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Final Site Observational Work Plan for the Shiprock, New Mexico, UMTRA Project Site

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Contents

Acronyms and Abbreviations xvii
Executive Summary xxi
1.0 Introduction 1–1
1.1 Purpose and Scope 1–1
1.2 UMTRA Project Programmatic Documents 1–1
1.3 Relationship to Site-Specific Documents 1–2
1.4 SOWP Revisions 1–2
2.0 Regulatory Framework 2–1
2.1 Uranium Mill Tailings Radiation Control Act 2–1
2.2 EPA Ground Water Protection Standards 2–1
2.2.1 Subpart B: Standards for Cleanup of Land and Buildings 2–2
2.2.2 Natural Flushing Standards 2–2
2.2.3 Subpart C: Implementation 2–3
2.2.4 Supplemental Standards 2–3
2.3 Cooperative Agreement 2–4
2.4 National Environmental Policy Act 2–4
2.5 Other Federal Regulations 2–4
2.6 Tribal Regulations and Requirements 2–5
2.7 DOE Orders 2–6
3.0 Site Background 3–1
3.1 Physical Setting and Climate 3–1
3.2 Site History 3–2
3.2.1 Pre-Milling Site Conditions 3–2
3.2.2 Milling-Era History 3–3
3.2.3 Surface Remedial Action 3–6
3.2.4 Sources of Ground Water Contamination 3–7
3.3 Present and Anticipated Land and Water Use 3–7
3.4 Explanation of Aerial Photographs 3–9
4.0 Site Characterization Results 4–1
4.1 Investigation Methods 4–6
4.1.1 Drilling 4–6
4.1.2 Subsurface Sampling 4–7
4.1.3 Lithologic Logging 4–7
4.1.4 Well Installation and Development 4–8
4.1.5 Packer Tests 4–8
4.1.6 Water Level Measurements 4–15
4.1.7 Ground Water Sampling and Analysis 4–15
4.1.8 Aquifer Tests 4–16
4.1.9 Surveying 4–16
4.2 Geology 4–22
4.2.1 Geologic Mapping 4–22
4.2.1.1 Mancos Shale 4–22
4.2.1.2 Quaternary Material 4–24
4.2.2 Joint Measurements 4–28
4.2.3 Borehole Stratigraphic and Structural Results 4–33
4.2.4 Geophysical Survey Results 4–41

4.3 Hydrology 4–41
4.3.1 Surface Water 4–42
4.3.1.1 San Juan River 4–42
4.3.1.2 Water Supply 4–45
4.3.1.3 Bob Lee Wash 4–46
4.3.1.4 Many Devils Wash 4–46
4.3.1.5 Additional Washes 4–47
4.3.1.6 Seeps and Springs 4–47
4.3.2 Ground Water 4–47
4.3.2.1 Floodplain Alluvium 4–48
4.3.2.2 Terrace Ground Water System 4–61
4.4 Geochemistry 4–76
4.4.1 Surface Water Chemistry 4–76
4.4.1.1 San Juan River 4–76
4.4.1.2 Floodplain 4–78
4.4.1.3 Terrace 4–78
4.4.2 Ground Water Chemistry 4–85
4.4.2.1 Floodplain 4–85
4.4.2.2 Terrace 4–112
4.4.2.3 Organic Constituents 4–150
4.4.3 Contaminants in Soils and Sediments 4–150
4.4.3.1 Background 4–151
4.4.3.2 Soil and Sediment Study 1: Distribution of Contaminants at Widely Distributed Locations 4–152
4.4.3.3 Soil and Sediment Study 2: Detailed Investigation at the Base of the Escarpment 4–166
4.4.4 Determination of Distribution Ratios 4–173
4.4.4.1 Background 4–174
4.4.4.2 Sample Collection and Laboratory Methods 4–179
4.4.4.3 Results and Discussion 4–179
4.4.5 Composition of Salt Deposits 4–182
4.4.5.1 Background 4–182
4.4.5.2 Methods 4–185
4.4.5.3 Major Ion Composition of the Salt Deposits 4–185
4.4.5.4 Nitrate, Sulfate, and Uranium Concentrations of the Salt Deposits 4–185
4.4.5.5 Constituents Other than Nitrate, Sulfate, and Uranium 4–188
4.4.5.6 Mineralogy of Salt Deposits 4–195
4.4.6 Column Leaching of Alluvial Aquifer Sediment 4–195
4.4.6.1 Background 4–196
4.4.6.2 Methods 4–196
4.4.6.3 Results and Discussion 4–197
4.4.7 Fate and Transport 4–201
4.4.7.1 Ammonium 4–201
4.4.7.2 Antimony 4–201
4.4.7.3 Arsenic 4–202
4.4.7.4 Cadmium 4–202
4.4.7.5 Magnesium 4–202
4.4.7.6 Manganese 4–202

4.4.7.7 Molybdenum 4–203
4.4.7.8 Nitrate 4–203
4.4.7.9 Radium 4–204
4.4.7.10 Selenium 4–204
4.4.7.11 Sodium 4–204
4.4.7.12 Strontium 4–205
4.4.7.13 Sulfate 4–205
4.4.7.14 Thorium 4–205
4.4.7.15 Uranium 4–205
4.4.7.16 Vanadium 4–206
4.4.8 Mancos Shale and Associated Ground Water Chemistry 4–206
4.4.8.1 Geographic Distribution of Late Cretaceous Marine Shales 4–206
4.4.8.2 Chemistry of Ground Water Contacting Weathered Mancos Shale 4–207
4.4.8.3 Shiprock Wells West of U.S. Highway 666 4–215
4.4.8.4 Summary 4–215
4.5 Numerical Ground Water Modeling 4–216
4.5.1 Flow Model Construction 4–216
4.5.2 Model Calibration 4–224
4.5.3 Particle Tracking 4–229
4.5.4 Transport Modeling 4–229
4.5.5 Active Remediation Simulations 4–235
4.5.6 Conclusions 4–251
4.6 Ecological Field Investigations 4–252
4.6.1 Vegetation Characterization 4–253
4.6.1.1 Methods 4–253
4.6.1.2 Results 4–254
4.6.2 Sampling for Chemical Analysis 4–259
4.6.2.1 Surface Water and Sediment Sampling Methods 4–260
4.6.2.2 Plant Tissue Sampling Methods 4–265
5.0 Site Conceptual Model 5–1
6.0 Baseline Risk Assessment 6–1
6.1 Human Health Risks 6–1
6.1.1 Summary of 1994 BLRA Methodology and Results 6–1
6.1.1.1 Floodplain Ground Water 6–1
6.1.1.2 Terrace Ground Water 6–2
6.1.1.3 Floodplain Surface Water 6–3
6.1.2 BLRA Update 6–3
6.1.2.1 Floodplain Alluvium 6–5
6.1.2.2 Terrace Ground Water 6–7
6.1.2.3 Surface Water 6–8
6.1.3 Results 6–8
6.1.3.1 Floodplain Ground Water 6–8
6.1.3.2 Terrace Ground Water 6–11
6.1.3.3 Surface Water 6–15
6.1.3.4 Summary and Recommendations 6–21
6.2 Ecological Risk Assessment 6–23
6.2.1 Problem Formulation 6–26
6.2.1.1 Potentially Affected Habitats and Populations 6–26

6.2.1.2 Summary of the 1994 Ecological Risk Assessment Results..... 6–29
6.2.1.3 Update of the Ecological COPCs..... 6–31
6.2.1.4 Ecological Site Conceptual Model..... 6–39
6.2.1.5 Ecological Receptors 6–44
6.2.1.6 Management Goals and Endpoints 6–48
6.2.2 Analysis..... 6–49
6.2.2.1 Exposure Assessment..... 6–49
6.2.2.2 Effects Characterization..... 6–53
6.2.3 Risk Characterization..... 6–58
6.2.3.1 Area A: The Distributary Channel and Tributaries..... 6–58
6.2.3.2 Area B: The San Juan River..... 6–61
6.2.3.3 Area C: Shiprock Floodplain 6–65
6.2.3.4 Area D: Bob Lee Wash 6–68
6.2.3.5 Area E: Many Devils Wash 6–72
6.2.3.6 Area F: Upland Terrace 6–75
6.2.3.7 Potential Risks from Salt Crust Ingestion..... 6–76
6.2.3.8 Potential Risks from Radionuclides..... 6–78
6.2.3.9 Comparison of Surface Water Data to Livestock Drinking Water Standards 6–78
6.2.3.10 Potential Risks to Sensitive Species 6–79
6.2.3.11 Ecological Risk Summary..... 6–79
6.3 Summary of Risk Assessments..... 6–82
7.0 Ground Water Compliance Strategy..... 7–1
7.1 Compliance Strategy Selection Process..... 7–1
7.2 Shiprock PEIS Compliance Selection Framework Analysis 7–1
7.2.1 Floodplain Strategy..... 7–2
7.2.2 Terrace Strategy..... 7–2
7.2.2.1 Compliance Strategy for Terrace East 7–2
7.2.2.2 Compliance Strategy for Terrace West..... 7–13
7.3 Risks Assessments 7–14
7.3.1 Human Health 7–14
7.3.2 Ecological Risk 7–14
7.4 Explanation of Floodplain Strategy-Active Remediation and Natural Flushing .. 7–17
7.4.1 Active Remediation 7–17
7.4.2 Natural Flushing..... 7–21
7.5 Explanation of Terrace Compliance Strategies..... 7–22
7.5.1 Terrace East—Active Remedial Action..... 7–22
7.5.2 Terrace West—Supplemental Standards With No Remediation 7–23
7.6 Future Monitoring Activities 7–24
7.6.1 Floodplain Monitoring Requirements..... 7–24
7.6.1.1 Monitor Chemistry in the San Juan River..... 7–24
7.6.1.2 Monitor Chemistry From Other Floodplain Surface Locations... 7–24
7.6.1.3 Monitor Water Levels and Analyze for List of Constituents in Observation Wells Associated with the Extraction Wells 7–24
7.6.1.4 Monitor Water Levels Versus Time and Analyze for List of Constituents from Extraction Wells..... 7–24
7.6.2 Terrace Monitoring Requirements..... 7–25
7.6.2.1 Monitor Water Levels and Chemistry in Wells Around the Disposal Cell and West of U.S. Highway 666..... 7–25

7.6.2.2 Monitor Chemistry in Seeps at the Base of the Terrace, in Bob Lee and Many Devils Washes, and in Seeps West of U.S. Highway 666 ... 7–25
7.6.2.3 Monitor Flow Rates in Washes and Seeps Along the Base of the Escarpment, in Bob Lee Wash, and (if possible) in Many Devils Wash ... 7–27
7.6.2.4 Monitor Storm Event Flows in Many Devils Wash ... 7–27
7.6.2.5 Monitor Volume Versus Time From Extraction Wells ... 7–27
7.6.2.6 Gather Data From the Weather Station ... 7–27
7.6.2.7 Measure Runoff From the Disposal Cell During Heavy Rains ... 7–27
7.6.2.8 Evaluate Lysimeter Installation at the Site ... 7–27
7.7 Completed Interim Actions ... 7–27
7.7.1 Bob Lee Wash ... 7–28
7.7.2 Many Devils Wash ... 7–29
7.7.3 Seeps 425 and 426 ... 7–30
7.7.4 Notes on Interim Actions ... 7–30
7.8 Institutional Controls ... 7–30
7.8.1 Floodplain Controls ... 7–31
7.8.2 Terrace East Controls ... 7–31
7.8.3 Terrace West Controls ... 7–31
7.9 Anticipated Future Land Use ... 7–32
8.0 Development and Evaluation of Active Remediation Alternatives ... 8–1
8.1 Process for Development and Evaluation of Technologies and Alternatives ... 8–2
8.1.1 Overview of the Process ... 8–2
8.1.2 Evaluation Criteria ... 8–3
8.1.2.1 Effectiveness ... 8–3
8.1.2.2 Implementability ... 8–5
8.1.2.3 Cost ... 8–6
8.2 Evaluation of Technologies ... 8–6
8.2.1 Technologies Considered for Remediation ... 8–6
8.2.2 Extraction Technologies ... 8–7
8.2.2.1 Conventional Vertical Wells ... 8–7
8.2.2.2 Horizontal Wells ... 8–8
8.2.2.3 French Drains ... 8–8
8.2.2.4 Choice of Extraction Technology ... 8–8
8.2.3 Effluent Discharge Technologies ... 8–9
8.2.3.1 No Discharge ... 8–9
8.2.3.2 Discharge to Surface Water ... 8–9
8.2.3.3 Injection of Treated Water ... 8–9
8.2.3.4 Effluent Discharge Technologies Recommended for Detailed Evaluation ... 8–9
8.2.4 Treatment Technologies ... 8–9
8.2.4.1 Evaporation Systems ... 8–10
8.2.4.2 Distillation Systems ... 8–10
8.2.4.3 Through-Medium Processes ... 8–11
8.2.4.4 Biological Processes ... 8–11
8.2.4.5 Chemical Treatment ... 8–13
8.2.4.6 Membrane Separation Processes ... 8–13

8.3 Evaluation of Alternatives 8–14
8.3.1 Extraction Alternative 8–14
8.3.1.1 Extraction Technology—Vertical Extraction Wells 8–14
8.3.1.2 Effectiveness 8–14
8.3.1.3 Implementability 8–17
8.3.1.4 Cost 8–17
8.3.2 Treatment Alternatives 8–17
8.3.2.1 Treatment Alternative 1—Solar Evaporation With Spray Enhancement 8–17
8.3.2.2 Treatment Alternative 2—Distillation 8–19
8.3.2.3 Treatment Alternative 3—Biological Denitrification With Membrane Separation 8–21
8.4 Comparative Evaluation of Alternatives 8–24
8.4.1 Comparative Effectiveness 8–24
8.4.1.1 Remediation Time Frame 8–24
8.4.1.2 Conformance with Project Treatment Standards (40 CFR 192) and Goals 8–24
8.4.1.3 Short-Term Effects 8–24
8.4.1.4 Disposal of Treatment Residuals 8–25
8.4.2 Comparative Implementability 8–25
8.4.2.1 Ease of Construction 8–25
8.4.2.2 Ease of Operation and Maintenance 8–26
8.4.2.3 Expected Reliability 8–26
8.4.2.4 Ability To Handle Changes in Influent Composition 8–27
8.4.2.5 Ability To Handle Variations in Extraction Capacity 8–27
8.4.3 Comparative Cost 8–27
8.4.4 Comparative Summary 8–28
8.5 Proposed Remediation Process 8–29
8.5.1 Description of the Proposed Remediation Process 8–29
8.5.1.1 Proposed Extraction System 8–29
8.5.1.2 Proposed Spray-Evaporation Treatment System 8–29
8.5.2 Summary 8–30
8.5.3 Limitations of Proposed Alternative 8–30
9.0 References 9–1

Figures

Figure 1–1. Site Location 1–3
Figure 3–1. 1935 Aerial View of Shiprock, New Mexico, Area 3–13
Figure 3–2. Late 1954 or Early 1955 View to the Southeast of the Early Navajo Mill Operation 3–15
Figure 3–3. Schematic of Navajo Mill Buildings in 1957 3–17
Figure 3–4. August 1962 View of Navajo Mill 3–19
Figure 3–5. July 1965 View Southeast of Navajo Mill Area 3–21
Figure 3–6. May 1980 View Southeast of Millsite Decommissioning 3–23
Figure 3–7. July 1986 View Southeast of Millsite Remediation 3–25
Figure 3–8. Summer 1987 Oblique Aerial Photograph of Completed Disposal Cell-View Northwest 3–27

Figure 4–1. Ground Water and Surface Water Sample Locations for February 2000 Sampling Event and New Wells Sampled in April 2000, Shiprock Site 4–17
Figure 4–2. Location Map of Pumping Tests Completed in the Floodplain Alluvial and the Terrace Ground Water System, Shiprock Site.....4–19
Figure 4–3. Well Cluster Cross Sections for Pumping Tests at (a) the Floodplain Aquifer, (b) the Weathered Mancos Shale, and (c) the Terrace Alluvium at the Shiprock Site 4–21
Figure 4–4. Contour Map of Top of Siltstone Bed in Mancos Shale 4–25
Figure 4–5. Orientation of Vertical Joints Along Escarpment 4–29
Figure 4–6. Rose Diagram of Joint Orientations (azimuths) for Locations Shown on Figure 4–5 4–31
Figure 4–7. Contour Map of the Mancos Shale Bedrock Surface.....4–35
Figure 4–8. Hydrograph of the San Juan River Near Shiprock.....4–43
Figure 4–9. Approximate February 2000 Contours of Equipotential Surface for Both Floodplain Alluvial Aquifer and the Terrace Ground Water System, Shiprock Site 4–49
Figure 4–10. Hydrographs of Selected Floodplain Wells, Shiprock Site.....4–51
Figure 4–11. Hydrographs of Selected Floodplain Wells, Shiprock Site.....4–53
Figure 4–12. Aquifer Test Data Collected While Pumping from Well 858 at 60 gpm: (a) Drawdown in Relation to Time for Observation Well 859, (b) Residual Drawdown in Relation to Dimensionless Time for Observation Well 859, (c) Drawdown in Relation to Time for Observation Well 612, and (d) Residual Drawdown in Relation to Dimensionless Time for Observation Well 612 4–57
Figure 4–13. Generalized Depiction of Flow Components for Alluvial Aquifer, Shiprock Site 4–59
Figure 4–14. Hydrographs of Selected Terrace Alluvial Wells at the Shiprock Site.....4–63
Figure 4–15. Saturated Thickness in Alluvial Material for Floodplain Alluvial Aquifer and Terrace Ground Water System, Shiprock Site 4–65
Figure 4–16. Aquifer Pumping Test Data for Pumping Well 818 Discharge of 1.86 gpm and Pumping Well 817 Discharge of 0.25 gpm: (a) Residual Drawdown in Relation to Dimensionless Time at Pumping Well 818, (b) Drawdown in Relation to Time at Observation Well 604, (c) Residual Drawdown in Relation to Dimensionless Time for Observation Well 604, and (d) Residual Drawdown in Relation to Dimensionless Time at Pumping Well 817 at the Shiprock Site 4–71
Figure 4–17. Piper Diagram of San Juan River Water (March 1999 data) 4–79
Figure 4–18. Uranium Concentrations in the Millsite Floodplain Ground Water.....4–81
Figure 4–19. Spatial Distribution of Concentrations of Sulfate, Uranium, Nitrate, and Total Dissolved Solids in San Juan River Water (June 1999 and February 2000 data) 4–83
Figure 4–20. Nitrate Concentrations in Millsite Floodplain Ground Water.....4–87
Figure 4–21. Sulfate Concentrations in Millsite Floodplain Ground Water 4–89
Figure 4–22. Manganese Concentrations in Millsite Floodplain Ground Water.....4–91
Figure 4–23. Selenium Concentrations in Millsite Floodplain Ground Water.....4–93
Figure 4–24. Uranium Concentrations Over Time in the Millsite Floodplain Ground Water ..4–97
Figure 4–25. Nitrate Concentrations Over Time in the Millsite Floodplain Ground Water 4–99
Figure 4–26. Sulfate Concentrations Over Time in the Millsite Floodplain Ground Water ...4–101
Figure 4–27. Concentrations of Uranium, Sulfate, Nitrate, and Total Dissolved Solids Over Time in the Millsite Floodplain Ground Water.....4–103

Figure 4–28. Concentrations of Uranium, Sulfate, Nitrate, and Total Dissolved Solids Over Time in Ground Water from the Base of the Escarpment on the Millsite Floodplain and Ground Water from the Adjacent Terrace 4–105
Figure 4–29. Uranium Concentrations in the Millsite Floodplain Ground Water from Test Pit Samples 4–107
Figure 4–30. Piper Diagram of Millsite Floodplain and Terrace Ground Water and San Juan River Water 4–109
Figure 4–31. Cross Section F–F' Showing Distribution of U, SO_4, NH_4, NO_3, and pH in Ground Water. February 2000 Sampling; Samples not Analyzed in February 2000 are Indicated by an Alternate Sampling Date. Location of F–F' is shown on Plate 3. 4–113
Figure 4–32. Cross Section G–G' Showing Distribution of U, SO_4, NH_4, NO_3, and pH in Ground Water. February 2000 Sampling; Samples not Analyzed in February 2000 are Indicated by an Alternate Sampling Date. Location of G–G' is shown on Plate 3. 4–115
Figure 4–33. Cross Section H–H' Showing Distribution of U, SO_4, NH_4, NO_3, and pH in Ground Water. February 2000 Sampling; Samples not Analyzed in February 2000 are Indicated by an Alternate Sampling Date. Location of H–H' is shown on Plate 3. 4–117
Figure 4–34. Piper Diagram of Artesian Well 648 Water, Shallow Ground Water in the Bob Lee Wash Area, and Ground Water in the Southeast and Northwest Portions of the Floodplain 4–119
Figure 4–35. Changes in Concentrations of U, SO_4, NO_3, and TDS Over Time for Artesian Well 648 Water, Shallow Ground Water in the Bob Lee Wash Area, and Ground Water in the Northwest Portion of the Floodplain 4–121
Figure 4–36. Uranium Concentrations in the Terrace System Ground Water....4–125
Figure 4–37. Changes in Concentrations of U, SO_4, NO_3, and Total N Over Time for Ground Water from Selected Terrace System Wells and from Floodplain Well 735 4–127
Figure 4–38. Nitrate Concentrations in the Terrace System Ground Water....4–129
Figure 4–39. Ammonium Concentrations in the Terrace System Ground Water 4–131
Figure 4–40. Sulfate Concentrations in the Terrace System Ground Water 4–133
Figure 4–41. Manganese Concentrations in the Terrace System Ground Water 4–135
Figure 4–42. Selenium Concentrations in the Terrace System Ground Water....4–137
Figure 4–43. Vanadium Concentrations in the Terrace System Ground Water 4–139
Figure 4–44. Molybdenum Concentrations in the Terrace System Ground Water 4–141
Figure 4–45. Piper Diagram Comparing Composition of Escarpment Seep Water to Terrace System Ground Water and Many Devils Wash Surface Water 4–145
Figure 4–46. Piper Diagram Showing Composition of Water in High School Wells Compared to Other Terrace System Ground Water 4–147
Figure 4–47. Soil and River Sediment Sample Locations....4–155
Figure 4–48. Nitrate (mg/kg) in Soil and River Sediment....4–157
Figure 4–49. Sulfate (mg/kg) in Soil and River Sediment 4–159
Figure 4–50. Uranium (mg/kg) in Soil and River Sediment 4–163
Figure 4–51. Uranium Concentrations (mg/kg) in Shallow and Deep Soil Samples on the Floodplain....4–167
Figure 4–52. Distribution of Concentrations in Soils Related to Distance From the Escarpment—1 Foot Depth 4–169

Figure 4–53. Distribution of Concentrations in Soils Related to Distance From the Escarpment—Water Table 4–171
Figure 4–54. Nitrate Concentrations (mg/kg) in Shallow and Deep Soil Samples on the Floodplain....4–175
Figure 4–55. Sulfate Concentrations (mg/kg) in Shallow and Deep Soil Samples on the Floodplain....4–177
Figure 4–56. Sample Locations for Salt Deposit Study 4–183
Figure 4–57. Nitrate Concentrations in Salt Deposits 4–189
Figure 4–58. Sulfate Concentrations in Salt Deposits 4–191
Figure 4–59. Uranium Concentrations in Salt Deposits 4–193
Figure 4–60. Column Leaching Results 4–199
Figure 4–61. U-234/U-238 Ratio in Ground Water in Alluvium and Mancos Shale....4–209
Figure 4–62. U Concentration in mg/L....4–211
Figure 4–63. Geographic Extent of Numerical Model and Lines of Schematic Cross Sections for the Shiprock Site 4–217
Figure 4–64. Schemes of Vertical Discretization for the Shiprock Flow and Transport Model (a) schematic geologic cross section from south to north and crossing the disposal cell, (b) south to north geologic cross section with deformed grid superimposed, (c) schematic geologic cross section from west to east , (d) west to east geologic cross section with deformed grid superimposed. 4–219
Figure 4–65. Summary of Hydrologic History of the Shiprock Site 4–223
Figure 4–66. Hydraulic Head and Residuals from Calibrated Four-Layer Model of Terrace Alluvial System and Floodplain Alluvial Aquifer 4–225
Figure 4–67. Plot of Flow-Model Calibration Data, Shiprock Site 4–227
Figure 4–68. Summary of Sensitivity Analysis for Recharge Zones 1 and 3, Shiprock Site ..4–227
Figure 4–69. Results of Particle Tracking Simulation for the Terrace Alluvial Ground Water System and the Floodplain Alluvial Aquifer, Shiprock Site 4–231
Figure 4–70. Model Simulation of Present Nitrogen Concentrations in Ground Water in Layer 1 of the MT3D Transport Model for the Shiprock Site 4–235
Figure 4–71. Model Simulation of Present Uranium Concentrations in Ground Water 4–239
Figure 4–72. Map of Area where Remedial Actions were Simulated for the Shiprock Site...4–241
Figure 4–73. Simulated Decline in Water Level in Feet Near the Disposal Cell 4–245
Figure 4–74. Projection of Simulated Uranium Plume 13.8 Years After Pumping Begins in the Floodplain Alluvial Aquifer 4–247
Figure 4–75. Projection of Simulated Nitrate Plume 13.8 Years After Pumping Begins in the Floodplain Alluvial Aquifer 4–249
Figure 4–76. Shiprock Millsite Floodplain Vegetation Map....4–255
Figure 4–77. Surface Water, Sediment, and Vegetation Sample Locations for the Ecological Risk Assessment....4–261
Figure 5–1. Physiographic Block Diagram of the Shiprock Site....5–3
Figure 5–2. Site Conceptual Model of the Shiprock Site 5–5
Figure 6–1. Ground Water and Surface Water Exposure Intake Equations and Default Assumptions 6–4
Figure 6–2. Ecological Risk Model for the Shiprock Site....6–24
Figure 6–3. Shiprock Ecological Site Conceptual Model 6–41
Figure 6–4. Generalized Food Web for Shiprock Ecological Receptors....6–45
Figure 7–1. Ground Water Compliance Selection Framework for the Floodplain 7–3
Figure 7–2. Ground Water Compliance Selection Framework for Terrace East....7–7
Figure 7–3. Ground Water Compliance Selection Framework for Terrace West 7–11

Figure 7–4. Approximate Boundary of the Terrace East and Terrace West Ground Water Systems, and Proposed Extraction Well and French Drain Locations.....7–15
Figure 7–5. Surface and Well Sampling Locations for Future Monitoring at the Shiprock Site 7–19
Figure 8–1. Proposed Extraction Wells and French Drain Systems for the Floodplain Alluvial Aquifer and the Terrace Ground Water System at the Shiprock Site 8–15

Tables

Table 2–1. Maximum Concentration Limits of Inorganic Constituents in Ground Water at UMTRA Project Sites 2–2
Table 4–1. Follow-up Action to Meet Additional Data Needs 4–2
Table 4–2. Summary of Tasks Completed in 1998 With Each Drilling Rig.....4–6
Table 4–3. Construction Details for Monitor Wells at the Shiprock Site 4–9
Table 4–4. Surface Water Quality Parameters for Selected Analytes Monitored at U.S. Geological Survey Gauge 09368000 at Shiprock 4–45
Table 4–5. Water Balance for the Floodplain Alluvial Aquifer at the Shiprock Site 4–55
Table 4–6. Estimate of the Minimum Volume of Ground Water in Alluvial Material in the Buried Ancestral River Channel Section of the Terrace Ground Water System South of the Disposal Cell.....4–67
Table 4–7. Summary of Packer Test Results, Shiprock Site.....4–68
Table 4–8. Preliminary Water Balance for the Terrace Ground Water System at the Shiprock Site 4–73
Table 4–9. Visible Ground Water Discharge Along the Escarpment 4–75
Table 4–10. Constituents Evaluated in the Geochemistry Section 4–77
Table 4–11. Background Concentrations in the San Juan River (upgradient) 4–77
Table 4–12. Summary of Concentrations in Surface Water Samples for Selected Contaminants.....4–85
Table 4–13. Background and Concentration Range of Selected Contaminants in Ground Water.....4–86
Table 4–14. Analysis of Water from the Neutron Hydroprobe Ports on Top of the Disposal Cell 4–96
Table 4–15. Water Quality of Samples from Artesian Well 648 (June 1998 sampling) 4–112
Table 4–16. Concentrations in mg/L of Kjeldahl Nitrogen, Nitrate (as N), and Ammonium (as N), Sampled in December 1998. 4–144
Table 4–17. Concentrations of Radium and Thorium Isotopes, February 2000 Sampling (filtered samples).....4–149
Table 4–18. Alkenes, Cycloalkenes, and Unidentified Compounds Detected in the NECA Pond (849) and Terrace Wells.....4–151
Table 4–19. Concentrations of Constituents (mg/kg) in Soils and Sediments (5-percent HCl leach) 4–154
Table 4–20. Concentration of Constituents in Soil and Sediments Adjacent to the Escarpment.....4–166
Table 4–21. Summary of R_d Determinations 4–180
Table 4–22. Summary of R_d Determinations Omitting Outliers 4–180
Table 4–23. R_d Values for Shiprock Floodplain, Adjusted Using Grain Size Distributions .4–180
Table 4–24. Concentrations (%) of Major Ions in the Salt Deposit Samples [a] 4–186

Table 4–25. Concentrations of Nitrate, Sulfate, and Uranium in Salt Deposit Samples (ESL Data)[a] 4–187
Table 4–26. Concentrations (mg/kg) of Trace Elements in the Salt Deposit Samples[a] 4–188
Table 4–27. Analyses (mg/L) of Ground Water from Terrace Wells in Irrigated Area....4–215
Table 4–28. Analyses (mg/L) from Terrace Wells in a Nonirrigated Area....4–215
Table 4–29. Summary of Hydrologic Parameters used in the Flow Model 4–221
Table 4–30. Summary of Residuals and Calibration Statistics for the Calibrated Ground Water Flow Model, Shiprock Site....4–230
Table 4–31. Summary of Residuals and Calibration Statistics for the Nitrogen Transport Model, Shiprock Site....4–237
Table 4–32. Time Stepping Schedule for Transient Simulations of Remedial Action at the Shiprock Site 4–238
Table 4–33. Summary of Withdrawal Rates for the Simulations of Active Remediation at the Shiprock Site 4–243
Table 4–34. Relevé Plant Cover for the Shiprock Millsite Floodplain and Floodplain Reference Area[a]....4–257
Table 4–35. Relevé Plant Cover for the Shiprock Millsite and Reference Wetlands[a] 4–258
Table 4–36. Summary of Surface Water Sampling Parameters—1998[a]....4–264
Table 4–37. Summary of Sediment Sampling Parameters-1998 and 1999 4–265
Table 4–38. Summary of Biota Sampling Parameters (1998 and 1999)....4–266
Table 5–1. Estimated Magnitudes of Major Flow Components for Floodplain Alluvial Aquifer and Terrace Ground Water System at the Shiprock Site 5–7
Table 6–1. Floodplain Alluvial Aquifer Data Summary 6–6
Table 6–2. Floodplain Aquifer Residential Ground Water Ingestion Risk Calculations....6–9
Table 6–3. Non-Carcinogens—Inhalation through Water Use in Residential Setting—Floodplain 6–10
Table 6–4. Floodplain Ground Water Incidental Ingestion/Dermal Exposure Pathways....6–12
Table 6–5. Meat/Milk Exposure Calculations—Floodplain Ground Water 6–13
Table 6–6. Terrace Ground Water Residential Ground Water Ingestion Risk Calculations..6–14
Table 6–7 Incidental Terrace Ground Water Ingestion/Dermal Exposure Pathways....6–16
Table 6–8. Meat/Milk Exposure Calculations—Terrace Ground Water....6–17
Table 6–9. Floodplain Surface Water Incidental Ingestion/Dermal Exposure Pathways 6–18
Table 6–10. Meat/Milk Exposure Calculations—Floodplain Surface Water 6–19
Table 6–11. Incidental Surface Water Ingestion/Dermal Exposure Pathways—Bob Lee and Many Devils Washes....6–20
Table 6–12. Summary of Potential Human Health Risks 6–21
Table 6–13. Meat/Milk Exposure Calculations—Terrace Surface Water (seeps and washes) 6–22
Table 6–14. Summary of T&E Survey Results....6–29
Table 6–15. Summary of Ecological Contaminants of Potential Concern in Ground Water, Surface Water, and Sediments (1994 BLRA) 6–30
Table 6–16. Summary of Ecological Contaminants of Potential Concern in Area A: the Distributary Channel and Tributaries (based on data collected from 1998 through 2000) 6–34
Table 6–17. Summary of Ecological Contaminants of Potential Concern in Area B: the San Juan River (based on data collected from 1998 through 2000)....6–35
Table 6–18. Summary of Ecological Contaminants of Potential Concern in Area C: the Shiprock Floodplain (based on data collected from 1998 through 2000) 6–36

