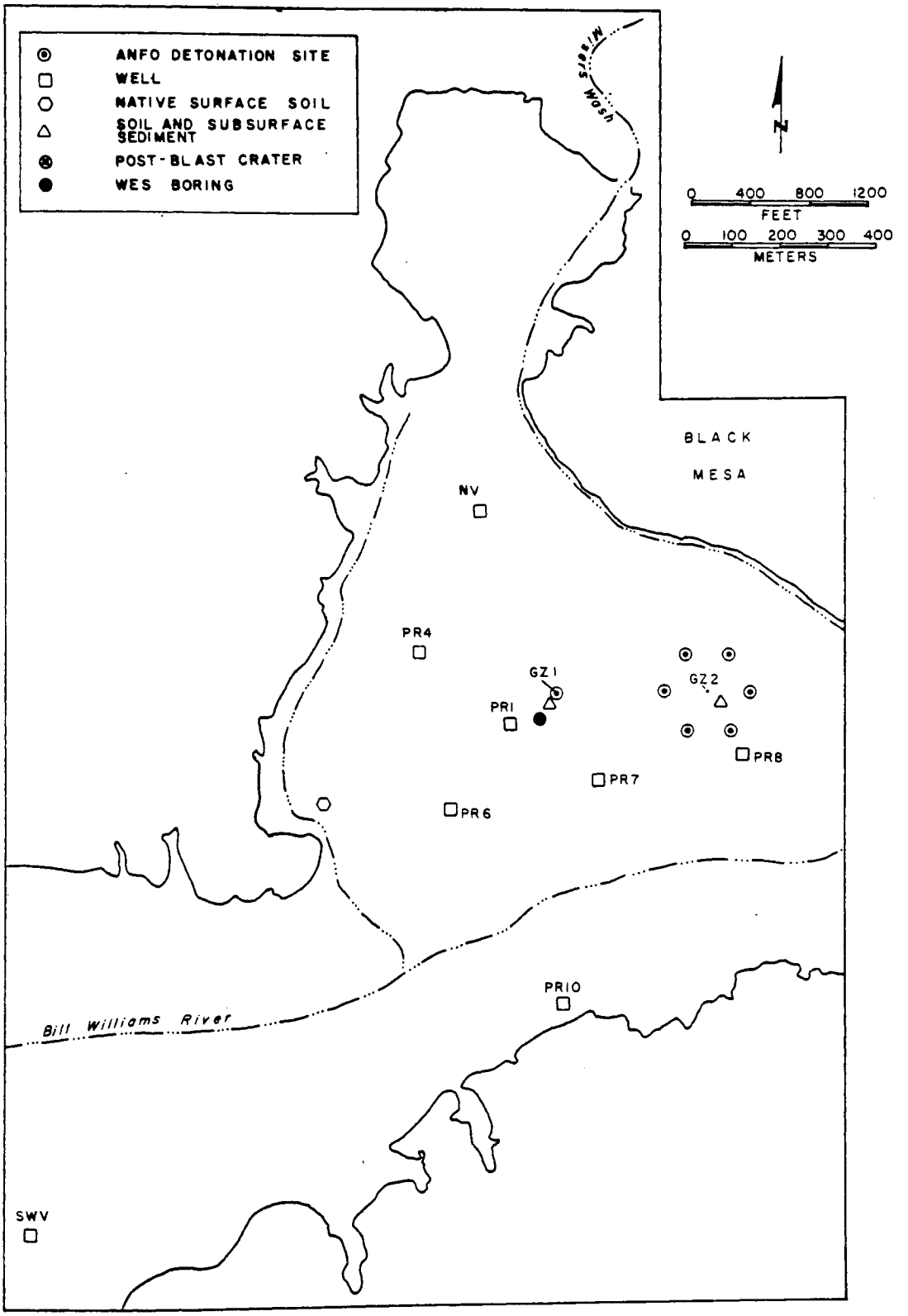

Figure 5. Location of soil and ground water sampling

materials collected near the test sites, with particular consideration being given to NO_3-N and NH_3-N. Because the physical effects of explosive cratering were not well understood and because soil constituents were expected to vary spatially, as many samples as possible were obtained.

Four sets of samples were obtained at each GZ along transects starting 9 m from the center of each GZ. Successive sampling locations were 3 m apart along the transect. Soil samples were collected at each location in 30-m increments to a depth of 3 m or to a restricting layer. For GZ1, the sampling locations were designated DS1, DS2, DS3, and DS4. Corresponding locations at GZ2 were designated DSB1, DSB2, DSB3, and DSB4.

Soil and subsurface sediment samples were obtained with a hand barrel auger and placed in paper bags. Samples were then transported to the Water Resources Research Center (WRRC) laboratory of The University of Arizona for analysis.

Post-blast Crater Material. The objective of this sampling was to obtain materials from the craters produced by ANFO detonation to use in column experiments conducted later in this study. One set of samples was collected after the first ANFO detonation on June 29, 1978, and another following the multiple ANFO detonations on August 31, 1978.

The first ANFO detonation produced a crater with an average radius of 15 m and a maximum depth of 5.3 m. A conspicuous mound, approximately 1.5 m high at its center, characterized the base of the crater. Visual observation suggested that this portion of the crater represented the epicenter of explosive activity and that this was the material that came in most intimate contact with the gaseous ANFO detonation products.

A bulk composite sample of this material was obtained by beginning at the top of the mound and sampling laterally down the slope of the mound to the base (Figure 6). The upper 15 cm of the mound was sampled.

Roddy (1979) stated, after sampling was completed, that the material that composed the upper 2 cm of this mound was probably in the upper 2 m of the land surface before detonation. Material deeper than 2 cm on the mound was probably 2 to 5.3 m below the original land surface in the immediate crater area.

The second crater was sampled the day after the second ANFO detonation. The objective of this sampling program was to secure additional material that had been affected by ANFO detonation in order to compare the analytical results of saturated extracts of this material with those of material derived from the first detonation. Only one of the six craters produced was sampled. Samples

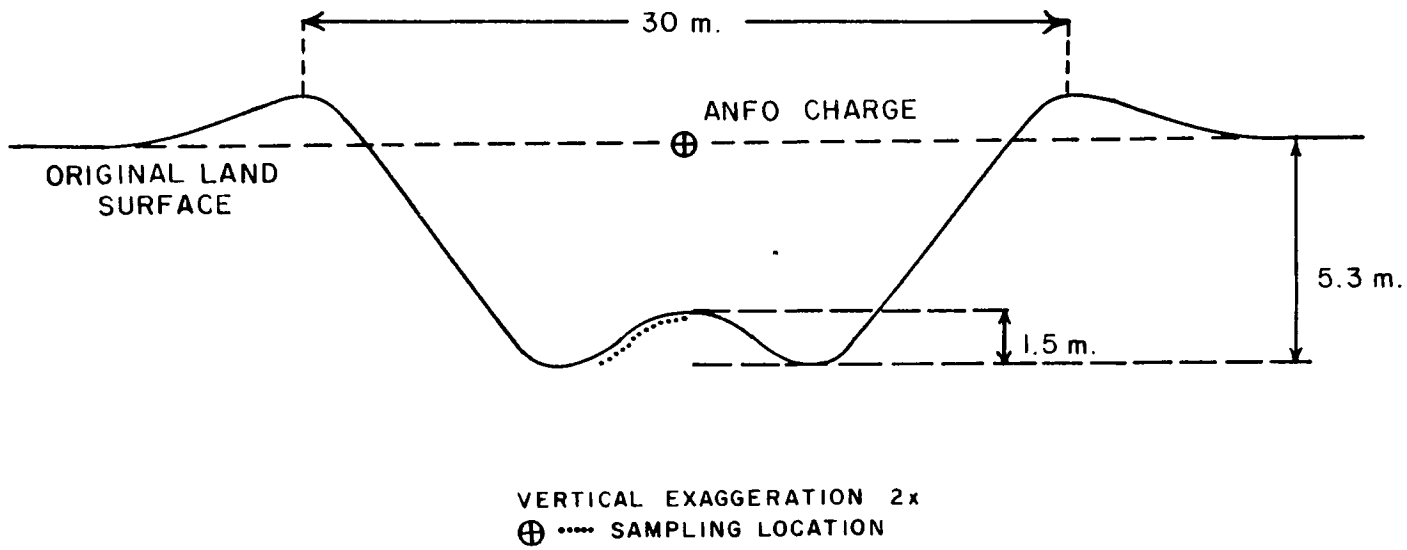

Figure 6. Location of crater sampling

were acquired in the same manner as those secured after the first detonation.

Water was found standing at the base of the crater and a hole was observed which transmitted the water to the surface. Apparently, the forces generated by this detonation were sufficient to rupture a confining layer at depth.

Both bulk crater samples were obtained with a shovel and placed in large plastic containers. A subsample of each of the crater materials was spread on a plastic sheet and mixed to insure homogeneity. A 10-g subsample of each of these materials was then placed in a glass jar to which 100 ml of sodium acetate was added as a preservative. These subsamples were subsequently used for the determination of NH_3-N.

Native Surface Soil. The day after the first detonation, a bulk composite sample of native surface soil was taken to a depth of 75 cm from an area in the valley unaffected by ANFO detonation or its fallout (Figure 5). This sample was also obtained with a shovel, and preparation procedures were the same as for the crater samples. This material was used in column experiments.

Chemical Analyses. Soil and subsurface-sediment samples obtained from the sampling conducted prior to the ANFO detonations were immediately placed on paper plates at the WRCC laboratory and dried until air-dry equilibrium

was reached as recommended by Pratt (1978). This procedure has been shown to preserve the nitrogen forms. Samples were then ground, passed through a 2-mm sieve, and thoroughly mixed.

Saturation extracts were prepared using techniques described in "Methods of Soil Analysis" (Black, 1965). The pH was measured directly on the saturated paste. The soil-water solution was extracted by applying suction through a Buchner funnel-vacuum flask arrangement. Soil-water extracts were used to determine electrical conductivity (EC_e), NO_3-N, Ca, Mg, K, Cl, SO_4, HCO_3, and CO_3. The NH_3-N was determined on the surface soils using steam distillation techniques. Specific details of these analytical techniques are described by Black (1965).

Samples obtained in bulk for laboratory column studies were placed on large plastic sheets and dried until air-dry equilibrium was reached. They were then grounded, sieved, and thoroughly mixed. They were placed back in their respective plastic containers and sealed tightly with plastic sheets to preclude contamination and stored for use in the laboratory phase of this study.

