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B

APPENDIX B – WATER RESOURCES AND HYDROLOGY

B.1 GROUNDWATER QUALITY

B.1.1 Chemical Waste Landfill Analysis

B.1.1.1 Site History and Monitoring Results

Disposal operations began at the Chemical Waste Landfill (CWL) in 1962 and continued until 1985. An estimate of disposal quantities was derived based on a detailed disposal inventory for the period from 1975 through 1982 and the assumption that landfill use did not change significantly over the period of operation. Based on the disposal quantities, sampling results under the CWL, and the U.S. Environmental Protection Agency (EPA) drinking water standards (maximum contaminant levels [MCLs]), trichloroethylene (TCE) and chromium were identified as the predominant organic and inorganic contaminants of concern (DOE 1992d).

Recent quarterly groundwater sampling results from two monitoring wells upgradient of the CWL and seven monitoring wells downgradient of the CWL showed the presence of TCE in groundwater. In some instances, the measurements were above the TCE MCL of 0.005 mg/L, as shown in Table B.1–1 (SNL 1997d). TCE was not found in the upgradient wells, indicating that its presence is due to the CWL.

Table B.1–2 shows that chromium was also found in two monitoring wells during the third quarterly sampling in 1996. Chromium, measured at levels above the MCL of

Table B.1–1. Trichloroethylene Measured at the Chemical Waste Landfill (1996)

CWL MONITORING WELL	CONCENTRATION RANGE (mg/L)
<i>MW2A</i>	0.010 to 0.026
<i>MW2BU</i>	0.004 to 0.024
<i>MW3A</i>	0.002 to 0.004
<i>MW5L</i>	0.002 to 0.015
<i>MW5U</i>	0.002 to 0.007
<i>MW6L</i>	0.006 to 0.010

Source: SNL 1997d
CWL: Chemical Waste Landfill
mg/L: milligrams per liter

Table B.1–2. Chromium Measured at the Chemical Waste Landfill (1996)

CWL MONITORING WELL	CONCENTRATION (mg/L)
<i>BW3</i>	0.16
<i>MW2A</i>	0.11

Source: SNL 1997d
CWL: Chemical Waste Landfill
mg/L: milligrams per liter

0.1 mg/L, was present in both upgradient and downgradient wells. Although the chromium source has been found in the vadose zone down to about 75 ft below ground surface, its presence in saturated groundwater samples is attributed to dissolution of stainless steel in monitoring wells (SNL/NM 1995d). Such dissolution is a well-known phenomenon (Hewitt 1992, Oakley & Korte 1996). Furthermore, if the chromium in the aquifer resulted from vertical transport of the CWL contamination, chromium contamination would be continuously seen in the vadose zone down to the water table. Chromium contamination is not found in the lower 410 ft of the vadose zone. Iron and nickel were also found in the groundwater above MCLs during the same monitoring period. Both metals were present at similar concentrations in upgradient and downgradient wells, indicating that they are background concentrations, although the nickel may also be a result of dissolution of stainless steel in monitoring wells.

B.1.1.2 Modes of Contaminant Transport

Three modes of TCE transport to the water table were considered: vapor phase, organic phase, and water (aqueous) phase. Vapor phase transport, by way of diffusion of TCE volatilizing in the vadose zone, is responsible for the levels presently measured in the groundwater. This is suggested by three pieces of evidence.

- Application of Henry's law, which governs the partitioning of the TCE between vapor and liquid phases, indicates that the vapor and liquid are near equilibrium, with liquid being slightly less than predicted by Henry's law (DOE 1992d).
- A thin layer of contamination exists at the water table, characteristic of mass transport from vapor to liquid occurring at the water surface (DOE 1992d).

- Vapor phase transport by way of diffusion is the only mechanism by which the TCE could have reached the water table in the relatively short time period between TCE disposal and appearance of contamination at the water table (DOE 1992d).

Organic liquid phase transport, movement of the organic liquid TCE toward the water table under its driving head at disposal, would not have reached the water table. Water phase transport, movement of TCE dissolved in natural recharge (from precipitation), would not have reached the water table either (DOE 1992d). The rate of vertical transport of natural recharge may be determined by four different methods: chloride mass balance, stable oxygen isotope, bomb-pulse tritium, and bomb-pulse chlorine-36 from aboveground remote atmospheric testing. Using these methods, it was determined that it would likely take recharging precipitation approximately 9,000 years to reach the water table at the CWL. A lower-bound water phase transport time, based on the tritium method, which is known to be affected by downward vapor fluxes, is 1,250 years (SNL/NM 1995d).

B.1.1.3 Modeling

The unsaturated zone thickness beneath the CWL is approximately 480 ft. The saturated zone (aquifer) at this location consists of interbedded, low permeability, silty clay layers that confine sand layers of relatively high conductivity. The uppermost water-bearing unit in the saturated zone consists of 39 ft of silty clay. Three sand layers are present within the upper 135 ft of the aquifer. The horizontal velocity in the silty clay layer is small, the pore velocity being 0.07 ft per year. Vertical flow through this layer to the sandy layer, at a pore velocity of 0.03 ft per year, would occur prior to meaningful horizontal flow. The dominant flow direction in this layer is vertical to the sandy layer. Because this is neither a recharge nor discharge area, flow through the sand layers is assumed to be primarily horizontal, away from the CWL.