Table 6–19. Summary of Ecological Contaminants of Potential Concern in Area D Bob Lee Wash (based on data collected from 1998 through 2000) 6–37
Table 6–20. Summary of Ecological Contaminants of Potential Concern in Area E: Many Devils Wash (based on data collected from 1998 through 2000) 6–38
Table 6–21. Summary of Ecological Contaminants of Potential Concern at Areas Associated with the Shiprock Millsite Ground Water Based on Most Recent Analytical Data 6–39
Table 6–22. Management Goals, Assessment Endpoints, and Measurement Endpoints for the Evaluation of Ecological Risks at the Shiprock Site 6–48
Table 6–23. Uptake Model Parameters and Bioaccumulation Factors for Ecological Contaminants of Potential Concern 6–52
Table 6–24. Exposure Parameters for Livestock and Wildlife Receptors 6–54
Table 6–25. Surface Water and Sediment Quality Benchmarks for Ecological Contaminants of Potential Concern for the Protection of Freshwater Aquatic Life 6–55
Table 6–26. Plant Toxicity Benchmarks for Ecological Contaminants of Potential Concern 6–56
Table 6–27. Mammal Toxicity Benchmarks for Ecological Contaminants of Potential Concern 6–57
Table 6–28. Avian Toxicity Benchmarks for Ecological Contaminants of Potential Concern 6–57
Table 6–29. Summary of Surface Water and Sediment Data from the Distributary Channel and Tributaries (Area A) 6–59
Table 6–30. Hazard Quotients for Aquatic Organisms, Benthic Organisms, and Plants at the Distributary Channel and Tributaries (Area A) Based on Comparisons of Media Concentrations to Water Quality, Sediment Quality, and Plant Toxicity Benchmarks 6–59
Table 6–31. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors at the Distributary Channel and Tributaries (Area A) 6–60
Table 6–32. Hazard Quotients for Riparian/Wetland Wildlife Receptors at the Distributary Channel and Tributaries (Area A) 6–61
Table 6–33. Summary of Surface Water and Sediment Data from the San Juan River (Area B) 6–62
Table 6–34. Hazard Quotients for Aquatic Organisms, Benthic Organisms, and Plants at the San Juan River (Area B) Based on Comparisons of Media Concentrations to Water Quality, Sediment Quality, and Plant Toxicity Benchmarks 6–62
Table 6–35. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors at the San Juan River (Area B) 6–64
Table 6–36. Hazard Quotients for Riparian/Wetland Wildlife Receptors at the San Juan River (Area B) 6–65
Table 6–37. Summary of Surface Water, Sediment, and Soil Data from the Shiprock Floodplain (Area C) 6–66
Table 6–38. Hazard Quotients for Aquatic Organisms, Benthic Organisms, and Plants at the Shiprock Floodplain (Area C) Based on Comparisons of Media Concentrations to Water Quality, Sediment Quality, and Plant Toxicity Benchmarks 6–67
Table 6–39. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors at the Shiprock Floodplain (Area C) 6–68
Table 6–40. Hazard Quotients for Riparian/Wetland Wildlife Receptors at the Shiprock Floodplain (Area C) 6–69

Table 6–41. Summary of Surface Water, Sediment, and Soil Data from Bob Lee Wash (Area D) 6–69
Table 6–42. Hazard Quotients for Aquatic Organisms, Benthic Organisms, and Plants at Bob Lee Wash (Area D) Based on Comparisons of Media Concentrations to Water Quality, Sediment Quality, and Plant Toxicity Benchmarks 6–70
Table 6–43. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors at Bob Lee Wash (Area D) 6–71
Table 6–44. Hazard Quotients for Riparian/Wetland Wildlife Receptors at Bob Lee Wash (Area D) 6–72
Table 6–45. Summary of Surface Water and Sediment Data from Many Devils Wash (Area E) 6–73
Table 6–46. Hazard Quotients for Aquatic Organisms, Benthic Organisms, and Plants at Many Devils Wash (Area E) Based on Comparisons of Media Concentrations to Water Quality, Sediment Quality, and Plant Toxicity Benchmarks 6–73
Table 6–47. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors at Many Devils Wash (Area E) 6–74
Table 6–48. Hazard Quotients for Riparian/Wetland Wildlife Receptors at Many Devils Wash (Area E) 6–75
Table 6–49. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors on the Upland Terrace (Area F) 6–76
Table 6–50. Summary of Salt Crust Analysis Data from Areas A, C, D and E 6–77
Table 6–51. Hazard Quotients for Terrestrial and Wetland Wildlife and Livestock Receptors Based on the Assumption of 100% Salt Crust Ingestion at Areas A, C, D, and E 6–77
Table 6–52. Hazard Quotients for Radiological COPCs 6–78
Table 6–53. Summary of Potential Ecological Risks at the Shiprock Site (see text for definition of risk categories) 6–81
Table 6–54. Floodplain Human Health Risk COPC Updates 6–83
Table 6–55. Terrace Human Health Risk COPC Updates 6–83
Table 6–56. Summary of Ecological Contaminants of Potential Concern at the Shiprock Millsite Based on the Ecological Risk Screening Results 6–84
Table 7–1. Explanation of the Compliance Strategy Selection Process for the Floodplain Alluvial Aquifer 7–5
Table 7–2. Explanation of the Compliance Strategy Selection Process for Terrace East Ground Water 7–9
Table 7–3. Explanation of the Compliance Strategy Selection Process for Terrace West Ground Water 7–13
Table 7–4. Summary of Monitoring Requirements for the Floodplain 7–25
Table 7–5. Summary of Monitoring Requirements for the Terrace 7–26
Table 7–6. Surface Water Contamination in Bob Lee and Many Devils Washes 7–28
Table 7–7. Surface Water Contamination in Bob Lee Wash 7–29
Table 8–1. Average Composition of Shiprock Contaminant Plumes 8–4
Table 8–2. Average Composition of Shiprock, Tuba City, and Monument Valley Contaminant Plumes 8–7
Table 8–3. Performance of Distillation System 8–19
Table 8–4. Removal Efficiency for RO and Nanofiltration Processes 8–21
Table 8–5. Predicted Effluent Concentration from Nanofiltration System 8–22
Table 8–6. Costs of Treatment Processes 8–28
Table 8–7. Comparative Ranking of Treatment Alternatives 8–28

Plates will be provided upon request. Click plates to request

Plate 1. Shiprock Base Map
Plate 2. October 1997 Aerial Photograph and Selected Features from Shiprock Base Map
Plate 3. Shiprock Geologic Map
Plate 4. Shiprock Geologic Cross Sections
Plate 5. Ecological Risk Assessment Areas

Appendices (Volume II) will be provided upon request. Click appendices to request

Appendix A. Lithologic and Monitor Well Completion Logs A–1
Appendix B. Summary of Surface Water and Ground Water Analytical Results, 1997 through April 2000 Sampling.....B–1
Appendix C. Surface Water Analytical Results .....CD-ROM
Appendix D. Ground Water Analytical Results and Water Level Data.....CD-ROM
Appendix E. Sediment, Soil, and Salt Crust Sample Analytical Results.....CD-ROM
Appendix F. Graduated-Symbol Plots for Ground Water Analyses, February 2000 Sampling, or Most Recent Sampling F–1
Appendix G. Vegetation (Ecological) Sample Analytical Results.....CD-ROM
Appendix H. Data and Calculations for Ecological Risk Assessment H–1

Acronyms and Abbreviations

ACL	alternate concentration limit
AEC	U.S. Atomic Energy Commission
AML	abandoned mine lands
AR	activity ratio
ASTM	American Society for Testing and Materials
AWQC	ambient water quality criteria
BAF	bioaccumulation factor
BCF	bioconcentration factor
BLRA	baseline risk assessment
BLS	below land surface
CFR	*Code of Federal Regulations*
cfs	cubic feet per second
cm/s	centimeters per second
CoC	chain of custody
COC	contaminants of concern
COPC	contaminants of potential concern
CV	coefficient of variation
DOE	U.S. Department of Energy
EA	environmental assessment
EHPA	di(2-ethylhexyl) phosphoric acid
EPA	U.S. Environmental Protection Agency
ERA	ecological risk assessment
ESC	expedited site characterization
ESL	Environmental Sciences Laboratory
FR	*Federal Register*
ft	foot (feet)
ft^2	square feet
ft^3	cubic feet
ft/day	foot (feet) per day
ft^2/day	square feet per day
ft^3/day	cubic feet per day
ft^3/yr	cubic feet per year
g	gram
GCAP	Ground Water Compliance Action Plan
GCMS	gas chromatography/mass spectroscopy
GIS	geographic information system
GJO	Grand Junction Office
gpm	gallons per minute
GPS	global positioning system
HCl	hydrochloric acid
HDPE	high-density polyethylene
HI	hazard index
HQ	hazard quotient
in.	inches
ITRD	Innovative Treatment Remediation Demonstration
K_d	distribution coefficient

LOAEL	lowest-observed-adverse-effect level
MAP	management action process
MCL	maximum concentration limit
MDRD	minimum detectable relative difference
MGD	million gallons per day
mg	milligram
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
mi	mile(s)
mi^2	square miles
mL	milliliters
mL/g	milliliters per gram
µg/L	micrograms per liter
µm	micrometer
µS/cm	microsiemens per centimeter
mm	millimeter
NABIR	Natural and Accelerated Bioremediation Research (Program)
NAPI	Navajo Agricultural Products Industries
NDEM	Navajo Department of Emergency Management
NECA	Navajo Engineering and Construction Authority
NEPA	National Environmental Policy Act
NFWD	Navajo Fish and Wildlife Department
NGVD	National Geodetic Vertical Datum
NIA	Navajo Irrigation Authority
NNEPA	Navajo Nation Environmental Protection Agency
NNSWQS	Navajo Nation Surface Water Quality Standard
NOAEL	no-observed-adverse-effect-level
NRC	U.S. Nuclear Regulatory Commission
NTUA	Navajo Tribal Utility Authority
NWCA	Navajo Water Code Administration
pCi/L	picocuries per liter
PEIS	Programmatic Environmental Impact Statement (for the UMTRA Ground Water Project)
PVC	polyvinyl chloride
Ra-226	radium-226
Ra-228	radium-228
RAP	remedial action plan
RBC	risk-based concentration
Rd	distribution ratio
RfD	reference dose
RO	reverse osmosis
RRM	residual radioactive material
SDWA	Safe Drinking Water Act
SOWP	site observational work plan
SX	solvent extraction
TBP	tributyl phosphate
T&E	threatened and endangered
TDS	total dissolved solids
TEL	threshold effect level

Th-230	thorium-230
Th-232	thorium-232
U-234	uranium-234
U-238	uranium-238
U_3O_8	uranium oxide
UCL_{95}	95 percent upper confidence limit
UIC	Underground Injection Control (Program)
UMTRA	Uranium Mill Tailings Remedial Action (Project)
UMTRCA	Uranium Mill Tailings Radiation Control Act
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
V_2O_5	vanadium oxide
VCA	Vanadium Corporation of America
ZVI	zero-valent iron

Executive Summary

Shiprock, New Mexico, was the site for a uranium and vanadium ore-processing mill operated from 1954 through 1968. During milling operations, contaminants infiltrated the terrace ground water system (alluvial material and weathered Mancos Shale) where the mill was located and the adjacent floodplain alluvial aquifer along the San Juan River. The two tailings piles at the site were combined and stabilized in one on-site disposal cell along with material from the nearby raffinate ponds and the adjacent floodplain. Cleanup of surface contamination and placement of this material in the disposal cell was completed in 1986. This remediation was conducted in accordance with U.S. Environmental Protection Agency (EPA) criteria in "Health and Environmental Protection Standards for Uranium and Thorium Mill Tailings" (40 CFR Part 192 Subpart A) as part of the Uranium Mill Tailings Remedial Action (UMTRA) Surface Project.

As part of the UMTRA Ground Water Project, additional characterization conducted in 1998 and early 1999 by the U.S. Department of Energy (DOE) Grand Junction Office was presented in the 1999 Site Observational Work Plan (SOWP), Revision 1. It revealed that contamination from former milling operations was more extensive than previously known. Contamination affected the floodplain aquifer and the terrace ground water system immediately adjacent to the disposal cell, and extended about 1 mile northwest into an irrigated area and about 0.6 mile southeast to Many Devils Wash. Additional characterization was proposed to further define the extent of contamination and to better quantify ecological risks. These additional tasks were conducted from summer 1999 to spring 2000, and results of all the characterization and risk evaluations conducted since 1998 are included in this SOWP.

The site is geographically and hydrologically divided into the floodplain and terrace regions. The floodplain is in hydrologic contact with the San Juan River and receives inflow from the terrace system; therefore, any remediation must accommodate these connections. The terrace is divided into Terrace East and Terrace West, reflecting different degrees of contamination and sources of ground water.

Contaminants of concern (COCs) in floodplain ground water are nitrate, uranium, sulfate, selenium and manganese. In addition, ammonium and strontium are of potential concern to the ecology in this area. Concentrations of COCs are generally highest near the escarpment base just north of the disposal cell, and the plume of contamination continues northward in an arc toward the San Juan River. Concentrations of COCs are lowest in the northwest and southeast areas of the floodplain. In the northwest, surface water from Bob Lee Wash, containing relatively clean ground water from flowing artesian well 648, naturally flushes the ground water in the floodplain aquifer. In the southeast, clean water from the San Juan River recharges the aquifer.

Nitrate concentrations in ground water are generally between 2,000 and 3,500 milligrams per liter (mg/L) in samples collected along the escarpment base just north of the disposal cell and north to the river. Concentrations of uranium exceed 2 mg/L in ground water samples obtained along the base of the escarpment just north of the disposal cell and reach almost 4 mg/L in samples collected north near the San Juan River. In the west part of the floodplain, both uranium and nitrate concentrations in ground water samples drop below their respective UMTRA Project maximum concentration limits (MCLs) in the area flushed by water from Bob Lee Wash. Sulfate concentrations are about 10,000 mg/L in samples collected along the base of the escarpment, but exceed 25,000 mg/L in samples obtained north near the river. Selenium concentrations are generally 0.1 to 1.0 mg/L in samples obtained along the escarpment base; however, these higher

concentrations do not extend northward toward the San Juan River. Manganese concentrations are generally from 5 to 10 mg/L in samples collected along the base of the escarpment and north to the river; these concentrations are higher than background floodplain concentrations of about 2 mg/L. The ecological risk evaluation concluded that ammonium and strontium concentrations in floodplain surface water and sediment are elevated and may pose a low to medium-low potential risk to aquatic receptors.

Contaminants of potential concern (COPCs) in the terrace ground water system are nitrate, sulfate, uranium, selenium, ammonium, manganese, and strontium. Highest concentrations of these COPCs are generally in ground water samples obtained around the former millsite, the disposal cell, and Bob Lee and Many Devils Washes. Irrigated areas to the northwest have much lower concentrations because of the natural flushing effects of irrigation. Nitrate concentrations range up to 7,250 mg/L in recent ground water samples collected in the areas west and south of the disposal cell; concentrations decrease in samples from the irrigated area but still exceed the UMTRA Project standard of 44 mg/L in places. No ground water standards have been established for sulfate; however, concentrations exceed 10,000 mg/L in samples collected as far as 2,500 feet west and 3,000 feet southeast of the disposal cell and decrease to generally less than 5,000 mg/L in samples from the irrigated area. Maximum concentrations of uranium in recent ground water and surface water samples are about 3 mg/L between the disposal cell and Bob Lee Wash but decrease rapidly to the west and south and are near the MCL of 0.044 mg/L in the irrigated areas to the northwest. High selenium concentrations (over 6 mg/L) occur in ground water samples from an area about 2,000 feet southwest and 3,500 feet west of the disposal cell. Farther west in the irrigated area, the selenium concentrations in ground water samples decrease to less than 1 mg/L but still exceed the Safe Drinking Water Act standard of 0.05 mg/L in most locations. No ground water standards have been established for ammonium and manganese; however, concentrations reach nearly 2,000 mg/L and 30 mg/L, respectively, in samples from areas adjacent to the disposal cell. As in the floodplain, the ecological risk evaluation concluded that ammonium and strontium concentrations in surface water in Bob Lee and Many Devils Washes and in the distributary channel may pose a low to medium-low potential risk to aquatic receptors.

DOE's goal at the Shiprock site is to implement a ground water remediation strategy that complies with EPA ground water standards and protects human health and the environment. The requirements for ground water compliance at UMTRA Project sites, including the Shiprock site, are in the Uranium Mill Tailings Radiation Control Act (42 *United States Code* §7901 *et seq*.). The compliance selection framework was developed in the *Final Programmatic Environmental Impact Statement for the Uranium Mill Tailings Remedial Action Ground Water Project* (DOE 1996b). Three compliance strategies are proposed for the Shiprock site—one each for the floodplain, Terrace East, and Terrace West.

The proposed strategy for the floodplain surficial aquifer is active remediation in combination with natural flushing. Ground water will be pumped for 14 years from extraction wells located in the most contaminated area of the floodplain and piped to a lined pond on the terrace south of the disposal cell, where it will be evaporated by jet spray. The remainder of the contaminant plume in the floodplain will undergo natural flushing. Numerical modeling of ground water flow and transport indicates that the COCs will diminish to acceptable levels within 100 years if no continued source of contamination exists. DOE will monitor and sample the floodplain and terrace systems for 5 years after remedial action begins to evaluate drainage of residual moisture from the disposal cell and the possibility that a continued source exists. At the end of this period,

DOE will report findings and consult with stakeholders and determine future actions, if any, for the site. During this period, institutional controls and interim actions will protect humans and the environment from potential risks posed by the contaminants.

The proposed strategy for Terrace East, the area around the disposal cell, including Bob Lee Wash and Many Devils Wash and west approximately to U.S. Highway 666, is active remediation. The lack of ground water in terrace alluvium upgradient of the former millsite, tailings piles, and raffinate ponds indicates that this area was dry prior to milling operations. Process solutions used in milling migrated vertically downward and saturated the base of the alluvium and the underlying weathered Mancos Shale, creating an artificial ground water regime with high concentrations of COPCs. Active remedial action is proposed for this region and consists of pumping the most contaminated ground water from an extraction well system, piping it to a lined pond south of the disposal cell, and treating it by spray evaporation. This treatment will continue until the terrace ground water system is hydrologically disconnected from the washes and seeps along the escarpment causing the escarpment seeps and seeps in the washes to dry up. This strategy will succeed only if no continued source of recharge is present. If a continuing source is present, numerical modeling predicts that contamination will recur if pumping is discontinued. Therefore, DOE will monitor and sample for 5 years after the start of remedial action to evaluate the nature of any drainage of residual moisture from the disposal cell. At the end of this period, DOE will report findings, consult with stakeholders, and reevaluate the implemented compliance action. During the period of remediation, interim actions in Bob Lee Wash, Many Devils Wash, and seeps 425 and 426 will protect humans and the environment from surface occurrences of contaminated ground water.

The proposed strategy for Terrace West, the area generally west of U.S. Highway 666, is no remediation and application of supplemental standards based on the criterion of limited use ground water. This area is underlain by Mancos Shale, and although milling activities may have contributed contamination, most of the contaminant mass is due to leaching of the Mancos Shale by irrigation water, and concentrations will remain elevated as long as irrigation continues. Therefore, the ground water system contains widespread ambient contamination—from uranium, selenium, and sulfate—not due to milling activities and that cannot be cleaned up using methods reasonably employed in public water systems. The presence of nitrate in this region may be due to milling activities or may have other anthropogenic sources. Pumping water from Terrace East will further isolate this region from the millsite, but continued irrigation will release additional naturally occurring uranium, selenium, and sulfate from the Mancos Shale. DOE will continue to monitor and sample ground water for COPCs in this area for at least the next 5 years to ensure that it is not being adversely affected by former millsite-related activities.

Complete pathways for exposure to contaminated ground water existed in the upper part of Bob Lee Wash, the lower part of Many Devils Wash, and at escarpment seeps 425 and 426. Interim actions designed to protect humans and the environment from contaminated ground water that surfaces at several locations were completed in 2000. A first phase was construction of fences around Bob Lee Wash, Many Devils Wash, and seeps 425 and 426. Signs indicating that water should not be used for drinking were placed on all fencing. A second phase was placement of riprap in the bottoms of Bob Lee Wash and Many Devils Wash where pooling was known to occur. An initial layer of cobbles was overlain with geotextile netting, which was overlain by a second layer of larger cobbles. This will effectively limit access to any surfacing ground water by animals and humans. The last interim action was surface netting along seeps 425 and 426 in the floodplain to limit access to the water, especially by birds.

In conclusion, the Shiprock site exhibits three distinct regions where ground water is contaminated due to uranium milling activities. They are the floodplain, Terrace East, and Terrace West regions based on physiography, source of recharge, proximity to the former millsite, and levels of contamination. DOE proposes three compliance strategies: 1) floodplain—active remediation and natural flushing, 2) Terrace East—active remediation to prevent contaminated ground water from surfacing in washes and seeps, and 3) Terrace West—no action based on supplemental standards due to widespread ambient contamination. Interim actions—fencing, riprapping, and netting—were implemented in Many Devils Wash, Bob Lee Wash, and at two seeps along the floodplain to protect humans and animals from contacting contaminated ground water surfacing in these locations. DOE believes that these three compliance strategies will address ground water contamination at the Shiprock site and will be protective of humans and the environment.

1.0 Introduction

1.1 Purpose and Scope

The Shiprock Uranium Mill Tailings Remedial Action (UMTRA) Project site is on the Navajo Indian Reservation (Navajo Nation) in northwestern New Mexico, approximately 1 mile (mi) south of Shiprock and about 30 mi west of Farmington (Figure 1–1). The site is just south of the San Juan River and east of U.S. Highway 666, on an elevated gravel-covered terrace overlooking the river and its floodplain.

The U.S. Department of Energy (DOE) completed remedial action of surface and near-surface contamination in 1986. Contaminated materials were stabilized on site in a disposal cell that covers approximately 76 acres. However, ground water affected by the uranium-ore processing at the site contains constituents in concentrations exceeding ground water protection standards established by the U.S. Environmental Protection Agency (EPA) in Title 40, Part 192 of the *Code of Federal Regulations* (40 CFR 192). Affected ground water is within the terrace material and weathered bedrock south of the San Juan River and also within an alluvial aquifer in the floodplain below.

DOE's goal is to implement a compliance strategy that is protective of human health and the environment by remediating contaminated ground water at the Shiprock site to meet the EPA standards. This final site observational work plan (SOWP) documents the data collection and data evaluation leading to the selection of an overall compliance strategy and remedial alternative that meets the regulatory requirements for ground water. This document is also a source of information for stakeholders for their input to the process of selecting remedial alternatives.

Compliance requirements for meeting the regulatory standards at the Shiprock site are presented in Section 2.0, "Regulatory Framework." Site background information, including an overview and history of the former milling operation and current and anticipated water and land use, are reviewed in Section 3.0, "Site Background." Results of characterization activities conducted at the site are presented in Section 4.0, "Site Characterization Results." The site conceptual model is presented in Section 5.0, "Site Conceptual Model." Summaries of potential human health and ecological risks associated with ground water and surface water contamination are presented in Section 6.0, "Baseline Risk Assessment." The selected compliance strategies are presented in Section 7.0, "Ground Water Compliance Strategy," and a remedial alternatives evaluation and the proposed alternative are presented in Section 8.0, "Development and Evaluation of Active Remediation Alternatives." References are listed in Section 9.0, "References." Appendices include lithologic and well completion logs, summary of recent water sample analyses, analytical results of all sampling, concentration plots based on analytical results of ground water samples, and risk assessment data.

1.2 UMTRA Project Programmatic Documents

Programmatic documents that guide the SOWP include the *UMTRA Ground Water Project Management Action Process Document (MAP)* (DOE 1999i) and the *Final Programmatic Environmental Impact Statement for the Uranium Mill Tailings Remedial Action Ground Water Project* (PEIS) (DOE 1996b). The MAP states the mission objectives of the UMTRA Ground

Water Project and provides a technical and management approach for conducting the project. The PEIS is the programmatic decision-making framework for conducting the UMTRA Ground Water Project. DOE follows PEIS guidelines to assess the potential programmatic impacts of the Ground Water Project, to determine site-specific ground water compliance strategies, and to prepare site-specific environmental impact analyses more efficiently.

1.3 Relationship to Site-Specific Documents

The surface remedial action plan (RAP) (DOE 1985) provides early site characterization information. However, no ground water protection strategy was determined for the Shiprock disposal site because the RAP was conditionally approved by the U.S. Nuclear Regulatory Commission (NRC) in 1985, before the proposed EPA ground water standards were finalized. The characterization information in the RAP was used in developing the SOWP to strengthen the site conceptual model. After the ground water compliance strategy and remedial alternatives are selected for this site, a ground water compliance action plan (GCAP) will be prepared to document the remediation decision.

In 1994, DOE prepared a baseline risk assessment (BLRA) (DOE 1994) and supplement (DOE 1996d) that identified potential public health and environmental risks at the site. Potential risks identified in the BLRA are considered and updated in this SOWP to ensure that the proposed compliance strategy is protective of human health and the environment.

After a proposed compliance strategy is identified in the SOWP and described in the GCAP, a site-specific National Environmental Policy Act (NEPA) document (e.g., an environmental assessment) will be prepared, as required by the NEPA process, to determine the potential effects, if any, of implementing the proposed compliance strategy.

1.4 SOWP Revisions

This SOWP was a multiyear process of sequenced field data-collection activities and document preparation that consisted of three versions: Revision 0 (considered a draft), Revision 1, and Revision 2 (considered a final). The draft SOWP was prepared in 1995 and included all previous information about the site. The draft SOWP presented a proposed compliance strategy and defined additional data that were necessary to support the most likely compliance strategy. DOE prepared a work plan detailing characterization activities (DOE 1998d) and, in conjunction with stakeholder review, conducted fieldwork in 1998 and early 1999 to address the data gaps identified in the draft SOWP. Following the evaluation of the new data, additional data gaps were identified in the SOWP, Revision 1, in Section 4.7 “Summary of Additional Data Needs.” These data needs were primarily related to the extent of contamination in the terrace area and floodplain. Most data needs were investigated with stakeholder input during fieldwork in late 1999 and early 2000. Those additional data are evaluated and the proposed ground water compliance strategy and remedial alternatives are updated and presented in Revision 2 (this document).

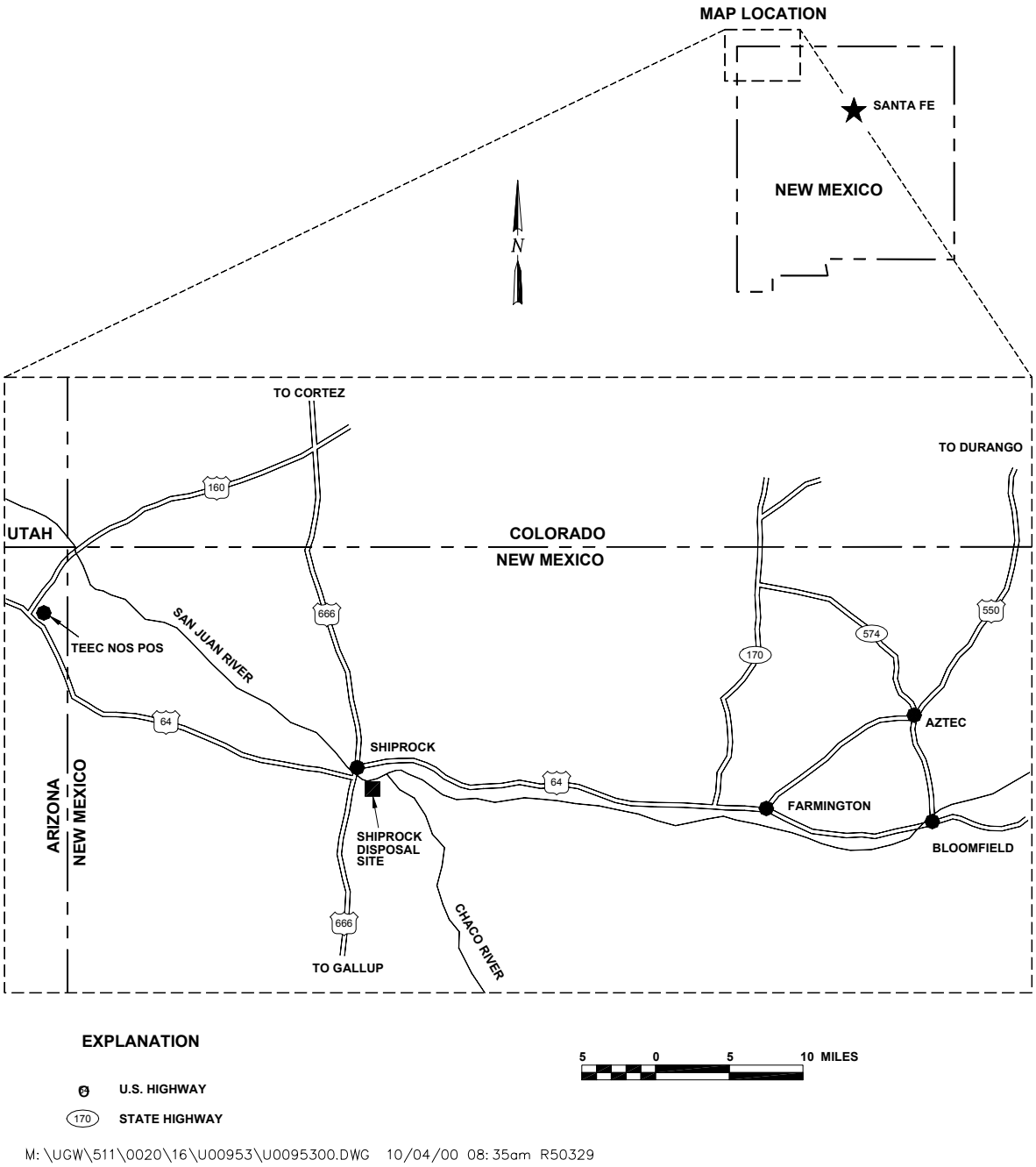

Figure 1–1. Site Location

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2.0 Regulatory Framework

This section identifies the regulatory framework to be applied to the proposed ground water compliance strategy at the former Shiprock millsite to achieve compliance with Subpart B of EPA's "Health and Environmental Protection Standards for Uranium and Thorium Mill Tailings" (40 CFR Part 192).

2.1 Uranium Mill Tailings Radiation Control Act

The United States Congress passed the Uranium Mill Tailings Radiation Control Act (UMTRCA) (42 U.S.C. §7901 *et seq.*) in 1978 in response to public concerns about potential health hazards from long-term exposure to uranium mill tailings. UMTRCA authorized DOE to stabilize, dispose of, and control uranium mill tailings and other contaminated materials at inactive uranium-ore processing sites.

The Shiprock site is designated under Title I of the three UMTRCA titles that apply to uranium-ore processing sites. Title I designates 24 inactive processing sites for remediation. It directs EPA to promulgate standards, mandates remedial action in accordance with these standards, stipulates that remedial action be selected and performed with the concurrence of the NRC and in consultation with the states and Indian tribes, directs NRC to license the disposal sites for long-term care, and directs DOE to enter into cooperative agreements with the affected states and Indian tribes.

In 1988, Congress passed the Uranium Mill Tailings Remedial Action Amendments Act (42 U.S.C. §7922 *et seq.*), authorizing DOE to extend without limitation the time needed to complete ground water remediation activities at the processing sites.

2.2 EPA Ground Water Protection Standards

UMTRCA requires EPA to promulgate standards for protecting public health, safety, and the environment from radiological and nonradiological hazards associated with uranium-ore processing and the resulting residual radioactive materials (RRM). On January 5, 1983, EPA published standards (40 CFR Part 192) for RRM disposal and cleanup. The standards were revised and a final rule was published January 11, 1995 (60 FR 2854).

The standards address two ground water contamination scenarios: (1) future ground water contamination that might occur from tailings material after disposal cell construction, and (2) the cleanup of residual contamination from the milling process at the processing sites that occurred before disposal of the tailings material. The UMTRA Surface Project (completed in 1996) was designed to control and stabilize tailings and contaminated soil. The UMTRA Ground Water Project addresses ground water contamination at the processing sites (after surface cleanup) and is regulated by Subparts B and C of 40 CFR 192.

2.2.1 Subpart B: Standards for Cleanup of Land and Buildings

Subpart B, "Standards for Cleanup of Land and Buildings Contaminated with Residual Radioactive Materials from Inactive Uranium Processing Sites," requires documentation that ground water remediation meets the requirements of supplemental standards or the following standards:

- Background levels, which are concentrations of constituents in nearby ground water not contaminated by ore-processing activities.
- Maximum concentration limits (MCLs), which are limits set by EPA for certain contaminants in ground water and are specific to the UMTRA Project (Table 2–1).
- Alternate concentration limits (ACLs), which are concentration limits for contaminants that do not pose a substantial hazard (present or potential) to human health or the environment as long as the limit is not exceeded.

Table 2–1. Maximum Concentration Limits of Inorganic Constituents in Ground Water at UMTRA Project Sites

Constituent	Maximum Concentration[a]
Arsenic	0.05
Barium	1.0
Cadmium	0.01
Chromium	0.05
Lead	0.05
Mercury	0.002
Molybdenum	0.1
Nitrate (as N)	10.0[b]
Selenium	0.01
Silver	0.05
Combined radium-226 (Ra-226) and radium-228 (Ra-228)	5 pCi/L
Combined uranium-234 (U-234) and uranium-238 (U-238)	30 pCi/L[c]
Gross alpha-particle activity (excluding radon and uranium)	15 pCi/L

[a]Concentrations reported in milligrams per liter (mg/L) unless otherwise noted.
[b]Equivalent to 44 mg/L nitrate as NO_3.
[c]Equivalent to 0.044 mg/L, assuming secular equilibrium of U-234 and U-238.

pCi/L = picocuries per liter.
Reference: 60 FR 2854.