Ground Water

Elevations. A total of eight wells were utilized for water-level measurements and ground-water sampling during the course of this study. Four of the regularly

sampled wells were polyvinyl chloride (PVC) observation wells constructed by FUGRO and were designated PR6, PR7, PR8, and PR10 (Figure 5). Two were steel-cased irrigation wells designated the north valley (NV) well and the southwest valley (SWV) well (Figure 5). The NV well was constructed at least 20 years prior to this study in anticipation of irrigation in the valley. When irrigation proved unacceptable it was abandoned. The additional wells, constructed of PVC, were discovered on the last sampling occasion and designated PR1 and PR4 (Figure 5).

Ground-water levels were measured in wells PR6, PR7, PR8, PR10, and the NV well during each sampling visit. Other organizations involved with explosive testing also monitored ground-water levels. When acquired from these agencies, these data provided a water-level record of the five wells from December 1977 to March 1979. Frequency of measurement ranged from one day to more than five months. During the last ground-water sampling visit in March 1979, the locations and water levels in the two additional wells (PR1 and PR4) were recorded.

Ground-water elevation data were used to interpret the nature of the ground-water system in the study area. A hydrograph was constructed of a well near the explosive test sites. Additionally, two water-level elevation maps were prepared in order to define the distribution of equipotential surfaces over the study area prior to

releases from Alamo Dam on January 27, 1978 and after releases had begun to recharge the alluvial deposits of the study area on March 6, 1979. These maps were used to provide an estimate of NO_3-N travel time to wells within the study area.

Sampling. Ground-water samples were obtained on five occasions. They were obtained prior to explosive testing on March 20, 1978, the day after each ANFO detonation, and approximately one and six months after the second ANFO detonation, September 29, 1978, and March 6, 1979, respectively.

Samples from PVC wells were derived by a hand bailer or positive displacement pump. The pump was also used for development of the wells. The NV well was equipped with a pump from the DOD Nevada Test Site for all samplings, except the final, when a hand bailer was used. The SWV well was equipped with a diesel pump for all samplings. Pumps were operated for a sufficient duration to ensure removal of stagnant water.

Three samples were collected from each well and stored in polyethylene containers. One sample, for the determination of CN^-, was put in a 1000-ml bottle to which NaOH was added to increase the pH to 12 in order to prevent CN^- volatilization. A second sample from each well was

preserved with $HgCl_2$ for NO_3-N determination. The third sample was untreated and used for routine chemical analyses.

In the interest of quality control, duplicate samples were collected from selected wells during all sampling visits. These samples were analyzed along with the other samples without the knowledge of the analyst to insure reproducibility of the data and also to serve as a check on the operator's analytical techniques. All samples were stored in an ice chest in the field and during transport to the WRRC laboratory in Tucson.

Chemical Analyses. The unpreserved ground-water samples were used for laboratory determination of the following parameters: electrical conductivity (EC_w), pH, Ca, Mg, Na, K, HCO_3, SO_4, and Cl. Procedures for these determinations were those recommended by the U.S. Environmental Protection Agency (1975) and the U.S. Geological Survey (Brown, Skougstad, and Fishman, 1970). The techniques used for the analysis of NO_3-N and NH_3-N were essentially those used on the saturated extracts (Black, 1965).

The CN^- determinations were done within 24 hours on all ground-water samples preserved with NaOH. The procedures used were those recommended by the U.S. Environmental Protection Agency (1975). Cation-anion balances

were compared in milliequivalent/liter (meq/l), to help check the accuracy of the analyses.

Laboratory Study

Column leachate studies were conducted to compare the mass and pattern of removal of NO_3-N and NH_3-N from soil affected by ANFO detonation, to surface and subsurface materials unaffected by detonation within the study area. Chloride removal was also quantitated in this study, because NO_3-N and Cl are known to be similarly mobile (Bohn et al., 1979).

These relationships were studied by percolating water through clear acrylic columns filled with the following three different materials: materials derived from the crater produced by the first detonation, subsurface materials from a borehole in the vicinity of GZ1 constructed prior to ANFO detonation, and native surface soil of the study area.

Column Fill Materials

The crater material used in the columns was considered by Roddy (1979) to be a mixture of soil material that came in intimate contact with the products of ANFO detonation and whose original location prior to detonation was 0 to 5.3 m below the original land surface.

The pre-blast subsurface soil was from a depth range of 4.9 m to 7.0 m, and was collected in 0.3 m

increments. The soil was provided by the WES laboratory in Vicksburg, Mississippi and was derived from a large test boring constructed prior to the first detonation at a site 75 m southwest of GZ1 (Figure 5). Equal masses of material from each increment were composited, mixed, and sieved in the same manner as the crater material and native surface soil samples.

The native surface soil was collected 670 m southwest of GZ1 (Figure 1) in an area that does not become inundated by present-day flows of the Bill Williams River. However, this site does lie on the periphery of a topographic low that channels surface water during some years from the alluvial hills lying to the north and northwest of the test sites (Figure 5). Observations of the ground surface after the blast showed that this site was not physically affected by crater ejecta propelled by ANFO detonation. Also, a comparison of the chemical analysis of material taken both before and the day after the detonation showed no change with respect to NO_3-N and NH_3-N at this site, leading to the assumption that this area was not chemically affected by fallout from the ANFO detonation.

Table 3 shows physical and chemical characteristics of the column fill materials used in the laboratory experiment. Based on the percentage of clay and silt-size particles, one would expect the rate of water movement through the three column materials to be the highest for the

Table 3. Physical and chemical characteristics of column fill materials

	Crater Material	Pre-blast Subsurface Soil	Native Surface Soil
Textural Class	Loam	Sand	Sandy Loam
Sand	39.6%	95.1%	70.2%
Silt	44.6%	1.0%	23.6%
Clay	15.8%	3.3%	6.2%
PH (paste)	8.2	8.0	8.0
EC (extract)	2010 µmhos/cm	200 µmhos/cm	1340 mhos/cm
Soluble Salts	1407 mg/l	140 mg/l	938 mg/l
Na	69.2 mg/l	28.5 mg/l	48.3 mg/l
Cl	401 mg/l	14 mg/l	81 mg/l
Total N	189.2 mg/l	61.1 mg/l	403.0 mg/l
Org-N	175.5	59.1	381.2
NH_3-N	8.6	0.9	2.8
NO_3-N	4.2	1.0	18.0
NO_2-N	1.0	0.1	1.0
Organic Matter	0.39%	0.00%	0.62%

pre-blast subsurface soil and lower for the native surface soil and crater soil, in that order.

The soluble salt content of the crater material is greater than either the native surface soil or the pre-blast subsurface soil (Table 3). This could possibly be due to the concentration of salts in this material by evaporation, as a result of the high temperature produced by ANFO detonation, or it may be solely attributed to spatial variability of soil constituents in the study area.

The organic N content of the native surface soil is greater than either that of the crater material or the pre-blast subsurface soil (Table 3). This is due to the fact that organic matter from native plant material has been incorporated into the soil over the years.

Soil Columns

Soil columns were constructed of clear, acrylic pipe 8.7 cm inside diameter and 32.0 cm long. Inlet tubing was placed 10.3 cm from the top of the column. A perforated disc with 2-mm openings was secured 2 cm from the bottom of the column to allow the leachate to drip onto a beveled Plexiglas column bottom and out a 0.95-cm hole drilled to accommodate the outflow tubing.

Procedures

Six soil columns were used in this experiment, two for each of the three soils. Two repetitions of this experiment were performed.

Soil columns were prepared as follows. To prevent soil losses through leaching, a thin layer of glass wool was placed at the bottom of the columns. Columns were then labeled and weighed. Homogeneous mixtures of the three column materials were packed as uniformly as possible to minimize channeling effects along the outer walls. About 60 cm^3 of soil was initially filled through a funnel with a rubber hose and packed with an aluminum rod. Packing continued until each column attained a total flow length of 17.5 cm. Columns were then reweighed to determine the mass of soil added to each column. Bulk density was then determined. Glass wool matting was then placed on the top of the soil to keep soil in place. Figure 7 shows a schematic diagram of a packed soil column.

All six columns were mounted on a wooden frame. A plastic-pipe, constant-head manifold was used to supply the columns with double-deionized water made to contain 1% formaldehyde by volume. This solution was used to preclude microbial activity for the duration of the experiments (Warrick, 1979). The manifold was mounted in the upper rear of the wooden frame with the center of the manifold 4.5 cm below the top of the columns to supply a

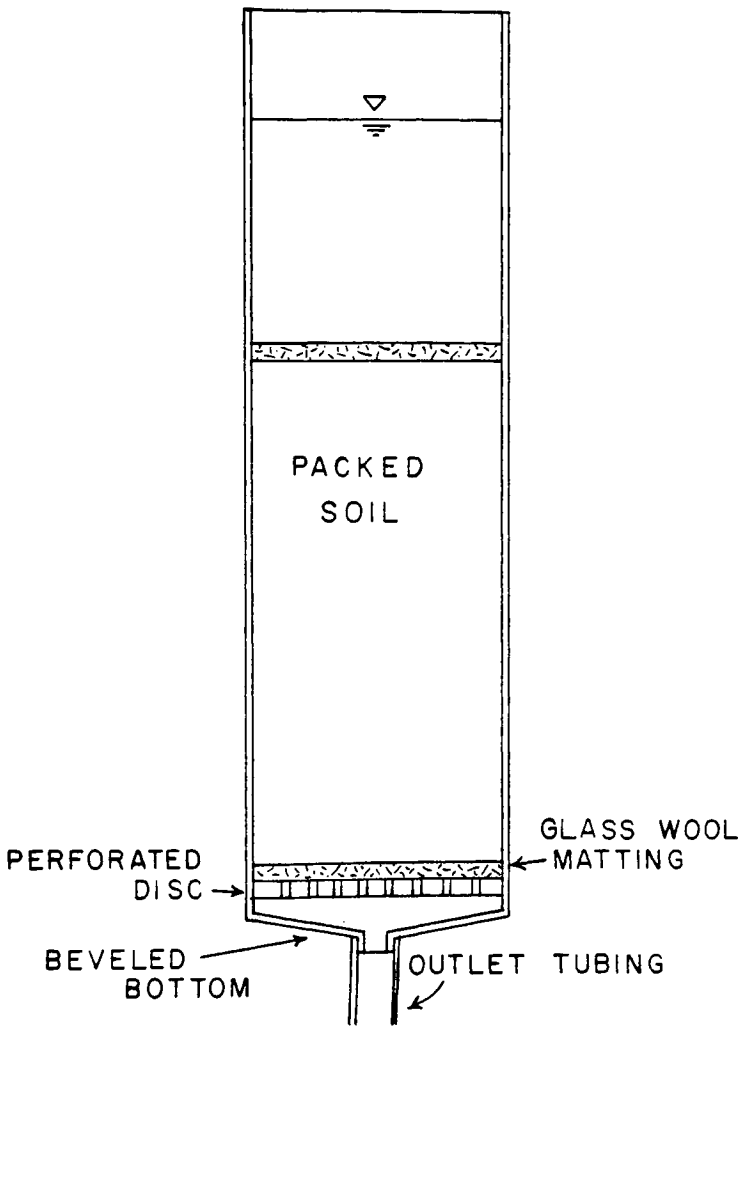

Figure 7. Schematic diagram of packed soil column

constant pressure head of 8.5 cm to all columns, once flooding of the manifold and columns commenced. As the leaching front moved down the initially unsaturated columns a total hydraulic head difference of 26 cm was eventually maintained for all columns. Overflow from the constant-head manifold was collected in a 24-l plastic container and delivered back to the water-supply reservoir located above the manifold by a small centrifugal pump. Water-supply transport and hydraulic connection of the system was accomplished through polyethylene tubing. Figure 8 shows a schematic diagram of the experimental apparatus.