TCE advective transport away from the CWL would be in the sandy layers underlying the silty clay layer to which the TCE is being released. Modeling has shown that it may take hundreds of years for transport through porous material from the silty clay layer to the sandy layer. However, TCE has been found in the sandy layer. It is thought that this might be from discontinuities in monitoring well grout seals or in joints and cracks in the polyvinyl chloride (PVC) casing of the well. Nevertheless, a natural preferential pathway has not been ruled out (SNL/NM 1995d).

Even if there was a natural preferential pathway, the release rate of TCE to the sandy layer would be attenuated. For the purposes of projecting TCE concentrations in the sandy layer, it was conservatively assumed that the TCE released to the silty clay layer would be instantaneously transported to the sandy layer without attenuation.

Downgradient concentrations resulting from subsequent transport away from the CWL through the sandy layer were estimated using the *Multimedia Environmental Pollutant Assessment System (MEPAS)* model (PNL 1989). The *MEPAS* model integrates standard calculation methodologies for source term, environmental transport, and exposure. The groundwater module of *MEPAS* simulates vertical transport (one-dimensional advection–one-dimensional dispersion) through the unsaturated soil and horizontal transport (one-dimensional advection–three-dimensional dispersion) through a single saturated zone.

The *MEPAS* model accounts for the major mechanisms of constituent mobility (adsorption/desorption), persistence (degradation or decay), advection, and dispersion. Mobility is described by a distribution coefficient that assumes instantaneous adsorption/desorption between the soil matrix and pore water. Persistence is described by a first-order degradation/decay coefficient. Advection is described by constant, unidirectional flow in the vertical direction in the unsaturated zone and in the horizontal direction in the saturated zone. Dispersion is described in one dimension (vertical) in the unsaturated zone and in all three dimensions in the saturated zone. Although three-dimensional dispersion is likely in the unsaturated zone, the assumption of one-dimensional dispersion results in the highest concentration at the water table and reflects current data showing TCE contamination directly under the CWL. The model was applied to both the aqueous phase transport of chromium and the TCE vapor release to the saturated zone.

The site conditions used in *MEPAS* to calculate downgradient contaminant concentrations are shown in Table B.1–3 (taken from SNL/NM 1995d unless otherwise indicated). The unsaturated zone parameters represent the site directly beneath the CWL. The saturated zone parameters represent the site along the projected contaminant plume trajectory, from the CWL to the nearby municipal well field (Ridgecrest), located approximately 7 mi north of the CWL. The trajectories and saturated groundwater velocities of contaminants released from the CWL were obtained from a simulation using the *MODPATH* model in conjunction with a three-dimensional simulation of flow of groundwater beneath Kirtland Air Force Base (KAFB) using the *MODFLOW*

Table B.1–3. Chemical Waste Landfill Transport Analysis Parameters

PARAMETER	VALUE
UNSATURATED ZONE PROPERTIES	
<i>Chromium source to water table depth, ft</i>	410
<i>TCE distribution coefficient, mL/g</i>	0.012
<i>Chromium (as chromic acid) distribution coefficient, mL/g</i>	0 ^a
<i>Dry bulk density, g/cm³</i>	1.8
<i>Total porosity</i>	0.34
SATURATED ZONE PROPERTIES	
<i>Aquifer thickness, ft</i>	100 ^b
<i>Pore velocity, CWL to TA-III boundary ft/yr</i>	430 ^b
<i>Pore velocity, TA-III boundary to Ridgecrest-5, ft/yr</i>	991 ^b
<i>Aquifer porosity</i>	0.2
<i>Longitudinal dispersivity, ft</i>	100
<i>Lateral dispersivity, ft</i>	3
<i>Vertical dispersivity, ft</i>	0.114 ^c
<i>TCE distribution coefficient, mL/g</i>	0.012
<i>Chromium (as chromic acid) distribution coefficient, mL/g</i>	0 ^a
<i>Dry bulk density, g/cm³</i>	1.8