2.2.2 Natural Flushing Standards

Subpart B allows the use of natural flushing to meet EPA standards. Natural flushing allows natural ground water processes to reduce the contamination in ground water to acceptable standards (background levels, MCLs, or ACLs) under certain conditions. Natural flushing must allow the standards to be met within 100 years. In addition, institutional controls and an adequate monitoring program must be established and maintained to protect human health and the environment during the period of natural flushing. Institutional controls would prohibit inappropriate uses of the contaminated ground water. The ground water also must not be a

current or projected source of drinking water for a public water system during the period of natural flushing, and beneficial uses of ground water must be protected.

2.2.3 Subpart C: Implementation

Subpart C provides guidance for implementing methods and procedures to reasonably ensure that standards of Subpart B are met, including consultation with affected states and tribes. Subpart C requires that the standards of Subpart B are met on a site-specific basis using information gathered during site characterization and monitoring, which, for the Shiprock site, is summarized in this SOWP. The plan to meet the standards of Subpart B must be stated in a site-specific remedial action plan, known as a GCAP. The plan must contain a compliance strategy and a monitoring program, if necessary, and is approved by the NRC following completion of site-specific NEPA documentation.

2.2.4 Supplemental Standards

Under certain conditions, DOE may apply supplemental standards to contaminated ground water in lieu of background levels, MCLs, or ACLs (40 CFR Part 192). Supplemental standards may be applied if any of the following conditions are met:

- Remedial action necessary to implement Subpart A or B would pose a significant risk to workers or the public.

- Remedial action to meet the standards would directly produce environmental harm that is clearly excessive, compared to the health benefits of remediation, to persons living on or near the sites, now or in the future.

- The estimated cost of remedial action is unreasonably high relative to the long-term benefits, and the RRM does not pose a clear present or future hazard.

- There is no known remedial action.

- The restoration of ground water quality at any processing site is technically impractical from an engineering standpoint.

- The ground water is classified as limited-use ground water. Subpart B of 40 CFR 192 defines limited-use ground water as ground water that is not a current or potential source of drinking water because total dissolved solids (TDS) exceed 10,000 milligrams per liter (mg/L); there is widespread ambient contamination that cannot be cleaned up using treatment methods reasonably employed in public water supply systems; or the quantity of water available to a well is less than 150 gallons per day. When limited-use ground water applies, supplemental standards ensure that current and reasonably projected uses of the ground water are preserved.

- Radiation from radionuclides other than radium-226 (Ra-226) and its decay products is present in sufficient quantity and concentration to constitute a significant radiation hazard from RRM.

2.3 Cooperative Agreement

UMTRCA requires that remedial action include consultation with states and Indian tribes that own land containing uranium mill tailings. UMTRCA also directs DOE to enter into cooperative agreements with the states and Indian tribes.

DOE and the Navajo Nation entered into a cooperative agreement on the UMTRA Ground Water Project in February 1999. The cooperative agreement sets forth the scope, schedule, and budgets for activities on Navajo Nation lands and is consistent with the DOE's American Indian Policy (being revised as of June 2000).

2.4 National Environmental Policy Act

UMTRCA is a major federal action that is subject to the requirements of NEPA (42 U.S.C. §4321 *et seq.*). Council on Environmental Quality regulations (to implement NEPA) are codified in 40 CFR Part 1500; these regulations require each federal agency to develop its own implementing procedures (40 CFR §1507.3). DOE NEPA regulations are contained in 10 CFR Part 1021, "National Environmental Policy Act Implementing Procedures." DOE guidance is provided in *Recommendations for the Preparation of Environmental Assessments and Environmental Impact Statements* (DOE 1993).

Pursuant to NEPA, in 1994 DOE drafted a PEIS for the UMTRA Ground Water Project. The PEIS document was made final in October 1996. The purpose of the NEPA document was to analyze the potential impacts of implementing four programmatic alternatives for ground water compliance at the designated processing sites. The preferred alternative for the UMTRA Ground Water Project was published in a Record of Decision in 1997 and contains the framework for site-specific NEPA documentation. All subsequent action on the UMTRA Ground Water Project will comply with the Record of Decision.

In some cases, prior to assessment of remediation alternatives, activities to complete characterization and interim actions may require review, and some may be categorically excluded in accordance with 10 CFR 1021.

2.5 Other Federal Regulations

In addition to UMTRCA EPA ground water standards and NEPA, DOE must also comply with other federal regulations and executive orders that may be relevant to the UMTRA Project sites. Examples include regulations that require protection of wetlands and floodplains, threatened or endangered species, migratory birds, and cultural resources. Other regulations, for which the State or Tribe may be delegated authority, include requirements for water discharge and waste management. Executive orders include those related to pollution prevention, environmental justice, floodplains and wetlands, and government-to-government relations with Indian tribes.

DOE is working with the Army Corps of Engineers to determine the need for 404 permitting at the Shiprock site. To date, the activities conducted by DOE have met the criteria for a nationwide permit.

DOE has established routine communications with the U.S. Fish and Wildlife Service (USFWS), Albuquerque Office, pertaining to the Endangered Species Act and other sensitive species requirements. The USFWS is also an integral team member in the determination of potential ecological risks and has provided guidance to DOE.

2.6 Tribal Regulations and Requirements

Tribal regulations must also be complied with when federal authority has been delegated to the Navajo Nation, or where the Navajo Nation has exercised sovereignty. Examples include the right of the Navajo Nation to require water-use permits and permits to drill wells, Clean Water Act regulations, cultural resources permits, tribal endangered species issues, and land use authorization. In cases where the Navajo Nation does not have authority for implementation of a regulatory program (e.g., underground injection permitting), EPA Region 9 has maintained jurisdiction. DOE and its contractors work closely with the Navajo UMTRA compliance specialist on a broad scope of regulatory issues on a regular basis.

DOE has also established routine communication with the Navajo Nation Environmental Protection Agency (NNEPA), Navajo Water Code Administration (NWCA), Navajo Fish and Wildlife Department (NFWD), Navajo Department of Emergency Management (NDEM), Navajo Cultural Resources Program, and district and chapter grazing boards.

Key organizations within the NNEPA include the Water Quality Program (which regulates surface waters and wetlands) and the Underground Injection Control (UIC) Program. The Water Quality Program works in conjunction with the federal agencies to administer surface water standards and to address wetland issues and 404 permitting requirements. The UIC program works in concert with EPA Region 9 to address underground injection issues related to DOE's proposed activities.

The NWCA administers drilling permits, water use permits, and water use agreements. Navajo UMTRA, NWCA, and DOE are in the process of finalizing a water use agreement to compensate the Navajo Nation for water used at the site.

The NFWD works closely with the DOE and USFWS to identify and mitigate any potential adverse impacts to sensitive plant and wildlife species. In addition, the NFWD is consulted on a regular basis concerning ecological risk issues.

The NDEM is consulted as necessary if chemicals or substances are to be stored at the site during compliance activities. The Cultural Resources Program is consulted on a regular basis to determine the need and locations for investigations where surface disturbance may be required.

The Shiprock Chapter grazing representative and the grazing district has been consulted on a regular basis to determine the need for grazing agreements and restrictions.

2.7 DOE Orders

Several environmental, health and safety, and administrative DOE orders apply to work conducted under the UMTRA Ground Water Project. DOE orders prescribe the manner in which DOE will comply with federal and state laws, regulations, and guidance, and the manner in which DOE will conduct operations that are not prescribed by law. DOE guidance for complying with federal, state, and tribal environmental regulations is given in the DOE Order 5400.1 series, partially superseded by DOE Order 231.1. DOE Order 5400.5 requires protection of the public from radiation hazards. DOE guidance pertaining to NEPA is in DOE Order 451.1, and specific guidance pertaining to environmental assessments (EAs) is provided in *Recommendations for the Preparation of Environmental Assessments and Environmental Impact Statements* (DOE 1993).

3.0 Site Background

The Shiprock UMTRA Project site is on the Navajo Indian Reservation in San Juan County in the northwest corner of New Mexico (Figure 1-1). The UMTRA Project site is accessible by Uranium Boulevard, which extends from U.S. Highway 666 eastward about 0.5 mi to the Navajo Engineering and Construction Authority (NECA) facility. The site of the former uranium mill, which operated from 1954 to 1968, is on the NECA facility. Immediately east of the NECA facility is the 76-acre UMTRA Project disposal cell, a stabilization completed in 1986 of two former tailings piles. An overview of the site's physical setting and climate, a history of the former milling operation and other site activities, sources of ground water contamination, and current and future land and water uses are presented in the following sections.

3.1 Physical Setting and Climate

The Shiprock site is in the northwest part of the San Juan Basin on the Four Corners Platform. Bedrock formations in this part of the basin are flat lying or gently dipping. This arid area in the southeast part of the Colorado Plateau has generally low local relief and is characterized by broad, desolate uplands and wide valleys partly covered by vegetation. Ship Rock, the prominent landmark about 10 mi southwest of the site, is a volcanic neck that rises about 1,700 feet (ft) above the upland area.

The disposal cell and surrounding physical and cultural features of the site area are shown in Plate 1, the site base map. Selected features from Plate 1 are shown on Plate 2, which is an October 1997 black-and-white aerial photograph.

The disposal cell and adjacent former millsite are on an elevated terrace south of the San Juan River at an elevation of about 5,000 ft. About 50 to 60 ft below the terrace is the San Juan River floodplain that extends 1,500 ft in width north of the millsite and south of the river. An escarpment south of the river forms the boundary between the floodplain and the nearly flat terrace. The floodplain area immediately north of the disposal cell (millsite floodplain) ends at the U.S. Highway 666 bridge to the northwest and ends to the southeast at about 1,500 ft downstream from the confluence of Many Devils Wash with the San Juan River. About 1,000 ft upstream from Many Devils Wash confluence, another floodplain appears south of the river and continues for about 1.5 mi to the confluence with the Chaco River. A terrace of varying width is present upstream of the disposal cell from Many Devils Wash eastward to the Chaco River area. Bob Lee Wash and Many Devils Wash are two minor north-northeast trending drainages that cut through the terrace south of the river.

Downstream from the U.S. Highway 666 bridge, the floodplain south of the river resumes, but its southern edge is mainly defined by a distributary channel of the river. The terrace area continues westward from the U.S. Highway 666 bridge and is cut by two minor north-trending drainages, 1st and 2nd Washes, and a northwest-trending drainage, 3rd Wash. About 0.75 mi west of the U.S. Highway 666 bridge, the height of the escarpment at the north edge of the terrace begins decreasing westward and it is not present in the area north of Stokely Elementary School. In this area of the site, the main terrace area slopes gently northward north of U.S. Highway 64 to a low terrace where the Sewage Treatment Plant is located.

The Shiprock area along the San Juan River valley has a desert climate, receiving approximately 7 inches (in.) of annual precipitation (Stone and others 1983). Precipitation is heaviest in summer

and early fall (July through October) during the Southwest monsoon, in which high intensity, short duration storms produce downpours. Late spring months of May and June are the driest time of the year. Annual snowfall is low, averaging less than 10 in.; it usually occurs from November through March.

The dry climate ensures large diurnal temperature variations of about 35 °F. Summer maximum and minimum temperatures during June through August average in the 90s °F and 50s °F, respectively. Winter maximum and minimum temperatures during December through February average in the 40s °F and teens °F, respectively. Nighttime temperatures fall below freezing generally from November through March. All-time extreme temperatures range from a low of -26 °F to a high of 109 °F (Maker and others 1973).

Surface water evaporation is high owing to the high percentage (about 80 percent) of clear days, the low annual precipitation, and the frequency of strong winds, which cause dust storms, particularly in the dry spring months of March through May. The annual average pan evaporation rate is approximately 70 in. (Robson and Banta 1995), for a potential evaporation-to-precipitation ratio of about 10:1. Wind direction is most frequently from the southeast; however, stronger winds associated with frontal systems are typically from the southwest, west, and northwest.

Meteorological data for Shiprock (station 298284) have been collected sporadically since 1931, mainly from a location about 1 mi east of the center of the town of Shiprock. Recently (1996 to 1997), the recording station for Shiprock was moved to Diné College about 2 mi north-northwest of the UMTRA Project site, a location where more continuous and comprehensive data will be available.

3.2 Site History

3.2.1 Pre-Milling Site Conditions

Dry conditions prevailed in the Shiprock area south of the San Juan River in the 1930s and early 1940s before the start of irrigated farming, housing developments, business developments, a helium processing plant, and a uranium mill. Only two houses are shown south of the San Juan River in the area of the site (within a mile upstream and downstream of the U.S. Highway 666 bridge) in the U.S. Geological Survey (USGS) topographic map (Chimney Rock SW) surveyed in 1933 and 1934. [**Note**: All figures in Section 3 are presented at the end of the section and are preceded by an explanation of the aerial photographs in Section 3.4.] Figure 3–1, a 1935 aerial photograph of the site area, shows a dry environment with little vegetation, particularly in the floodplain. Sand dunes are prevalent on the floodplain area south of the San Juan River about 1 mi upstream from the site. The (millsite) floodplain just north of the site is barren except for some vegetation immediately adjacent to the river. Only one small irrigated tract is evident in the photo south of the river; it was watered from a small canal off a distributary channel of the river.

Significant quantities of helium-an important wartime commodity-were found along with nitrogen in oil and gas fields in the area in the early 1940s. A helium processing plant (Navajo Plant) was constructed in 1944 by the U.S. Bureau of Mines on the site of the present Shiprock Shopping Center. A self-contained, rectangular-shaped community of 54 houses and streets was constructed for workers just south of the plant (Foster 1945). Water for the processing plant and housing area was taken from the south bank of the San Juan River at infiltration galleries just

west of the U.S. Highway 666 bridge at the head of the distributary channel. Wastewater from the plant and housing area drained to the northwest to a pond (sewage lagoon) in the 3rd Wash off the terrace west of the U.S. Highway 666 bridge (U.S. Department of Health, Education, and Welfare 1962). The Navajo Plant operated during the later part of World War II in 1944 and 1945, and then was on standby status until 1952 when a high level of production began in response to the Korean War.

In the early 1950s, the Shiprock area experienced dramatic growth resulting from uranium and oil and gas exploration. In January 1952, the U.S. Atomic Energy Commission (AEC) established a uranium-ore buying station at the Shiprock site. American Smelting and Refining Company, an AEC contractor, operated the station until November 1954 when construction of the uranium mill, built by Kerr-McGee Oil Industries, Inc., was completed just east of the buying station (Albrethsen and McGinley 1982).

3.2.2 Milling-Era History

The uranium mill, known as the Navajo Mill, was operated by Kerr-McGee from November 1954 to March 1963 when it was sold to the Vanadium Corporation of America (VCA). VCA operated the mill until August 1967 when the company merged with Foote Mineral Company, which continued operation until milling ended in August 1968. Before and during the milling operations, the site was leased from the Navajo Nation. In 1973, the lease expired and the site ownership reverted back to the Navajo Nation.

Figure 3–2 is an oblique low-altitude aerial photograph showing the early mill operation in late 1954 or early 1955. The layout of mill buildings in 1957 is shown in Figure 3–3. An aerial photograph of the mill and surrounding area in August 1962 is shown in Figure 3–4. An oblique low-altitude aerial photograph of the mill and surrounding area in July 1965 is shown in Figure 3–5.

During its life, the mill processed about 1.5 million tons of ore, which contained an average of 0.26 percent uranium oxide (U_3O_8) and 1.16 percent vanadium oxide (V_2O_5). Uranium recovery averaged about 94 percent and vanadium recovery was only about 58 percent, resulting in production of about 7.9 million pounds of U_3O_8 and 35.4 million pounds of V_2O_5 (Albrethsen and McGinley 1982). The mill was initially designed to treat mainly uranium ores containing carnotite and roscoelite from the Salt Wash Member of the Morrison Formation in the Lukachukai Mountains of northeast Arizona. These ores had low lime and high vanadium contents and were initially treated using an acid cure process. However, as the mill capacity increased from about 300 to 500 tons of ore per day and the source of ore changed (because of a decrease in the vanadium market) to a high lime-low vanadium content, the acid cure was converted to a conventional agitation leach in 1955. For several years after 1955, only uranium was recovered and vanadium-rich solutions were placed in the raffinate lagoons for possible later recovery of vanadium. After VCA took over mill operation in 1963, more than half the ore supplied to the mill was from mines in the Uravan Mineral Belt, 100 to 150 mi to the north.

In 1956, Kerr-McGee added a solvent extraction (SX) circuit for uranium recovery on a trial basis to supplement the agitation leach/ion exchange process circuit. The SX circuit operated successfully and the process was expanded and adapted to include vanadium recovery. By 1957, the mill had converted from the ion exchange process after leaching to a two-stage SX process where uranium was recovered first in a separate SX circuit and vanadium was recovered second in another SX circuit. In this milling process, ore was crushed and ground to less than 35 mesh,

then subjected to a strong acid leach in two stages. A high concentration of acid was required in the second stage to improve vanadium recovery. The strong acid solution produced in the second stage was recirculated to the first stage for partial neutralization by the entering ore slurry. In addition to ore, after VCA assumed operation of the mill in March 1963, millfeed also consisted of dried slimes concentrates and chemical precipitates produced by the VCA concentrating plants near the Monument No. 2 mine in Monument Valley, Arizona. During the second stage of leaching, old tailings containing vanadium that had not been extracted during uranium processing in the early years of milling were added.

After leaching, the sands and slimes entered a countercurrent washing system in which the sands were washed in classifiers and the slimes were washed in thickeners. Uranium and vanadium were then removed from the pregnant liquors by the two SX circuits. Organic solvents used in the SX process were di(2-ethylhexyl) phosphoric acid (EHPA) and tributyl phosphate (TBP) in a base of high flash point kerosene. Also, alcohol was likely added as a modifying agent (DOE 1997). Both nitrate and ammonium complexes were used as ion exchange strippers to concentrate the uranium, and ammonia was used to adjust the pH of the slurry during milling. Additional details of the leaching and SX processes are in Merritt (1971).

Tailings from the washing circuit were pumped to ponds on the two tailings piles. Raffinate from the SX operation was allowed to evaporate in up to 10 unlined raffinate ponds (Figure 3–4, Figure 3–5, and Plate 1), south and southwest of the tailings piles. Water for the milling process was pumped from the San Juan River from an intake about 0.6 mi south-southeast of the mill (Figure 3–4).

During the milling period, the Shiprock area south of the San Juan River and west of the Navajo Mill gained population, and agricultural use increased. These changes required water, and the availability of water changed the character of the terrace area and the area along the San Juan River floodplain. In 1956, the Bureau of Indian Affairs completed the construction of an irrigation project in the terrace area west of the U.S. Bureau of Mines' Navajo (helium) Plant (Young 1961). For this project, a siphon was constructed west of U.S. Highway 666 to bring irrigation water from the Hogback Canal (diverted from the San Juan River about 8 mi east of Shiprock) southward to the terrace area and distributed by means of the Helium Lateral Canal. By 1960, irrigated farming was well established in this area, both north and south of U.S. Highway 64.

In 1961, a well was drilled as an oil and gas test to a depth of 1,850 ft on the terrace about 0.4 mi northwest of the mill. Known in the UMTRA Project as artesian well 648 (Navajo tribal well 12T–520), the well was not plugged and has since flowed at a rate of approximately 64 gallons per minute (gpm) from a screened zone in the Morrison Formation. For several years after the well was drilled, water from the well is believed to have flowed in a ditch to the northeast and down the escarpment to the floodplain. Evidence for this flow is in an aerial photograph from August 1962 (Figure 3–4) showing a line of vegetation northeast from the well. Flow from the artesian well to the east-southeast toward Bob Lee Wash began sometime between August 1962 and June 1974; an aerial photograph taken in June 1974 shows vegetation along both northeast and east-southeast drainage routes away from the well. Vegetation was thicker and more continuous along the northeast drainage route, suggesting that well drainage was mostly to the northeast.

Vegetation increased dramatically on the San Juan River floodplain north of the millsite during the milling period in response to increased availability of water. As early as the summer of 1955, drainage of mill effluent northward onto the floodplain was evident by the presence of a pond at the mouth of a small arroyo incising the terrace and leading north from the mill area. This pond and several smaller ones to the north are present on the floodplain, as shown in the August 1962 aerial photograph in Figure 3–4. By that time, vegetation on the southern part of the floodplain had increased from the pond area westward to the mouth of Bob Lee Wash and to the point farther west where artesian well 648 water drained to the floodplain. This vegetation contrasts with the sparsity of vegetation at the same time in the floodplain south of the San Juan River about 1 mi upstream from the millsite. A similar increase in vegetation is noted in the August 1962 photo in the floodplain area west of the U.S. Highway 666 bridge along the distributary channel (Figure 3–4). This vegetation is in response to irrigation return flow water and wastewater draining from the Navajo (helium) Plant.

In 1963 the Navajo Dam was completed on the San Juan River, forming Navajo Lake about 75 mi upstream and east of Shiprock. Before the dam, the river flow fluctuated greatly through the year from extreme low flows in the fall and winter to sometimes extreme high flows in the spring and early summer in response to snowmelt conditions at the headwaters. In most years, the runoff was high enough to cover the floodplain for periods of several days to weeks. These periodic high flows scoured much of the vegetation off the floodplain and created numerous drainage and distributary channels. After the 1963 control by the dam, fluctuations in river stage have been less extreme. High flows that cover the floodplain are rare and occur only about once every 10 years-the last flood was in June 1995 when water covered the floodplain for only a few days; an earlier flood occurred in May 1987. This control of the river has nearly prevented scouring during flood events and has allowed vegetation to become established along much of the floodplain area upstream and downstream from the site.

During milling, large amounts of mill process water were added to the terrace area in the unlined raffinate ponds and on the tailings piles, as shown in the aerial photograph in July 1965 (Figure 3–5). In August 1960, a large volume of acidic waste effluent was spilled from the west end of the raffinate ponds and flowed down Bob Lee Wash to the floodplain. The effects of this spill and of the long-term conditions resulting from millsite effluent seeping into the San Juan River were described in a report by the U.S. Department of Health, Education, and Welfare (1962). Several seeps were noted and sampled along the escarpment from upstream of the site just below the mouth of Many Devils Wash to downstream on the first wash (1st Wash) west of the U.S. Highway 666 bridge. Also, the presence of a pond was noted that contained piped mill-cooling water, which was at times contaminated with overflow of contaminated process waters. This pond discharged northwestward into Bob Lee Wash.

Some of the mill buildings and most of the equipment were dismantled and placed in the west tailings pile from the time that milling ended in 1968 to the expiration of the Foote Mineral Company lease in 1973. During this period, in about 1972, Shiprock Community Development completed several large housing projects on the terrace about 0.75 mi to 1 mi southwest of the millsite. City water and sewer lines to support this development greatly increased the potential amount of water available to the shallow ground water system south and west of the millsite.

3.2.3 Surface Remedial Action

In 1973 when the millsite and tailings property reverted to control of the Navajo Nation, NECA obtained a lease for the site, occupied the former plant office and shop buildings, and began operating a training school on the site to train Navajo students to operate earth moving equipment. Soon after acquiring the site in 1973, the Navajo Tribal Chairman asked officials from EPA and other federal agencies for assistance in stabilizing the tailings piles (FBDU 1977). In response, EPA conducted radiation surveys around the site in April 1974 to determine the extent of windblown and water-transported tailings. Following this evaluation, EPA recommended decontaminating the site and stabilizing the tailings, and EPA and AEC prepared a work plan to accomplish these objectives (AEC 1974). The decontamination work began in January 1975 and was conducted primarily by NECA trainees under EPA guidance. These activities continued with the trainees until mid-1978, and with other NECA personnel until 1980.

Some moving of the tailings and filling of drainages by the NECA trainees had already occurred by June 1974, as evidenced by a June 1974 aerial photograph that shows reworking of the west (south) tailings pile and partial filling in of the small drainage north of the millsite area. During the early part of the tailings pile stabilization work, a broadcast irrigation system was installed on the south pile to reduce wind erosion; this system was dismantled in 1980. Filling in of the drainages northwest and east of the disposal cell occurred during the significant decommissioning work and recontouring in the mid- to late-1970s. The axes of these filled drainages are shown on Plate 1. A pond, presumably constructed to hold surface water drainage from the NECA buildings area, was present just northwest of the NECA yard from the mid-1970s to about 1984. This pond, at the site of an earlier pond that had held contaminated mill process waters, was in a small drainage that flowed into the east side of Bob Lee Wash.

As shown in the May 1980 aerial photo (Figure 3–6), the pond on the millsite floodplain just north of the escarpment had been filled in, as had the small drainage to the south from the ore storage and millsite area that fed the pond. An aerial photograph from August 1980 shows that upper Bob Lee Wash (above the well 648 outflow) was much more vegetated than at present. This presence of vegetation indicates an abundance of water still available at that time in the terrace system from previous milling and processing activities. Also shown in this photograph, water from Bob Lee Wash that entered the floodplain was channeled by ditch northward to an old distributary channel and then westward to the San Juan River; a wetland area was not present.

By 1980 the extensive changes to the site caused by decommissioning activities and the changes in remedial action criteria brought by UMTRCA legislation in 1978 made it necessary to prepare a revised site engineering assessment (FBDU 1981). This was followed by the surface and ground water characterization studies that were conducted prior to the development of the RAP and Site Conceptual Design for Stabilization of the Site, completed in June 1985 (DOE 1985). These characterization studies included an aerial radiometric survey conducted in December 1980 (EG&G 1981), a geochemical investigation (DOE 1983), a radiologic characterization (Allen and others 1983), a processing site characterization report (DOE 1984b), and an EA of remedial action (DOE 1984a). Mention was made in the geochemical investigation report (DOE 1983) of the use of contaminated soil from the ore storage area to fill (in the late 1970s) a wash on the river bluff (escarpment). The wash referred to is probably the drainage that went north from the old millsite area to the floodplain. No deep radiologic contamination was identified in this filled area during the radiologic characterization; however, it appears that none

of those characterization boreholes (Allen and others 1983) penetrated the filled drainage. Therefore, the presence of contaminated soil in the filled drainage remained questionable.

Site remediation occurred during late 1985 and 1986 and consisted of consolidation of the two tailings piles (stabilization in place) into one disposal cell. An excellent photographic record of remediation activities and disposal cell construction during the 1985-1987 period are archived at the DOE Grand Junction Office (DOE–GJO); additional information on construction activities is in the Remedial Action Completion Report for the Shiprock Site (MK Ferguson 1987).

September 1985 aerial photos show that the wetlands on the floodplain had not yet formed and that the high school to the west in the irrigated area was under construction. March 1986 aerial photos show the radon cover borrow material (loess) being excavated south of the disposal cell and remediation occurring on the floodplain south of the east-northeast trending fence; three ponds were created in the remediated area on the floodplain for waterfowl. A July 1986 aerial photo (Figure 3–7) shows additional remediation on the floodplain and the waterfowl (duck) ponds, which were filled in about a year later because the ponds contained highly contaminated water; ponded water (which could be the ground water surface or water used to control dust) is shown in the northwest end of the radon cover borrow pit. In July 1986, the millsite floodplain was fenced off to prevent grazing. Also in 1986, construction started on the shopping center. A summer 1987 aerial photo (Figure 3–8) shows the completed disposal cell, and a white efflorescent (salt) deposit has appeared on the floodplain in the recently disturbed (scraped) and remediated ground surface from Bob Lee Wash southeast along the base of the escarpment. The NECA pond was constructed in about 1987 in the north portion of the NECA yard after completion of the disposal cell. In 1994 a long-term surveillance plan was prepared for the Shiprock disposal site (DOE 1994). Following approval of this plan, NRC issued a license in September 1996 to the DOE–GJO for the long-term care of the site.

3.2.4 Sources of Ground Water Contamination

During active milling, water usage was approximately 270 gpm. Water with tailings from the washing circuit and from yellow-cake filtration was pumped to the disposal area. Although excess solutions were recycled to the plant during winter months, raffinate was also disposed of by evaporation in separate holding ponds (Merritt 1971). Ground water contamination at the site is believed to have resulted from infiltration of these fluids and leaching of ore and uranium mill tailings constituents by mill water and rainwater. An estimate of the amount of ground water contamination that could have resulted from the ore processing is presented in Section 4.3.2.2, “Terrace Ground Water System.”

3.3 Present and Anticipated Land and Water Use

The current population of rapidly growing Shiprock is about 15,000. This sprawling, unincorporated community is the largest in the Navajo Nation and the largest Native American town in the United States. Several thousand people live south of the San Juan River in the south part of Shiprock. The disposal cell and the floodplain immediately to the north are just east of the south part of Shiprock. Fencing around the disposal cell prevents public access to it, and the gated fence on the road at the mouth of Bob Lee Wash and the natural 50- to 60-ft-high escarpment effectively preclude public access to the uninhabited millsite floodplain area.

A variety of land uses occur in the area underlain by contaminated ground water west and south of the disposal cell. Some of these land uses are shown in Plate 1 and are more clearly shown on the aerial photograph in Plate 2. Immediately west of the disposal cell is the NECA facility (accessed from the west by Uranium Boulevard), which includes offices, equipment repair shops, and equipment and material storage. Also within the fenced NECA facility is an Indian Health Service Office of the U.S. Public Health Service and the Shiprock Field Office of the Navajo Abandoned Mine Lands (AML) Reclamation Department. Several of the NECA facility buildings were former millsite buildings. Southeast of the disposal cell is the fenced NECA gravel pit, which extends nearly to the mouth of Many Devils Wash and includes gravel mining and crushing equipment. South of the disposal cell is the fenced radon cover borrow pit from which loess (silt-sized material) was removed and used for construction of the thick radon barrier in the disposal cell in 1986. West of the fenced NECA facility is the large fairgrounds area north and south of Uranium Boulevard. This is the site of the annual Northern Navajo Shiprock Fair held around October 1 and attended by about 70,000 people.

Commercial and administrative developments line both sides of U.S. Highway 666 south of the San Juan River around the junction of U.S. Highway 64. The largest commercial facility in the area (and in the entire town of Shiprock) is the Tsé Biť aí (Shiprock) shopping center. Included in the shopping center is the Shiprock Regional Business Development Office that administers business lease tracts. East and northeast of the shopping center are several fast food restaurants and small businesses. South of the shopping center are a few small businesses, a senior citizens center, the post office, and a day care center.

Various housing areas are scattered on the terrace and upland areas southwest, west, and northwest of the disposal cell. Most of the housing is in several high density government-funded developments; however, several areas of houses are on individual residential tracts administered by the Navajo Land Department, mainly south and west of the disposal cell, northwest of Bob Lee Wash, and south of the irrigated area (south of Helium Lateral Canal). Two schools, Shiprock High School (and its stadium and athletic fields) and Stokely Elementary School, are in the irrigated area south of U.S. Highway 64.

Irrigated agricultural areas, where mainly alfalfa is grown, are west of U.S. Highway 666, both north and south of U.S. Highway 64. These irrigated areas are east of the high school, the Diné College farm area, and the Blueeyes Ranch north of the irrigation return flow ditch. Water for these irrigated areas is supplied by the buried siphon (constructed in 1956) that takes water from the Hogback Canal north of the San Juan River and discharges it into the Helium Lateral Canal. Water flows through this irrigation system during the growing season, generally from April through October.

Grazing (through a system of permits) of mainly sheep and goats and a few cattle occurs in the open lands southeast of NECA gravel pit and in the upland area south of the disposal cell. A grazing permit is held for the floodplain area north of the disposal cell, but grazing has not been allowed there since 1986. Several acres of sewage pits, where septic tanks are drained, are in the grazing area south of the upland along the west fork of Many Devils Wash (Plate 1); these pits are fenced to prevent livestock entry. Cows and horses also graze in the alfalfa fields on the Blueeyes Ranch. A few livestock (cows and horses) also graze around the scattered residences just west of Bob Lee Wash and southwest of the disposal cell.

No ground water from the floodplain aquifer is being used in the site area. The only known use of ground water from the terrace system in the site area is from well 847 at the north edge of the Shiprock High School property. Water from this well is used for irrigating the school grounds. A small amount (several gallons per minute) of ground water from artesian well 648 is piped to the nearby fairgrounds to water stock for a few days each year. In the fall of 1999, a small cistern-like pond was constructed near the eastern end of the outflow ditch from well 648. All water from the outflow ditch is diverted to the small fenced pond, which presumably was constructed as an access point to fill water tanks for livestock or irrigation.

Water from the San Juan River is taken by NECA just downstream from the mouth of Many Devils Wash; this water is used at the NECA gravel pit for dust control and gravel processing. The Navajo Tribal Utility Authority (NTUA) provides treated water to most of the residents south of the San Juan River through a municipal water supply system that is piped from the Farmington area. The intake structure (Plate 1) on the north bank of the San Juan River just east of the U.S. Highway 666 bridge is operable but takes water out of the river only during emergency situations.

Planned land use changes in the Shiprock site area include:

- Movement of the fairgrounds facility by about 2001 or 2002 to a location about 4 mi to the south.
- Construction of a hotel and several other new businesses in the area of the former fairgrounds.
- Construction of a multipurpose cultural center and a Bureau of Indian Affairs office south of the senior citizens center. The center will include a library, welcome center, youth center, small museum, auditorium, amphitheater, gymnasium, and sports fields.
- Construction of a new Diné College facility in the tract east of the Shiprock High School.
- Construction of the Tabaaji Recreational Vehicle Park on the floodplain just north of the San Juan River and west of U.S. Highway 666.
- Return of the millsite floodplain north of the disposal cell to grazing use after remediation is completed.
- Possible expansion of the NECA gravel pit westward to the area of the radon cover borrow pit after remedial action is completed.
- Probable expansion of housing developments and individual homes.