After completion of each repetition of this experiment, the columns were drained to the soil surface and weighed. The weight of the water contained in the soil volume was used to calculate porosity and pore volume.

Table 4 lists some of the physical characteristics of the soil columns. As is evident, the mass of column material was not the same for individual replicates of each column fill material. Instead, the soil volume and flow length of each column were constant for each replicate. Larger bulk density values are associated with a larger mass of column material. Also, porosity and pore volume were not the same for the three column materials or for individual replicates (Table 4). The higher percentages of clay and silt-size particles in the crater and native surface soil materials (Table 4) effected

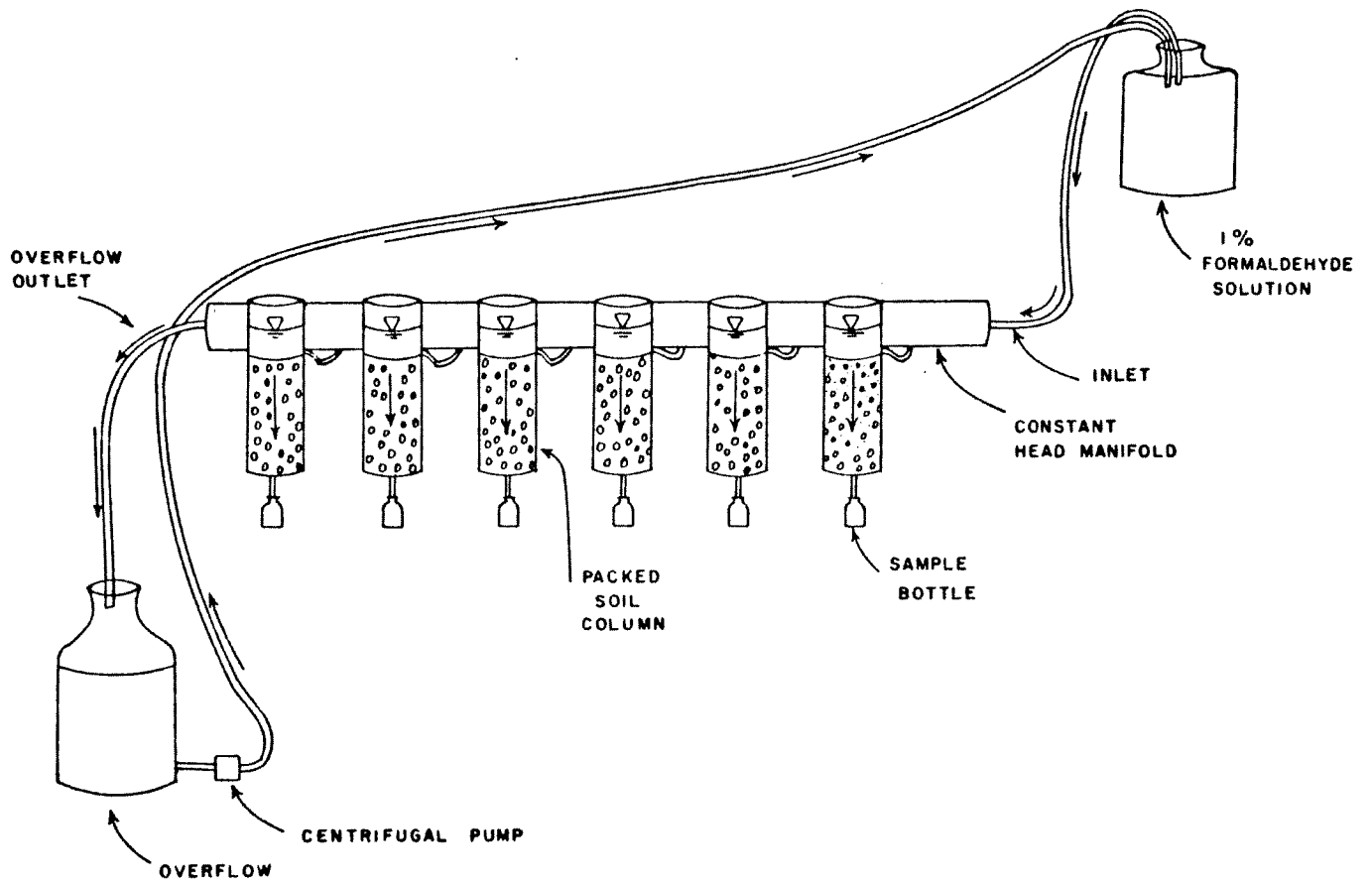

Figure 8. Schematic diagram of experimental apparatus

Table 4. Physical characteristics of soil columns

Soil	Repetition	Replicate	Mass Soil (kg)	Bulk Density (g/cm^3)	Porosity (%)	Pore Volume ml
Crater Material	I	1	1.726	1.62	35.07	373
		2	1.752	1.64	35.54	378
	II	3	1.807	1.70	36.79	392
		4	1.842	1.73	36.77	391
Pre-blast Subsurface Soil	I	1	2.022	1.90	29.98	319
		2	1.983	1.86	30.50	325
	II	3	1.991	1.87	31.67	337
		4	1.934	1.82	31.06	331
Native Surface Soil	I	1	1.909	1.79	34.45	367
		2	1.911	1.80	34.54	368
	II	3	1.959	1.84	35.28	376
		4	1.916	1.80	34.58	368

higher values of porosity and pore volume for these materials.

Chemical Analyses

Leachate was removed in 60-ml increments for analysis. The NO_3-N, EC_w, and pH were monitored during the course of the experiment to provide an indication of equilibrium conditions between the column-fill material and the percolating solution. When NO_3-N concentration of the leachate fell below 2 mg/l, removal of leachate was discontinued. The pH and EC_w were determined using a standard glass electrode pH meter and a conductivity bridge, respectively.

To study the effects outlined previously, the leachates were analyzed quantitatively for NO_3-N, NH_3-N, and Cl. The NO_3-N and NH_3-N were determined by Bremner's steam distillation technique (Black, 1965). Chloride was determined using the mercuric nitrate method (American Public Health Association, 1971).

Statistical Design

Six main effects were analyzed statistically (Marx, 1979). These effects included the difference in total mass of NO_3-N and NH_3-N removed from the crater material and the controls, the pre-blast subsurface soil, and the native surface soil. Because the mass of column material was not the same for the three soils or for

individual replicates, chemical concentrations of the constituents were normalized per kilogram of column material.

Chemical concentration was assumed to follow a negative exponential relationship with increasing volume of solution passed through the columns. In the following function,

$$y = \alpha e^{-\beta x}$$

y equals the chemical concentration, x is the cumulative volume of solution passed through a column, α represents the y-intercept, and β represents the degree of descent of the curve. The parameters α and β were estimated using nonlinear, least-squares, regression analysis. Functions were then integrated to obtain the total mass of the particular constituent eluted from the column.

RESULTS AND DISCUSSION

Field Study

Preliminary Soil and Subsurface Sediments

Several months prior to the ANFO detonations, soil and subsurface sediment samples were obtained from borings made near each testing site. Four borings were made at each testing site. Table 5 shows the concentrations of selected ionic constituents in saturated extracts of soil and subsurface sediments derived from borings made near GZ1. Table 6 shows similar data from borings made near GZ2.

Data for samples from borings at both detonation sites indicate that chemical constituents vary both vertically within the profiles and spatially in the vicinity of the GZs. Borings near GZ1 suggest an accumulation of salts from 61-152 cm below the land surface. This accumulation represents the illuviation of salts from the surficial silt layer to the underlying sand layer by ponded water derived from overland flow. The NO_3-N values ranged from less than 1 to 59.8 mg/l in all samples collected at GZ1. The highest NO_3-N concentrations occurred in the 30-61-cm and the 91-122-cm horizons of samples DS3 and DS4. Data for samples at GZ2 also varied vertically and spatially, although

Table 5. Concentrations of selected ionic constituents in saturated extracts from soil and subsurface sediment borings near GZ1

Sample	Depth	Ca	K	Na	NO_3-N	Cl	SO_4
	cm	----------	----------	mg/l	----------	----------	----------
DS1	0- 30	39	21	32	1.0	16	0
	30- 61	15	11	116	1.7	39	30
	61- 91	20	9	133	1.3	42	80
	91-122	47	12	106	1.3	42	30
	122-152	200	11	50	1.0	16	25
DS2	0- 30	30	19	50	0.7	32	0
	30- 61	49	31	100	7.3	35	65
	61- 91	420	174	106	5.8	433	1960
	91-122	330	112	122	20.0	878	1750
	122-152	60	17	134	11.0	274	85
	152-183	198	24	128	14.6	698	75
	183-213	120	24	146	13.2	281	50
	213-244	35	10	89	4.7	82	25
	244-274	52	20	112	6.3	149	50
	274-305	51	9	90	4.5	107	65
DS3	0- 30	76	54	83	8.2	60	10
	30- 61	310	42	48	54.8	125	125
	61- 91	550	40	116	4.4	548	215
	91-122	490	63	120	50.8	681	800
	122-152	245	49	125	22.8	528	95
	152-183	152	31	105	15.1	296	40
	183-213	95	20	84	11.2	232	25
DS4	0- 30	70	24	46	3.7	52	0
	30- 61	340	30	102	59.8	229	410
	61- 91	780	74	115	13.1	614	1250
	91-122	390	64	127	25.5	665	180
	122-152	230	40	138	14.8	753	74
	152-183	190	35	126	17.1	458	40
	183-214	167	31	63	14.9	232	25