Sources: ^aDOE 1992d, ^bDOE 1997a, ^cPNL 1995, SNL/NM 1995d

ft: feet

ft/yr: feet per year

g/cm³: grams per cubic centimeter

g/yr: grams per year

m: meters

mg/L: milligrams per liter

mL/g: milliliters per gram

TCE: trichloroethylene

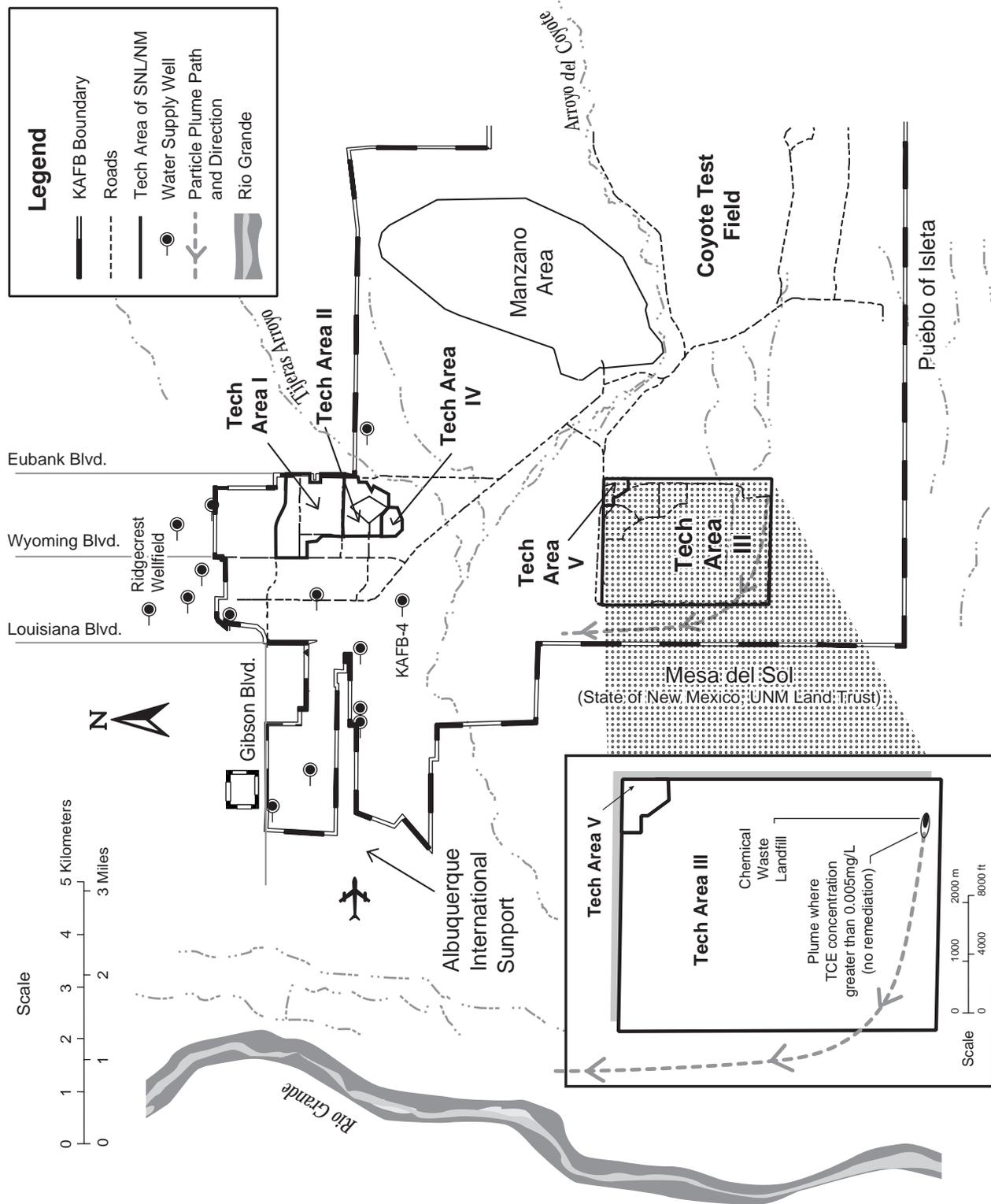
model (DOE 1997a). The aquifer thickness was conservatively chosen to include only the most conductive sandy zone, the ancestral Rio Grande lithofacies. The nearest drinking water supply well to the CWL is KAFB-4. This well, which lies along the projected plume trajectory, is located approximately 4 mi north of the landfill (SNL/NM 1995d). Although recent measurements indicate that TCE is degrading at the source (Ardito 1998), no degradation of the vapor phase TCE was assumed. This is intended to account for uncertainty of the degradation rate and the short travel time of this phase. Degradation products have not been detected in the TCE plume in the aquifer.

A previous modeling analysis (SNL/NM 1995d) of the vapor phase transport in the vadose zone from the source to the water table indicated that, in the absence of significant advection (such as, in the upper silty clay unit), nearly circular isopleths would result because diffusion is the dominant transport mechanism. The analysis showed that concentrations from 0.036 to 0.050 mg/L (in groundwater) could be expected within a nearly circular region, 349 ft in diameter. The source conditions used in the present analysis assumed an equivalent source area and a conservative concentration of 0.050 mg/L (the maximum concentration measured in 1996, as shown in Table B.1–1, was 0.026 mg/L). The release rate of TCE to the aquifer that duplicated these assumed conditions was 33.3 g per year; this release rate was used for subsequent calculations.