Future use of the ground water may include additional use of the terrace ground water west of U.S. Highway 666 where construction of the multipurpose cultural center, the new Diné College facility, and other buildings will result in landscaping that requires irrigation. Ground water for use other than irrigation is not planned or anticipated because of the availability of a municipal water system.

3.4 Explanation of Aerial Photographs

Figure 3–1: 1935 Overhead aerial photograph of the Shiprock, New Mexico, area. Dry conditions are evident from scant vegetation south of the San Juan River. Sand dunes are present in the floodplain background area (1), vegetation is sparse in the

main millsite floodplain area (2), one small irrigated plot (3) is near the distributary channel of the river, and terrace gravel outcrops (4) are distinguishable by their darker color. Only two houses are present south of the river.

Figure 3–2: Winter 1954 to 1955 oblique aerial photograph of the Navajo Mill-view southeast. The mill had just begun operation in November 1954. The raffinate ponds (1) had just been constructed and many ore piles (2) were present; tailings piles had not yet been generated. Sulfuric acid was stored in the horizontal tanks (3) in the center, and to the right are the change house (4), office (5), control lab (6), and warehouse and shops (7). The main uranium and vanadium mill buildings are just left of the sulfuric acid tanks, and the sampling plant (8) and crusher (9) are farther left.

Figure 3–4: August 1962 overhead aerial photograph of the Navajo Mill area. After nearly 8 years of milling operations, the east (1) and west (2) tailings piles and the raffinate ponds (3) are well established. Vegetation has appeared in Bob Lee Wash and on the floodplain just north of the escarpment. On the floodplain just north of the escarpment, a pond (4) is present at the mouth of a small arroyo draining the area of the mill and east tailings pile. Water from artesian well 648 (5), drilled a year earlier, has drained northeast (from the line of vegetation) to the escarpment. The Navajo (helium) Plant (6) and the housing area (7) are present and their process water and wastewater were sent to a pond (8) near the escarpment. Water from the Hogback Canal has been siphoned southward and used to create an irrigated farming area (9). Irrigation return flows (10) have supplied water to support vegetation in the floodplain along the distributary channel of the San Juan River.

Figure 3–5: July 1965 oblique aerial photograph of the navajo mill area-view southeast. Abundant milling process water is evident from the full raffinate ponds (1) and ponded water on the east (2) and west (3) tailings piles.

Figure 3–6: May 1980 oblique aerial photograph of millsite decommissioning–view southeast. Decontamination efforts in the late 1970s by the EPA and NECA removed contaminated material from the mill (1), raffinate ponds (2), and ore storage (3) areas and moved it to the lower tailings pile (4). Contaminated material from the ore storage area was used to rebuild the dike around the upper tailings pile (5) and to fill a drainage (6) along the escarpment. A pond (7) had been constructed just northwest of the NECA area to hold surface water drainage. The NECA gravel pit (8) has begun operating. A wetland area is not present on the floodplain at the mouth of Bob Lee Wash; infrequent flow of water from well 648 eastward to Bob Lee Wash is indicated by sparse vegetation along this flow path.

Figure 3–7: July 1986 oblique aerial photograph of millsite remediation-view southeast. Construction of the disposal cell is under way with much of the thick radon barrier material emplaced and some of the cobble blanket cover in place. Loess (silt) material has been excavated from the radon cover borrow pit to construct the radon barrier. The NECA gravel pit in the upper left is in operation. Surface remediation on the floodplain has occurred mainly south of the fence. The duck ponds (1) were created as part of the remediation. The small arroyo (2) that drained the mill area has been filled in. Vegetation is thick along the river bank and has taken over much of

the floodplain (outside the remediated area). Water (3) is present in the northwest (low end) of the radon cover borrow pit.

Figure 3–8: Summer 1987 oblique aerial photograph of the completed disposal cell-view northwest. Remediation has been completed. Housing area (1) for the former Navajo (helium) Plant is still present, but the plant has been removed and a shopping center (2) has just been completed. To the upper left beyond the irrigated fields, the Shiprock High School (3) is under construction. Much of the millsite floodplain is covered by vegetation north of the fence. Efflorescence, shown by white crust, is evident on the millsite floodplain from the mouth of Bob Lee Wash southeastward along the base of the escarpment.

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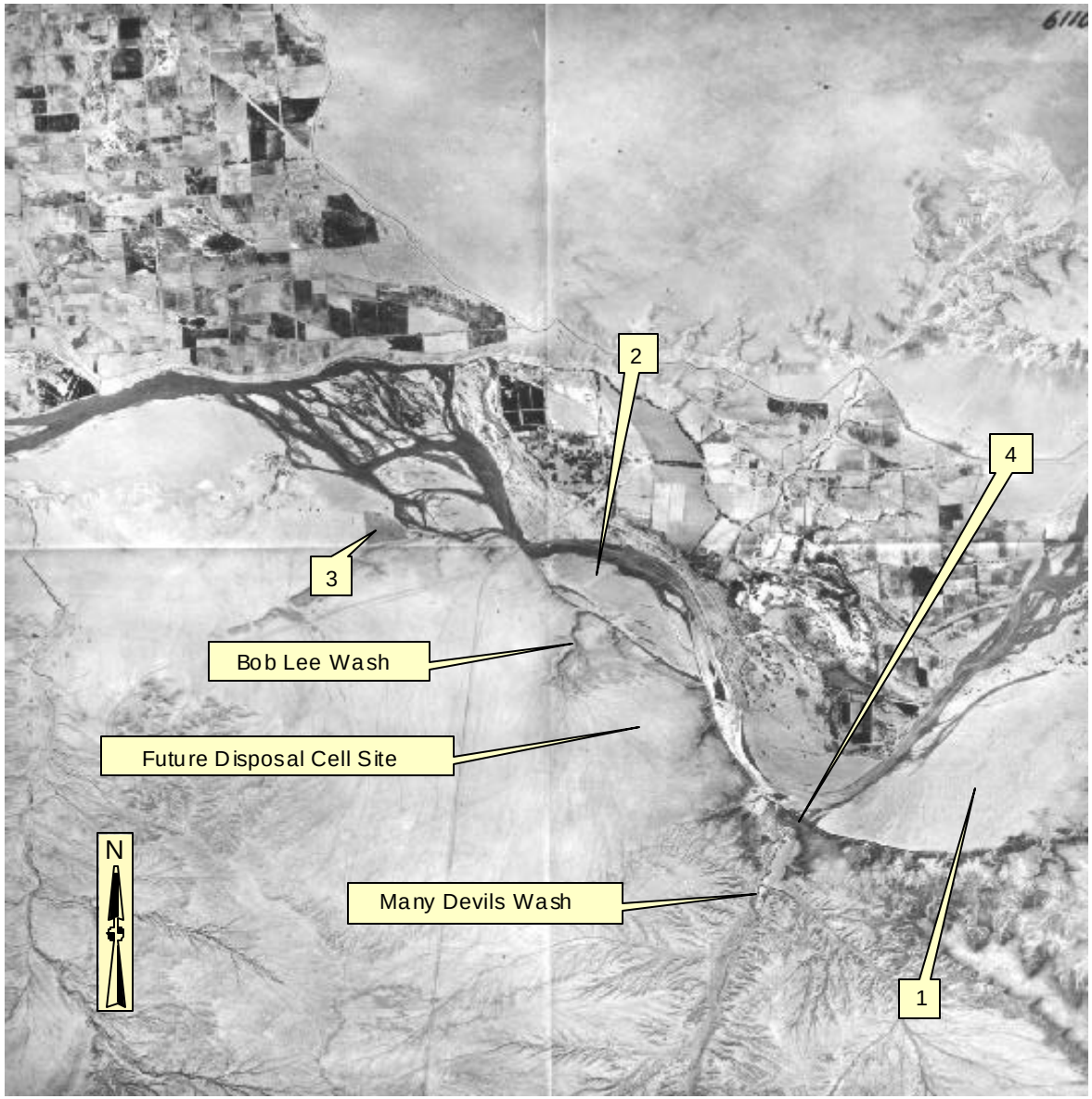

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Figure 3–1. 1935 Aerial View of Shiprock, New Mexico, Area

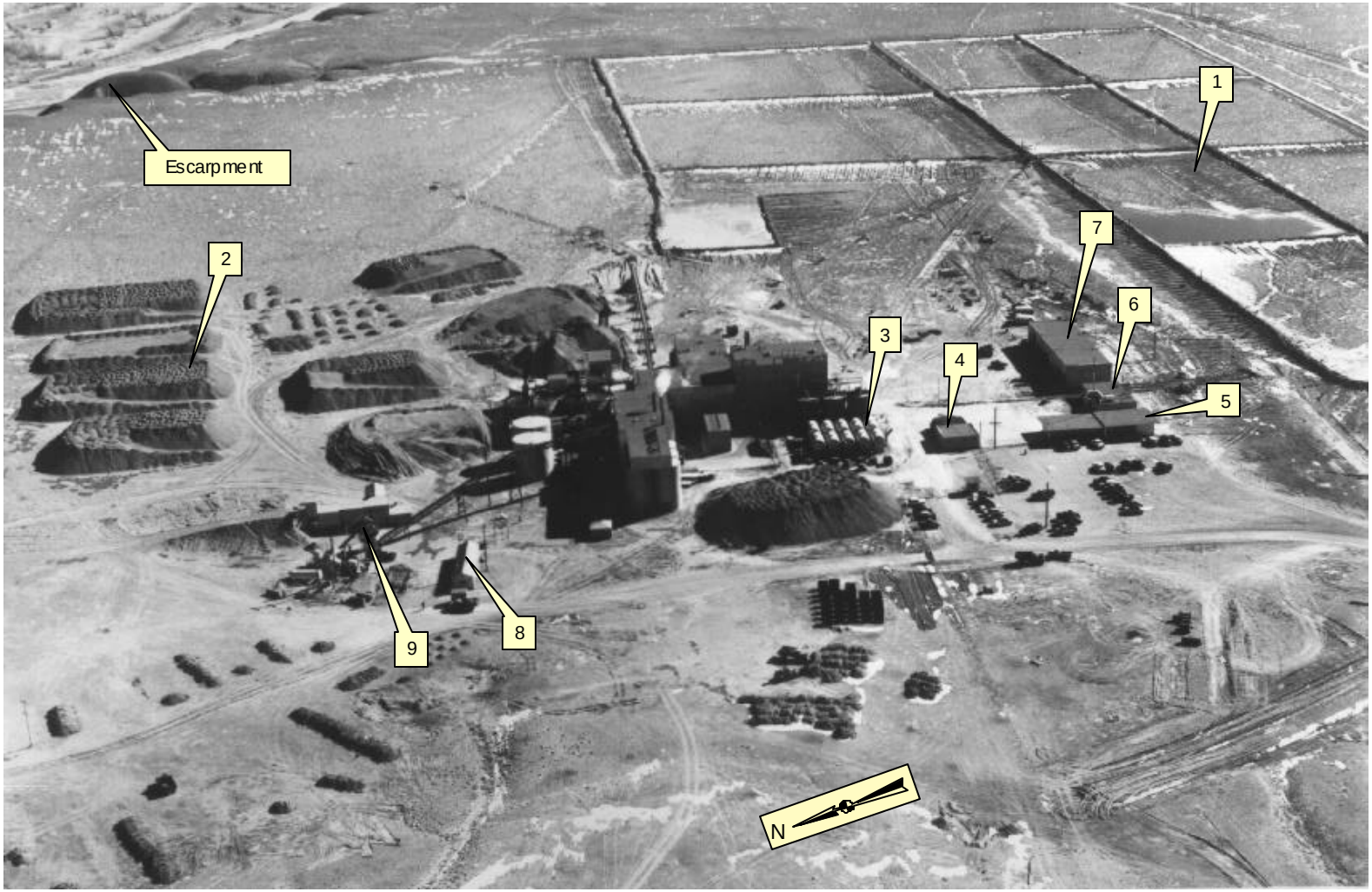

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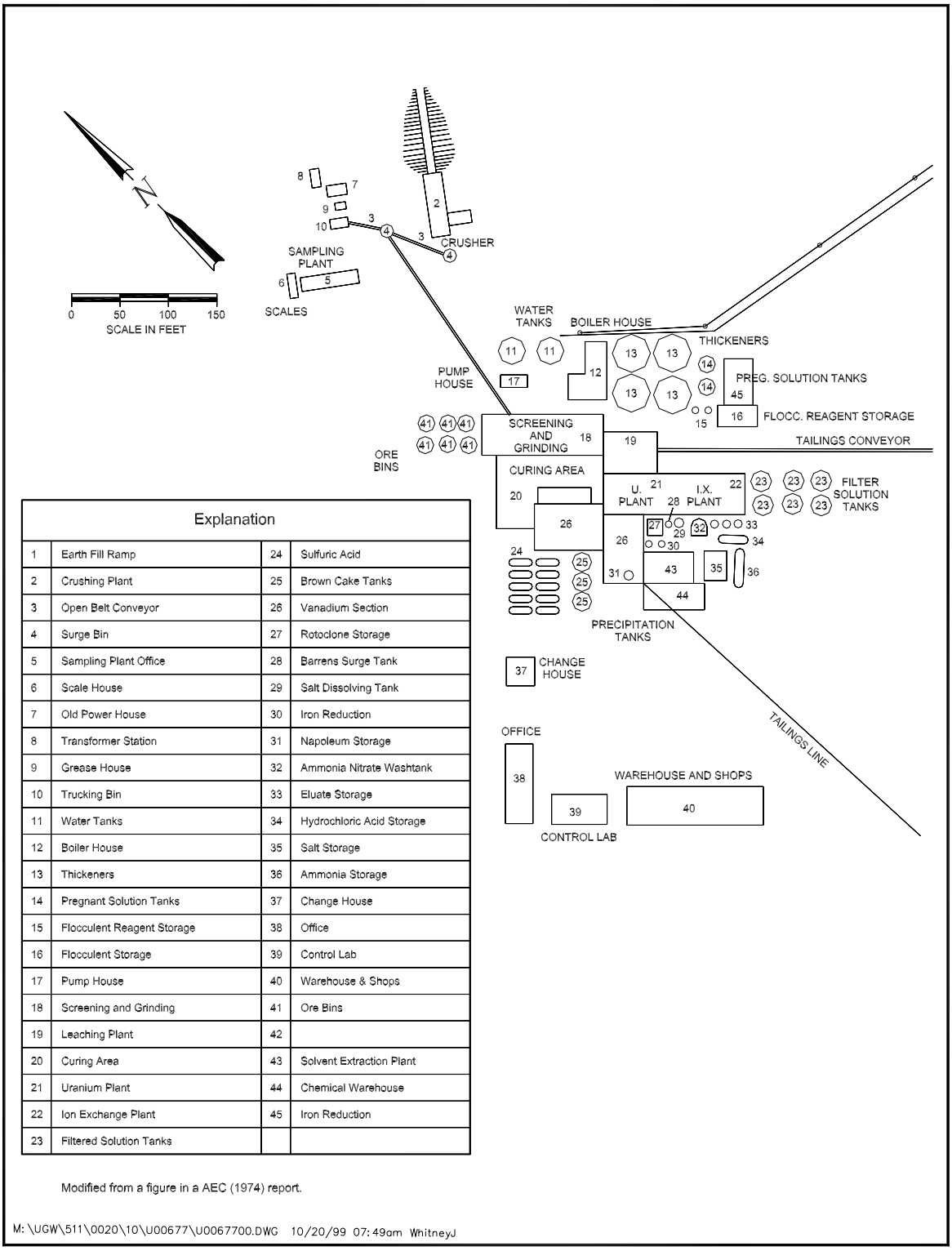
Figure 3–2. Late 1954 or Early 1955 View to the Southeast of the Early Navajo Mill Operation



Explanation

1	Earth Fill Ramp	24	Sulfuric Acid
2	Crushing Plant	25	Brown Cake Tanks
3	Open Belt Conveyor	26	Vanadium Section
4	Surge Bin	27	Rotoclone Storage
5	Sampling Plant Office	28	Barrens Surge Tank
6	Scale House	29	Salt Dissolving Tank
7	Old Power House	30	Iron Reduction
8	Transformer Station	31	Napoleum Storage
9	Grease House	32	Ammonia Nitrate Washtank
10	Trucking Bin	33	Eluate Storage
11	Water Tanks	34	Hydrochloric Acid Storage
12	Boiler House	35	Salt Storage
13	Thickeners	36	Ammonia Storage
14	Pregnant Solution Tanks	37	Change House
15	Flocculent Reagent Storage	38	Office
16	Flocculent Storage	39	Control Lab
17	Pump House	40	Warehouse & Shops
18	Screening and Grinding	41	Ore Bins
19	Leaching Plant	42	
20	Curing Area	43	Solvent Extraction Plant
21	Uranium Plant	44	Chemical Warehouse
22	Ion Exchange Plant	45	Iron Reduction
23	Filtered Solution Tanks		

Modified from a figure in a AEC (1974) report.

Figure 3–3. Schematic of Navajo Mill Buildings in 1957

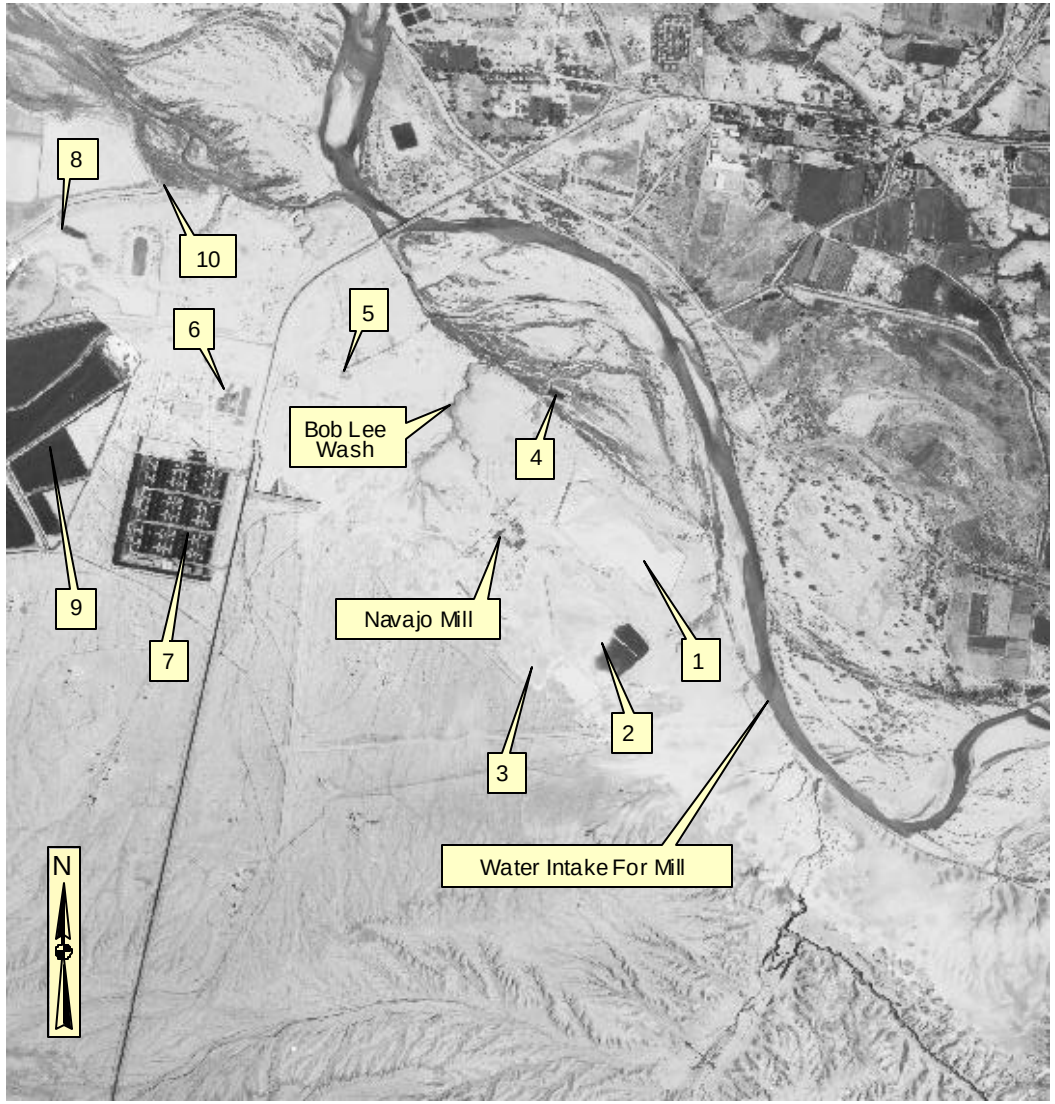


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Figure 3-4. August 1962 View of Navajo Mill

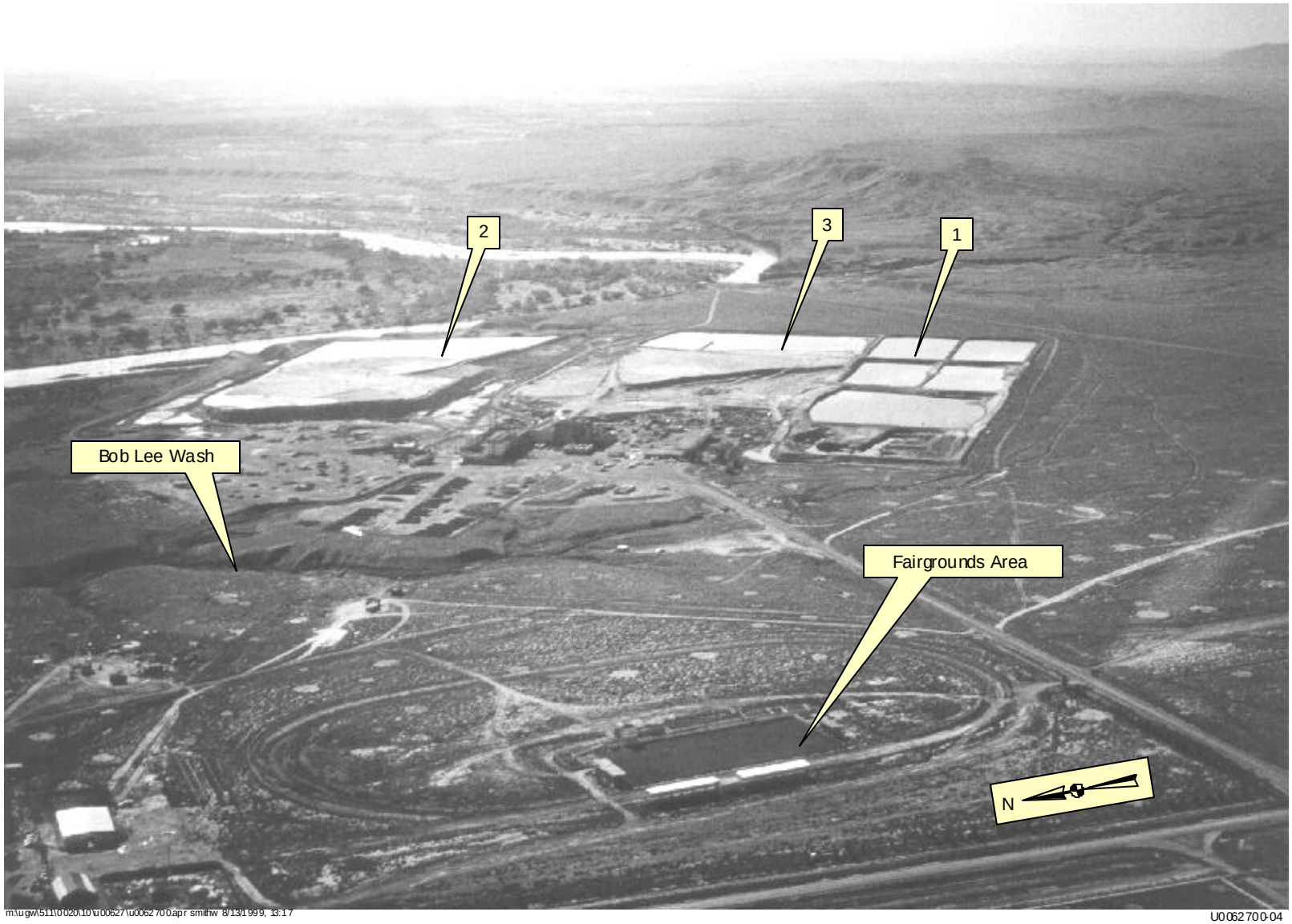

Figure 3-5. July 1965 View Southeast of Navajo Mill Area

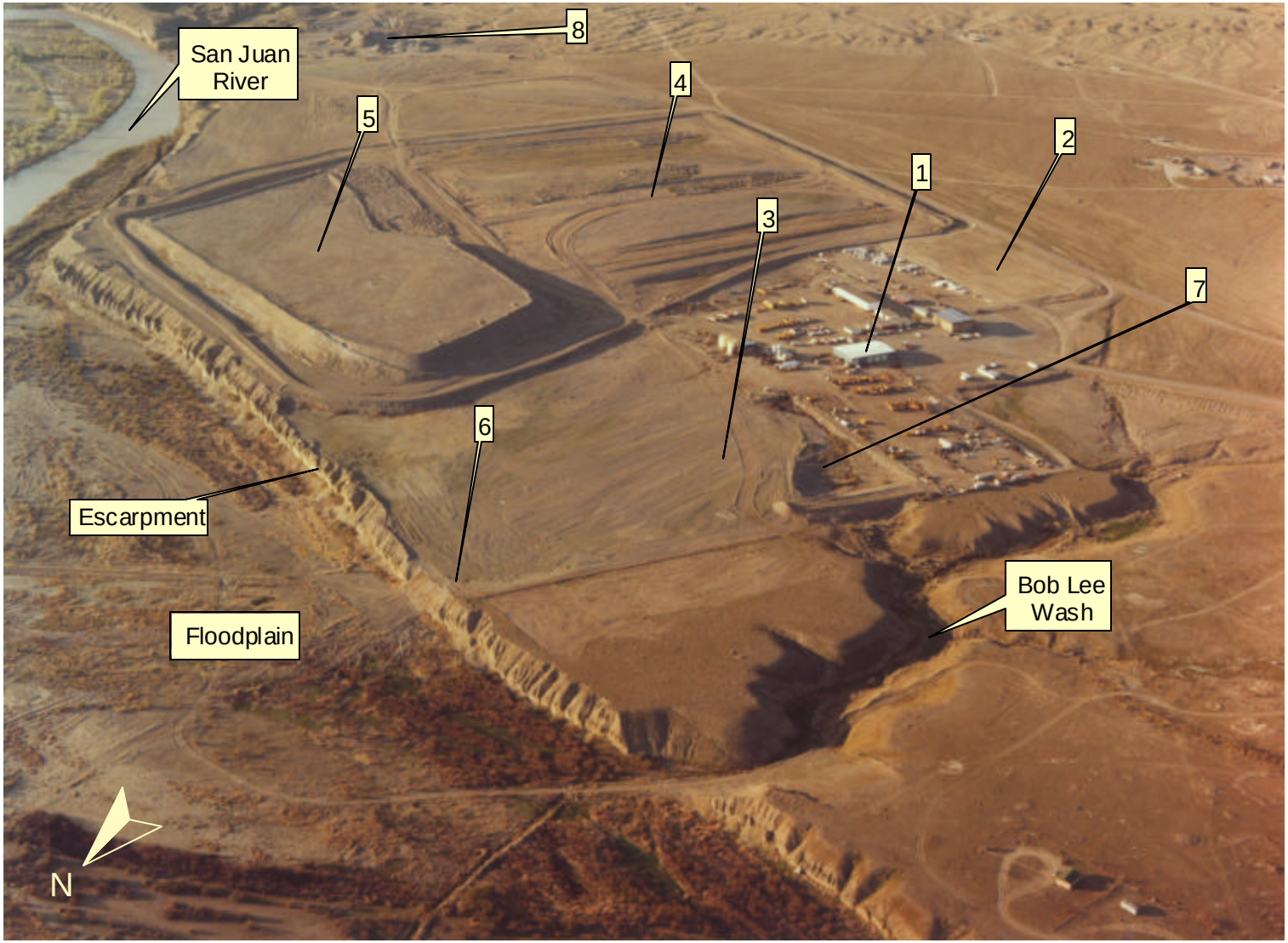

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Figure 3–6. May 1980 View Southeast of Millsite Decommissioning

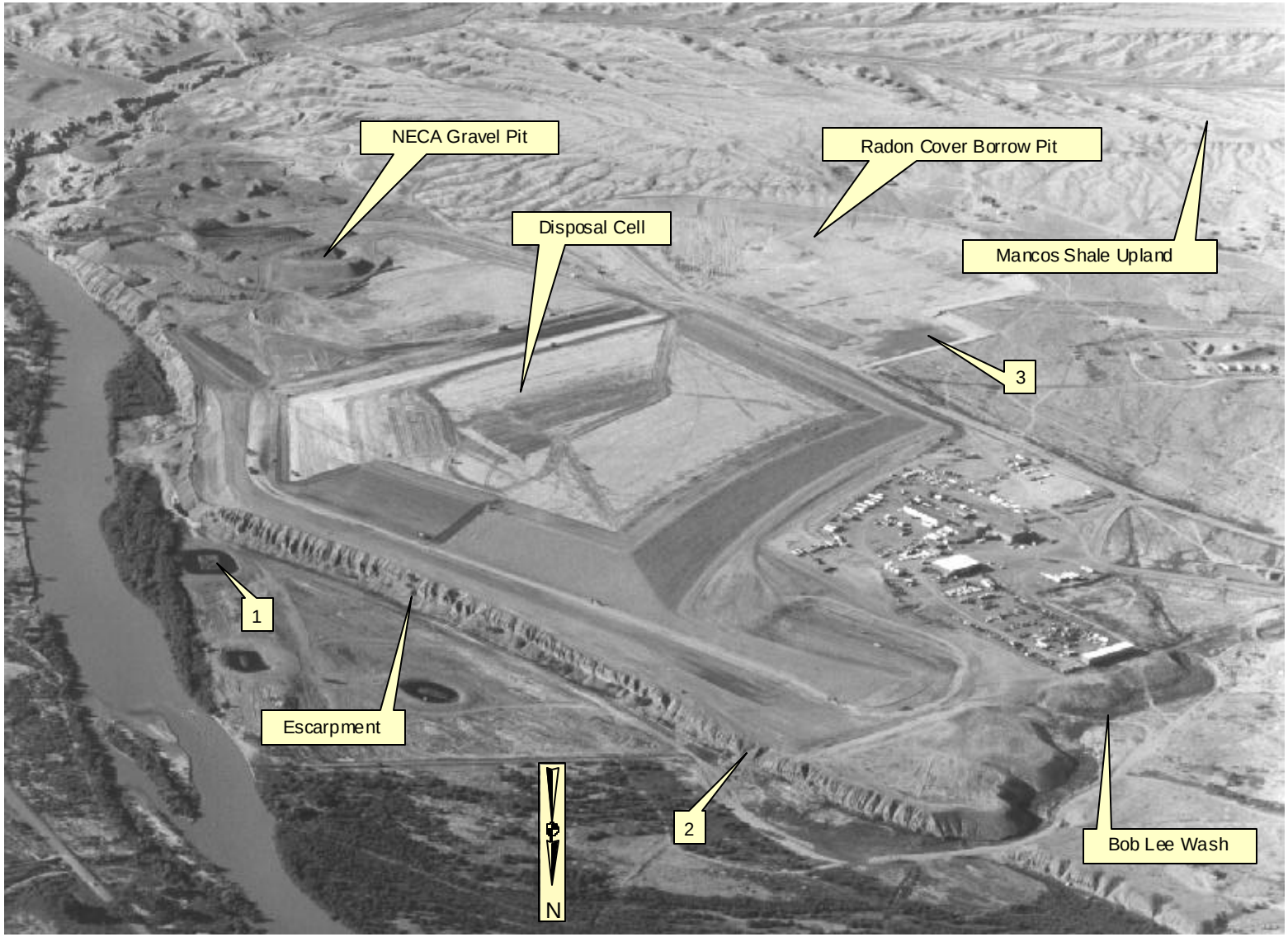

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Figure 3-7. July 1986 View Southeast of Millsite Remediation

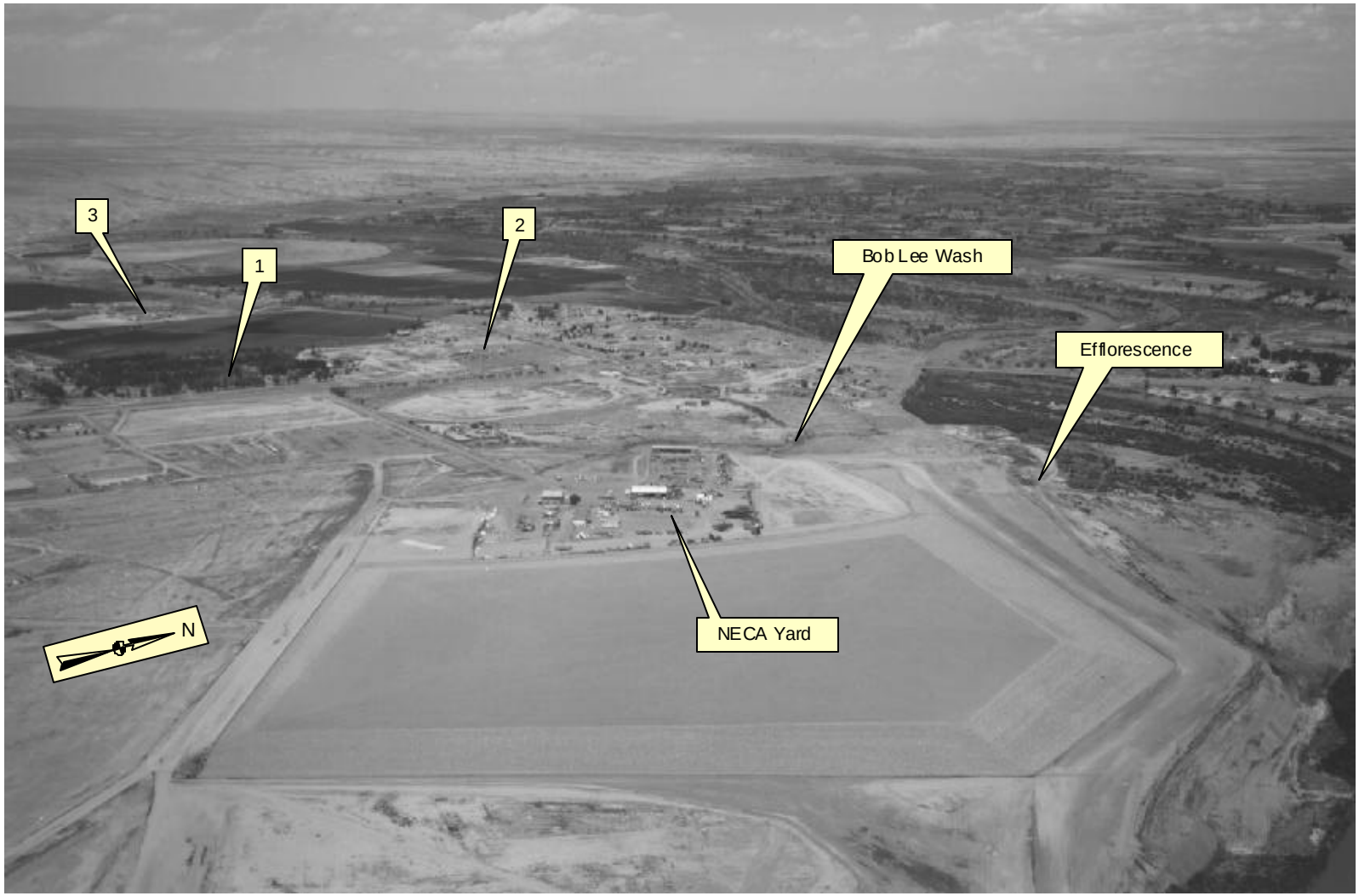

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Figure 3-8. Summer 1987 Oblique Aerial Photograph of Completed Disposal Cell-View Northwest

4.0 Site Characterization Results

The SOWP, Rev. 0 (DOE 1995), provided a summary of site conditions based on characterization data available at that time, presented a site conceptual model, identified likely compliance strategies, and proposed additional data collection activities to address uncertainties. Several of the proposed data collection activities were conducted at the site in early 1996 under the direction of the DOE Albuquerque Operations Office. Stakeholder review of the SOWP identified significant additional site characterization data needs. After programmatic responsibilities for the UMTRA Ground Water Project were transferred to DOE-GJO in late 1996, existing site characterization data were evaluated along with additional stakeholder concerns. To address the data gaps, additional characterization activities were identified and presented in the *Work Plan for Characterization Activities at the Shiprock UMTRA Project Site* (DOE 1998d). The principal goals of the additional data collection were (1) to investigate the extent of ground water contamination in the terrace system, (2) to evaluate the hydraulic interconnection between the terrace and alluvial ground water systems, (3) to evaluate the hydraulic interconnection between the alluvial ground water and the San Juan River, and (4) to select a corrective action for the site. Associated data deficiencies that needed to be addressed by additional characterization include (1) hydrogeologic properties of the floodplain and terrace ground water systems, (2) further definition of the nature and extent of contamination in the floodplain, (3) determination of background water quality in the floodplain and assessment (and quality) of ground water conditions at a terrace background site, (4) contribution of ground water from the upland area south of the site to the terrace system, and (5) evaluation of potential ecological risks.