Table 6. Concentrations of selected ionic constituents in saturated extracts from soil and subsurface sediment borings near GZ2

Sample	Depth	Ca	K	Na	NO_3-N	Cl	SO_4
	cm	mg/l					
DSB1	0- 30	460	198	188	34.1	370	1880
	30- 61	890	233	186	13.8	3262	2360
	61- 91	285	47	120	22.6	1159	1080
	91-122	170	30	190	15.4	704	300
DSB2	0- 30	495	152	128	4.9	76	2120
	30- 61	450	165	192	3.7	98	2720
	61- 91	225	91	109	8.8	471	1800
	91-122	405	45	195	11.6	1214	1120
	122-152	415	51	128	11.2	1069	640
DSB3	0- 30	125	91	113	5.5	166	295
	30- 61	77	59	158	2.8	491	200
	61- 91	165	42	187	5.0	730	860
	91-122	440	41	109	10.9	800	2240
	122-152	430	41	167	17.5	1197	345
DSB4	0- 30	340	98	128	9.2	75	1600
	30- 61	460	60	125	48.5	375	2000
	61- 91	240	79	114	32.1	401	680
	91-122	210	33	130	13.5	503	185
	122-152	230	25	172	17.6	577	285

the variability of most chemical constituents is not as pronounced as in samples collected near GZ1. This is attributed to the fact that the surficial silt layer pinches out in the vicinity of GZ2, resulting in a more uniform soil texture throughout the profiles, and GZ2 does not become inundated by waters derived from overland flow. The NO_3-N values ranged from 2.8 to 48.5 mg/l in all samples collected at GZ2.

It can be concluded that considerable NO_3-N existed at some depths in the desert alluvium in the vicinity of GZ1 and GZ2 prior to the detonation of ANFO.

Ground Water

Elevations. During the winter and spring of both 1978 and 1979, water was released from Alamo Dam. These releases contributed significant recharge to the ground water, which resulted in large annual fluctuations of ground-water levels in the study area.

This fluctuation is shown in a hydrograph of observation well PR6 located 320 m southwest of GZ1 (Figure 9). The hydrograph shows that the lowest water levels occurred in January and February of 1978, and then rose to a maximum in late May due to releases from Alamo Dam. The hydrograph indicates that after the peak in May the water table began to decline to another minimum level in early 1979 and rose to another high level in 1979 in response to the next

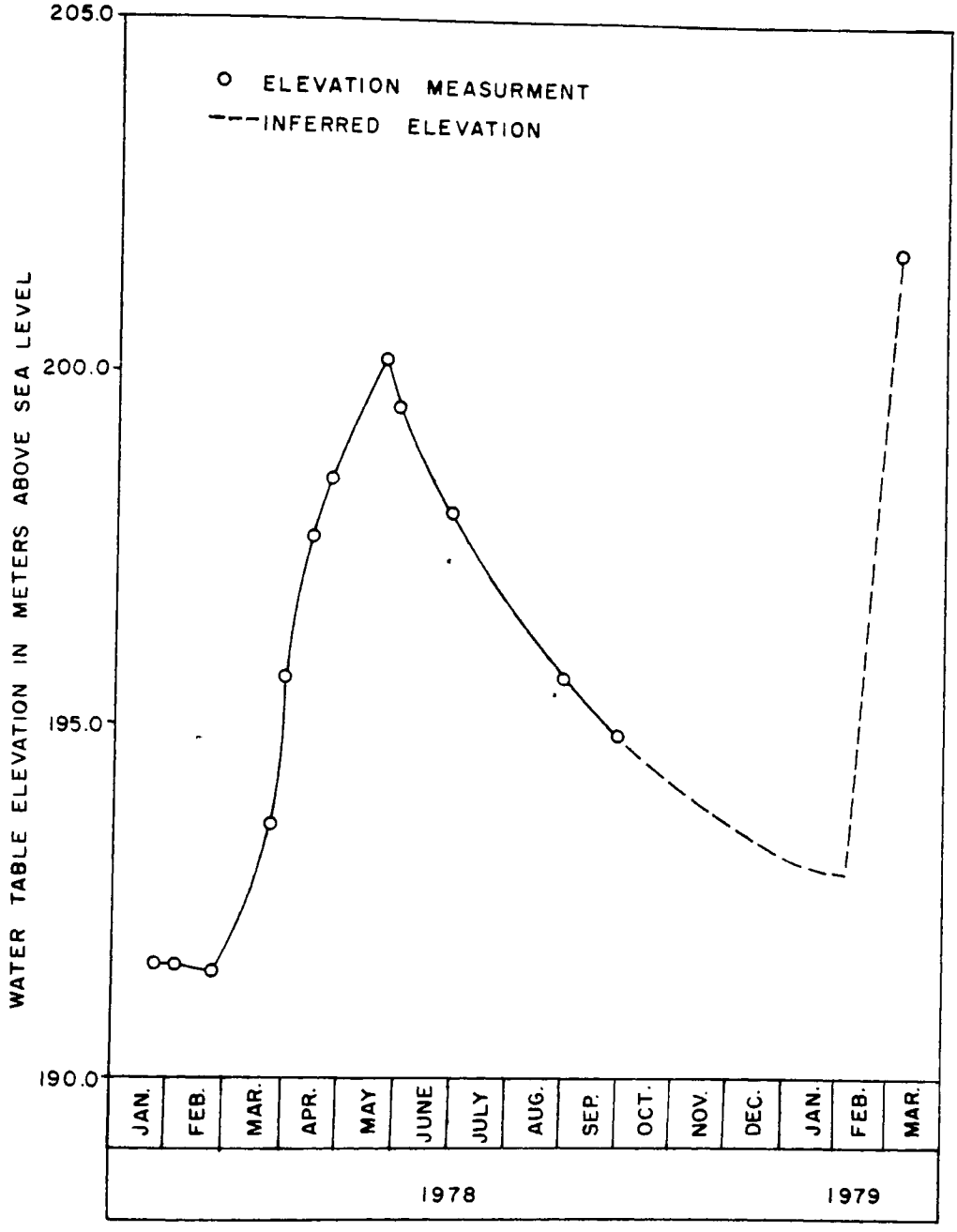

Figure 9. Well hydrograph of PR6

releases from the dam. Total measured water-level fluctuation in PR6 was 8.7 m.

It is estimated that on June 29, 1978, the day after the single ANFO detonation, the ground-water elevation was 2 to 3 m below the crater base. On the day after the multi-charge detonation, August 31, 1978, it is estimated that the ground-water elevation was 3 to 4 m below the bases of the six craters produced.

As is evident from the hydrograph, water levels continued to decline beneath the craters following the detonations until about the beginning of February 1979. Therefore, alluvial materials that had been disturbed by ANFO detonation and had come in contact with its detonation products did not become saturated with ground water until sometime in early February 1979.

Borings augered at GZ1 on March 6, 1979 showed that the ground-water level was 1.5 m below the land surface. Given the aforementioned crater depth of 5.3 m at this location, at least 3.8 m of ANFO-affected material had become saturated with ground water by this date.

To understand the ground-water flow system within the alluvial fill of the valley during the course of this study, two water-level contour maps were prepared. The first (Figure 10) represents the water-level contour on January 27, 1978 prior to that year's releases from Alamo Dam and hence the most steady ground-water levels of the

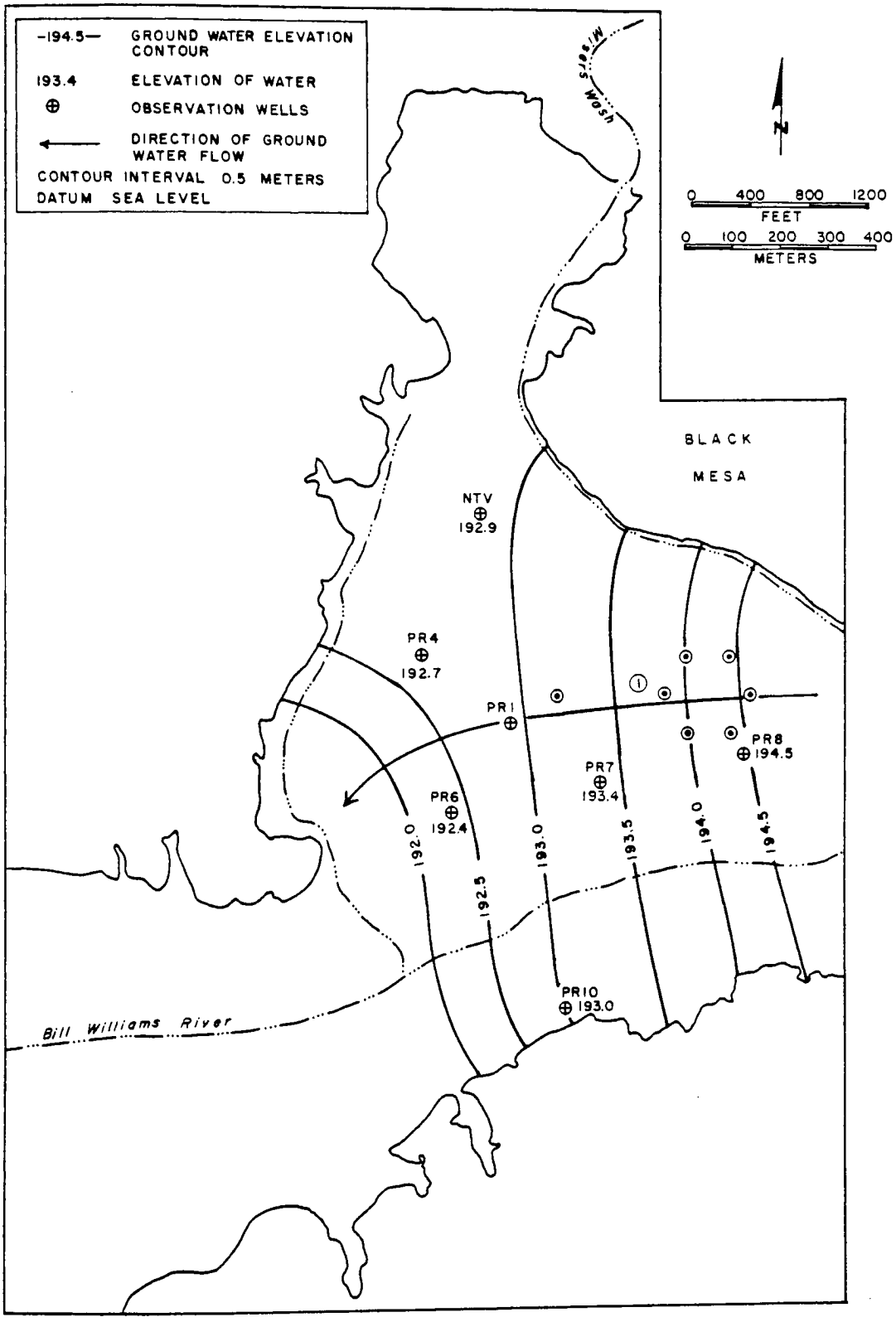

Figure 10. Water level contour map of January 27, 1978

water year. The second (Figure 11) represents the water-level contour on March 6, 1979, the day of the last ground-water sampling visit. This sampling was performed after releases from the dam had begun to recharge the study area and ground-water levels were therefore very unsteady.