Because the vapor has already reached the aquifer, the release rate of 33.3 g per year is not likely to decrease immediately after source remediation. Consequently, that rate was used for the 10-year period from when TCE was first found in the aquifer (May 1990) until remediation, allowing for the 1-year period following a 1991 pump test in which no TCE was detected in the groundwater. The model results indicate that the maximum concentrations in the sandy aquifer (through which the contaminants are transported from the landfill and from which the drinking water wells draw their water) will always be less than the drinking water standard 410 ft downgradient from the source (covering an area of 1.7 ac), and will be an order of magnitude less than this standard at the Technical Area (TA)-III boundary (Figure B.1–1). The MCL concentration at its farthest downgradient extent will be reached approximately 5 years after introduction into the sandy layer, and will begin to decrease approximately 10 years thereafter as a result of source remediation.

B.1.1.4 Remediation Efficiency

A sensitivity study was performed for the Site-Wide Environmental Impact Statement (SWEIS) that considers the ultimate fate of the TCE for remediation of 50, 90, and 95 percent of the source. After remediation, downgradient concentrations due to the vapor phase source would be expected to quickly decline. Concentrations at the maximum unremediated downgradient extent of the MCL would decrease below this value about 10 years after introduction into the sandy layer. The maximum downgradient distance within which the MCL would be exceeded would decrease to 190 ft after 50 percent remediation and to



Source: Original

Figure B.1–1. Location and Maximum Extent of Projected Trichloroethylene Contamination in Groundwater at the Chemical Waste Landfill
The maximum calculated extent of trichloroethylene contamination above 0.005 mg/L is 410 ft from the Chemical Waste Landfill.

3 ft after 90 percent remediation (a minimum expected remediation efficiency [Ardito 1998]) and would not exceed the MCL for a remediation efficiency of 95 percent. Table B.1–4 shows the maximum downgradient concentrations along the plume path for the smallest (50 percent) and largest (95 percent) remediation efficiencies considered. Preremediation concentrations (10 years of preremediation releases conservatively followed by 50 percent remediation) are also given.

The liquid organic phase of the TCE currently resides totally in the unsaturated zone. The aquifer is presently not being affected as a result of unsaturated transport of this phase. The inventory of this phase (which was taken as the total disposed, less the inventory in the vapor phase) was estimated as 3.10×10^7 g. The initial percolation of the TCE is to a depth of 33 ft below ground surface (SNL/NM 1995c). The liquid organic phase will tend toward residual liquid levels in the vadose zone and, given a sufficiently small release or sufficiently thick vadose zone, will cease to move as an integral phase (EPA 1991, EPA 1993). Calculations have been performed that indicate that the unsaturated zone is sufficiently thick beneath the CWL so that the organic phase liquid will not reach the aquifer prior to reaching residual concentration levels (at which the liquid is retained in the soil pores by capillary forces) and that the dominant mode of liquid transport will be by way of the aqueous phase (DOE 1992d, SNL/NM 1995d).

Recently, measurements have been taken that indicate degradation of the liquid organic TCE (Ardito 1998). Degradation will result in aqueous phase TCE reaching the water table at levels far below the MCL. This is demonstrated by noting that the saturation concentration of TCE, 1,100 mg/L, would require 17.75 half-lives to degrade to the MCL level of 0.005 mg/L. The lower end of the possible range of travel times of this TCE to the water table is 1,250 years (SNL/NM 1995d). Therefore, a half-life for environmental degradation of less than 70.4 years (1,250 years divided by 17.75 half-lives) would result in TCE reaching the aquifer at levels below the MCL. The longest half-life presented for environmental degradation of TCE in any of various media is 4.5 years for anaerobic degradation (Howard et al. 1991). Even if the degradation rate was a factor of 10 slower than this; that is, a 45-year half-life, the concentration of the TCE as it reaches the water table, prior to dilution with aquifer water, would be 4.8×10^{-6} mg/L, a factor of 1,000 less than the MCL. This indicates that, in addition to the source reduction by remediation and volatilization, degradation would likely result in undetectable TCE concentrations prior to reaching the water table.

Chromium was disposed of in the form of chromic acid; this chromium presently resides totally in the unsaturated zone, to a depth of 75 ft from ground level. Although not presently affecting the saturated zone, this chromium may reach the saturated zone in the future. The EPA has presented studies that show that hexavalent chromium is frequently reduced to trivalent chromium in the environment (Palmer and Puls 1994). The latter has relatively low toxicity and very low mobility. The EPA has also indicated that hexavalent chromium can be expected to adsorb to soil, although not as strongly as trivalent chromium (EPA 1996b). Site documents, however, indicate that the disposed form of the chromium is not retarded by site soils (DOE 1992d). The analysis conservatively assumes that the chromium remains in the hexavalent state in which it was disposed and does not undergo soil adsorption.