Field investigations were conducted according to the Work Plan (DOE 1998d) from September 1998 through May 1999. The drilling and well installation part of the investigation extended from September to December 1998. Miscellaneous surface sampling and surveying investigations occurred generally from January to June 1999. These surface investigations included ecological sampling and mapping; sediment, soil, and salt crust sampling; surface water sampling; geologic mapping; and land surveys of new and old wells and other features. The sequence of 1998 drilling field activities was approximately as follows: (1) coring and installation of monitor well nests, (2) installation of boreholes in upland Mancos Shale, (3) installation of monitor wells in the terrace and floodplain background areas, (4) installation of monitor wells and boreholes to determine the extent of the contaminant plume in the terrace system, and (5) installation of monitor wells in the floodplain aquifer. Information from each of these drilling activities was integrated with existing data to continually revise the site conceptual model and to revise and refine the data collection needs.

Additional data needs were identified in Table 4–30 of the SOWP, Rev. 1 (DOE 1999g). Follow-up characterization activities to fulfill most of these data objectives were conducted from October 1999 to April 2000. The follow-up action (or action status) for each data objective identified in the table of additional data needs is shown in Table 4–1. The sequence of main field activities was as follows: (1) excavation and sampling of test pits and installation of well points on the floodplain in late October 1999, (2) excavation and sampling of test pits on the floodplain and installation of monitor wells and boreholes by Geoprobe in the terrace in mid-December 1999, (3) surface water ecologic sampling in the floodplain in March 2000, and (4) excavation and sampling of test pits on the floodplain and installation of monitor wells on the floodplain and terrace from late March to mid-April 2000.

Table 4–1. Follow-up Action to Meet Additional Data Needs

Data Objective	Proposed Action	Action Status
Identify ground water pathways where contaminated terrace water feeds the floodplain alluvial aquifer	• Install additional well nests: one on floodplain north of disposal cell near wells 613 and 614 and one on terrace immediately to the south • Bore into the filled-in drainages on terrace just west of well 735 and east of well 827 and complete as wells with screens near the contact with Mancos Shale	• Pair of well nests installed in March–April 2000, one nest (wells 1000 and 1001) on floodplain adjacent to wells 613 and 614 and one nest (wells 1002 through 1004) on terrace immediately to the south. • Three wells installed in March–April 2000 in the two largest filled drainages?well 1011 in drainage just east of well 827 and wells 1006 and 1007 in drainage northwest of well 735.
Evaluate if a source is present in the floodplain aquifer	• Backhoe will be used to collect three soil samples between the land surface and the water table (approx. 5 ft) at as many as 30 locations on a grid. The GJO mobile laboratory will be used to acid leach the samples and to perform preliminary uranium analyses. As many as 10 samples with the highest uranium concentrations will be sent to the GJO Analytical Chemistry Laboratory for further analysis and evaluation.	• Samples of alluvial material were collected by backhoe in December 1999 from 23 locations on a grid on the floodplain and one location in the floodplain background area. Two samples were collected at each location? one at the surface (0–1 ft) and one at the ground water surface. A ground water sample from each location was analyzed for uranium, sulfate, nitrate, and ammonia at the on-site mobile laboratory. A bulk sample was collected at the floodplain background location and at three locations on the main floodplain where uranium concentrations in ground water were relatively low, medium, and high. Sieve analysis (grain size) and column leach studies were conducted on the bulk samples at the GJO Environmental Sciences Laboratory.
Evaluate how the source can be isolated	• Engineering evaluation of the technologies that could be used to isolate, contain, or control the source of contamination in the floodplain alluvial aquifer	• Any continued source of contamination will be evaluated during the remedial action phase of the Shiprock cleanup. Contaminated ground water will be extracted and treated at Shiprock and the results of this treatment will be monitored. If a continued source is verified, DOE will address the issue at that time.
Evaluate the extent of the floodplain contaminant plume that extends northward to the San Juan River	• Drill and complete one new monitor well between wells 619 and 854	• Well points 766, 768, and 775 were installed in this part of the floodplain contaminant plume in October 1999. Well 1008 was installed near well 854 in April 2000.
Confirm the flow rates for natural flushing in the floodplain	• Conduct tracer tests	• No tracer tests were conducted. Ground water flow paths and rates in the floodplain are well understood and tracer tests were deemed unnecessary.

Table 4–1 (continued). Follow-up Action to Meet Additional Data Needs

Data Objective	Proposed Action	Action Status
Evaluate the infiltration potential through radon cover borrow pit	• Use Geoprobe to measure thickness of residual loess in bottom of borrow pit • Evaluate effect of diverting runoff from borrow pit	• A backhoe pit was dug in the northwest (lowest) part of the radon cover borrow pit in October 1999 and loess thickness was measured. Loess thickness under rest of pit can be determined from nearby borehole lithologic logs. • Diverting runoff away from the borrow pit area was deemed unnecessary because the extra recharge is insignificant in comparison to the recharge area of the entire terrace.
Identify the ground water flow path from the disposal cell to Many Devils Wash	• Install as many as five additional wells to be completed near the top of the siltstone bed in Mancos Shale; predict target completion depth at each location on the basis of surveyed elevation and estimated structure contour of the top of the siltstone bed	• Three monitor wells (1057 through 1059) were installed in March–April 2000 along the ground water flow path southeast from the radon cover borrow pit to Many Devils Wash.
Identify the eastern limit of the terrace contamination	• Use Geoprobe to bore in as many as 20 locations east and southeast of Many Devils Wash • Sample formation fluids, if present, for mill-related constituents NO_3, SO_4, and U	• Fourteen boreholes were drilled by Geoprobe in December 1999 in the area of Many Devils Wash where contaminated ground water reaches the surface. Twelve of the boreholes were dry and two were completed as monitor wells (1048 and 1049). • Ground water samples from monitor wells 1048 and 1049 were analyzed for uranium, sulfate, nitrate, and ammonia at the on-site mobile laboratory. These wells were sampled later during the February 2000 water sampling event.
Identify the western limit of the terrace contamination	• Install as many as five well points using a backhoe in the island area; sample ground water for mill-related constituents NO_3, SO_4, and U • Ecological sampling of leaf tissues and soils in the island area • Collect additional surface water samples in area of surface water drainage from gravels north of high school • Add one or two additional monitor wells in area north of high school	• Three well points (782 through 784) were installed by backhoe in the south part of the island area in October 1999. Initial ground water samples from these wells were analyzed for uranium, sulfate, and nitrate at the GJO Environmental Sciences Laboratory. These well points were sampled later during the February 2000 water sampling event. • Ecologic sampling on the island area was deemed unnecessary because of the low contaminant concentrations in analyses of ground water samples from well points 782 through 784. • Surface water that emerges from the terrace system gravel at location 942 was sampled during the June 1999 and February 2000 sampling events. Surface water at locations 1063 and 1064 was sampled once in December 1999. • With the establishment of surface sample point 942 and the other existing wells to the north, additional wells were deemed to be not necessary.

Table 4–1 (continued). Follow-up Action to Meet Additional Data Needs

Data Objective	Proposed Action	Action Status
Improve water balance for numerical modeling of terrace	• Measure the discharge off the escarpment and from irrigated areas • Results of flux measurements would be used to perform source-term modeling	• Discharge rates were not measured, but parameters from the source-term modeling were included. • Source-term modeling and a four-layer model that hydrologically connects the terrace to the floodplain are in progress.
Evaluate nature and extent of terrace background	• Perform regional reconnaissance of other equivalent terraces both upstream and downstream of site and on both sides of San Juan River • Check Rattlesnake Wash for surface water and salt deposits • Check Many Devils Wash upstream of west fork confluence for ground water and salt deposits	• Age-equivalent terraces north of the San Juan River upstream and downstream from the site contain ground water originating from irrigation. • Surface water or shallow ground water does not appear to be present in Rattlesnake Wash; salt deposits are absent also. • Four boreholes were drilled by Geoprobe in December 1999 in Many Devils Wash south of the confluence of the West Fork. All boreholes were drilled through loess and into Mancos Shale; no ground water was found. No salt deposits were noticed along the floor of Many Devils Wash south of the West Fork.
Identify top-of-bedrock elevation at selected areas on terrace	• Redrill and complete one new monitor well near well 834.	• Monitor well 1060 was drilled into Mancos Shale bedrock and installed in April 2000.
Map plant communities and habitat types west of the U.S. Highway 666 bridge	• Collect plant relevé data in riparian and wetland areas west of the bridge that are influenced by site-related ground water • Delineate and map plant communities and habitat types	• Relevé data were collected in three riparian areas in the terrace ground water system west of U.S. Highway 666. • Plant communities were noted in the vicinity of the three areas in which relevé data were collected.
Expand the ecological risk assessment to include areas west of the U.S. Highway 666 bridge	• Refine the conceptual risk model and food web • Collect additional surface water, sediment, and vegetation samples from seeps and the floodplain west of the bridge • Evaluate chemical analyses against appropriate ecological benchmarks	• The ecological risk section was revised and contains a site-specific food web and risk model. • Vegetation was sampled in September 1999 in three riparian areas in the terrace ground water system west of U.S. Highway 666. Surface water was sampled in March 2000 (in low-flow conditions) at two new locations west of the U.S. Highway 666 bridge. • Analyses of the vegetation and surface water will be compared to ecological benchmarks in the ecologic risk assessment.
Classify and map landscape units with respect to evapotranspiration rates	• Classify and delineate vegetation with respect to differences in evapotranspiration • Assign evapotranspiration ranges to mapping units based on literature values	• Vegetation types were mapped, but evapotranspiration rates were not ascribed to vegetation. This may be conducted as part of the EA to refine the risk model.

Table 4–1 (continued). Follow-up Action to Meet Additional Data Needs

Data Objective	Proposed Action	Action Status
Evaluate potential changes in ecological risk associated with ground water remediation alternatives	• Identify possible changes in the conceptual risk model associated with the remediation alternatives. • Collect any needed additional field data • Revise risk evaluation as appropriate; for example, assess potential effects of natural flushing on endangered fish habitat in the San Juan River	• Possible changes in the risk model associated with remedial action alternatives are addressed in this SOWP.
Evaluate potential changes in ecological risk associated with future land-use alternatives	• Determine potential future land-use alternatives based on consultation with the Navajo Nation and appropriate regulatory agencies • Identify possible changes in the conceptual risk model • Collect any needed additional field data and revise risk evaluations as appropriate	• The site base map was updated to show current land uses, and these were noted in this SOWP. • Most of the land-use changes occur on the terrace and have little effect on the risk model. The floodplain will continue to be used for grazing after remedial action removes most of the contaminated ground water. • Livestock forage on the floodplain was sampled and results will be included in the EA.

Results of additional characterization (and the methods used) conducted since the 1995 SOWP was completed are presented in the following subsections. The subsections include discussion and interpretation of the characterization results. These interpreted characterization results from the major disciplines are integrated and presented in Section 5, "Site Conceptual Model." Also included in the following subsections are surveying results (in Section 4.1, "Investigation Methods").

4.1 Investigation Methods

Field investigations were performed from 1998 to May 2000. Investigation methods included subsurface drilling of test borings and monitor well installation; collection of soil, rock core, soil crust, sediment, ecologic, ground water, and surface water samples; water level measurements; and aquifer testing. Methods for these investigations are described in this Section (4.1); methods used for some of the geochemical sampling and for the numerical ground water modeling are included in Sections 4.4 and 4.5, respectively.

4.1.1 Drilling

The three drilling rigs used during the 1998 drilling project were a Schramm T–660W air rotary with casing driver, a CME–75 wireline, and a CME–55 all terrain drill. The Schramm drill was used to penetrate gravel and cobbles both on the terrace and in the floodplain areas, to drill the deep holes for well nests, and to drill deep holes in the upland area and the terrace background area. A casing hammer was used to drive casing through the gravel, and a center bit was advanced through the casing to remove cuttings from the hole. The CME–75 was used primarily for coring the Mancos Shale, and the CME–55 was used for drilling in loose-sand areas and for well development. Table 4–2 presents a summary of the tasks that were completed with each drilling rig. The 1998 drilling produced 49 new monitor wells, three new production wells, and 10 test borings.

Table 4–2. Summary of Tasks Completed in 1998 With Each Drilling Rig

Task	Schramm T–660W	CME–75 Wireline	CME–55 All Terrain Drill
Test Borings	✓		
2-in. Monitor Wells	✓	✓	✓
5-in. Production Wells	✓		
Coring		✓	
Reaming	✓		
Packer Tests		✓	
Well Development			✓

A Geoprobe, Model 5400, mounted on the back of a four-wheel drive truck, was used to drill 18 boreholes (1039 through 1056, Plate 1) in the Many Devils Wash area. The Geoprobe was used to penetrate thick loess overlying Mancos Shale bedrock. Maximum depth achievable (depth to refusal) by the Geoprobe was usually within the first 2 ft of weathered, but firm, Mancos Shale. The mid-December 1999 Geoprobe drilling produced two new monitor wells and 16 test borings.

A rotasonic drill rig (Hawker Siddeley Super Drill 150) was used in March and April 2000 to penetrate gravel and cobbles both on the terrace and in the floodplain areas and to drill deep

holes into underlying Mancos Shale bedrock for well nests. The rotasonic drilling was used for 17 new monitor wells and one test boring. Four of the monitor wells on the floodplain were 4-in.-diameter wells, installed mainly for the purposes of the Natural and Accelerated Bioremediation Research (NABIR) Program, and the remaining 13 wells were 2-in. diameter.

4.1.2 Subsurface Sampling

Soil samples were collected during the 1998 drilling for lithologic descriptions and for geochemical tests and analyses; core samples were also collected for lithologic information and for selection of packer-test intervals from fracture data. During the air-rotary drilling, bulk samples were lifted to the ground surface with compressed air at 10-ft intervals and placed in plastic bags for archival, testing, and analyses. The CME–75 core samples were cut in 10-ft runs and retrieved using an NX wireline coring system. The core samples were placed in core boxes, labeled, and archived at the DOE-GJO core storage area at the Cheney (Colorado) Repository site. Coring was performed at holes 820, 823, 860, 862, and terrace-background holes 800 and 802 (Plates 1 and 3). Coring was attempted for approximately 360 ft of drilling; overall, core recovery was approximately 90 percent. Split samples of the core and soil samples were also retrieved for distribution coefficient (K_d) analyses. The coring was accomplished using the guidelines published in Designation 2113–83 (ASTM 1993a).

Soil samples of alluvial material were collected by backhoe during excavation of 24 test pits (locations 1015 through 1038, Plate 1) on the floodplain in mid-December 1999. One of the test pits was in the background area. The test pit samples were collected to evaluate whether a continued source of contamination is present in the floodplain soils just below the disposal cell. At each backhoe test pit, two 1-gallon samples were collected in plastic bags—one at a depth of 0 to 1 ft and one at the ground water surface. A ground water sample collected at each test pit was analyzed for uranium, sulfate, nitrate, and ammonia at an on-site mobile laboratory. A bulk sample (15 gallons) was collected at the floodplain background location, and bulk samples were collected from three representative test pits on the main floodplain where analyzed uranium concentrations were relatively low, medium, and high. The bulk samples were returned to the GJO Environmental Sciences Laboratory (ESL) for sieve (grain size) analysis and column leach studies; results are in Section 4.4, “Geochemistry.”

Continuous samples of drilled material were provided by the Geoprobe during drilling of the 18 holes. This material was used in preparing the lithologic description of each hole. During the rotasonic drilling of 18 holes, a continuous sample in the form of a core about 3.5 in. in diameter was provided. Samples of this material were taken at 5-ft intervals, or where lithologic changes occurred, from most boreholes, placed in labeled plastic bags, and archived at the DOE–GJO core storage area. This core-like material was also used in preparing the lithologic description of each hole.

4.1.3 Lithologic Logging

Samples of rock and soil material were described as they were collected. Descriptions of the soil and rock material were prepared on the basis of guidelines established in ASTM D 2488–93 and ASTM D 2487. Soil (Quaternary material) color was described on the basis of comparison to the Munsell Soil Color Charts (GretagMacbeth 1994), and color of bedrock and cored material was described using the Rock-Color Chart (GSA 1975). The lithologic logs are in Appendix A of this report.

4.1.4 Well Installation and Development

Well installation in 1998 consisted of 49 new 2-in. monitor wells and three new 5-in. production wells. Wells in both the terrace alluvium and the floodplain alluvium were normally completed by drilling to the top of bedrock and advancing the borehole slightly into the bedrock. However, several new wells in 1998 were also drilled without reaching bedrock. In those wells, the screen was installed at the desired depth and the annular space was backfilled while the drill string was extracted from the hole. The two wells installed by Geoprobe in December 1999 were 1-in. diameter. Wells installed by rotasonic drilling in 2000 consisted of 13 2-in. wells and four 4-in. wells. Locations of all wells are shown in Plate 1, and well-completion information is summarized in Table 4–3.

For all wells, flush-joint polyvinyl chloride (PVC) was used for well casings, and well screen with 0.010-in. slots was installed. The only exception was the 1998 well 819, which has a stainless steel screen to monitor for organic constituents in the ground water.

For nested wells in the Mancos Shale, nominal 2.5-ft screens were used to obtain discreet head measurements. In 1998, the two pairs of well nests in the Mancos Shale were installed with 2.5-ft screens to obtain discreet head measurement. Screen length was 5 ft for the pair of well nests installed in March–April 2000 in Mancos Shale. Other wells drilled in 1998 to 2000 typically had 5-ft or 10-ft screened intervals. Two of the wells (1010 and 1013) installed as part of the NABIR Program, had 15-ft screened lengths. Natural formation cave-in material was used as filter pack in most 1998 wells drilled in the floodplain alluvium, and 20-40 fraction sand was used as the filter sand in most of the other borings. The technical approach to the well installation was based on ASTM D 5092–90 (reapproved 1995). Well completion diagrams are in Appendix A of this report.

Each new monitor well was allowed to sit undisturbed for at least 24 hours after final completion before it was developed. Development was performed according to the Work Plan (DOE 1998d).

4.1.5 Packer Tests

Packer tests are conducted in a borehole after the hole is cored and flushed with clear water. The method consists of lowering the testing apparatus into the borehole, inflating the packers so that they fit snugly against the wall of the borehole, and then injecting water under pressure into the test interval. The flow of water into the test interval is measured with a flow meter; the flow rate is measured as a function of the injection pressure. This test provides an estimate of the hydraulic conductivity of the rock formation.

Packer tests were performed in 1998 on boreholes 820, 823, 860, and 862 (Plates 1 and 3). The tests began at the deepest part of the borehole and proceeded upward until representative parts of the formation were tested. The test intervals were selected on the basis of visual observations of the rock core retrieved from each borehole. The intervals were chosen in highly fractured, moderately fractured, and unfractured rock; each interval was 5 ft long. The diameter of the cored borehole was nominally 3 in. A gauge pressure of 40 pounds per square inch was used for the injection tests, and a test duration of 20 minutes was used whenever practicable.

Table 4-3. Construction Details for Monitor Wells at the Shiprock Site

Location Code	Install Date	North Coord. (ft State-Plane)	East Coord. (ft State-Plane)	Ground Elev. (ft NGVD)	Borehole Depth (ft BLS)	Borehole Diameter (inches)	Top of Casing Elevation (ft NGVD)	Well Depth (ft BLS)	Casing Diameter (inches)	Top of Screen Depth (ft BLS)	Screen Length (ft)	Top of Bedrock Depth (ft BLS)	Zone of Completion	Status
Wells Installed Since 1/1998														
Floodplain (SHP01)														
766	10/1999	2103964.44	250686.97	4888.68	9.0	–	4892.55	9.00	2.0	6.3	2.50	–	AL	Active
768	10/1999	2103147.09	250340.45	4889.28	7.3	–	4892.33	7.33	2.0	4.6	2.50	–	AL	Active
773	10/1999	2101742.40	251394.19	4891.50	6.8	–	4894.87	6.75	2.0	4.0	2.50	–	AL	Active
775	10/1999	2103476.13	250663.37	4888.92	7.0	–	4892.20	7.00	2.0	4.3	2.50	–	AL	Active
779	10/1999	2103162.67	251034.71	4890.93	9.8	–	4893.86	9.75	2.0	7.0	2.50	–	AL	Active
782	10/1999	2105138.22	247772.85	4882.24	6.8	–	4885.68	6.75	2.0	4.0	2.50	–	AL	Active
783	10/1999	2105116.90	247564.91	4881.86	7.6	–	4884.48	7.60	2.0	4.9	2.50	–	AL	Active
784	10/1999	2105420.96	247021.94	4879.22	7.3	–	4882.21	7.25	2.0	4.5	2.50	–	AL	Active
850	10/1998	2098486.21	256685.04	4904.99	20.0	8.0	4907.51	15.60	2.0	5.6	9.80	19.00	AL	Active
851	10/1998	2098473.35	256679.18	4904.63	13.0	8.0	4906.45	12.30	2.0	6.0	5.00	–	AL	Active
852	10/1998	2098472.49	256707.25	4904.61	13.0	8.0	4907.37	12.60	2.0	6.4	5.00	–	AL	Active
853	10/1998	2102501.58	251196.38	4888.81	16.5	8.0	4891.41	15.30	2.0	10.0	5.00	16.00	AL	Active
854	10/1998	2103848.58	250820.77	4888.35	13.0	8.0	4890.75	11.80	2.0	9.1	2.50	–	AL	Active
855	10/1998	2103849.57	249057.21	4885.59	17.8	8.0	4888.18	15.10	2.0	4.9	10.00	17.60	AL	Active
856	10/1998	2104395.65	249110.63	4884.83	24.5	8.0	4887.57	24.10	2.0	18.8	5.00	24.00	AL	Active
857	10/1998	2103029.83	251160.35	4891.61	19.2	8.0	4894.02	18.50	2.0	13.2	5.00	19.00	AL	Active
858	09/1998	2101963.30	251540.03	4891.38	25.3	8.75	4893.50	20.60	5.0	10.2	10.00	21.00	AL	Active
859	09/1998	2101971.57	251528.87	4891.37	24.5	8.75	4893.68	19.90	2.0	14.5	5.00	21.00	AL	Active
860	10/1998	2102538.99	250576.01	4889.50	91.0	5.88	4892.28	87.24	2.0	84.6	2.50	14.00	KM	Active
861	11/1998	2102546.90	250570.59	4889.80	138.5	5.875	4891.32	138.35	2.0	135.5	2.50	14.00	KM	Active
862	11/1998	2101451.27	251713.33	4890.73	91.8	5.875	4893.83	91.57	2.0	88.9	2.50	8.50	KM	Active
863	11/1998	2101459.13	251711.10	4890.85	137.7	5.875	4893.00	137.70	2.0	135.1	2.50	8.50	KM	Active
1000	04/2000	2102013.35	250969.35	4890.27	38.4	6.0	4892.17	38.40	2.0	33.1	5.00	10.50	KM	Active
1001	04/2000	2102020.79	250960.99	4890.25	28.0	6.0	4892.44	27.90	2.0	22.6	5.00	12.00	KM	Active
1008	04/2000	2103812.23	250769.64	4888.72	17.2	8.0	4890.80	17.20	4.0	6.9	10.00	15.50	AL	Active
1009	04/2000	2102533.18	250818.64	4890.29	17.7	8.0	4892.10	17.70	4.0	7.4	10.00	17.00	AL	Active
1010	04/2000	2103016.57	251086.63	4890.25	19.0	8.0	4892.32	19.00	4.0	3.7	15.00	18.50	AL	Active
1013	04/2000	2102517.14	251129.67	4889.00	22.3	8.0	4890.89	22.30	4.0	7.0	15.00	16.00	AL-KM	Active
1062	04/2000	2101439.45	251715.33	4890.66	36.0	6.0	4892.51	36.00	2.0	30.7	5.00	8.00	KM	Active

Table 4-3 (continued). Construction Details for Monitor Wells at the Shiprock Site

Location Code	Install Date	North Coord. (ft State-Plane)	East Coord. (ft State-Plane)	Ground Elev. (ft NGVD)	Borehole Depth (ft BLS)	Borehole Diameter (inches)	Top of Casing Elevation (ft NGVD)	Well Depth (ft BLS)	Casing Diameter (inches)	Top of Screen Depth (ft BLS)	Screen Length (ft)	Top of Bedrock Depth (ft BLS)	Zone of Completion	Status
Terrace (SHP02)														
800	09/1998	2097118.68	261458.17	4993.14	65.0	8.75	4995.76	62.46	2.0	52.3	10.00	14.00	KM	Active
801	11/1998	2096236.35	260359.85	4993.22	68.0	8.25	4995.29	65.00	2.0	54.8	10.00	16.00	KM	Active
802	09/1998	2096472.78	259469.34	4992.80	65.0	8.75	4996.01	61.56	2.0	51.4	10.00	20.00	KM	Active
803	11/1998	2097915.13	261956.47	4992.10	68.0	8.25	4994.40	65.00	2.0	55.0	9.80	15.00	KM	Active
804	10/1998	2098659.62	252260.86	4934.73	70.5	5.875	4936.93	70.00	2.0	59.8	10.00	24.00	KM	Active
805	10/1998	2097803.99	252157.62	4950.34	50.9	5.875	4953.14	49.90	2.0	39.7	10.00	3.50	KM	Active
810	09/1998	2095925.14	247626.49	5050.27	100.0	5.875	5049.58	90.00	2.0	79.9	10.00	28.00	KM	Active
812	10/1998	2098339.51	248308.83	5002.16	61.5	5.875	5004.98	61.50	2.0	51.3	10.00	55.00	AL-KM	Active
813	10/1998	2099346.57	248023.06	4984.52	51.0	5.875	4984.37	51.00	2.0	40.8	10.00	47.00	AL-KM	Active
814	11/1998	2100474.01	247414.84	4968.37	36.5	5.875	4968.12	34.00	2.0	23.8	10.00	29.00	AL-KM	Active
815	11/1998	2101610.39	247426.75	4953.79	36.0	5.875	4953.67	32.50	2.0	22.3	10.00	27.00	AL-KM	Active
816	11/1998	2103511.60	247952.70	4935.37	31.0	5.875	4937.92	25.30	2.0	20.1	5.00	23.00	AL-KM	Active
817	10/1998	2100885.97	249770.34	4957.77	36.0	8.875	4957.34	32.00	5.0	21.6	10.02	12.00	KM	Active
818	10/1998	2098534.26	249199.65	4995.40	64.5	8.875	4998.25	62.00	5.0	52.0	9.50	62.00	AL	Active
819	10/1998	2101176.66	249753.77	4956.42	31.2	5.875	4955.76	26.00	2.0	15.7	10.00	12.00	KM	Active
820	11/1998	2102191.62	250374.05	4954.14	153.0	5.875	4954.95	151.89	2.0	149.0	2.50	12.00	KM	Active
821	11/1998	2102200.62	250370.62	4954.21	104.0	5.875	4955.46	101.89	2.0	99.0	2.50	12.00	KM	Active
822	11/1998	2102192.54	250363.65	4953.85	205.0	5.875	4954.42	201.66	2.0	199.0	2.50	12.00	KM	Active
823	09/1998	2101289.48	251528.73	4956.53	122.0	5.875	4957.65	100.34	2.0	97.5	2.50	26.00	KM	Active
824	10/1998	2101288.61	251538.80	4956.75	201.1	5.875	4958.21	201.10	2.0	198.5	2.50	24.00	KM	Active
825	10/1998	2101298.38	251534.90	4956.94	151.0	5.875	4958.68	150.45	2.0	147.8	2.44	27.00	KM	Active
826	10/1998	2101938.33	249596.17	4948.09	31.0	5.875	4950.73	20.17	2.0	10.0	10.00	12.00	AL-KM	Active
827	11/1998	2102444.90	249873.25	4943.91	31.3	5.875	4946.92	30.03	2.0	19.9	10.00	22.00	AL-KM	Active
828	10/1998	2101524.12	249145.90	4946.67	41.0	5.875	4949.34	15.47	2.0	5.3	10.00	7.00	AL-KM	Active
829	10/1998	2102758.77	249544.67	4939.54	62.0	5.875	4941.94	50.20	2.0	40.0	10.00	16.00	KM	Active
830	11/1998	2099901.80	251233.69	4957.75	23.5	5.875	4960.77	17.80	2.0	7.7	10.00	9.00	KM	Active
832	11/1998	2100815.04	245788.84	4964.91	37.0	5.875	4964.65	31.30	2.0	21.1	10.00	28.00	AL-KM	Active
833	12/1998	2102760.52	245623.02	4938.15	41.0	5.875	4940.52	35.00	2.0	24.9	10.00	35.00	AL	Active
835	12/1998	2104159.66	246020.38	4927.75	35.5	5.875	4930.48	32.00	2.0	21.9	10.00	32.00	AL	Active
836	12/1998	2103969.34	241957.93	4898.74	43.0	5.875	4901.74	36.90	2.0	26.8	10.00	37.00	AL	Active
837	12/1998	2105185.63	243678.55	4886.45	32.0	5.875	4889.54	27.20	2.0	17.0	10.10	27.00	AL	Active
838	12/1998	2102498.85	244738.77	4934.66	39.0	5.875	4937.70	32.00	2.0	21.9	10.00	32.00	AL	Active
839	11/1998	2102521.32	247357.45	4943.46	31.0	5.875	4943.21	28.30	2.0	18.1	10.00	27.00	AL-KM	Active

Table 4-3 (continued). Construction Details for Monitor Wells at the Shiprock Site