Despite the different ground-water conditions represented by these two contour maps, the flow path through the ANFO detonation sites is clearly similar for the two conditions. Under both conditions, observation well PR1 is along the flow path of ground water that had passed through alluvial material physically disturbed by ANFO detonation and in close contact with its detonation products.

Chemical Analyses. Major cations and anions were analyzed in water samples. Ground water of the study area can be characterized as calcium, sodium, magnesium, bicarbonate, chloride, and sulfate water. Appendix A contains results of individual water analyses. The overall chemical quality of ground water is excellent, with concentrations of all constituents well below recommended limits for both agricultural and municipal uses.

Table 7 lists the results of NO_3-N and NH_3-N analyses of water samples from wells PR6, PR7, PR8, PR10, NV, and SWV for the five sampling visits. Also shown are the NO_3-N and NH_3-N analyses for wells PR1 and PR4, which were

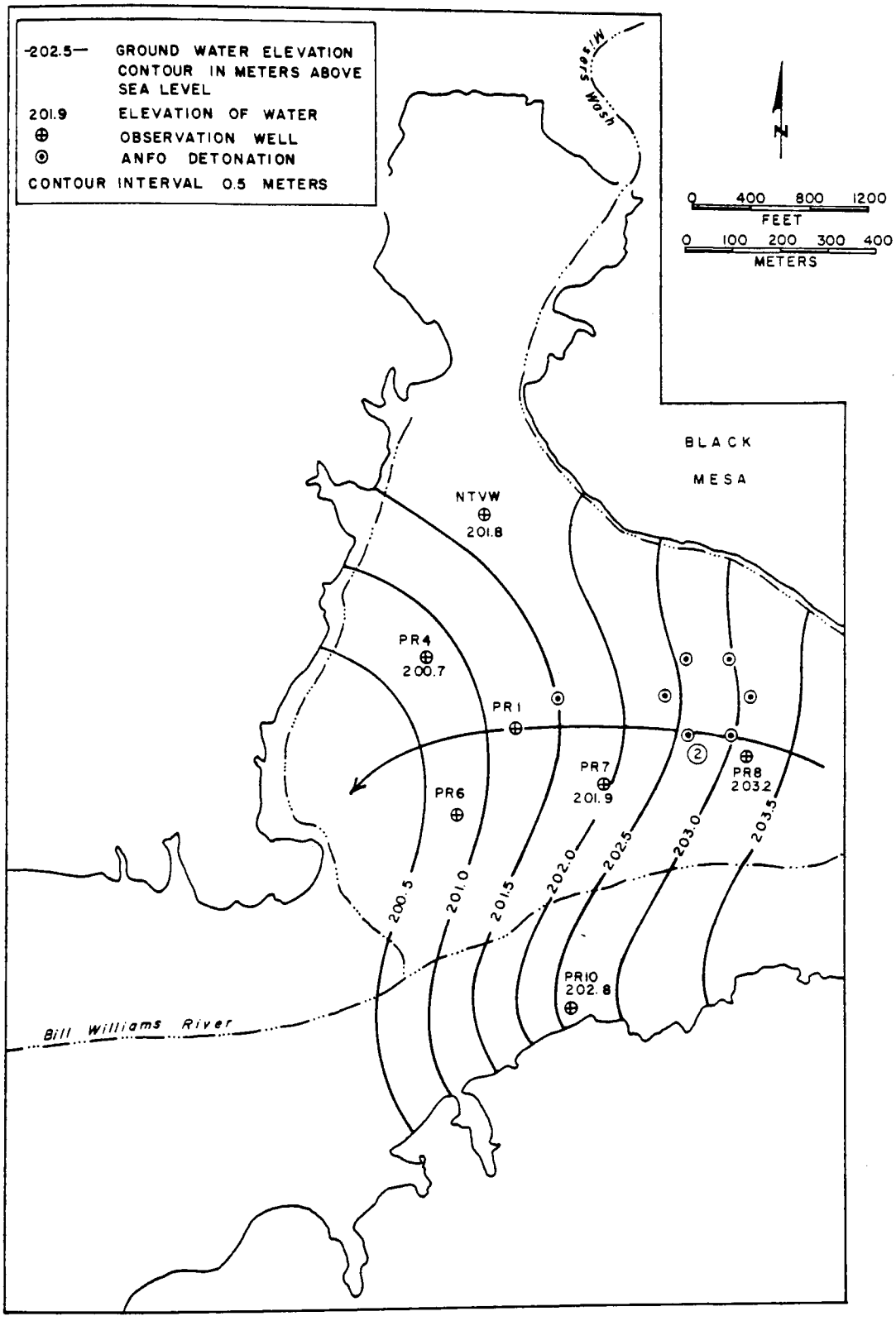

Figure 11. Water level contour map of March 6, 1979

Table 7. NO_3-N and NH_3-N analyses for all wells

Date	Analysis	Well							
		PR1	PR4	PR6	PR7	PR8	PR10	NV	SWV
		mg/l							
3-20-78	NO_3-N	--	--	0.2	0.0	0.0	0.2	0.1	0.2
	NO_3-N	--	--	0.0	0.0	0.0	0.0	0.0	0.0
6-29-78	NO_3-N	--	--	3.7	6.0	1.1	1.1	1.0	--
	NH_3-N	--	--	0.3	0.0	0.0	0.4	0.0	--
8-31-78	NO_3-N	--	--	4.9	2.5	0.0	1.5	0.6	0.9
	NH_3-N	--	--	0.7	0.0	0.2	0.2	0.0	0.4
9-29-78	NO_3-N	--	--	5.4	1.8	0.3	1.6	0.6	0.9
	NH_3-N	--	--	0.0	0.1	0.0	0.1	0.0	0.0
3-06-79	NO_3-N	22.0	0.9	2.5	3.4	1.0	1.1	0.8	1.6
	NH_3-N	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

only sampled on March 6, 1979. The data for CN^- analysis are not included because all samples were below the detection limit of 15 ppb.

The NH_3-N concentration in all wells ranged from 0 to 0.7 mg/l, with 74 percent of the samples containing no detectable NH_3-N (Table 7). This indicates that the redox potential (Eh) of the ground water is probably high and that the nitrifying bacteria have readily effected the transformation of NH_3-N to NO_3-N.

The highest NO_3-N concentration observed during this study was from the sample obtained from well PR1 (Table 7). As previously noted, PR1 was the only well expected to receive ground water that had traveled through the alluvium affected by the ANFO detonations, based on the water-level contour maps alone.

Travel time of ground water was computed between points at two of the detonation sites of the multi-charge event and well PR1 along the flow lines of Figures 10 and 11. Travel times were determined using Darcy's equation in the following form,

$$t = \frac{(L^2)\ n}{(K)\ \Delta H}$$

where t = travel time, days
L = travel distance along flow line, m
n = porosity, dimensionless
K = hydraulic conductivity, m/day
ΔH = head loss between points on flow line, m

Travel times were calculated using an average value for K of 951 m/day as reported by Turner (1962) for Planet Valley alluvium and two different estimates of porosity. An estimated low value of porosity of 0.15, equal to the specific yield of Bill Williams River alluvium as reported by Wolcott et al. (1956), was chosen to yield a low estimate of travel time. The mean porosity of coarse sand, 0.3 (Bouwer, 1978), was chosen as an estimate of the high value to approximate a high estimate of travel time.

Travel time of ground water from the westernmost detonation site (crater 1 in Figure 10) of the multi-charge event to PR1 along the flow line of Figure 10 was computed to be 17.35 days for the lower estimate and 34.70 days for the higher estimate. Travel time of ground water from the southwesternmost detonation site (crater 2 in Figure 11) of the multi-charge event to PR1 was determined from the flow line of Figure 11 to be 18.6 days for the lower estimate and 37.2 days for the higher estimate.

Calculated travel times for the two ground-water conditions represented by Figures 10 and 11 show close agreement. The occurrence of the high NO_3-N concentration of well PR1 on March 6, 1979 correlates well with the saturation of the ANFO-affected alluvium during the first or second week of February 1979 in the vicinity of the multi-charge detonation. Travel time calculations tend to indicate further that the high NO_3-N concentration of PR1

on March 6, 1979 is due to the travel of ground water through the ANFO-affected material.

Observation wells PR8 and PR10 and the NV and SWV wells showed little variation of NO_3-N concentration throughout the study. Observation wells PR6 and PR7 illustrated high values of 5.4 and 6.0 mg/l NO_3-N, respectively. These values may not be attributed to ANFO detonation and probably are the result of slow diffusion of NO_3-N through relatively impermeable strata that lie below areas where NO_3-N was leached by water derived from overland flow.

Laboratory Study

An analysis of variance was performed on the data from the soil column tests using groups with an equal number of replicates (Ostle, 1960). The values of F for α and β for the two measured constituents in the column leachate, NO_3-N and NH_3-N were calculated for the three soils. The F values were used to determine if the crater material differed from the two controls, comprising pre-blast subsurface soil and native surface soil, with respect to the leaching of NO_3-N and NH_3-N. In addition, the least significant differences (LSD) at the 5 percent probability level were calculated for each of the α and β associated with each NO_3-N and NH_3-N. The LSDs were used to detect the significant differences at the stated probability level among the mean values for α and β for each constituent and for each soil.

Also, an analysis of variance was performed on the total mass of NO_3-N and NH_3-N removed from the three soils, per kg of column fill used in each particular replicate. Again the values of F were calculated to determine if there were differences between the three soils. In addition, the values of LSD were calculated for the total amount of NO_3-N and NH_3-N.