The major vertical chromium incursion into the vadose zone has been found under the unlined chromic acid pit (SNL/NM 1992). The dissolved chromium under the pit reaches concentrations greater than 200 mg/L in soil moisture (SNL/NM 1998b). However, remediation is planned that will remove up to 20 ft of soil if the concentrations are greater than three times the background concentrations. In practice, this means that the entire upper 20 ft of soil at the CWL will be removed (Peterson 1998). This remediation will remove all of the areas in which chromium exceeds 200 mg/L (as well as much of the TCE source). The remaining chromium inventory, 9,050 g, was conservatively estimated based on the cross-section of maximum content, approximately 54 ft below ground surface. This cross-section was assumed to represent the vadose zone presence of the chromium between 20 and 75 ft. The moisture levels under the pit have been found to be equivalent to residual moisture levels (SNL/NM 1992). This indicates that although the initial head in the pit carried the chromium to 75 ft deep, remaining chromium movement would be by way of dissolution in percolating precipitation. Indeed, no evidence has been found of recent vertical chromium movement (SNL/NM 1998b).

Based on the expected vertical velocity found at the CWL, 0.05 ft per year, the chromium will take 7,900 years to reach the water table. Given the indicated inventory and vertical velocity and the site information indicated in Table B.1–3, *MEPAS* was used to calculate the chromium concentration 1 m downgradient of the chromium source. It was found that the maximum concentration would only be 0.005 mg/L, a factor of 20 less than the MCL. Even if the maximum vertical velocity calculated for the CWL (see Section B.1.1.2) was assumed, 0.40 ft per year, the chromium would take

Table B.1–4. Maximum Downgradient Trichlorethene Concentrations from Vapor Phase Source

DOWNGRAIDENT DISTANCE (ft)	NO REMEDIATION		50% REMEDIATION		95% REMEDIATION	
	CENTERLINE PLUME CONCENTRATION (mg/L)	LATERAL DISTANCE FROM PLUME CENTERLINE TO MCL (0.005 mg/L) (ft)	CENTERLINE PLUME CONCENTRATION (mg/L)	LATERAL DISTANCE FROM PLUME CENTERLINE TO MCL (ft)	CENTERLINE PLUME CONCENTRATION (mg/L)	LATERAL DISTANCE FROM PLUME CENTERLINE TO MCL (ft)
3	0.050	43	0.025	27	0.0025	-
33	0.026	43	0.013	28	0.0013	-
190	0.010	3	0.005	0	0.0005	-
410	0.005	0	0.0025	-	0.00025	-
980	0.002	-	0.001	-	0.0001	-
3,280	0.00086	-	0.00043	-	0.000043	-
7,100	0.0004	-	0.00023	-	0.000023	-
19,700	0.0001	-	0.000057	-	0.0000057	-
32,300	0.00006	-	0.000034	-	0.0000034	-
44,200	0.000045	-	0.000027	-	0.0000027	-
Area exceeding MCL (acres)	1.7		1.2		0	

Source: Calculations derived from PNL 1995

–: Plume does not reach concentration above MCL at this distance.

ft: feet

mg/L: milligrams per liter

MCL: maximum contaminant level

1,000 years to reach the water table and the concentration 1 m downgradient of the source would be 0.03 mg/L, a factor of 3 less than the MCL.

B.2 GROUNDWATER QUANTITY

Because discharge exceeds recharge for this portion of the Albuquerque-Belen Basin (USGS 1993), groundwater withdrawal by water supply wells for the city of Albuquerque and KAFB has resulted in significant changes to the direction of groundwater flow and levels of drawdown in the regional aquifer system over the past 30 years. Groundwater flow at KAFB has been altered from a principally westward direction to northwestward and northward along the western and northern portions of KAFB, chiefly in response to withdrawals by local city (Ridgecrest) and KAFB well fields (SNL/NM 1997a). Water levels in the Albuquerque-Belen Basin have been declining since the 1960s in response to significant increases in groundwater usage. Basin-wide declines range up to 160 ft, with the maximum located just north of KAFB (USGS 1993). Declines in the KAFB vicinity, in response to the local withdrawals, have been measured in the KAFB vicinity since 1985. Cumulative drawdowns are depicted in Figure B.2–1 (SNL/NM 1997a). Levels in the upper unit of the Santa Fe Group have recently declined by as much as 3 ft per year (36 ft over the 12-year period from 1985 through 1996) in the vicinity of the KAFB and Ridgecrest pumping wells, west of the onsite fault zone. Hydrographs of water levels within wells east of the natural flow barrier created by the fault zone show that water levels in these regions are not affected by water supply production from the regional aquifer system.

Projections of near-term (SNL/NM SWEIS operational period of 1998 through 2007) groundwater drawdown can best be estimated by comparison with the recent drawdown. Near-term drawdown is estimated by relating the projected groundwater withdrawals in the KAFB vicinity to the recent withdrawals and assuming a proportional aquifer level response. Table B.2–1 shows the quantity of water recently pumped from onsite KAFB wells and from Ridgecrest, the nearby Albuquerque well field. Groundwater levels in the KAFB vicinity are most dependent on these nearby wells.