Location Code	Install Date	North Coord. (ft State-Plane)	East Coord. (ft State-Plane)	Ground Elev. (ft NGVD)	Borehole Depth (ft BLS)	Borehole Diameter (inches)	Top of Casing Elevation (ft NGVD)	Well Depth (ft BLS)	Casing Diameter (inches)	Top of Screen Depth (ft BLS)	Screen Length (ft)	Top of Bedrock Depth (ft BLS)	Zone of Completion	Status
841	11/1998	2099895.06	246000.03	4981.43	57.0	5.875	4984.05	52.20	2.0	42.0	10.00	50.00	AL	Active
843	12/1998	2105743.99	244999.74	4880.60	30.0	5.875	4883.56	22.00	2.0	11.9	10.00	21.50	AL	Active
844	11/1998	2102036.39	246001.56	4948.66	43.0	5.875	4948.46	40.20	2.0	30.0	10.00	34.00	AL-KM	Active
845	11/1998	2100877.91	245146.72	4965.87	28.5	8.0	4969.20	28.33	2.0	18.2	10.00	–	AL	Active
846	12/1998	2102475.12	242268.43	4931.75	32.0	5.875	4934.57	28.00	2.0	17.9	10.00	25.00	AL-KM	Active
1002	03/2000	2101812.07	250892.78	4955.78	103.3	6.0	4957.63	103.30	2.0	98.2	4.80	12.00	KM	Active
1003	03/2000	2101818.32	250884.95	4955.93	92.8	6.0	4957.84	92.30	2.0	87.2	4.80	12.00	KM	Active
1004	04/2000	2101807.54	250884.49	4955.58	42.4	6.0	4957.61	40.00	2.0	34.9	4.80	12.00	KM	Active
1006	04/2000	2100599.95	251758.62	4960.13	38.3	6.0	4962.16	38.30	2.0	28.5	9.50	25.00	FL-KM	Active
1007	04/2000	2100457.81	251791.21	4960.03	46.6	6.0	4962.01	46.60	2.0	36.8	9.50	42.00	FL-KM	Active
1011	04/2000	2102537.67	249922.02	4943.93	26.3	6.0	4945.96	26.30	2.0	16.5	9.50	16.00	QA-KM	Active
1048	12/1999	2097481.46	252735.06	4921.48	9.0	2.2	4921.35	8.90	1.0	3.6	5.00	7.50	QA-KM	Active
1049	12/1999	2097350.35	252721.23	4924.09	10.0	2.2	4923.89	9.60	1.0	4.3	5.00	9.00	QA-KM	Active
1057	03/2000	2098222.99	250667.36	4978.94	40.0	6.0	4980.89	36.50	2.0	31.2	5.00	37.00	QA	Active
1058	03/2000	2098084.43	251464.48	4971.67	51.5	6.0	4973.58	51.50	2.0	41.7	9.50	26.50	KM	Active
1059	04/2000	2097603.88	252100.93	4968.55	50.0	6.0	4970.52	49.30	2.0	39.5	9.50	16.50	KM	Active
1060	04/2000	2100719.07	244446.80	4968.57	38.0	6.0	4970.62	37.00	2.0	27.2	9.50	35.50	QA-KM	Active
Wells Installed Before 1998														
Floodplain (SHP01)														
601	09/1984	2103195.24	251150.35	4890.00	6.0	–	4890.00	3.58	1.25	0.4	2.92	–	AL	Abandoned
602	09/1984	2102936.86	250749.31	4890.00	7.0	–	4890.00	3.58	1.25	0.4	2.92	–	AL	Abandoned
603	09/1984	2103099.48	250099.96	4888.00	5.0	–	4888.00	3.58	1.25	1.4	1.92	–	AL	Abandoned
604	09/1984	2103521.29	249651.66	4888.00	6.0	–	4888.00	3.58	1.25	0.4	2.92	–	AL	Abandoned
606	10/1984	2103248.20	249451.05	4887.67	5.3	–	4888.57	3.58	1.25	0.9	2.30	–	AL	Abandoned
607	10/1984	2102958.88	250249.39	4888.00	6.6	–	4890.00	3.58	1.25	0.9	2.30	–	AL	Abandoned
608	08/1985	2101434.86	251712.58	4891.67	19.0	8.75	4893.35	17.00	4.0	10.0	5.00	10.00	KM	Active
609	08/1985	2101450.02	251704.91	4890.97	14.0	8.75	4892.46	10.80	4.0	3.8	5.00	8.00	AL	Active
610	09/1985	2101686.65	251334.83	4892.24	15.0	8.75	4895.70	11.00	4.0	4.0	5.00	13.00	AL	Active
611	09/1985	2101693.09	251324.05	4892.35	22.0	8.75	4895.62	16.25	4.0	9.5	5.00	13.00	AL-KM	Active
612	09/1985	2101985.43	251560.91	4891.91	15.0	8.75	4893.35	12.00	4.0	5.0	5.00	14.50	AL	Active
613	09/1985	2101991.72	250943.68	4889.92	15.0	8.75	4893.19	12.00	4.0	5.0	5.00	14.00	AL	Active
614	09/1985	2101985.26	250953.07	4890.30	19.0	8.75	4892.79	17.00	4.0	10.0	5.00	14.00	AL-KM	Active
615	09/1985	2102542.15	250564.45	4890.83	14.0	8.75	4892.23	11.50	4.0	4.5	5.00	13.00	AL	Active
616	09/1985	2103008.96	251039.92	4890.28	14.0	8.75	4891.90	12.00	4.0	5.0	5.00	–	AL	Active

Table 4-3 (continued). Construction Details for Monitor Wells at the Shiprock Site

Location Code	Install Date	North Coord. (ft State-Plane)	East Coord. (ft State-Plane)	Ground Elev. (ft NGVD)	Borehole Depth (ft BLS)	Borehole Diameter (inches)	Top of Casing Elevation (ft NGVD)	Well Depth (ft BLS)	Casing Diameter (inches)	Top of Screen Depth (ft BLS)	Screen Length (ft)	Top of Bedrock Depth (ft BLS)	Zone of Completion	Status
617	09/1985	2102937.07	250761.09	4890.05	20.0	8.75	4891.90	12.00	4.0	5.0	5.00	19.80	AL	Active
618	09/1985	2102934.43	250748.52	4889.87	21.0	8.75	4891.51	18.00	4.0	11.0	5.00	20.00	AL	Active
619	09/1985	2103321.90	250401.87	4890.42	20.0	8.75	4892.19	15.00	4.0	8.0	5.00	18.00	AL	Active
620	08/1985	2102960.74	250243.13	4888.18	23.0	8.75	4889.72	20.00	4.0	13.0	5.00	17.00	AL-KM	Active
621	08/1985	2102960.06	250252.85	4888.33	19.0	8.75	4890.20	17.00	4.0	10.0	5.00	16.50	AL	Active
622	08/1985	2102958.94	250263.63	4888.51	16.0	8.75	4890.06	12.00	4.0	5.0	5.00	–	AL	Active
623	09/1985	2103409.01	250256.67	4889.27	23.0	8.75	4891.19	17.00	4.0	10.0	5.00	17.00	AL	Active
624	09/1985	2103396.91	250252.71	4889.29	24.0	8.75	4891.49	22.00	4.0	15.0	5.00	18.00	AL-KM	Active
625	09/1985	2103384.86	250249.62	4889.28	17.0	8.75	4891.23	11.50	4.0	4.5	5.00	–	AL	Active
626	09/1985	2103324.50	249941.38	4888.48	20.0	8.75	4891.40	16.50	4.0	9.5	5.00	19.00	AL	Active
627	09/1985	2103526.75	249650.71	4887.48	20.0	8.75	4889.41	15.00	4.0	8.0	5.00	17.00	AL	Active
628	09/1985	2103517.40	249660.32	4887.84	15.0	8.75	4889.87	12.00	4.0	6.0	4.00	–	AL	Active
629	09/1985	2103359.79	249378.67	4887.29	20.0	8.75	4887.49	17.00	4.0	10.0	5.00	13.00	AL-KM	Active
630	09/1985	2103349.44	249382.75	4887.65	15.0	8.75	4887.62	12.00	4.0	5.0	5.00	13.00	AL	Active
631	09/1985	2105158.16	249038.59	4888.21	23.0	8.75	4889.95	20.00	4.0	13.0	5.00	20.00	AL	Active
632	09/1985	2105146.77	249045.09	4888.17	20.0	8.75	4890.01	15.00	4.0	8.0	5.00	19.00	AL	Active
634	09/1985	2102727.63	252113.40	4896.20	24.0	–	4896.90	24.00	–	–	–	–	AL	Active
635	09/1985	2103503.93	251674.62	4893.01	12.0	–	4895.01	12.00	–	–	–	–	AL	Active
638	03/1987	2104780.10	248983.91	4882.17	5.0	–	4884.37	5.00	2.0	0.0	5.00	–	AL	Abandoned
639	03/1987	2104782.81	249952.79	4889.00	5.0	10.0	4890.07	5.00	8.0	0.0	5.00	–	AL	Active
640	03/1987	2104446.71	248636.45	4881.37	5.0	–	4883.97	5.00	2.0	0.0	5.00	–	AL	Abandoned
641	03/1987	2103910.58	249690.43	4884.21	5.0	–	4887.41	5.00	2.0	0.0	5.00	–	AL	Abandoned
642	03/1987	2104375.10	249931.82	4883.87	5.0	–	4886.37	5.00	2.0	0.0	5.00	–	AL	Abandoned
643	03/1987	2104440.83	249162.13	4882.73	5.0	–	4885.63	5.00	2.0	0.0	5.00	–	AL	Abandoned
644	03/1987	2104136.15	250519.01	4884.97	5.0	–	4886.96	5.00	2.0	0.0	5.00	–	AL	Abandoned
645	03/1987	2100670.51	252104.62	4898.70	5.0	–	4901.30	5.00	2.0	0.0	5.00	–	AL	Abandoned
646	03/1987	2100610.00	252118.00	4898.63	5.0	–	4902.33	5.00	2.0	0.0	5.00	–	AL	Abandoned
647	03/1987	2100547.36	252118.53	4898.02	5.0	–	4902.32	5.00	2.0	0.0	5.00	–	AL	Abandoned
670	01/1988	2104550.07	250560.69	4889.10	11.1	–	4892.67	11.05	2.0	7.0	3.50	–	AL	Active
671	01/1988	2104418.59	250662.29	4889.49	10.9	–	4892.65	10.90	2.0	6.9	3.50	–	AL	Active
672	01/1988	2103823.00	251489.00	4891.50	10.9	–	4894.41	10.88	2.0	6.9	3.50	–	AL	Active
732	03/1993	2099626.94	252632.79	4895.62	19.0	8.0	4897.55	19.00	2.0	7.0	10.00	12.00	AL-KM	Active
733	03/1993	2104885.18	249564.17	4887.78	15.0	6.0	4889.67	13.50	2.0	6.5	5.00	–	AL	Active

Table 4-3 (continued). Construction Details for Monitor Wells at the Shiprock Site

Location Code	Install Date	North Coord. (ft State-Plane)	East Coord. (ft State-Plane)	Ground Elev. (ft NGVD)	Borehole Depth (ft BLS)	Borehole Diameter (inches)	Top of Casing Elevation (ft NGVD)	Well Depth (ft BLS)	Casing Diameter (inches)	Top of Screen Depth (ft BLS)	Screen Length (ft)	Top of Bedrock Depth (ft BLS)	Zone of Completion	Status
734	03/1993	2104505.13	248608.49	4886.00	7.0	2.0	4886.55	7.00	2.0	2.0	2.00	–	AL	Active
735	03/1993	2099904.08	252193.67	4894.53	9.0	6.0	4895.85	9.00	4.0	3.0	5.00	–	AL	Active
736	03/1993	2104420.64	249808.04	4887.20	7.0	2.0	4887.99	7.00	2.0	3.0	2.00	–	AL	Active
Terrace (SHP02)														
600	01/1982	2102012.65	250674.90	4955.45	62.7	6.75	4955.87	48.80	4.0	29.0	19.80	13.80	KM	Active
601	06/1983	2099020.00	250616.00	4981.24	50.0	6.0	–	45.30	2.0	30.3	10.00	37.00	AL-KM	Abandoned
602	12/1981	2100887.57	249786.07	4957.89	96.7	6.75	4956.89	47.00	4.0	27.0	20.00	9.50	KM	Active
603	06/1983	2098739.34	251189.95	4977.61	42.0	6.0	4978.62	40.90	2.0	25.9	10.00	31.00	AL-KM	Active
604	05/1983	2098538.57	249216.98	4995.43	80.0	6.0	4995.87	77.70	2.0	62.7	10.00	58.00	KM	Active
605	10/1984	2102920.00	249219.00	4898.77	3.8	–	4898.77	3.58	1.25	0.9	2.30	–	AL	Abandoned
633	10/1985	2102392.61	249198.00	4915.99	3.4	5.88	4918.24	3.42	2.0	0.0	3.42	–	AL	Abandoned
648	02/1961	2102944.07	248019.38	4940.18	1850.0	12.0	4943.80	1850.00	12.0	1482.0	295.00	30.00	JM	Active
725	03/1993	2103010.18	249192.23	4906.29	20.0	6.0	4908.58	19.50	2.0	7.5	10.00	16.00	AL-KM	Active
726	03/1993	2102452.85	248972.56	4937.97	40.0	6.0	4939.95	39.20	2.0	27.2	10.00	9.00	KM	Active
727	03/1993	2101721.10	248674.51	4938.52	19.0	6.0	4940.65	18.70	2.0	6.7	10.00	6.50	KM	Active
728	03/1993	2100541.89	248356.21	4962.55	30.0	6.0	4964.46	29.00	2.0	17.0	10.00	23.00	AL-KM	Active
730	03/1993	2099429.89	249494.92	4977.81	40.0	6.0	4979.74	39.00	2.0	27.0	10.00	33.00	AL-KM	Active
731	03/1993	2098278.21	251390.35	4970.15	29.0	6.0	4972.15	29.00	2.0	17.0	10.00	23.00	AL-KM	Active
847	01/1995	2102987.81	243884.59	4924.17	–	–	4924.35	–	–	–	–	–	AL-KM	Active
848	01/1995	2101767.85	243482.71	4949.89	142.6	–	4949.91	142.58	4.5	45.0	97.58	–	AL-KM	Active
9003	01/1982	2100683.39	251603.22	4955.80	53.7	6.75	–	30.00	4.0	15.0	15.00	4.50	KM	Abandoned
9004	01/1981	2100403.17	250914.08	4970.60	47.6	6.75	–	29.40	4.0	25.4	4.00	27.00	AL-KM	Abandoned
9005	02/1982	2100373.08	250936.96	4970.00	87.4	6.75	–	56.00	4.0	35.0	19.00	29.40	KM	Abandoned
9006	12/1981	2101071.21	250410.78	4968.00	85.3	6.75	–	54.00	4.0	44.0	10.00	19.00	KM	Abandoned
9007	12/1981	2099416.57	250814.10	4973.50	92.5	6.75	–	48.00	4.0	29.0	19.00	24.00	KM-AL	Abandoned
9008	02/1982	2100285.75	249283.60	4966.70	87.6	6.75	–	64.00	4.0	36.0	27.00	31.00	KM	Abandoned
9009	12/1982	2100217.58	249326.32	4966.80	47.7	6.75	–	45.00	2.0	27.0	13.00	23.50	KM-AL	Abandoned
9010	01/1982	2100428.55	250324.17	4985.00	74.3	6.75	–	65.00	4.0	45.0	20.00	33.00	KM	Abandoned
9011	01/1982	2101128.78	251012.01	4986.40	71.3	6.75	–	70.58	4.0	49.1	20.50	45.00	KM	Abandoned
9012	03/1982	2098851.33	249632.81	4989.20	85.0	6.75	–	84.67	4.0	54.3	30.34	44.00	KM	Abandoned

Table 4-3 (continued). Construction Details for Monitor Wells at the Shiprock Site

Location Code	Install Date	North Coord. (ft State-Plane)	East Coord. (ft State-Plane)	Ground Elev. (ft NGVD)	Borehole Depth (ft BLS)	Borehole Diameter (inches)	Top of Casing Elevation (ft NGVD)	Well Depth (ft BLS)	Casing Diameter (inches)	Top of Screen Depth (ft BLS)	Screen Length (ft)	Top of Bedrock Depth (ft BLS)	Zone of Completion	Status
9013	05/1983	2102145.70	250174.21	4943.33	60.0	6.0	–	25.00	2.0	10.0	10.00	0.00	KM	Abandoned
9014	05/1983	2100104.96	251861.59	4962.90	60.0	6.0	–	38.00	2.0	23.0	10.00	18.00	KM	Abandoned
9015	05/1983	2099606.10	248675.35	4977.31	60.0	6.0	–	53.70	2.0	38.7	10.00	41.00	KM-AL	Abandoned
9016	06/1983	2098615.77	250779.82	4983.93	55.0	6.0	–	52.60	2.0	37.6	10.00	42.00	AL-KM	Abandoned
9017	06/1983	2099368.86	251287.21	4971.43	35.0	6.0	–	35.00	2.0	20.0	10.00	25.00	AL-KM	Abandoned
9018	06/1993	2098296.75	250955.20	4983.71	50.0	6.0	–	39.40	2.0	29.4	10.00	41.00	AL-KM	Abandoned
9019	06/1983	2099053.00	251494.48	4972.78	39.0	6.0	–	34.00	2.0	19.0	10.00	24.00	KM-AL	Abandoned
9020	01/1982	2100438.78	250269.22	4985.00	40.7	6.75	–	40.70	2.0	35.3	5.00	40.00	AL	Abandoned
DM7	01/1982	2099645.67	249944.02	4976.50	85.1	5.6	4974.50	54.00	4.0	38.0	15.00	29.00	KM	Active
MW1	–	2101488.51	251338.36	4956.91	–	–	4955.64	–	–	–	–	–	NR	Active

Zones of Completion:

AL—Alluvium
KM—Mancos Shale
BLS –below land surface
NGVD–National Geodetic Vertical Datum
FL—Fill material
NR—No recovery of data for classifying
JM—Morrison Formation, Westwater Canyon Member
QA—Quaternary

The depth to water was recorded before each sequence of tests in a borehole. All tests were performed below the water table. Computations of the hydraulic conductivity were made with the appropriate formulas (University of Missouri-Rolla 1981; U.S. Bureau of Reclamation 1974).

Each reported measurement was assumed to represent a constant flow rate averaged over the elapsed time increment. If the flow rate was so low that it could not be measured with the flow meter, the hydraulic conductivity result was assumed to be less than the detection limit, and the detection limit itself was reported. Raw data and computations of the hydraulic conductivity are presented in a calculation set (MACTEC–ERS 1999a).

4.1.6 Water Level Measurements

Water level measurements provided information on ground water flow directions, saturated thickness of the aquifer, and temporal changes in water levels. Measurements were made with a commercially available, weighted, electrical measuring tape. All measurements were taken with respect to a fixed point at the top of each PVC well casing. Water level measurements were collected in all wells in December 1998, March 1999, June 1999, and February 2000. The most recent measurements (February 2000) were used to prepare the water table maps presented in Section 4.3 of this report. For the new wells installed in March–April 2000, water levels measured in April 2000 were used for water table maps. Each measurement was made to the nearest 0.01 ft. Measurements of ground water began as early as 1984 for a subset of wells; these wells provide an opportunity to construct time series plots of ground water elevations. Manual measurements of the water levels were conducted using the guidance in the *Environmental Procedures Catalog*, LQ-2(T), "Standard Test Method for the Measurement of Water Levels in Ground Water Monitoring Wells" (DOE 1998b).

Electronic data loggers in selected monitor wells provide continuous water level records for the site. The data are collected at 4-hour intervals and are obtained by programming the electronic data loggers and periodically downloading the data files. The data logger measurements began on February 5, 1999, and are collected each time the water sampling crew visits the site (March 1999, June 1999, and February 2000).

4.1.7 Ground Water Sampling and Analysis

After the wells were developed, ground water samples were collected from the new monitor well network and selected existing wells and were submitted to the GJO Analytical Chemistry Laboratory for analyses. Figure 4–1 presents the locations where surface and ground water samples were collected during the most recent sampling round in February 2000. Also shown in Figure 4–1 are the 16 new monitor wells, which were sampled in April 2000 immediately after they were installed.

Ground water sampling was performed in accordance with the *Addendum to the Sampling and Analysis Plan for the UMTRA Ground Water Project* (DOE 1996a), the *Sampling and Analysis Plan for the UMTRA Ground Water Project* (DOE 1999h), and the *Environmental Procedures Catalog* (DOE 1998b). The following specific procedures from the *Environmental Procedures Catalog* were used for ground water sampling:

- GN-8(P), "Standard Practice for Sample Labeling"

- GN-9(P), "Standard Practice for Chain-of-Sample-Custody and Physical Security of Samples"
- GN-13(P), "Standard Practice for Equipment Decontamination"
- LQ-3(P), "Standard Practice for Purging of Monitoring Wells"
- LQ–11(P), "Standard Practice for Sampling Liquids"
- LQ–12(P), "Standard Practice for the Collection, Filtration, and Preservation of Liquid Samples"
- LQ–2(T), "Standard Test Method for the Measurement of Water Levels in Ground Water Monitoring Wells"
- LQ–4(T), "Standard Test Method for the Field Measurement of pH"
- LQ–5(T), "Standard Test Method for the Field Measurement of Specific Conductance"
- LQ–6(T), "Standard Test Method for the Field Measurement of the Oxidation-Reduction Potential (Eh)"

4.1.8 Aquifer Tests

Aquifer tests were performed in 1998 in each of the hydrostratigraphic units at the site. One aquifer test was completed in the floodplain alluvium and two tests were completed in the terrace unit. Figure 4–2 and Figure 4–3 show the locations and well configurations, respectively, for the tests. Electronic data loggers were used to capture time and drawdown measurements. The captured data were transferred onto computer files using the software provided by the manufacturer of the data loggers. The data files were copied into Excel 97 spreadsheets and then copied into Aquifer^{Win32} software (ESI 1999) for analysis and interpretation of the results. Detailed results and interpretation of the pumping test data are presented in a calculation set (MACTEC–ERS 1999b). Section 4.3 presents plots of the drawdown-versus-time data for the pumping tests.

The pumping tests were analyzed using Neuman (1972), the Theis unconfined approximation, and the Theis recovery test methods (Theis 1935). These analysis methods are contained in the Aquifer^{Win32} software package.

4.1.9 Surveying

Location and elevation surveying of key hydrogeologic features were performed in January 1999, May 1999, and May 2000. All surveying was referenced to USBR BM R–11–L (brass cap, elevation of 4,939.70 ft; local coordinates of North 10,000, East 10,000). Specific hydrogeologic features that were surveyed include all active monitor wells (all monitor wells installed previously by others were resurveyed), surface water and soil sample locations, location and elevation of the San Juan River at various points, location and elevation of a siltstone bed in the Mancos Shale, location and elevation of all test pits excavated from 1998 to 2000, location and elevation of all test borings drilled from 1998 to 2000, and location and elevation of seeps and springs along the escarpment. Locations and selected elevations were measured using global positioning system (GPS) methods. Critical elevations, specifically top-of-well casing, were established by running a level loop from the USBR BM R–11–L. All survey locations and elevations were then transferred to the geographic information system (GIS) database at GJO where they are stored.

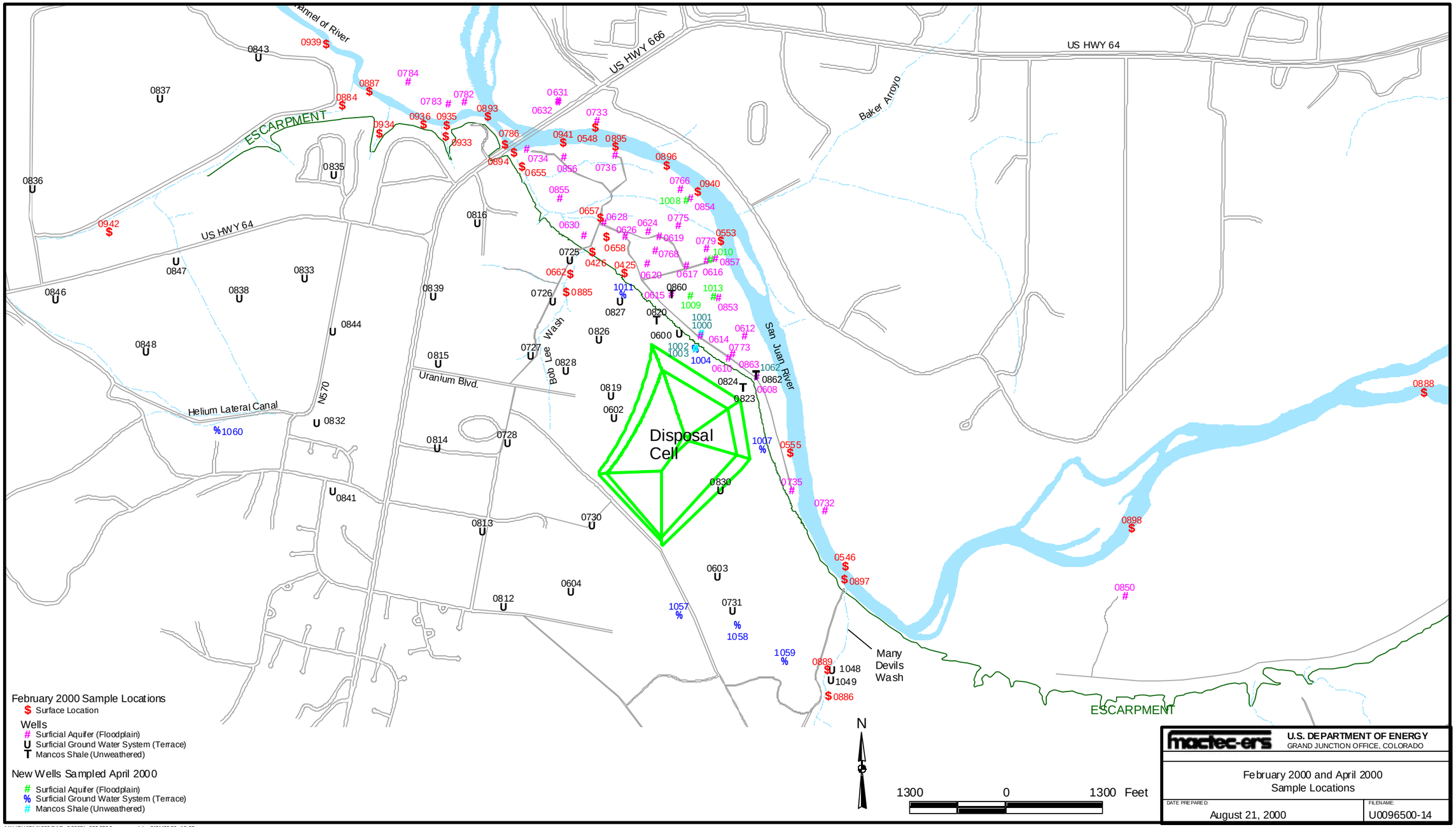


Figure 4-1. Ground Water and Surface Water Sampling Locations for Most Recent Sampling Event, Shiprock, New Mexico, UMTRA Site

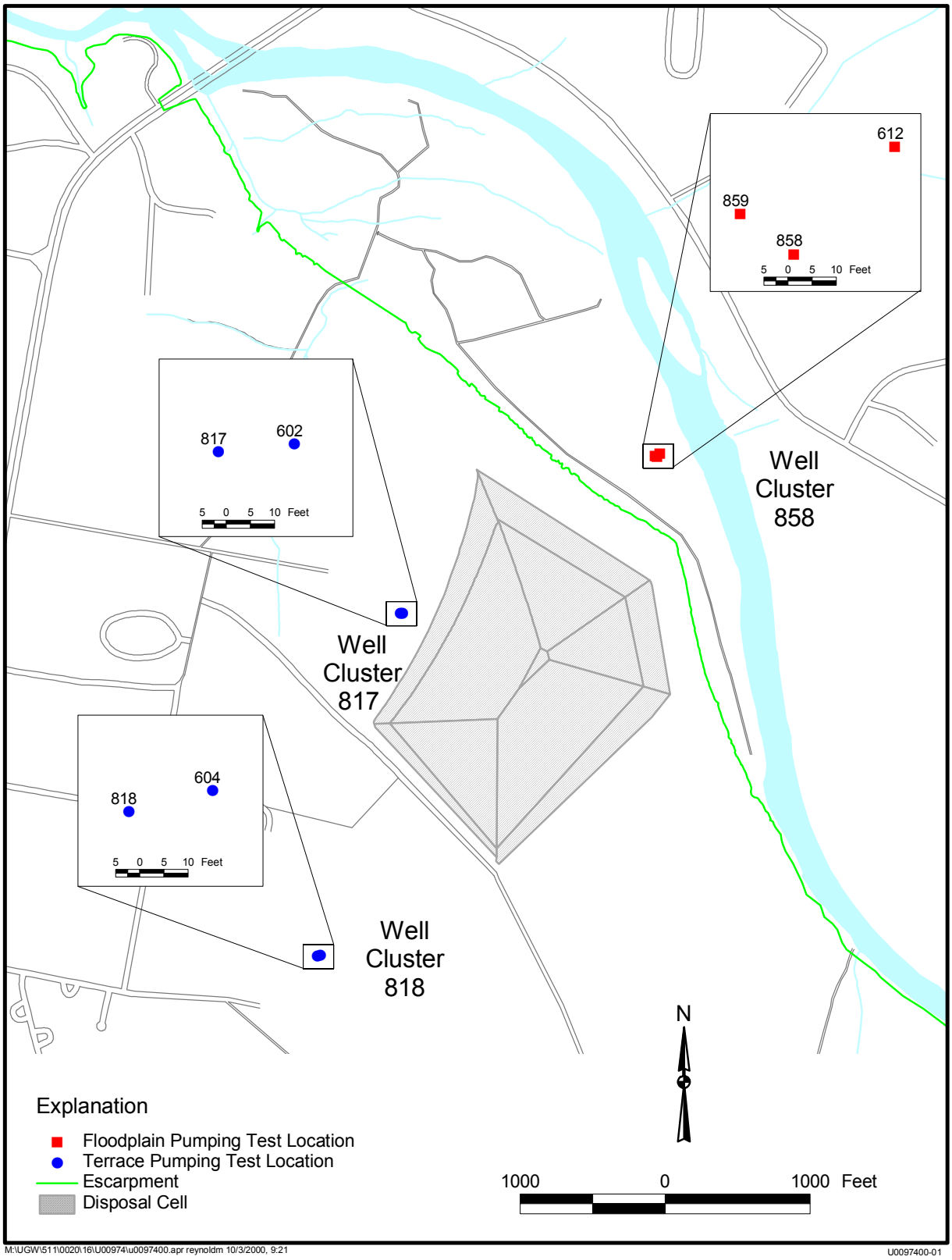


Figure 4–2. Location Map of Pumping Tests Completed in the Floodplain Alluvial Aquifer and the Terrace Ground Water System, Shiprock Site

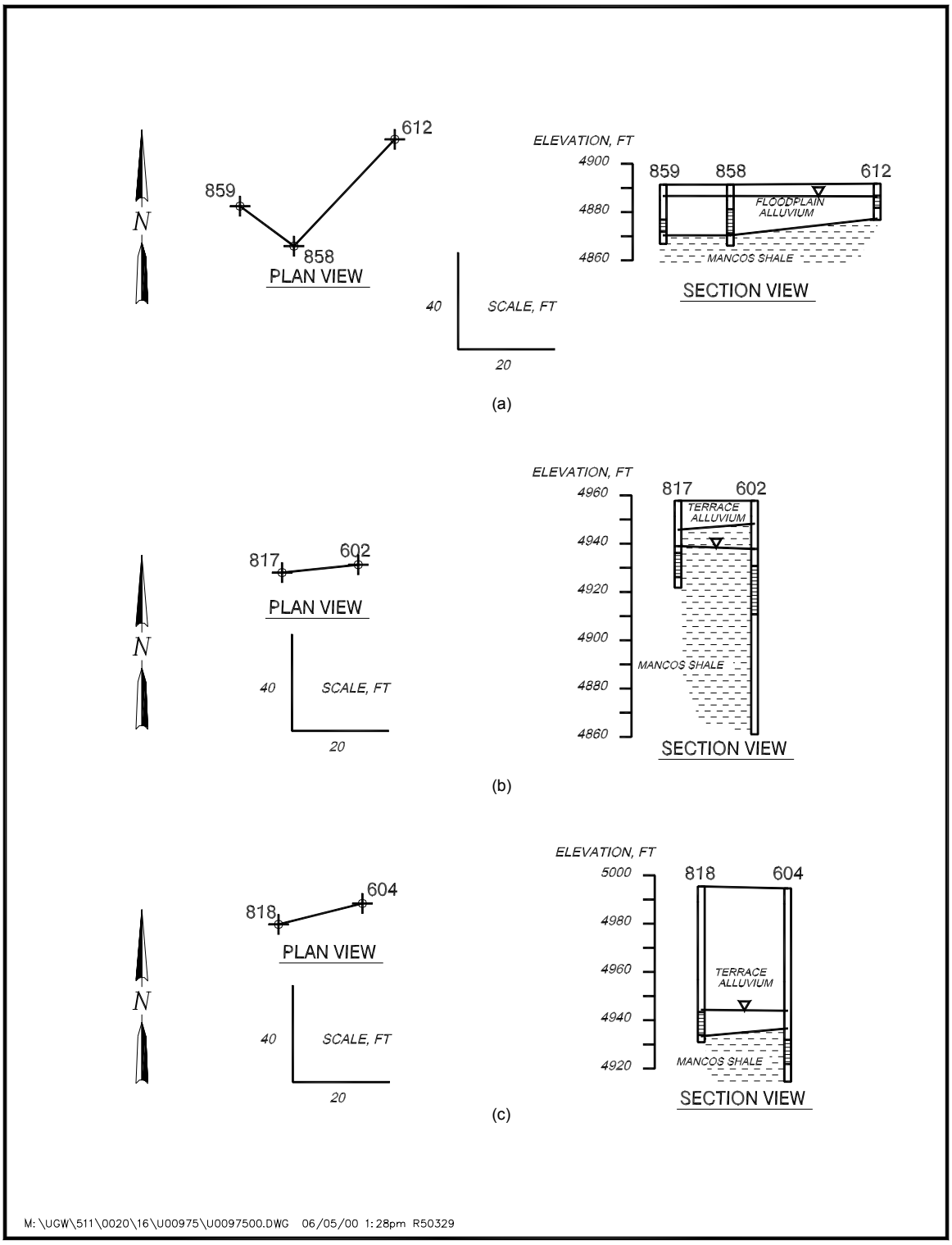


Figure 4–3. Well Cluster Cross Sections for Pumping Tests at (a) the Floodplain Aquifer, (b) the Weathered Mancos Shale, and (c) the Terrace Alluvium at the Shiprock Site

4.2 Geology

Bedrock underlying all the site area is the Late Cretaceous Mancos Shale that dips gently eastward. Unconsolidated Quaternary deposits consisting of terrace material, loess, and floodplain alluvium cover the bedrock in much of the area within 0.5 mi of the San Juan River. Detailed geologic maps of the site area have not been published; only small-scale geologic mapping by O'Sullivan and Beikman (1963) and Ward (1990) are available.