The mean relative concentration (C/C_{max}) was plotted versus pore volume, for each chemical constituent, and for each soil; C equals the concentration in mg/l/kg soil at any value of pore volume, and C_{max} equals α. Also, the mean concentration values of NO_3-N and NH_3-N observed in this experiment were tabulated in 0.1-pore volume increments of the leachate removed.

Chloride removal was also measured to compare the removal of Cl with that of NO_3-N. Although not statistically analyzed, the mean relative concentration of Cl was plotted versus pore volume, and the mean concentrations of Cl that were observed in this experiment were tabulated to effect this comparison.

Nitrate Nitrogen

The mean values of α, β, and the total mass of NO_3-N removed from the three soils are given in Table 8. Also given are the values of F and the values of LSD at the 5 percent probability level.

Table 8. The mean values of α and β, the mean total mass of NO_3-N removed from the three soils, the values of F, and the LSD at the 5% probability level

Soil	Composition	α	β	NO_3-N
		mg/l/kg soil	l/ℓ	mg/kg soil
I	Crater material	299.108	16.643	14.723
II	Pre-blast subsurface soil	122.070	85.400	1.125
III	Native surface soil	289.163	14.705	16.395
F		0.541	0.029	0.023
LSD(.05)		No significant difference	54.883	10.909

The calculated F values for β and total NO_3-N indicated that there were significant differences between soils. The β of the pre-blast subsurface soil (soil II) was significantly higher than the βs for the crater material (soil I) and the native surface soil (soil III). This indicates that the NO_3-N concentration of soil II was reduced by a smaller pore volume of solution than was required for soils I and III. The total NO_3-N removed from soil II was significantly lower than the amount removed from soils I and III, which showed no significant differences (Table 8).

Figure 12 illustrates the pattern of NO_3-N removal from the three soils. The pattern of NO_3-N removal is essentially the same for soils I and III. NO_3-N concentration was reduced by more than 80 percent of the original values by the 0.4 pore volume for soils I and III, whereas soil II only required 0.1 pore volume to reduce concentration to the same degree. Table 9 gives the reader an idea of the observed NO_3-N concentration reduction with increasing pore volume.

It is concluded that there was no significant difference between the total mass of NO_3-N removed from the crater material (soil I) and the native surface soil (soil III) nor was there a significant difference in the pattern of NO_3-N removal from these soils. Also, it is concluded that a significant difference existed between the mass of

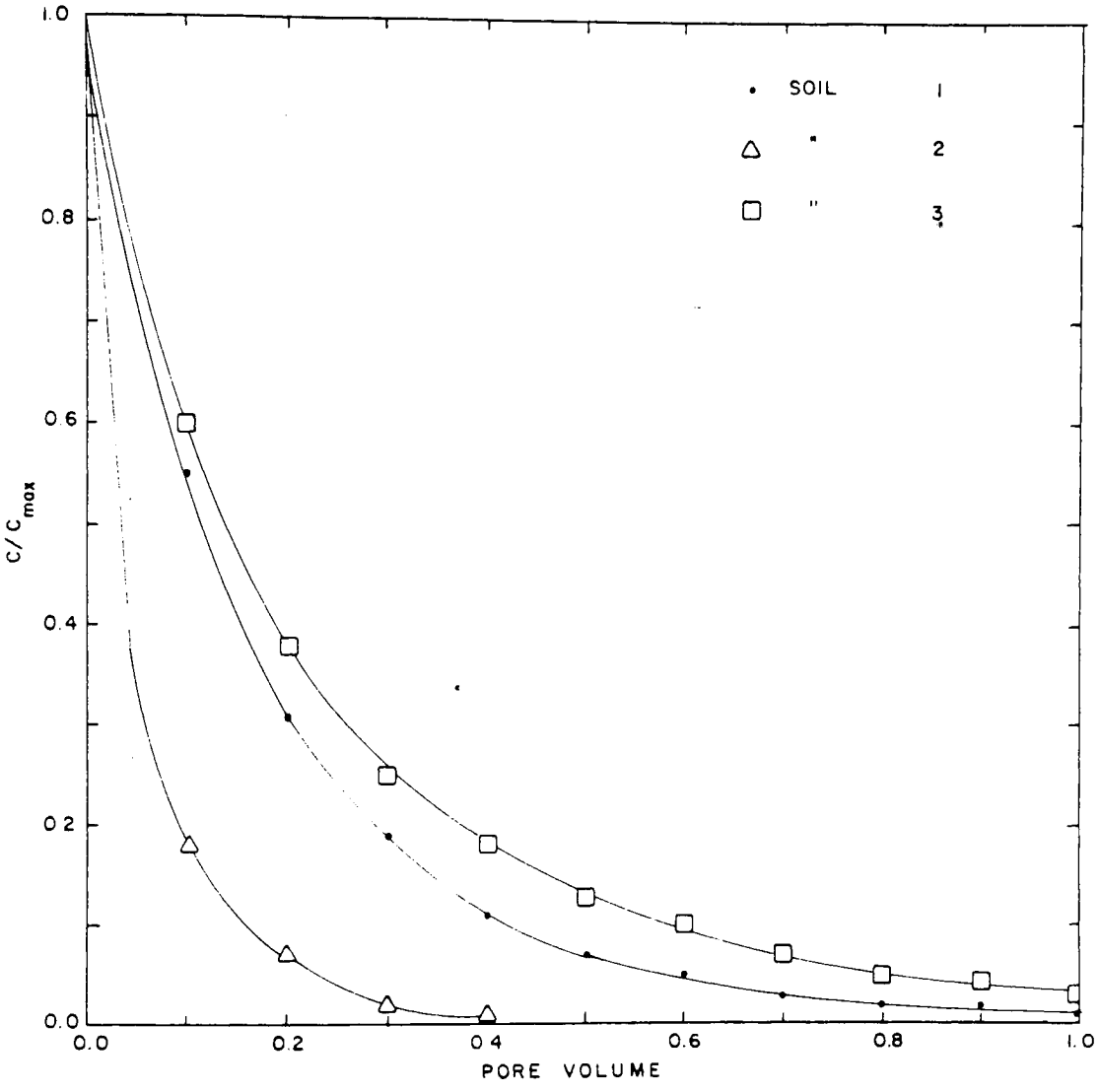

Figure 12. The mean relative concentration (C/C_{max}) of NO_3-N removed from the three soils

Table 9. The mean NO_3-N concentration values observed in leachate removed from the three soils

Soil	Composition	1/10 pore volume of the leachate removed									
		1	2	3	4	5	6	7	8	9	10
		mg/l/kg									
I	Crater material	132.8	61.5	29.9	15.3	8.2	4.6	2.7	1.7	1.1	0.7
II	Pre-blast sub-surface soil	4.6	1.1	0.4	0.1	0.0	0.0	0.0	0.0	0.0	0.0
III	Native surface soil	140.3	71.5	38.5	22.1	13.6	8.8	6.0	5.7	3.1	2.3

NO_3-N liberated from the pre-blast subsurface soil (soil II) and that eluted from the crater material (soil I) and the native surface soil (soil III), the former liberating a significantly lower mass of NO_3-N, with a significantly smaller pore volume of solution.

Ammonia Nitrogen

The mean values of α, β, and the total mass of NH_3-N removed from the soils and the values of F and LSD are given in Table 10. The calculated F values for all parameters, α, β, and total NH_3-N, suggest significant differences between all soils.

The mean values α and β of soil II were significantly higher than the values for soils I and III, which showed no significant differences. The mass of NH_3-N leached from soil II was significantly lower than that leached from soils I and III. Soils I and III do not show significant differences with respect to the total NH_3-N removed.

Figure 13 illustrates the relative concentration of the NH_3-N removed from the three soils. Table 11 shows the NH_3-N mean concentrations observed in this experiment. Both show similar patterns of NH_3-N removal for soils I and III, contrasting to that of soil II.

It is concluded that a significant difference did not exist between the total mass of NH_3-N removed from the

Table 10. The mean values of α and β, the mean total mass of NH_3-N removed from the three soils, the values of F, and the LSD at the 5% probability level

Soil	Composition	α	β	NH_3-N
		mg/l/kg soil	l/ℓ	mg/kg soil
I	Crater material	4.5650	10.3500	0.4725
II	Pre-blast subsurface soil	12.0250	76.9900	0.0800
III	Native surface soil	3.4350	7.5975	0.4500
F		0.004	0.001	0.012
LSD(.05)		3.0825	5.1068	0.2528