Projections of groundwater use through 2007 were based on the city of Albuquerque's goal of 30 percent reduction from 1994 levels in per capita water use by 2004 (COA n.d. [a]). This water conservation goal was assumed to be reached linearly over the 10-year period. In addition, a population growth factor of 1.5 percent

Table B.2–1. 1985 through 1996 Groundwater Withdrawals in the Immediate SNL/NM Vicinity

YEAR	KAFB WITHDRAWAL (10 ⁶ ft ³)	RIDGECREST WITHDRAWAL (10 ⁶ ft ³)
1985	232.3	274.1
1986	237.4	316.4
1987	210.1	374.2
1988	199.0	421.3
1989	258.1	422.8
1990	208.0	390.6
1991	219.7	385.3
1992	235.7	332.2
1993	201.2	454.5
1994	166.7	319.3
1995	151.7 ^a	375.5
1996	155.5 ^a	356.8
TOTAL (1985-1996)	2,475	4,466
Number of wells	14	5

Sources: ^aUSAF 1998b, USGS 1995

ft³: cubic feet

KAFB: Kirtland Air Force Base

per year, compounded, was applied to the Ridgecrest field (COA n.d. [b]). Table B.2–2 shows the year-by-year factors used to account for water conservation and population growth. The table indicates that water use will decline until 2004, when the city's goal is assumed to be met, and will begin to increase thereafter.

Table B.2–3 shows the total projected withdrawal for the SWEIS operational period of 1998 through 2007 from the well fields nearest to the site. Ridgecrest water withdrawal was particularly low in 1994; 1995 was used as the base year for this well field. The combined population growth and water conservation factor was applied to the 1995 Ridgecrest withdrawal of 375.5x10⁶ ft³. The KAFB withdrawals include the water used by SNL/NM. Although SNL/NM has committed to the 30 percent water conservation, an explicit projection of SNL/NM water use was conservatively assumed under the No Action Alternative (SNL/NM 1998c). This projection was subtracted from the KAFB withdrawals prior to

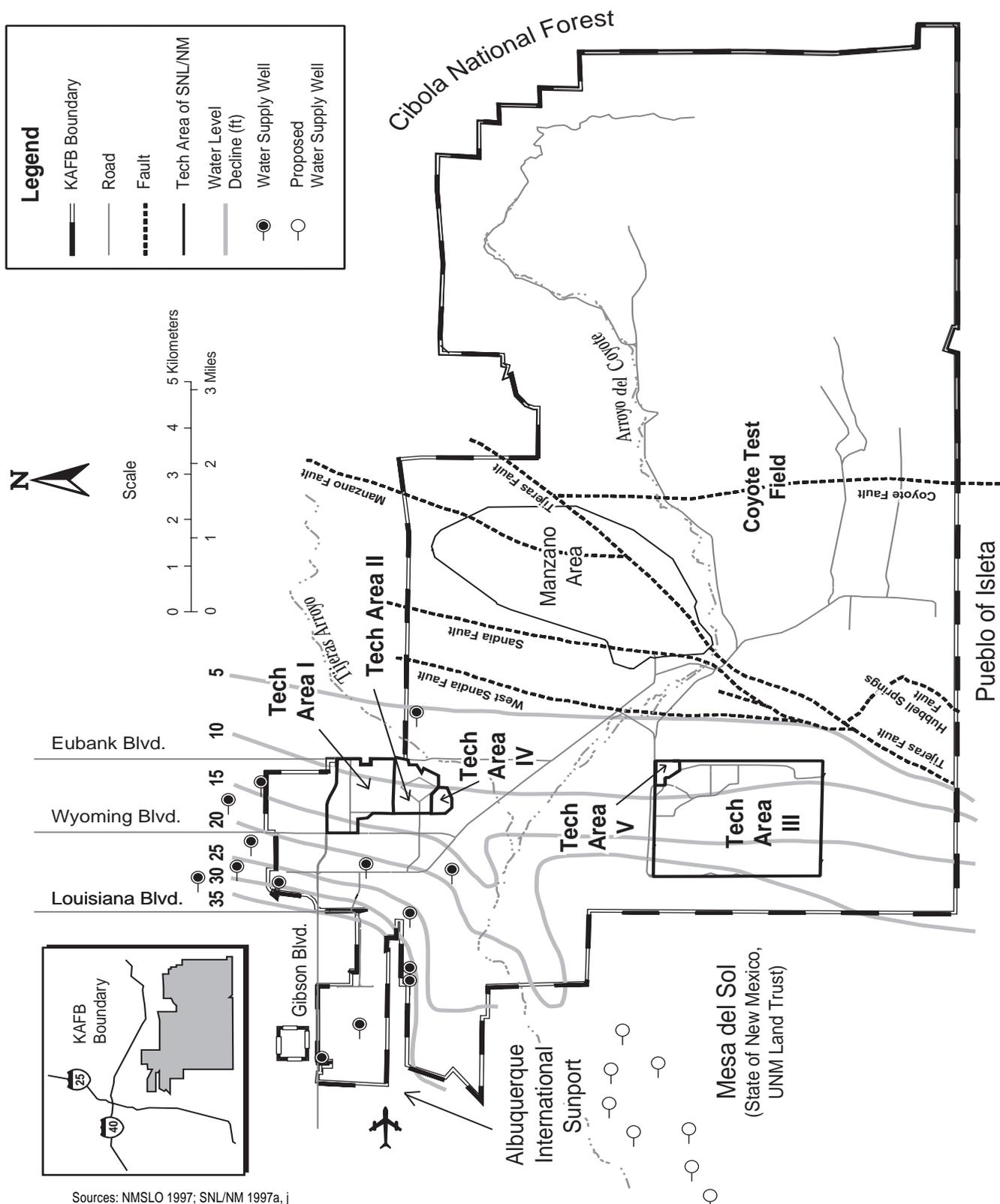


Figure B.2–1. Albuquerque-Belen Basin Groundwater Level Declines, 1985 through 1996