The Work Plan (DOE 1998d) presents summaries of the stratigraphy and structure of the site area as it was known from previous sources, namely the SOWP, Rev. 0 (DOE 1995), mapping of surficial material by Ward (1990), and geophysical surveys by DOE (1996c). Also identified in the Work Plan were geologic data needs, which, if provided, would improve the site conceptual model and refine the parameters necessary for use in ground water remediation. Data needs defined as tasks were (1) map the surface geology to identify the contact of weathered Mancos Shale bedrock and Quaternary material along the north side of the upland area, (2) measure the orientation and spacing of joints (fractures) in the escarpment where Mancos Shale is well exposed, (3) describe cuttings from proposed boreholes to improve the understanding of bedrock topography and thicknesses of overlying Quaternary geologic units, and (4) describe core from deep boreholes that penetrate into weathered and unweathered Mancos Shale to determine the degree of fracturing and the relative amounts of ground water. The results of these field investigations from 1998 to 2000 are discussed in Sections 4.2.1 through 4.2.4.

4.2.1 Geologic Mapping

The emphasis in geologic mapping of the site was to delineate the contact between the bedrock (Mancos Shale) and Quaternary material. This map, presented as Plate 3, does not distinguish weathered from unweathered Mancos Shale; however, Quaternary material is divided into four units. The location and orientation of joints in Mancos Shale were measured during the geologic mapping; Section 4.2.2 presents descriptions of these features. Also on the geologic map are lines showing the location of 10 cross sections that are presented in Plate 4.

Mapping for much of the site area was done on a base map made by enlarging the USGS 7.5 minute (1:24,000 scale) Shiprock topographic map with a contour interval of 20 ft. For the central part of the site, including the millsite/disposal cell and floodplain just to the north, mapping was done on a 2-ft contour topographic base map at a scale of 1:2,400. This map was produced by Morrison-Knudsen Engineers in June 1987 after the disposal cell was completed. A base map covering the site and surrounding area at a scale of 1:2,400 and a contour interval of 2 ft is needed to map detailed geologic characteristics and design remedial actions.

Descriptions of the surface features noted during mapping of the Mancos Shale and Quaternary units are presented in the following sections. Included are pertinent interpretations of these data as related to ground water hydrology of the site.

4.2.1.1 Mancos Shale

Drab gray to gray-tan exposures of Mancos Shale in the site area represent the upper part of this thick formation, deposited as an open marine mudstone in the Late Cretaceous Western Interior Seaway. Approximately 1,000 ft of the Mancos underlies the site. Most Mancos exposures in the upland area and other areas of low relief are weathered and resemble colluvium. This weathered material is soft, and bedding is only poorly to moderately exposed.

The 50- to 60-ft-high escarpment separating the San Juan River floodplain from the adjacent terrace contains the best Mancos Shale exposures in the site area. In several places, such as just upstream and downstream of the Many Devils Wash confluence with the San Juan River and downstream of the south end of the U.S. Highway 666 bridge, the escarpment plunges directly to the San Juan River. The shale exposed in the escarpment is well bedded and only slightly weathered. Another area of well-exposed Mancos Shale is along the lowermost 1,200 ft of Many Devils Wash, where the wash has incised its narrow channel up to 20 ft into the shale.

Thin bentonite layers are visible in several places along the escarpment, particularly at seep 427. Here, a soft, tan-orange to tan-brown layer about 0.5 to 1.0 in. thick appears to provide the pathway for the seep. Other thin bentonite layers are present along the cliff between seeps 425 and 426 and in the areas of seeps 786 and 935.

A continuous, distinctive, thin, resistant tan-to-orange, weathered, fossiliferous, calcareous siltstone bed about 1 ft thick forms a marker bed in the Mancos Shale in part of the site area. The bed is exposed mainly in the escarpment cliff north and east of the disposal cell, starting from the area of seep 427 and extending southeastward along the San Juan River to about 1,000 ft east of the confluence of Many Devils Wash (Plate 3). The position of the siltstone bed on the escarpment drops in elevation gradually from its westernmost exposure to its easternmost exposure, indicating that the Mancos Shale dips easterly at a low angle. The same siltstone bed is exposed in the lower part of Many Devils Wash, where it forms a knickpoint in the wash about 1,200 ft upstream from the confluence with the San Juan River. The determination was made that the siltstone bed in Many Devils Wash was the same as the bed exposed along the escarpment by following semicontinuous outcrops of the siltstone bed from the knickpoint downstream along the walls of the incised wash.

Slight undulations and small beaks in the siltstone bed on the east side of Many Devils Wash about 350 ft downstream from the knickpoint are the site of seepage—the most northerly of such occurrence in the wash. This structure lines up with a subtle topographic swale that strikes about N20E and parallels the incised wash below the knickpoint. This structure provides a likely pathway for ground water north-northeastward along the east side of the wash.

Surveyed elevations of the top of the siltstone bed at various locations indicate by contouring (Figure 4–4) that the strike of the Mancos Shale in the site area is approximately north (varies from an azimuth of 000 to 355). The eastward dip of the Mancos flattens eastward across the site and varies from about 1° just north of the disposal cell to about 0.3° east of Many Devils Wash (Figure 4–4). For the contouring in Figure 4–4, greater validity was given to the observable, surveyed siltstone bed locations than to the siltstone bed elevations derived from borehole lithologic logs.

Deposits of white salts (efflorescent crusts) of variable thickness are present in places on outcrops of Mancos Shale along the escarpment and in Many Devils Wash. Similar salt deposits are present on the surface in the Mancos Shale upland and other areas of low relief on the shale; however, these deposits occur as thin discontinuous veneers of powder. Thicker salt deposits, which occur along the escarpment and in Many Devils Wash often cover the surface, are white with an occasional yellow tinge and are up to 0.25 in. thick. The deposits form when water of high salt content evaporates and the salts precipitate on the surface. Salt deposits on the escarpment are thickest and most extensive where seeps occur. Salt deposits in Many Devils

Wash occur on the wash bottom for several hundred feet upstream from the siltstone bed knickpoint to just above the confluence of the East Fork. Downstream from the knickpoint, salts are deposited along the wash bottom for most of the distance to the San Juan River and along the sides of the wash downstream from the siltstone knickpoint for several hundred feet. Salt deposits extend only about 150 ft upstream on the East Fork. Infrequent rains dissolve the crust, but the crust reappears by evaporation after several days of dry conditions. This was observed in Many Devils Wash on March 28, 2000, when the crust disappeared after approximately a 0.75 in. rain. Dry conditions reestablished the crust within the following week. The composition of the salt deposits is described in Section 4.4, "Geochemistry." Evangelou and others (1984) describe the efflorescence (salt deposits) that commonly occur naturally in the Mancos Shale as containing a mixture of calcium, sodium, and magnesium sulfate evaporite mineral species.

4.2.1.2 Quaternary Material

Unconsolidated Quaternary material was divided into four units for mapping: (1) terrace material deposited by the ancestral San Juan River about 240 ft above the present San Juan River, designated Qt2; (2) terrace material deposited by the ancestral San Juan River about 50 to 60 ft above the present San Juan River floodplain, designated Qt1; (3) sand deposited in the present San Juan River floodplain, designated Qfps; and (4) loess deposited mainly by wind over terrace material, Mancos Shale, and possibly floodplain material, designated Ql.

Older terrace material (Qt2) caps only one small mesa in the site area (Plate 3). This material, about 20 ft thick capping the mesa crossed by Navajo Road N5072, is outwash from a Pleistocene glacial episode in the San Juan Mountains. Ward (1990) mapped the material as Q5.

Terrace material mapped as Qt1 is extensive and forms a prominent surface approximately 50 to 60 ft above the present floodplain of the San Juan River. The terrace is continuous south of the river from the NECA gravel pit westward to the Shiprock High School area (Plate 3). Most of the town of Shiprock south of the San Juan River sits on this terrace, including the disposal cell, NECA yard/old millsite, and NECA gravel pit. Remnants of the terrace occur in the area of the mouth of Many Devils Wash where incision has removed most of the terrace. About 1 mi east of the mouth of Many Devils Wash, the terrace resumes and extends about 1 mi eastward to the escarpment above the Chaco River. The Qt1 terrace is also present north of the San Juan River on top of an escarpment about 1 mi northeast of the disposal cell (Plate 3).

The Qt1 terrace material is typically 10 to 20 ft thick where exposed along the top of the escarpment and is generally mapped by Ward (1990) as Q6. The Qt1 material was deposited as glacial outwash during a period estimated by Tsosie (1997) from 88,000 to 150,000 (late-middle Pleistocene) years ago. The material was deposited during aggradation in a former San Juan River valley; later erosion and downcutting have left remnants of these deposits preserved as strath terraces. Clast-supported deposits of well-rounded gravel, cobbles, and boulders with a silty and sandy matrix compose much of the terrace material. The coarsest part of the deposit is typically at the base, where cobbles 1 ft in diameter are common, and the largest noted were 2.5 ft in diameter. The resistant cobbles and boulders typically consist of metamorphic rocks (quartzite and metaconglomerate) eroded from the San Juan Mountains. Locally mixed with these far-traveled deposits on the terrace are less coarse and more angular debris derived from nearby tributaries.

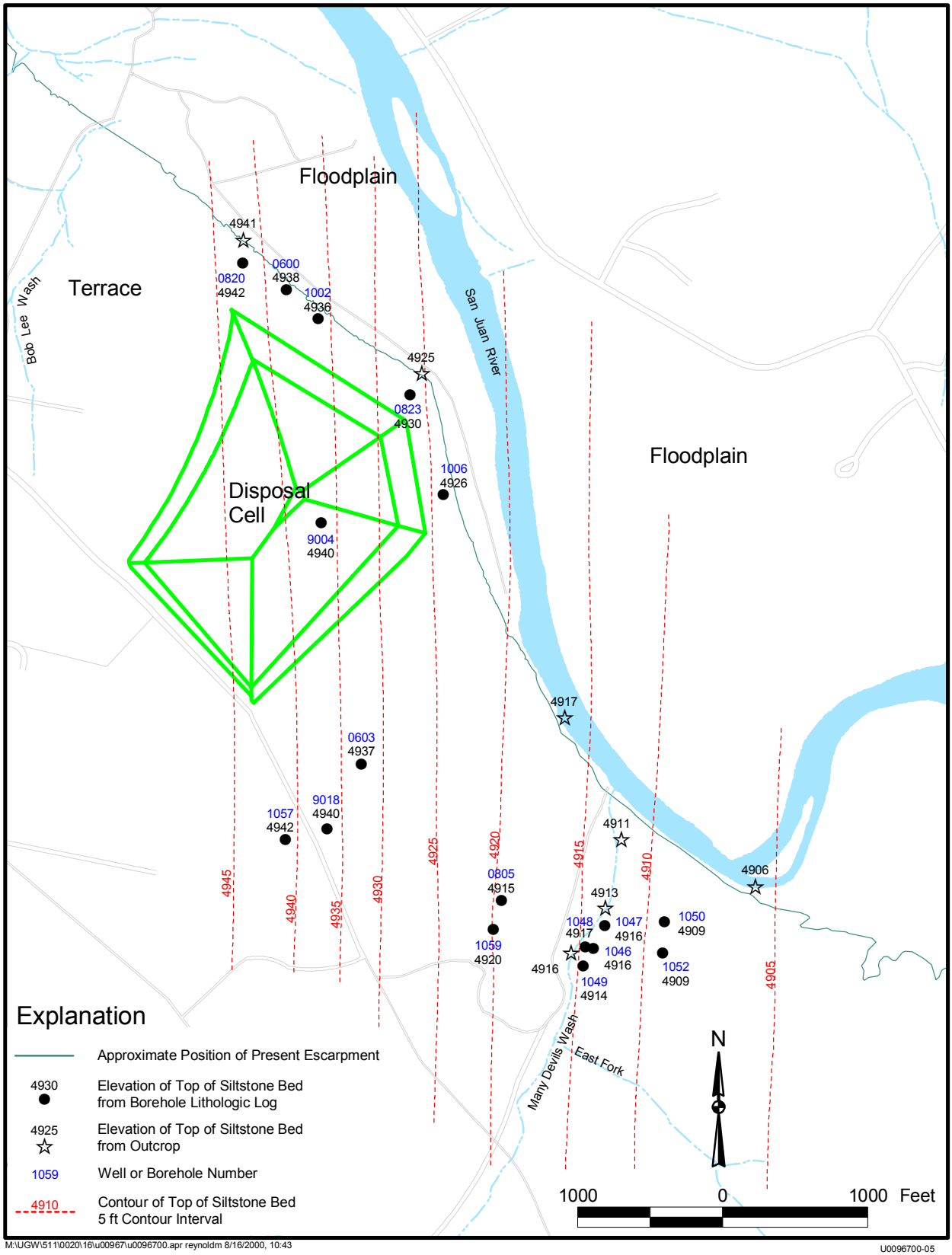


Figure 4–4. Contour Map of Top of Siltstone Bed in Mancos Shale

Alluvial deposits in the present San Juan River floodplain were mapped as Qfps. This designation identifies sand because it is the most common grain size of the material on the floodplain surface. Where undisturbed, the 10- to 20-ft-thick deposits typically consist of at least 5 ft of sand on the surface, underlain by coarser material composed mainly of gravel and cobbles. In some places on the floodplain where flood-scouring (as on the "island" area downstream from the U.S. Highway 666 bridge) or remedial action activities (as on the floodplain just north of the disposal cell) have occurred, the sand has been removed and gravel material is exposed. These areas are generally small and scattered and were not mapped separately. The surface of the floodplain area south of the San Juan River starting about 0.7 mi upstream from the disposal cell is covered largely by sand in stabilized to semistabilized dunes. The coarser material, generally in the basal part of the floodplain deposits, is shown in the cross sections in Plate 4 as Qfpg.

The floodplain deposits are at an elevation of 5 to 10 ft or less above the San Juan River. With one exception, the base of the escarpment forms the south edge of the floodplain deposits south of the river on the site. The exception is in the northwest part of the site just west of the distributary channel of the river (Plate 3) where a subtle rise of 3 to 4 ft defines the boundary of the floodplain. West and southwest of the rise, the area of cultivated fields on the Blueeyes Ranch is designated as a low terrace and is covered by loess. However, it is believed that the floodplain material underlies the loess and extends southward to the vicinity of the irrigation return flow ditch.

The coarse part of the floodplain alluvial material represents glacial outwash deposited during the most recent glaciation in the San Juan Mountains. This late Pleistocene deposition was estimated by Tsosie (1997) as occurring from 16,000 to 70,000 years ago.

Eolian deposits, mapped as loess (Ql), have filled in, draped over, and covered some of the landforms in the site area. The loess material occurs in a band from Many Devils Wash westward and northwestward to the Stokely Elementary School and high school area and to the irrigated farm lands on the low terrace (Plate 3). Except in the Many Devils Wash area, the loess generally contacts (indistinctly) weathered Mancos Shale that forms low uplands to the south. The Mancos Shale uplands become more pronounced as hills in the area just west of the Stokely Elementary School (as at Blueberry Hill). The color of the loess is typically gray-tan on the surface, and it forms a flat surface that slopes gently northward in the area west of the radon cover borrow pit. To the north, the loess-covered sloping surface indistinctly contacts the terrace material (Qt1). West of the radon cover borrow pit, where most of the loess material was removed, the terrace material is present in the subsurface and is covered by a north-thinning wedge of loess.

In the Many Devils Wash area, the tan-colored loess occurs on top of Mancos Shale and consists mainly of silt and very fine grained sand. In places, some thin layers of coarse-grained sand and small pebbles occur, indicating episodes of fluvial deposition. Erosion in the lower part of the wash is actively incising through the loess, leaving distinctive vertically standing remnants (towers) of loess up to 25 ft in height and creating extensive piping structures up to 25 ft in depth. The piping has facilitated gully-head recession southward in Many Devils Wash, where the southernmost incision point is several hundred feet beyond the remains of concrete-and-rock walls constructed across the wash in the early 1930s by federal programs to control erosion.

The distinctive piping and towers in the loess produce a pseudokarst topography, as described by Parker and Higgins (1990). The piping that causes this topography develops in material that has

high contents of smectite clay and salts. Wetting and drying of the smectite clay causes swelling and shrinking, leading to the formation of desiccation cracks that are infiltrated and enlarged by runoff water. High salt content, especially high exchangeable sodium in the soils, also causes swelling when wetted. Mancos Shale, from which much of the loess is derived, has a high salt content and contains large amounts of smectite and illitic clays.

Loess accumulated in low areas along ancestral drainages in locations on the north (or leeward) side of topographic features, sheltered from prevailing southerly winds. In the site area, this occurred primarily north of the Mancos Shale upland, where loess filled the south part of the ancestral San Juan River floodplain (on top of the Qt1 gravel and cobble deposits) after the river had downcut into the area of the present floodplain. Loess also filled in low areas along Many Devils Wash, which at that time had incised through the Qt1 deposits to allow it to drain into the San Juan River. Most of the loess was probably deposited during dry periods in late Pleistocene time, after the Qt1 material was deposited, and as late as the mid-Holocene dry period of 2,800 to 6,000 years ago (Love and Gillam 1991).

Fill material and the covered tailings pile, or disposal cell, have also been mapped in Plate 3. The fill material is mapped along the bottom of Bob Lee Wash, in four locations along the escarpment north and east of the disposal cell where small drainages have been filled, and in one area adjacent to the southwest corner of the disposal cell. Bob Lee Wash fill material was emplaced during and after milling operations; fill in the drainages was emplaced after milling from the mid-1970s to the 1985–1986 period of remediation, escarpment stabilization, and disposal cell construction. Grading and leveling of part of the old raffinate pond area in the 1970s and 1980s created the fill southwest of the disposal cell. Fill material, which may be up to 25 ft thick in the filled drainages, is probably uncompacted and probably does not consist of tailings according to the site completion report (MK-Ferguson 1987) and the radiologic characterization report (Allen and others 1983). However, another report on the geochemical investigation (DOE 1983) of the site indicated that contaminated soil from the ore storage area was used to fill a drainage that went north from the old millsite.

4.2.2 Joint Measurements

Joints (fractures) were investigated to evaluate what effect they might have on movement of ground water through the Mancos Shale and on location of seeps. The investigation focused on the escarpment where Mancos Shale is well exposed between the corner of the escarpment near wells 862 and 863 northwestward to the mouth of Bob Lee Wash. This escarpment area is immediately north of the disposal cell and is the site of seeps 425 through 427. Twenty-four joint orientation measurements were made with a Brunton compass. These measurements of joint strike are shown on Figure 4–5. The dip of all the joints measured was vertical, or within a few degrees of vertical. A rose diagram of joint orientation frequency is presented on Figure 4–6. This diagram shows that the principal joint strike direction is northeast. Tsosie (1997) noted the northeast direction of fracturing and indicated that most of the gullies cutting the escarpment edge were fracture induced.

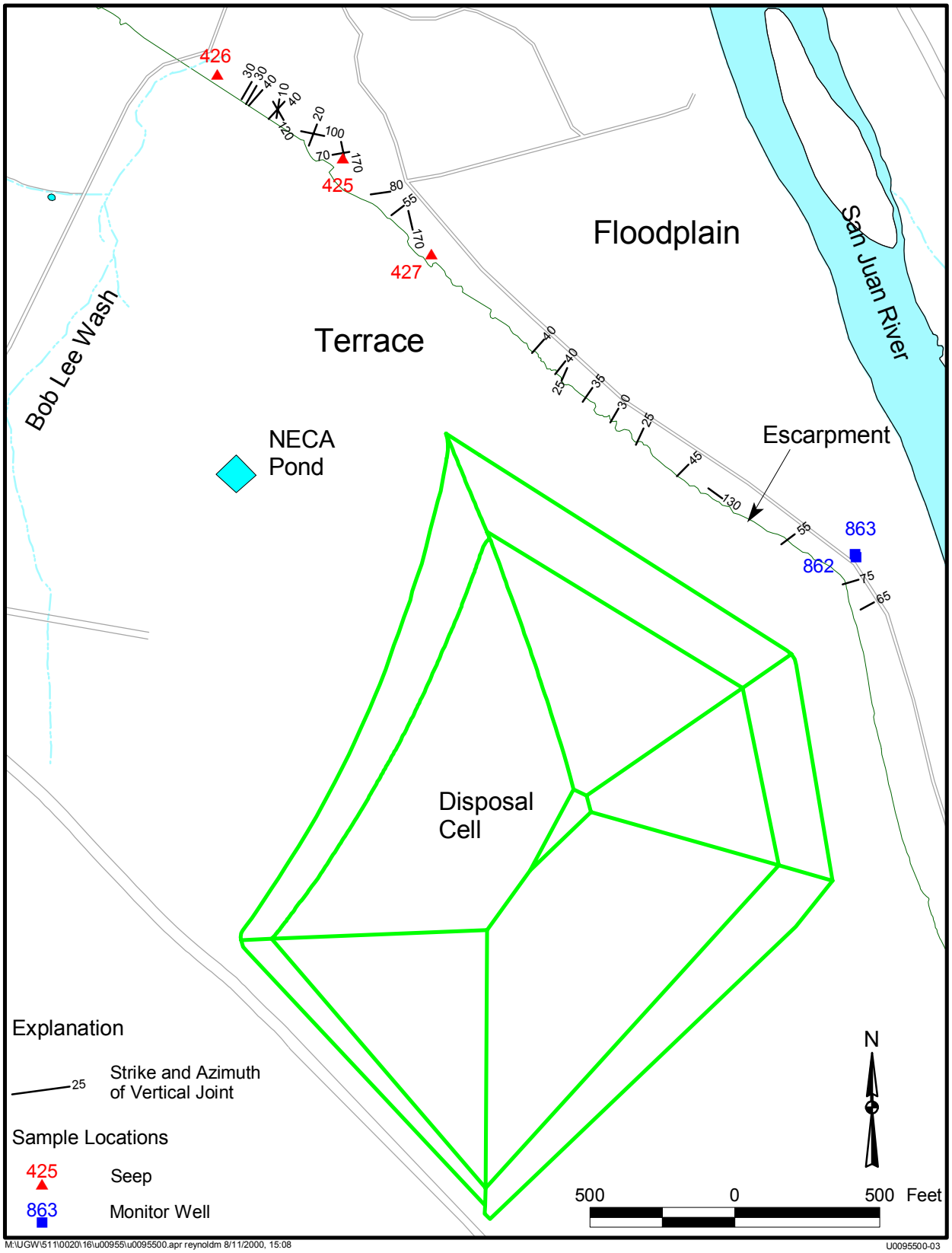
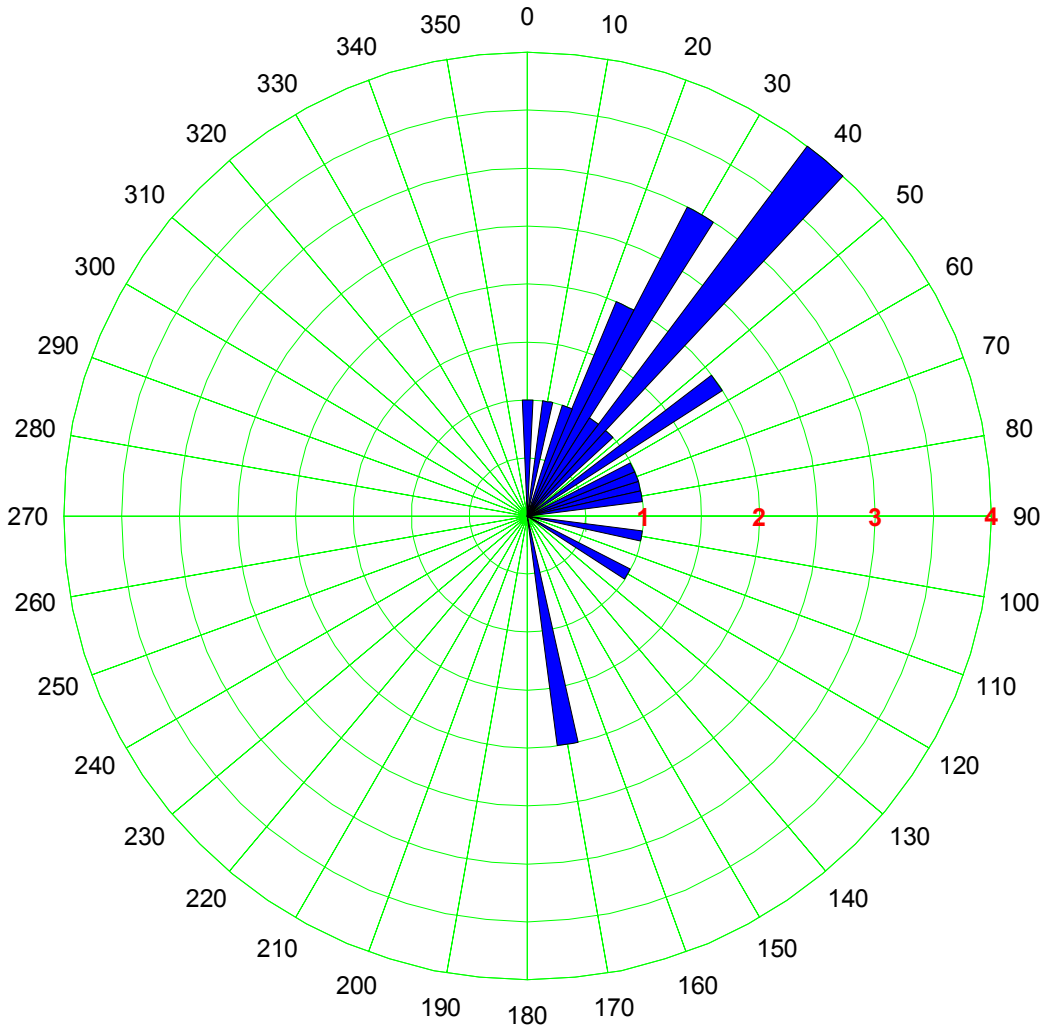


Figure 4–5. Orientation of Vertical Joints Along Escarpment



Explanation

1 Number of Orientation Measurements

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Figure 4–6. Rose Diagram of Joint Orientations (azimuths) for Locations Shown on Figure 4–5

Joints along the escarpment from seeps 425 through 427 and southeastward to the escarpment corner did not appear to be a significant factor in ground water movement. Instead, particularly at seeps 425 and 426, water appears in a less resistant horizontal layer that may represent a more permeable lithology (such as the presence of one or more thin bentonite beds) within the Mancos Shale, or the layer may contain numerous bedding plane fractures that promote water movement. Also, a seep area (786) in Mancos Shale just west of the U.S. Highway 666 bridge is in a less resistant horizontal layer, and water movement along vertical fractures is not apparent.

Joint measurements were made at two other locations along the escarpment; one was east of the NECA gravel pit and the other was west of the U.S. Highway 666 bridge. Joints are vertical in both locations. At the location east of the gravel pit, near salt crust sample location 922, the joint orientation is 035; west of the highway bridge, near seep location 935, joints have orientations of 000, 010, and 035. Ground water expressed as seeps in both of these locations appears to flow along horizontal bedding in the Mancos Shale, probably along a slightly more permeable layer similar to the occurrence at seeps 425 and 426.

4.2.3 Borehole Stratigraphic and Structural Results

Boreholes were drilled during September to December 1998, December 1999, and March–April 2000 to install monitor wells and collect stratigraphic and structural information. Depending on the drilling method and objectives for drilling each borehole, samples of material penetrated were brought to the surface by coring, split-barrel sampling, drill cuttings, and auger returns. Lithologic logs prepared in the field from each of the 62 boreholes drilled from September to December 1998, the 18 boreholes drilled in December 1999, and the 18 boreholes drilled in March–April 2000 were placed into gINT, a computer-generated borehole log system. The gINT logs for all 1998 to 2000 boreholes are presented in Appendix A. Also included in Appendix A are gINT logs for earlier boreholes and monitor wells (active and abandoned) for which lithologic logs are available. Information from both the new and old boreholes was used in this geologic site characterization.

Borehole lithologic information was used to prepare the geologic cross sections (Plate 4), the contour map of the top of the siltstone bed in the Mancos Shale (Figure 4–4), and the bedrock contour map (Figure 4–7). Subsurface characteristics of the Mancos Shale, the Mancos Shale bedrock surface, and overlying units noted as a result of drilling are described in this section.

Mancos Shale has been separated into upper and lower parts by the Gallup Sandstone in this part of New Mexico (Ward 1990). The Gallup Sandstone, present in part of the San Juan Basin to the west and south of the site area, pinches out several miles southwest of the town of Shiprock (Molenaar and others 1996). Northeast of the pinchout, a sporadic extension of this sandy interval has been called the “Stray” sandstone; more recently, this interval was named the Tocito Sandstone Lentil. The Tocito crops out about 4 mi west of the site along the San Juan River, and the unit is present in the subsurface of the site area. No boreholes drilled during site characterization were deep enough to penetrate the Tocito, but its presence and depth are known generally from the lithologic log of artesian well 648 (Appendix A), which was drilled as an oil and gas test to a depth of 1,850 ft from October 1960 to February 1961. The well produces water from the Morrison Formation of Late Jurassic age through a perforated zone from 1,482 to 1,777 ft in depth. Well 648 penetrated the Gallup Sandstone (now termed the Tocito Sandstone Lentil in this area) from depths of 248 to 330 ft. A projection of the east-dipping (about 1°) Mancos Shale westward to the west edge of the site around well 846 would place the depth to the

top of the Tocito at about 150 ft. As shown on cross section E–E' on Plate 4, the depth to the Tocito in the western part of the site is several tens of feet deeper than the approximate 150-ft total depth of well 848. Penetration of the Tocito Sandstone should be avoided, because ground water that may be present in the sandstone would be under artesian conditions.

During the 1998 to 2000 characterization, depth to bedrock (Mancos Shale) was recorded in all the boreholes drilled to sufficient depth on the terrace and floodplain. In addition to the 1998 to 2000 data, bedrock depths from earlier boreholes were also used to prepare the contour map of the bedrock surface shown on Figure 4–7. In cases where bedrock elevations from earlier boreholes differed greatly from 1998 to 2000 borehole bedrock elevations, preference (or weighting) was given to the more recent data in preparation of the bedrock surface map. The bedrock surface was considered as the top of the weathered Mancos Shale. The weathered Mancos Shale is typically 5 to 10 ft thick, but some characteristics of weathering occur as deep as 30 ft in places. Tan-orange limonitic staining that typically occurs on bedding plane surfaces within the uppermost few feet of the Mancos is a distinguishing feature of the soft, weathered shale.

Additional depth-to-bedrock data from the 1998 to 2000 boreholes have provided a different and more complete understanding of the terrace bedrock surface than what was presented in the SOWP, Rev. 0 (DOE 1995). The approximately northwest to southeast 2.3-mi extent of the terrace bedrock surface is shown on Figure 4–7. The map, using a 5-ft contour interval, was based on bedrock data from old and new boreholes. The bedrock surface gradually drops about 90 ft northwestward across the 2.3-mi distance. A buried escarpment bounds the bedrock surface to the south and west and forms the north boundary of the upland area. The approximate location of the buried escarpment is shown on Figure 4–7. The presence of this feature is evident from the difference in bedrock elevations between boreholes 808 and 812 or 806 and 1058. This buried escarpment, about 50 to 60 ft high, is similar to the present escarpment to the north that separates the terrace from the present floodplain. An unusual stratigraphic sequence in well 841 indicates that the escarpment may be vertical to overhanging in places. This borehole penetrated 10 ft of loess, then 16.5 ft of Mancos Shale, below which 23.5 ft of coarse sand, gravel, and cobbles were followed by more Mancos Shale bedrock at 50 ft. The Mancos Shale initially penetrated by the borehole could represent an overhanging cliff at the edge of the buried escarpment, or the shale could be a block of bedrock that fell from the nearby escarpment onto the outwash material in the former San Juan River channel.