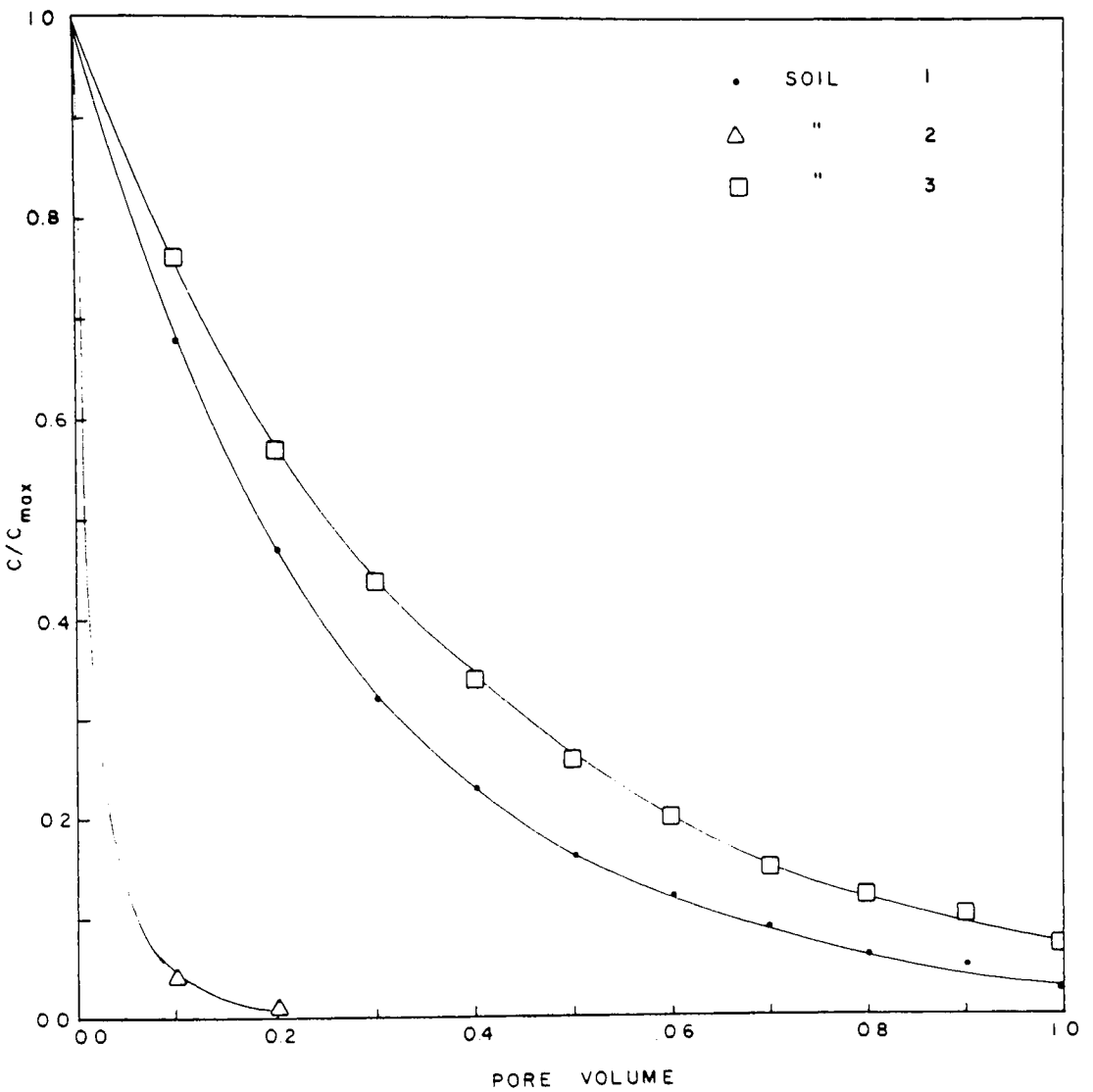

Figure 13. The mean relative concentration (C/C_{max}) of NH_3-N removed from the three soils

Table 11. The mean NH_3-N concentration values observed in leachate removed from the three soils

Soil	Composition	1/10 pore volume of the leachate removed									
		1	2	3	4	5	6	7	8	9	10
		mg/l/kg									
I	Crater material	3.07	2.09	1.43	1.00	0.70	0.50	0.37	0.27	0.20	0.13
II	Pre-blast sub-surface soil	0.51	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
III	Native surface soil	2.57	1.92	1.44	1.08	0.82	0.70	0.53	0.36	0.27	0.21

crater material (soil I) and the native surface soil (soil III) nor was there a significant difference in the pattern of removal of this constituent. Also, a significantly lower mass of NH_3-N was removed from the pre-blast subsurface soil (soil II) than was removed from the crater material (soil I) or the native surface soil (soil III).

Chloride

Figure 14 shows the mean relative concentration reduction of Cl in the three soils with increasing pore volume of water passed through the columns. Table 12 lists the mean Cl values observed in this experiment.

Soils I and III clearly show similar patterns of removal of Cl, whereas soil II required a larger pore volume of water to leach Cl, relative to the initial concentration (Figure 14 and Table 12).

Soils I and III also show similar patterns of NO_3-N removal. However, larger increments of pore volume were required to leach NO_3-N than Cl (Figure 12 and Table 9). These observations suggest the attenuation of NO_3-N in soils I and III, assuming that NO_3-N and Cl are similarly mobile. Bohn et al. (1968) indicated that the only difference between NO_3-N and Cl, with respect to their mobility in soils, is the amenability of NO_3-N to attenuation by microorganisms. As microorganisms were assumed to

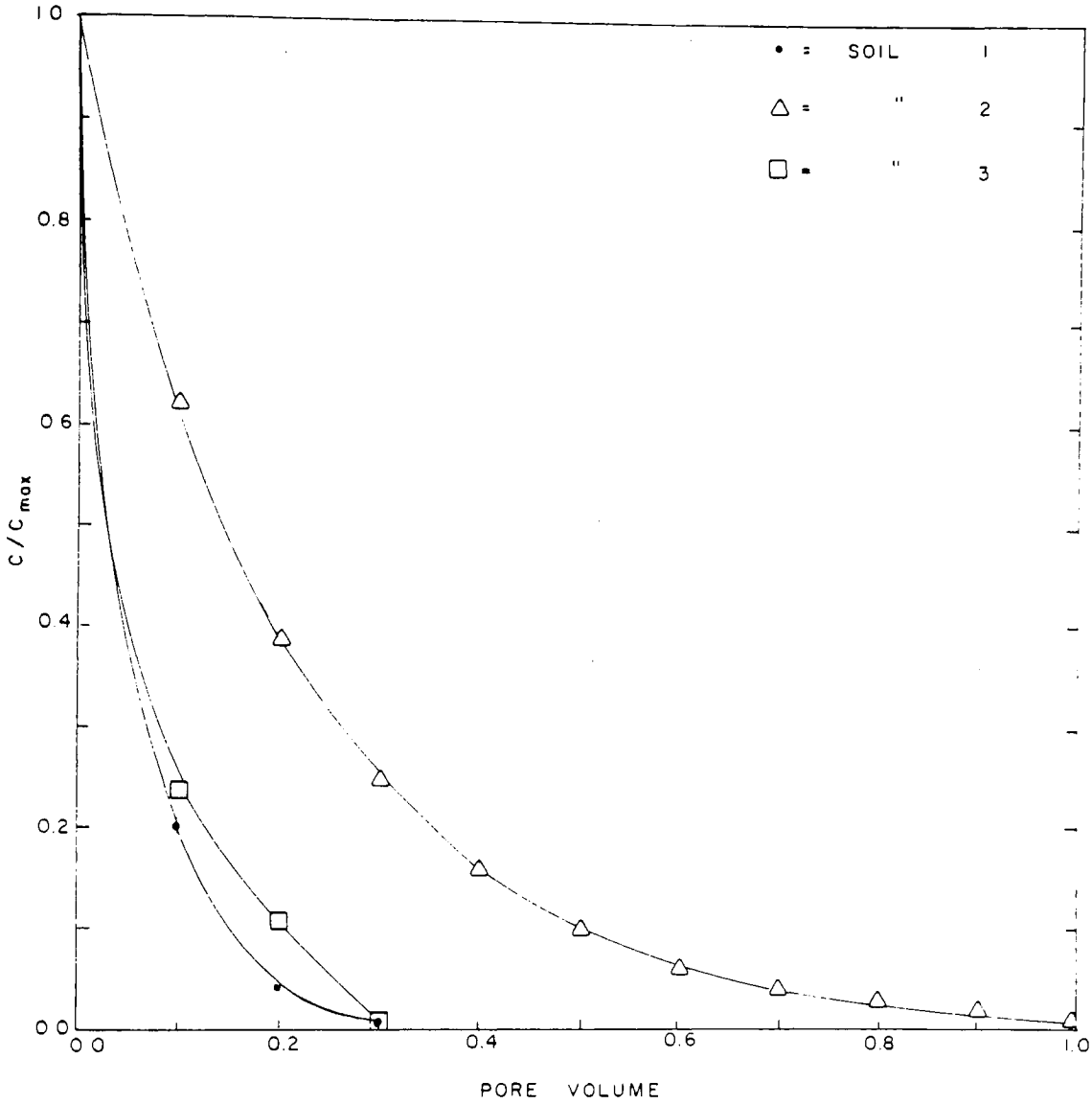

Figure 14. The mean relative concentration (C/C_{max}) of Cl^- removed from the three soils

Table 12. The mean Cl^- concentration values in leachate removed from the three soils

Soil	Composition	1/10 pore volume of the leachate removed									
		1	2	3	4	5	6	7	8	9	10
		mg/l/kg									
I	Crater material	5510.8	1118.2	232.2	49.2	10.6	2.3	0.5	0.1	0.0	0.0
II	Pre-blast sub-surface soil	21.4	13.1	8.1	5.0	3.1	2.0	1.3	0.8	0.5	0.3
III	Native surface soil	371.8	88.8	21.4	5.2	0.9	0.2	0.1	0.0	0.0	0.0

be exterminated by formaldehyde solution, this is not considered likely.

INTERRELATIONSHIP OF THE FIELD AND LABORATORY STUDIES

The results of the NO_3-N analyses of saturated extracts of materials collected in the proximate vicinity of the ANFO detonation sites prior to detonation evinced that considerable NO_3-N existed in these areas to at least a depth of 3 m.

Upon initiation of ANFO, these same materials, from at least a depth of 5.3 m, were subjected to the forces generated by gaseous expansion of the ANFO detonation products and were consequently ejected from their original locations. They were deposited in a clearly defined area of the valley.

This ejected alluvial material was used to fill the craters following military testing. This operation mixed alluvial material which was both locally high in NO_3-N concentration prior to military testing and also exposed to the products of ANFO detonation, which were largely nitrogenous.

Ground-water elevation, flow, and water-quality investigations conducted during this study showed that the occurrence of the high NO_3-N concentration of PR1 correlated with the saturation of the crater-fill material in the vicinity of GZ2 with ground water and with the

calculated travel times of ground water to reach this well from that area.

The source of the NO_3-N that occurred in well PR1, either natural soil NO_3-N present prior to ANFO detonation or NO_3-N introduced to the alluvial material of the study area as a residual of ANFO detonation, or both, is not easily determined. Laboratory study clearly showed that there was no significant difference between the mass of NO_3-N liberated by the crater material and that removed from the native surface soil, upon saturation of each with water. Conversely, the results of the laboratory study showed that there was a significantly higher mass of NO_3-N eluted from the crater material than from the pre-blast subsurface soil. The question of which alluvial material composed the background of the crater material, either high NO_3-N surface soil or low NO_3-N subsurface soil, is indeterminate.

If high NO_3-N surface soil composed the background of the material obtained from the crater, then one could conclude that there were no nitrogenous ANFO residuals deposited by the ANFO detonation. Conversely, if low NO_3-N subsurface soil composed the background of the material from the crater that came in contact with the ANFO detonation products, then one could conclude that a large mass of nitrogenous ANFO residuals were deposited by detonation. However, any conclusions as to the source of

NO_3-N observed in well PR1 that would ignore the origin of the alluvial material in the crater soil would be purely conjectural.

Therefore, the source of NO_3-N observed in well PR1 was NO_3-N removed by ground water from a mixture of surface and subsurface soil that was variably high in NO_3-N prior to detonation. Upon detonation, this material came in contact with gaseous ANFO detonation products, which may or may not have contributed nitrogenous residuals to the physically disturbed alluvial material.

However, it appears that the effect of any nitrogenous ANFO residuals that may have been produced by the detonation of 760 tons of ANFO during this study on the ground-water quality of the study area is no more than that which would occur if natural soil NO_3-N were leached from the soil to the ground water, for example, by irrigation. This postulate is based on the results of the laboratory study, which showed that insignificant differences of NO_3-N were removed from the crater material and the native surface soil.