During the period of 1985 through 1996, groundwater levels at KAFB declined in some places by more than 35 feet.

Table B.2–2. Annual Factors Applied to 1994 Water Withdrawal for Projecting Future Withdrawals

YEAR	POPULATION GROWTH FACTOR	WATER CONSERVATION FACTOR	COMBINED FACTOR
1994	1	1	1
1995	1.015	.97	.985
1996	1.030	.94	.968
1997	1.046	.91	.952
1998	1.061	.88	.934
1999	1.077	.85	.915
2000	1.093	.82	.896
2001	1.110	.79	.877
2002	1.126	.76	.856
2003	1.143	.73	.834
2004	1.161	.70	.813
2005	1.178	.70	.825
2006	1.196	.70	.837
2007	1.214	.70	.850
TOTAL (1998 through 2007)	11.4	7.63	8.64

Source: Original

application of the water conservation factor. Only the water conservation factor was applied to the 1994 KAFB withdrawal of 166.7×10^6 ft³. KAFB withdrawals have been and are projected to continue to be significantly below the amount allowed by their water rights, 278.7×10^6 ft³ (Bloom 1972).

It is expected that the San Juan/Chama Project (COA 1997b) will be on-line in approximately 2004. The project will allow the city of Albuquerque to meet its normal water demands from Rio Grande water. The river water would be replenished using water from city-owned water rights to the San Juan/Chama Diversion Project. Groundwater withdrawals will be used only to supplement these normal demands. All of the city wells will remain on-line and ready for operation. Which wells will be operated (and how often and how much) has not yet been determined. Although it is safe to say that the Ridgecrest well withdrawal would decrease substantially, the analysis given here conservatively assumes groundwater continues as the chief supplier of water to the region.

The proposed Mesa del Sol Project will be a potential major contributor to groundwater usage in the KAFB vicinity for the period of analysis (NMSLO 1997). For this projection, it was assumed that 20,000 persons (of the eventual 97,500 total) will be resident in 2007,

Table B.2–3. Projected Groundwater Withdrawal (1998 through 2007) in the KAFB Vicinity Under the No Action Alternative

YEAR	WITHDRAWAL (10 ⁶ ft ³)				ANNUAL TOTAL
	RIDGECREST	KAFB (EXCLUSIVE OF SNL/NM)	SNL/NM	MESA DEL SOL	
1998	350.7	95.6	59.4	0	505.7
1999	343.6	92.4	59.6	0	495.6
2000	336.4	89.1	59.9	10.7	496.1
2001	329.3	85.8	60.2	32.0	507.3
2002	321.4	82.6	60.4	53.4	517.8
2003	313.2	79.3	60.7	74.7	527.9
2004	305.3	76.1	60.9	96.1	538.4
2005	309.8	76.1	61.2	117.4	564.5
2006	314.3	76.1	61.4	138.8	590.6
2007	319.2	76.1	61.7	160.1	617.1
TOTAL	3,243.2	829.2	605.4	683.2	5,360.0

Source: Original
ft³: cubic feet

KAFB: Kirtland Air Force Base
SNL/NM: Sandia National Laboratories/New Mexico

with the year 2000 being the beginning of residential groundwater usage. The city's post-30 percent reduction goal of 175 gal per day per person was assumed for Mesa del Sol. Projected Mesa del Sol withdrawals for the period 1998 through 2007 are shown in Table B.2–3. As with the city of Albuquerque, it is planned that Rio Grande water will be used to satisfy average Mesa del Sol usage requirements; groundwater wells will supplement this surface water. The withdrawals shown in the table conservatively assume that the entire community usage is from groundwater. Although Mesa del Sol's water usage is similar to SNL/NM's for the period of SWEIS performance, projected continued growth of Mesa del Sol beyond 2007 would result in significant increases in water usage.

Under the No Action Alternative, the total groundwater withdrawal in the KAFB vicinity, projected for the period from 1998 through 2007, is the sum of Kirtland, Ridgecrest, and Mesa del Sol withdrawals, 5,360 M ft³ (Table B.2–3). This withdrawal is 77.2 percent of the 6,941 M ft³ withdrawn during the 12-year period, 1985–1996 (Table B.2–1).