Also, it appears that the effect of any nitrogenous ANFO residuals produced by the detonation of 760 metric tons of ANFO during this study, on the ground-water quality could be no more than that actually observed in the field portion of this study. This hypothesis is obtainable if one assumes that most, if not all, of the NO_3-N was derived

as an ANFO residual and is analogous to the laboratory result that the crater material liberated a significantly higher mass of NO_3-N than did the pre-blast subsurface soil. It represents the largest possible effect of ANFO residuals on the ground-water quality of the study area.

This hypothesis will be discussed. As more ground water passes through the point sources of ANFO-affected alluvium, NO_3-N molecules will continue to be incorporated into the moving ground water. This will lead to exhaustion of any NO_3-N contributed as an ANFO residual at the point source of deposition. As ground water moves away from the point source of desposition, longitudinal and transverse dispersion effects within the alluvium will result in a decrease of NO_3-N concentration both along the flow path and normal to it. Longitudinal dispersion will reduce NO_3-N along the flow path owing to the differential macroscopic velocities at which individual NO_3-N molecules will traverse the aquifer matrix. Transverse dispersion will decrease the NO_3-N concentration normal to the flow path owing to the deflection of flow by solid particles which comprise the alluvial fill.

These mechanisms of concentration reduction of NO_3-N, both at the point sources of deposition of ANFO affected material and along and normal to the flow paths of ground water within the study area, coupled with the high amount of recharge that is contributed to the alluvial

materials by the Bill Williams River, should be sufficient to return NO_3-N levels within the study area back to the original levels within one or two months.

It is recognized that the hydrologic conditions that existed during this study, that is, high ground-water velocity and the availability of large amounts of recharge for dilution, would be likely to mitigate any effects that the deposition of nitrogenous ANFO residuals may have imposed on ground-water quality. Perhaps under less ideal hydrologic conditions and in various operations where ANFO is detonated frequently and over a long time period, the hydrologic consequences of ANFO detonation could be clearly measurable.

Therefore, further study is recommended, preferably in an area where ANFO is detonated regularly, as in coal-strip mining or open-pit mining regions. Inherent difficulties were encountered in this study with respect to the identification of nitrogenous ANFO residuals in a soil system in which appreciable amounts of NO_3-N existed prior to ANFO detonation. It is strongly recommended that attempts at studies of this nature be employed with experimental procedures designed to facilitate the correlation of soil NO_3-N to NO_3-N occurring in ground water. Kreitler and Jones (1975) devised methods to delineate unique isotopic nitrogen ranges for NO_3-N from different sources. By comparing the nitrogen isotope ratios of these sources to

the nitrogen isotope ratios of ground-water samples, he was able to estimate the source of NO_3-N in the ground water. Utilizing these methods, perhaps a better understanding could be obtained of the hydrologic consequences of ANFO detonation.

SUMMARY AND CONCLUSIONS

Field and laboratory studies were conducted to assess the effect, if any, of ANFO detonation on the ground-water quality within a portion of Planet Valley, Arizona. A single, 109-metric-ton charge of ANFO was detonated on the land surface on June 28, 1978, and an array of six 109-metric-ton charges on August 30, 1978.

Field studies consisted of soil and ground-water investigations. Soil samples were obtained from borings near the detonation sites to characterize the NO_3-N content of soil prior to ANFO detonation. Following detonation, materials which were exposed to products of ANFO detonation were acquired for laboratory study. Native surface and subsurface soils were also obtained for experimentation.

Ground-water investigations included the measurement of water-level elevations and the collection of water samples in eight wells to define the flow system of the study area and to monitor changes in ground-water quality, respectively.

Column leachate studies were performed to determine the mass of NO_3-N and NH_3-N removed from soils both exposed and unexposed to the products of ANFO detonation.

Chemical analyses of saturated extracts of soils collected near the detonation sites showed that considerable NO_3-N existed prior to ANFO detonation. After detonation, craters produced by the detonations were filled in part with material ejected from its original location by ANFO detonation.

Once saturated with ground water, NO_3-N contained within the crater fill was amenable to transport within the ground-water system. Chemical analyses of samples obtained from all wells, except one, showed NO_3-N levels were similar to background values. A flow system analysis showed that the NO_3-N levels that occurred in this well correlated with ground-water flow paths through the crater-fill material. A correlation also existed between the travel time of NO_3-N to this well and the probable time that crater-fill material became saturated.

The laboratory study clearly showed that the difference in NO_3-N leached from soil columns containing ANFO detonation material and from columns containing native surface soil were insignificant. Conversely, a significantly lower mass of NO_3-N was leached from soil columns containing subsurface material collected prior to detonation than from columns containing material exposed to the products of ANFO detonation.

The original location in the soil profile, and hence the composition, of material that came in contact

with ANFO detonation products is indeterminate. Therefore, in light of the results of the column studies, the source of the NO_3-N that occurred in the one well could be either natural soil NO_3-N present in the soil profile prior to ANFO detonation or NO_3-N introduced as a residual of ANFO detonation, or both.

The hydrologic consequences of ANFO detonation on ground-water quality are no more than would occur if natural soil NO_3-N were leached to the ground water, for example, by irrigation.

APPENDIX A

RESULTS FOR INDIVIDUAL WATER ANALYSES

As noted in the text, the major cations and anions of the ground water were analyzed, along with pH and EC_w. Tables A.1, A.2, A.3, A.4, A.5, and A.6, show the results of the analyses for observation wells PR6, PR7, PR8, and PR 10, and for the NV and SWV wells, respectively, for all sampling occasions. Table A.7 lists the results of the analyses for wells PR1 and PR4 on March 6, 1979.

Table A.1. The results of water analyses for well PR6

Analysis* (mg/l)	Date				
	3/20/78	6/29/78	8/31/78	9/29/78	3/6/79
Ca	68	42	96	46	40
Mg	22	15	10	13	12
Na	62	55	21	48	60
K	9	8	8	5	4
NH_3-N	0	0.25	0.67	0	0
HCO_3	250	214	193	211	191
SO_4	80	36	80	--	44
Cl	85	60	41	45	41
NO_3-N	0.18	3.7	4.9	5.4	2.5
pH	7.4	7.1	7.1	7.4	7.8
EC_w	769	620	640	550	570

*All analyses are in units of mg/l except pH, which is dimensionless, and EC_w, which is in units of μmhos/cm.

Table A.2. The results of water analyses for well PR7

Analysis* (mg/l)	Date				
	3/20/78	6/29/78	8/31/78	9/29/78	3/6/79
Ca	70	32	85	34	32
Mg	21	10	8	9	9
Na	70	57	22	51	62
K	10	7	6	4	7
NH_3-N	0	0	0	0.1	0
HCO_3	264	191	189	198	178
SO_4	90	25	76	--	41
Cl	81	47	37	36	37
NO_3-N	0	6.0	2.5	1.8	3.4
pH	7.4	7.2	7.4	7.6	8.0
EC_w	778	530	550	460	570

*All analyses are in units of mg/l except pH, which is dimensionless, and EC_w, which is in units of µmhos/cm.

Table A.3. The results of water analyses for well PR8

Analysis* (mg/l)	Date				
	3/20/78	6/29/78	8/31/78	9/29/78	3/6/79
Ca	75	44	96	49	38
Mg	23	15	10	13	11
Na	77	62	25	58	62
K	8	7	7	6	6
NH_3-N	0	0	0.21	0	0
HCO_3	270	278	276	284	204
SO_4	100	35	77	--	41
Cl	100	54	37	38	40
NO_3-N	0	1.1	0	0.3	1.0
pH	7.3	7.1	7.0	7.3	7.8
EC_w	850	650	770	580	570

*All analyses are in units of mg/l except pH, which is dimensionless, and EC_w, which is in units of μmhos/cm.

Table A.4. The results of water analyses for well PR10

Analysis* (mg/l)	Date				
	3/20/78	6/29/78	8/31/78	9/29/78	3/6/79
Ca	65	27	90	46	32
Mg	18	9	7	11	9
Na	56	46	20	48	46
K	8	6	9	4	4
NH_3-N	0	0	0.2	0.1	0
HCO_3	268	173	195	207	158
SO_4	65	23	78	--	36
Cl	30	38	37	48	33
NO_3-N	0.2	1.1	1.5	1.6	1.1
pH	7.5	7.1	7.4	7.5	7.8
EC_w	707	450	620	520	460

*All analyses are in units of mg/l except pH, which is dimensionless, and EC_w, which is in units of µmhos/cm.

Table A.5. The results of water analyses for the NV well

Analysis* (mg/l)	Date				
	3/20/78	6/29/78	8/31/78	9/29/78	3/6/79
Ca	62	57	128	69	16
Mg	22	24	20	22	19
Na	120	104	44	102	120
K	11	10	8	6	0.5
NH_3-N	0	0.4	0	0	0
HCO_3	255	283	272	275	166
SO_4	150	109	223	--	95
Cl	153	138	124	131	145
NO_3-N	0.1	. 1.0	0.6	0.6	0.8
pH	7.6	7.3	7.3	7.6	8.3
EC_w	1080	1080	1115	910	930

*All analyses are in units of mg/l except pH, which is dimensionless, and EC_w, which is in units of µmhos/cm.

Table A.6. The results of water analyses for the SWV well

Analysis * (mg/l)	Date				
	3/20/78	6/29/78	8/31/78	9/29/78	3/6/79
Ca	63	--	127	55	54
Mg	21	--	13	17	17
Na	82	--	34	80	85
K	9	--	6	6	6
NH_3-N	0	--	0.35	0	0
HCO_3	253	--	247	250	228
SO_4	95	--	146	--	85
Cl	103	--	91	86	88
NO_3-N	0.2	--	0.9	0.9	1.6
pH	7.6	--	7.4	7.7	8.0
EC_w	877	--	950	710	825

*All analyses are in units of mg/l except pH, which is dimensionless, and EC_w, which is in units of µmhos/cm.

Table A.7. The results of water analyses for wells PR1 and PR4 for March 6, 1979

Analysis* (mg/l)	Well	
	PR1	PR4
Ca	84	64
Mg	25	20
Na	60	108
K	8	7
NH_3-N	0	0
HCO_3	259	266
SO_4	75	142
Cl	58	122
NO_3-N	22	0.8
pH	7.3	7.8
EC_w	940	1040

*All analyses are in units of mg/l except pH, which is dimensionless, and EC_w, which is in units of μmhos/cm.

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