Assuming a linear relationship between local water use and drawdown, projected drawdowns over the 10-year period, 1998–2007, would be 77.2 percent (5,360 M ft³/6,941 M ft³) of the drawdowns shown in Figure B.2–1. Figure B.2–2 shows these projected drawdowns across KAFB for the 10-year period, 1998–2007. The maximum drawdown on KAFB during the 1985–1996 period was 36 ft, which would correspond to a maximum projected drawdown over the period, 1998–2007, of 27.8 ft (77.2 percent of 36 ft.)

The projected SNL/NM water use for the period 1998 through 2007 varies 10 percent (12 percent including the Microsystems and Engineering Sciences Applications [MESA] Complex configuration in the Expanded Operations Alternative) or less among the three alternatives, being 605.4, 628 (635 M ft³ including MESA), and 570.7 M ft³ under the No Action, Expanded Operations, and Reduced Operations Alternatives, respectively. The SNL/NM water use corresponds, therefore, to approximately 11 percent (12 percent under the Expanded Operations Alternative) of the projected withdrawal in the KAFB vicinity, and 3 ft of water level decline over 10 years.

B.3 SURFACE WATER QUANTITY

The following section describes calculations and assumptions used to estimate the contribution of SNL/NM storm water runoff to surface water quantity in the Rio Grande. This set of calculations estimates excess precipitation runoff from the presence of relatively impermeable surfaces at SNL/NM. The excess precipitation runoff applies to the No Action, Expanded Operations, and Reduced Operations Alternatives, as no significant variation of input parameters is expected under each of the alternatives.

The Montessa Park gaging station, operated by the U.S. Geological Survey (USGS), is located on Tijeras Arroyo, 0.8 mi downstream from where Tijeras Arroyo exits KAFB. The drainage area at this point is 122 mi² (USGS 1998). Impervious surfaces covered by SNL/NM

include buildings (0.595 mi²) and parking lots (0.125 mi²) (SNL/NM 1997j). The total SNL/NM area covered by impervious surfaces is 0.72 mi². This number would remain the same under each of the alternatives. A comparison of the runoff potential of this area in its natural state with its developed state is in Table B.3–1.

Comparing the 5.3 percent effective increase in watershed area resulting from the presence of SNL/NM with measured flows at the Montessa Park gaging station allows an estimate of the SNL/NM contribution to runoff within Tijeras Arroyo. Assuming (conservatively) that all flow at the Montessa Park gaging station will reach the Rio Grande (5 mi downstream), the percentage contribution to Rio Grande flow can then be calculated (Table B.3–2).

Note that the volumes in Table B.3–2 are annual totals. Flow at the Montessa Park gaging station was measured

Table B.3–1. Comparison of Natural and Developed Runoff Potential at SNL/NM

PARAMETER KEY	PARAMETER DESCRIPTION	PARAMETER VALUE
A	Natural runoff percentage (conservative estimate) ^a	10%
B	Developed area runoff percentage (conservative estimate)	100%
C	Ratio of developed area runoff percentage to natural runoff percentage (B/A)	10
D	Current developed (impervious) area ^b	0.72 mi ²
E	Size of natural area for equivalent runoff (D x C)	7.2 mi ²
F	Effective increase in drainage area (E - D)	6.48 mi ²
G	Montessa Park drainage area ^c	122 mi ²
H	Effective percentage increase in watershed area (F/G)	5.3%

Sources: ^aSNL/NM 1997a, ^b1997j, ^cUSGS 1998
mi²: square miles

Table B.3–2. Values Used for Calculation of SNL/NM Storm Water Runoff Contributions to Tijeras Arroyo and Rio Grande Flow

YEAR	MONTESSA PARK FLOW VOLUME (ft ³)	SNL/NM CONTRIBUTION (5.3%) TO FLOW (ft ³)	RIO GRANDE FLOW VOLUME (ft ³)	SNL/NM CONTRIBUTION TO RIO GRANDE FLOW (percent)
1993	1.84x10 ⁶	97,520	5.97x10 ¹⁰	0.00016
1994	13.1x10 ⁶	694,300	5.41x10 ¹⁰	0.0013
1995	6.5x10 ⁶	344,500	6.78x10 ¹⁰	0.00051

Source: USGS 1998
ft³: cubic feet
SNL/NM: Sandia National Laboratories/New Mexico

only on 20 days in the 1993 through 1995 period, all during summer storm events. During these periods, the SNL/NM contribution to Rio Grande flow was likely higher than the percentages calculated above. However, these storm events would also contribute to higher Rio Grande flow because of runoff in surrounding areas, particularly the large paved areas of Albuquerque. For example, on the day during the 1993 through 1995 period when the greatest flow in Tijeras Arroyo was

measured at the Montessa Park gaging station, Rio Grande flow increased by nearly 400 percent from the previous day (USGS 1998). Because the major SNL/NM contribution to surface water quantity is discharge to the water reclamation plant (Section 5.3.4), and this discharge amount will remain relatively constant regardless of Rio Grande flow, the total SNL/NM percentage contribution to Rio Grande flow may actually decrease during storm events.

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