Characteristics of the terrace bedrock surface, or strath terrace formed by the ancestral floodplain of the San Juan River, affect ground water movement. The disposal cell sits on an elevated and nearly flat bedrock surface. This low-relief surface extends south-southeast from the disposal cell to the buried escarpment. Wells 603 and 731 are on this surface, which forms a low divide that separates steeply sloping surfaces to the east from gently sloping surfaces to the west. Also, extending westward from the disposal cell area is a low ridge about 1 mi long that is defined by bedrock elevations found in wells 728, 814, and 832. North of this ridge, the bedrock surface drops gradually to the northwest, and south of the ridge is a shallow valley that slopes gently to the west and northwest (Figure 4–7). The south edge of this shallow valley is the buried escarpment. Wells 604, 818, 812, 813, 841, and 1060 are situated in the shallow valley. Borehole 834 and well 1060 are at the west end of the shallow valley; north of this point, the bedrock slope abruptly steepens and the valley appears to extend northwestward to the area of borehole 831.

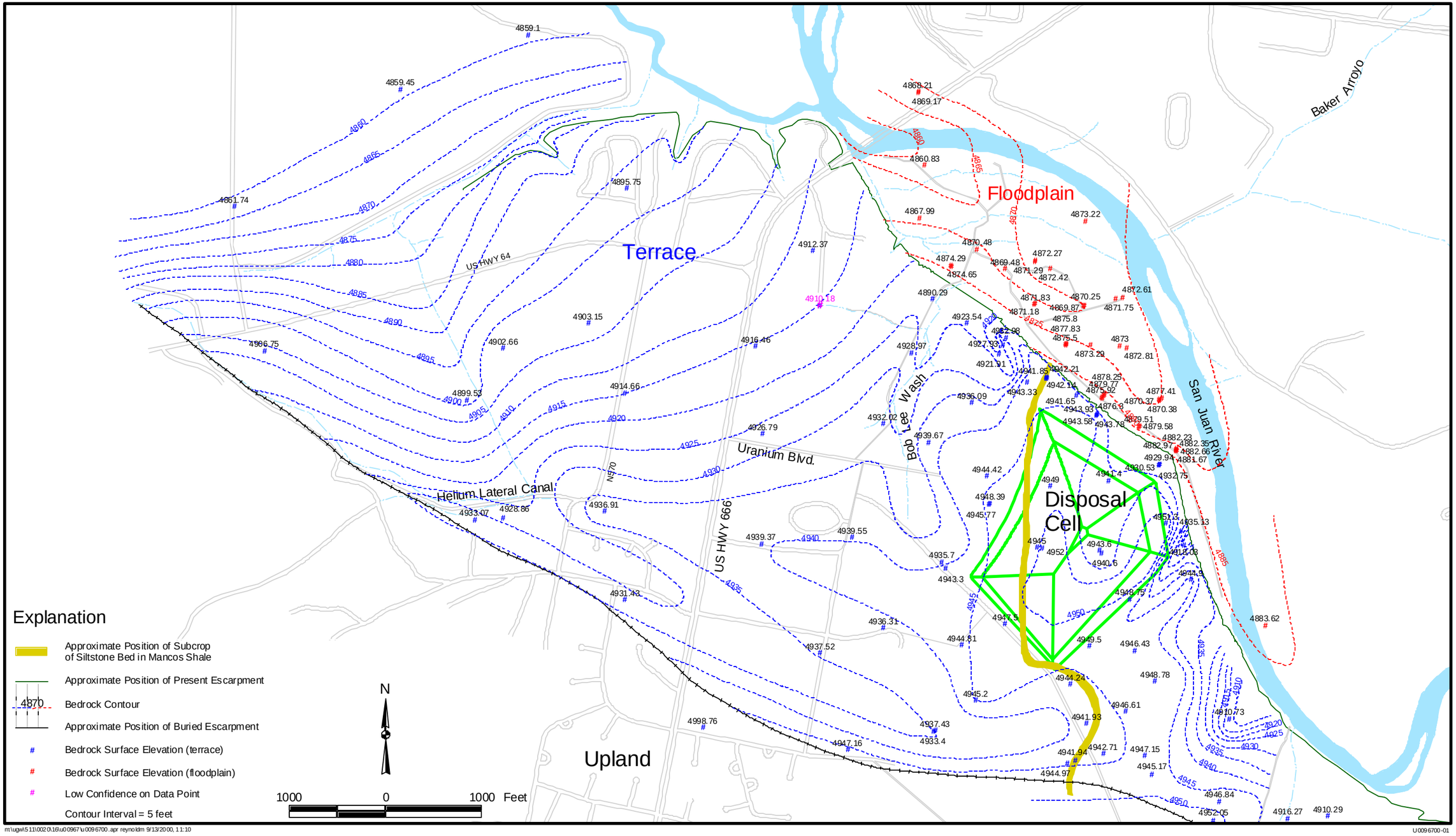

Figure 4-7. Contour Map of the Mancos Shale Bedrock Surface

Ground water laden with raffinate pond effluent during milling (and for years afterward) likely moved south and southwest into the shallow bedrock valley. The flat to gently sloping valley promoted only slow westward movement of this water. A large area of contaminated water is still present in this low bedrock valley area between wells 818 and 841. Ground water from the east end of the raffinate ponds could also have moved southward along the nearly flat bedrock divide. There, in the area of wells 731, 603, and 1058, movement of ground water also could be slow.

Another feature shown on the bedrock surface contour map (Figure 4–7) that affects ground water movement is the approximate position of the subcrop of the 1-ft-thick siltstone bed in the Mancos Shale. This bed dips about 1° eastward, and its subcrop extends across the mostly low relief bedrock surface from the north end of the disposal cell southward to the buried escarpment south of borehole 807 and well 1060. The position of this resistant siltstone bed may be the reason that the relatively flat bedrock surface is present. The orientations of the siltstone subcrop and the high, flat bedrock area are roughly coincident. In addition to providing a resistant lithology to "hold up" the high bedrock area, the siltstone bed provided a relatively low permeability barrier to downward ground water movement east of its subcrop. Ground water east of the siltstone subcrop could percolate down through weathered Mancos Shale until it reached the siltstone bed, then move downdip eastward along this perched layer to seeps along the escarpment (such as expressed at sample location 922) and along Many Devils Wash. Salt crust has been sampled along the damp cliff face at location 922, but sufficient water for a sample has not been available. Ground water in well 1059, just north of the buried escarpment, is a reflection of the pathway of ground water moving through weathered Mancos Shale eastward toward Many Devils Wash.

Several narrow drainages have incised into the bedrock surface north and east of the nearly flat bedrock surface in the disposal cell area. The most prominent of these is Bob Lee Wash; less noticeable are several short, narrow drainages that were filled during remediation in the 1970s and 1980s. The position of axes of these small drainages cut into bedrock are shown on Figure 4–7, on Plate 1, and also on the site geologic map on Plate 3. The three bedrock drainages north and east of the disposal cell provided potential pathways for effluent-laden ground water in areas of the millsite and tailings piles to move down to the floodplain. The first drainage drained the north part of the mill area, and its mouth cuts through the escarpment between seeps 425 and 427. Borehole 1005 and well 1011 probed this drainage in 2000, and its precise location is known from old aerial photographs (Figures 3–1 and 3–4) and a 1960 topographic map. A second drainage is at the corner of the escarpment just north of the northeast corner of the disposal cell. This drainage was probed in 1998 by wells 823, 824, and 825 (the east terrace nest). The third drainage is just east of the southeast corner of the disposal cell and enters the floodplain north and west of well 735. Its head is near the former raffinate ponds. Wells 1006 and 1007 probed this drainage in 2000, and its location is known from a 1960 topographic map.

Ten boreholes drilled in 1998 and seven boreholes drilled in 2000 on the floodplain north of the disposal cell penetrated the alluvial material and contacted the top of the Mancos Shale bedrock. These boreholes provided a more complete understanding of the floodplain bedrock surface. The floodplain bedrock surface map shown on Figure 4–7 is different from the bedrock surface map presented in the SOWP, Rev. 0 (DOE 1995). The present interpretation on Figure 4–7 is simplified and shows a shallow swale that parallels the escarpment (about 500 ft north of it). The swale, which represents an ancestral channel of the San Juan River, is bounded on the north by a

low ridge. The edge of the ridge may have as much as 10 ft of topographic relief in places, as shown in the area of the cluster of wells 858, 859, and 612. From the bedrock surface map presented on Figure 4–7, the mainly subtle bedrock topography does not appear to present barriers to a normal northwestward movement of ground water through the floodplain.

Terrace material (Qt1) overlying the Mancos Shale is typically about 20 ft thick. As shown in Plate 4, the terrace material thickness varies from less than 10 ft at wells 831, 844, and 846 to about 35 ft at well 818. Terrace material appears to be the thickest along the ancestral channel of the San Juan River just north of the buried escarpment (Plate 4, cross section A–A'). Thickness of the terrace material around well 835 is about 30 ft. This area may be the site of another ancestral river channel. Near the escarpment and in the millsite area, the terrace material is only about 10 to 15 ft thick. This lesser thickness is probably the result of removal of some material during remedial action.

Sandy material, shown in the cross sections on Plate 4 as terrace sand (Qts), overlies the terrace material in several places in the subsurface in the south and west parts of the site. This sandy material, not exposed on the surface, occurs east and west of U.S. Highway 666 in different hydrogeologic settings. East of the highway, it occurs in wells 812 and 813 and in borehole 807 and well 1057. At these eastern locations the sand is brown, fine to medium grained, and about 5 ft thick. This sandy layer was not found in wells 818 and 604, so it is uncertain if the sand present at borehole 807 and well 1057 extends as a continuous layer westward to the area of the wells 812 and 813. The sand in these eastern locations is dry and is about 20 ft above the ground water surface in the terrace material or weathered Mancos Shale.

West of U.S. Highway 666 sandy material occurs in wells 833, 838, 844, and 1060, and in borehole 831. At these western locations, the sand is yellowish brown to grayish brown and is from 4 to 11 ft thick. The sand in this western area around the Diné College construction tract is probably continuous, and the ground water surface is either in the lower part of the sand or just below in the terrace gravel material. The sand in both locations east and west of the highway overlies the coarser-grained terrace material and was deposited during a low-energy regimen of the ancestral San Juan River before the river abandoned its terrace location and established its course in the present floodplain area.

Loess covering much of the terrace area typically overlies either the terrace gravel material or sandy material. In the low terrace area at the far northwest part of the site, loess covers floodplain gravel. The loess material is composed mainly of silt, with minor amounts of very fine-grained sand, clayey silt, and sandy clay. A finer-grained variant of the loess occurs in the lower terrace area where wells 831, 836, and 843 penetrated about 5 ft of sandy clay or clayey silt in the lower part of the loess sequence. The silt is mottled in places, calcareous, and contains a few thin, white layers, possibly caliche. Light yellowish brown is the most common color of the loess and brown and light brownish gray also occur.

Thickest loess occurrences are in the south part of the terrace area just north of the buried escarpment. Well 812 is in such a setting and penetrated 34 ft of loess. Similar thicknesses likely occur to the northwest in the high school area, and at least 25 ft of loess was removed from parts of the radon cover borrow pit. A backhoe pit dug in the lowest, northwest part of the radon cover borrow pit found only 3 ft of loess remaining. Loess is thinner in the terrace background area where wells 800 through 803 penetrated only about 5 to 10 ft of it. The ground water surface is

below the loess in all terrace locations except the low terrace area, where the lower part of the loess is saturated (wells 836, 837, and 843).

Loess directly overlies Mancos Shale in the Many Devils Wash area and along the north edge of the upland area. In mid-December 1999, 18 boreholes were drilled through the loess in the Many Devils Wash area to evaluate the eastward extent of contaminated ground water and to evaluate if ground water was present in the loess on Mancos Shale in the wash area south of road N5072. Except for boreholes 1048 and 1049, completed as monitor wells just east of the wash, no ground water was found in the boreholes. Loess in the dry boreholes typically increased in moisture content with depth (but not approaching saturation) until reaching the weathered Mancos Shale, which was dry. Loess up to 33 ft thick was present in the bench area east of Many Devils Wash (Plate 4, cross section E–E') and was present up to 35 ft thick south of road N5072. These dry boreholes in the loess and the underlying Mancos Shale indicate that ground water also is not present in the areas of the east and west forks of Many Devils Wash.

Alluvium in the San Juan River floodplain north of the disposal cell consists mainly of two types of material: (1) a lower, coarse-grained unit composed of sand, gravel, and cobble-sized material representing glacial outwash overlain by (2) a finer-grained unit consisting of silt, sand, and minor gravel. The coarse-grained unit is shown in cross sections (Plate 4) as Qfpg, and the finer-grained unit is shown on the geologic map (Plate 3) and cross sections as Qfps. The coarse-grained unit is thicker, and in some places in the eastern part of the floodplain (wells 853, 854, 858, 862, and 863) it is the sole alluvial unit present. The absence of the finer-grained unit in some of the eastern part of the floodplain may be a result of removal during surface remediation.

In 1998, 13 additional boreholes were drilled into the floodplain alluvial material north of the disposal cell, and seven more were drilled in 2000. Grab samples of the alluvial material were taken, typically at 5-ft intervals, during drilling of most of the boreholes. Lithologic description of this material and sampled intervals are in the gINT logs for each borehole in Appendix A. The alluvial material in the floodplain north of the disposal cell reaches as much as 24 ft in thickness; the typical thickness was 15 to 20 ft. Alluvial material of similar composition and thickness was present in boreholes for the three wells (850 through 852) installed in the floodplain background area, where 16 ft of sandy gravel was overlain by 4 ft of sand.

Four boreholes completed as terrace monitor wells in 1998 penetrated fill material. The fill at these locations was placed in small drainages near the terrace edge in the mid- to late-1970s. Wells 823 through 825 in the east terrace cluster penetrated about 26 ft of fill in an east-trending drainage (Plate 2 and cross section G–G' on Plate 4). Approximately 22 ft of fill was penetrated at well 827 (cross section B–B' on Plate 4), which was drilled in a northwest-trending drainage (Plate 3) that drained millsite effluent to a pond on the floodplain. During borehole drilling it became apparent that filled drainages had been penetrated at both borehole locations because the expected depth to bedrock was greatly exceeded. The existence and location of the drainages was later confirmed by their positions shown on a 1960 topographic map. The composition of the fill material in both drainages was similar to that of the terrace material (Qt1) adjacent to the drainages.

In April 2000, four additional boreholes were drilled into two of the filled drainages. The purpose was to complete the holes as monitor wells in ground water in fill material or weathered Mancos Shale (or both) and to determine if the drainages are acting as pathways for ground water movement from the terrace to the floodplain. Because it was uncertain if tailings were

present in the fill material in the drainages, the cored material recovered during rotasonic drilling in the drainages was checked with a gamma scintillometer. No gamma activity was measured above the background range of approximately 100–150 counts per second, indicating tailings were not present.

Borehole 1005 was drilled first and reached bedrock at a depth of 13 ft, indicating that the buried drainage channel probably was not intercepted. The hole was filled and abandoned and another borehole was drilled to the south between borehole 1005 and existing well 827. In this second borehole, bedrock was contacted at 16 ft, and ground water was found deeper in weathered Mancos Shale. The hole was completed as well 1011. For the three boreholes drilled in the northwest-trending drainage, well 827 penetrated to the thickest fill material and is closest to the axis of the drainage.

Boreholes 1006 and 1007 were drilled into the filled drainage just east of the southeast corner of the disposal cell. Bedrock contact was at 25 ft in borehole 1006, and it was completed as a well deeper in weathered Mancos Shale. This well is located near the edge of the drainage. Bedrock contact was at 42 ft in borehole 1007—likely near the axis of the filled drainage. This well was completed in ground water near the contact of fill material and weathered Mancos Shale.

Core (NX size) was recovered from Mancos Shale in six boreholes during the 1998 drilling. Four of the boreholes cored were from each of the terrace and floodplain well nests (wells 820 and 823 and wells 860 and 862). The other two boreholes cored were in the terrace background area (wells 800 and 802). Detailed description of the rock core is included in the gINT lithologic log (Appendix A) of each cored borehole. The labeled core is boxed by borehole and stored at the DOE–GJO core-storage area at the Cheney Repository site.

Coring in both the well nest and terrace background boreholes was conducted in weathered and unweathered Mancos Shale to evaluate the presence of ground water and its relation to fracturing and stratigraphic features. The amount of fracturing in the core, recorded in the core log, was the basis for selecting intervals to be packer tested for hydraulic conductivity in the terrace and floodplain well nest boreholes. A summary of the results of coring from a hydrogeologic perspective follow.

The Mancos Shale is generally light gray to dark gray and is calcareous throughout, but especially so in the lighter-colored, coarser-grained (silty) layers. Thin claystone layers up to several inches thick are common and are the darkest (dark gray); they swell when brought to the surface and appear to be excellent aquitards. Traces of carbonaceous material and finely disseminated pyrite were identified. Contorted bedding caused by bioturbation is common in these shales deposited in a shallow shelf environment. Wavy and planar bedding is also common. Fossils occur sporadically; the largest are flattened pelecypod shells preserved as white, fibrous, aragonite layers. Weathered Mancos Shale in the shallowest parts of the cored intervals is dark yellowish brown to light olive gray and contains some limonite staining and white calcite and gypsum fracture fillings. Fracturing decreases with depth, and bedding plane fractures are the most common. Only a few inclined or vertical fractures were identified; all were closed with no evidence of ground water movement along them.

The 1-ft-thick calcareous siltstone bed penetrated by coring in terrace background well 803 (Appendix A) is believed to be the same siltstone that crops out in Many Devils Wash and along the escarpment north and east of the disposal cell. The presence of this siltstone bed at an elevation of 4,937 ft indicates that the dip of the siltstone (and the Mancos Shale) is at a low

angle westward at well 803. This occurrence of the siltstone bed implies that a shallow, synclinal axis is present west of well 803 and east of Many Devils Wash. From the terrace background area, the Mancos Shale rises eastward on the flank of the Hogback anticline.

4.2.4 Geophysical Survey Results

Geophysical surveys were conducted in February 1996 by Geraghty and Miller, Inc. (DOE 1996c) on the floodplain north of the disposal cell and on the terrace in areas adjacent to the disposal cell. These surveys were conducted to address data needs identified in the SOWP, Rev. 0 (DOE 1995). Four other geophysical surveys were conducted from mid-1995 to mid-1996 on the floodplain north of the disposal cell. These surveys were conducted with EM 31 and EM 38 instrumentation, and the results show different configurations of the contaminant plume corresponding to different levels of the San Juan River (Tsosie 1997).

The Geraghty and Miller work consisted of electrical conductivity surveys with EM 31 instrumentation on the floodplain and EM 34 instrumentation on the terrace; seismic refraction surveys were also conducted in the floodplain. The floodplain EM 31 survey was intended to locate sulfate and nitrate contamination. Results of this survey showing areas of high conductivity (DOE 1996c, Figure 3) on the floodplain correspond closely to the present understanding of the configuration of the contaminant plume. The siting of well 854 was based on the position of the high-conductivity area shown in this EM 31 survey (DOE 1998d). Analyses of ground water samples from this well and from backhoe trenches in the nearby area verified that the contaminant plume extends northward across the floodplain to the San Juan River in the well 854 area. The EM 34 survey on the terrace was conducted to identify contaminant concentrations and bedrock fractures that might act as conduits for ground water movement. Results of this survey indicated that few fractures were present and none were of importance. Areas of high conductivity were identified adjacent to the disposal cell and NECA yard and extended southeast through the NECA gravel pit; a low conductivity area identified south of the disposal cell is probably the result of a thick layer of loess and terrace material covering the contaminant plume.

The refraction surveys were conducted to determine bedrock topography and its relationship to areas of high conductivity (high contaminant concentrations). Results indicated that bedrock depressions generally coincided with areas of high conductivity (DOE 1996c). However, present interpretation of bedrock topography based on additional borehole data does not indicate a correlation of high levels of contaminants with bedrock depressions.

4.3 Hydrology

This section presents the hydrologic characterization of the UMTRA Project Shiprock disposal site and the surrounding area. The surface water part of this section presents an overview of the San Juan River and its importance as a water supply in the region, as well as a description of surface water that comes from flowing well 648, seeps and springs that emerge from the escarpment, irrigation return flow, 1st and 2nd Washes, and wetlands on the floodplain at the mouth of Bob Lee Wash.

The ground water portion of the section describes the floodplain alluvium, the terrace alluvium, and the bedrock flow systems. The floodplain alluvium is a potentially significant ground water resource because it is hydraulically connected to the San Juan River. The hydrologic conditions

of the floodplain alluvium were investigated as part of this project. An aquifer pumping test was performed in the floodplain to obtain an estimate of the transmissivity of the system, and a water balance was developed for the floodplain as a whole. Numerical flow-and-transport modeling of the alluvial aquifer was also performed to evaluate compliance strategies for the system.

The terrace alluvium was described previously as a limited use ground water system (*Federal Register* January 11, 1995, p. 2863). The assumption of limited use formed the basis for the site conceptual model for a number of years, and no concerted effort was made to test its validity. The 1998 to 2000 investigation was geared toward (1) evaluating if the limited use designation is appropriate for the terrace alluvium; (2) assessing if ground water exists in terrace background areas near the disposal cell; (3) prospecting the upland areas south of the disposal cell to determine if they contain ground water and, if not, then delineating the boundary between the dry upland areas and the saturated terrace alluvium; (4) identifying the discharge areas for the terrace alluvial flow system; and (5) evaluating the hydrologic interaction between the terrace alluvium and the floodplain alluvial aquifer.

4.3.1 Surface Water

This section presents descriptions of the various surface water bodies and estimates of discharge and water use for those systems.

4.3.1.1 San Juan River

The San Juan River has a drainage area of approximately 12,900 square miles (mi^2) upstream from the town of Shiprock. Discharge records for the San Juan River at Shiprock are nearly continuous since February 1927. A river stage recorder (09368000) operated by the USGS is located on Shiprock's alternate-water-source intake structure about 300 ft east (upstream) of the U.S. Highway 666 bridge along the north side of the river (Plate 1). The river gauge was established at this location in 1995; formerly, the gauge was located about 3 mi west (downstream) of Shiprock. Data from the river gauge indicate that before 1963 extreme low and high flows ranged from less than 8 cubic feet per second (cfs) to about 80,000 cfs, respectively. After construction of the Navajo Reservoir (located 78 river mi upstream of Shiprock) was completed in 1963, the minimum and maximum flows moderated to about 80 cfs and 15,000 cfs, respectively. Average flow in the San Juan River at Shiprock is 2,175 cfs (Stone and others 1983). Figure 4–8 presents a hydrograph of the San Juan River at Shiprock. A stilling well has also been established at location 899 (Plate 1).

The Chaco River drains more than 4,000 mi^2 and empties into the San Juan River upstream about 2 mi east of the Shiprock site. It drains many areas in the San Juan Basin that contain coal and uranium (Stone and others 1983). Flow in the lower reach of the Chaco River ranges from 10 to 30 cfs during nonstorm-flow periods. Much of the flow is reported to be effluent from the Four Corners Power Plant, about 12 mi southeast of the Shiprock site (Stone and others 1983). Water quality standards have been promulgated by the Navajo Nation for surface waters within the reservation. The San Juan River is classified as a domestic water supply suitable for primary and secondary human contact, for livestock and wildlife watering (including migratory birds), for irrigation, and for a cold-water fishery. Consequently, stringent water quality standards are applicable to the San Juan River at Shiprock. These standards are described in terms of their significance to the Shiprock UMTRA site in Section 7.0, "Ground Water Compliance Strategy."

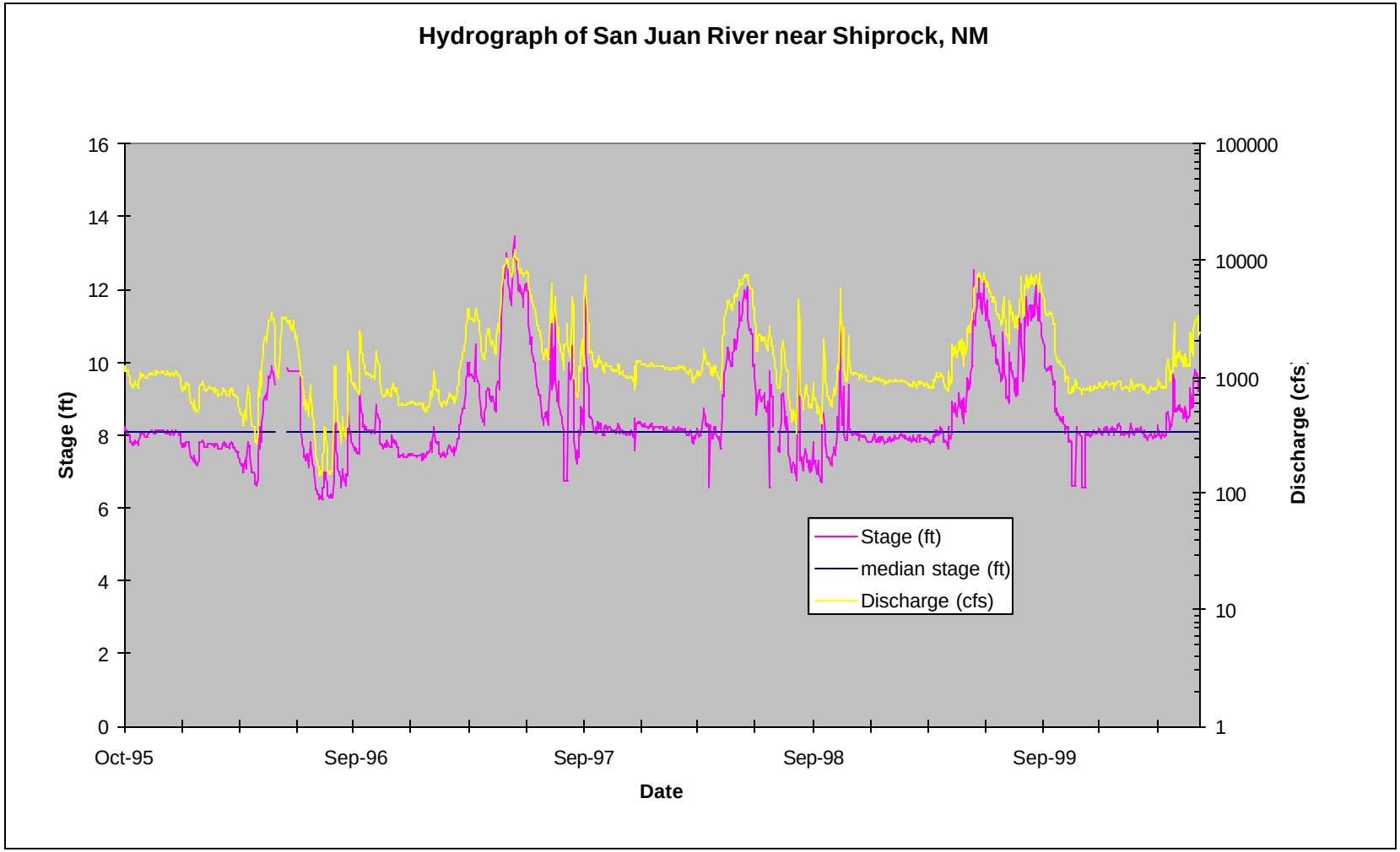

Figure 4-8. Hydrograph of the San Juan River Near Shiprock

Water quality is monitored by USGS at river gauge 09368000, the location of which is now shared with Shiprock's water intake structure. The water is also monitored by NTUA in conjunction with requirements of the Safe Drinking Water Act (SDWA). DOE monitors the San Juan River both upstream and downstream of the Shiprock millsite under the auspices of the UMTRA Project.

Table 4–4 presents results of quarterly water quality monitoring performed by USGS. These results indicate that for the varied flow rates reported, concentrations of the selected analytes are below the water quality standards for domestic and primary human-contact designated uses in the surface water quality standards of the Navajo Nation (Navajo Nation EPA Water Quality Program 2000). In conjunction with the analytical results of DOE monitoring, the results also indicate that millsite-related contaminants do not pose an immediate threat to the quality of the alternate water supply (see Section 4.3.1.2) at Shiprock. DOE's analytical results are discussed in Section 4.4.

Table 4–4. Surface Water Quality Parameters for Selected Analytes Monitored at U.S. Geological Survey Gauge 09368000 at Shiprock

Date	Discharge (cfs)	Sulfate (mg/L)	TDS (mg/L)	Nitrogen as NO_2 +NO_3 (mg/L)	Arsenic Total (mg/L)	Selenium Total (mg/L)	Uranium (mg/L)
Nov 17, 1994	996	170	410	0.410	0.002	0.0001	0.0022
Mar 02, 1995	1,460	170	392	0.390	Not Analyzed	Not Analyzed	Not Analyzed
May 03, 1995	4,210	65	199	0.090	0.002	<0.001	0.00068
Aug 08, 1995	1,280	100	260	<0.050	Not Analyzed	Not Analyzed	Not Analyzed

Notes: mg/L = milligrams per liter

4.3.1.2 Water Supply

NTUA maintains the town of Shiprock's water supply and has several potential sources of water available, all of which rely on the San Juan River. From Shiprock upstream toward Navajo Reservoir Dam, these sources are:

- **San Juan River at Shiprock (alternate water source)**: The Shiprock alternate water source consists of an octagonal (in plan view) intake structure set in the river channel next to the north bank of the river (Plate 1). The structure has four slide gates, each at a different elevation to allow operators to adjust intake elevation in response to changes in river stage. The capacity of the intake structure is calculated to be 2.6 million gallons per day (MGD). The 1997 maximum projected peak production for Shiprock was 2.6 MGD, and 3.1 MGD is projected by the year 2013 (Molzen-Corbin & Associates 1993). Therefore, the capacity of the intake structure is projected to be insufficient to supply the entire peak demand. The single biggest operation and maintenance problem with the Shiprock water intake is inadequate facilities to remove the suspended river sand (Molzen-Corbin & Associates 1993).

- **Navajo Irrigation Authority (NIA) Canal**: Hogback Canal (Plate 1) is an irrigation ditch designed to deliver 143 MGD to various tribal agricultural users in the San Juan River

Valley; the canal is operated and maintained by NIA. The intake for the canal is located 11 mi upstream from Shiprock on the north bank of the San Juan River. Canal deliveries usually occur between April and September. Chemical water quality in the canal is assumed to be similar to water pumped from the water intake structure; however, the suspended load is probably much lower. Hogback Canal is projected to be capable of meeting all municipal requirements through the year 2013 with only a 3 percent loss of carrying capacity (Molzen-Corbin & Associates 1993).

- **City of Farmington:** The City of Farmington has been selling water to NTUA through a purchase agreement that began in 1967. This is the principal source of municipal water for the town of Shiprock. The original purchase agreement had a 10-year term with options to renew for additional 10-year periods. The terms of the original purchase agreement were that NTUA would purchase at least 0.7 MGD and that the maximum quantity delivered on any day would be 3.0 MGD. The cost of the water is adjusted annually to reflect changes in the City of Farmington's actual cost basis. As of 1993, the City of Farmington believed that the contract with NTUA had expired but that there was enough surplus treatment capacity to enter into another long-term agreement. The 1993 cost of treated water was $0.98 per 1,000 gallons (Molzen-Corbin & Associates 1993).

- Other potential San Juan River diversions include the Navajo Agricultural Products Industries (NAPI) Irrigation Canal and the Proposed Navajo-Gallup Pipeline Project (Molzen-Corbin & Associates 1993). Both of these are additional potential sources of water supply for the town of Shiprock.

4.3.1.3 Bob Lee Wash

Discharge from flowing-well 648 accounts for almost the entire surface water flow in Bob Lee Wash. The flow at the mouth of the wash has not been measured with a weir, but during the winter of 1999, discharge from well 648 was measured with a flow meter at approximately 64 gpm. It is reasonable to assume that discharge at the mouth of the wash is equal to well 648 discharge during the winter; however, a small pond (Plate 1) constructed in the fall of 1999 along the outflow ditch from well 648 intercepts some of the water before it flows into Bob Lee Wash. During the summer, evapotranspiration may reduce the flow slightly en route to its discharge point at the mouth of the wash. Upstream of the confluence with well 648 discharge, seeps in Bob Lee Wash support saltgrass vegetation but no stream flow, even in winter. These seeps are contaminated with millsite effluent and issue forth from weathered Mancos Shale and terrace alluvial gravel, as described in Section 4.3.2, "Ground Water."

A wetland about 5 acres in size is located near the mouth of Bob Lee Wash. Discharge from the wetland flows slowly west to northwest along an abandoned distributary channel on the floodplain. Ultimately, the discharge from the wetland, and any intercepted ground water discharge, emerges from the floodplain near surface sampling location 894 on the San Juan River.

4.3.1.4 Many Devils Wash

Surface water in Many Devils Wash is confined largely to the northernmost 1,400 ft of the channel. The southernmost, or highest, occurrence of water in the channel appears to be spring flow that is controlled by a 1-ft-thick siltstone bed in the Mancos Shale. In the vicinity of sample locations 889 and 916, where the siltstone bed is exposed in Many Devils Wash (Plate 1), the

soil and shale bedrock are covered with a whitish efflorescence that occurs along both east and west banks of the wash. However, as described in Section 4.3.2.2, "Terrace Ground Water System," the source of water in the wash is quite likely derived from the saturated terrace alluvium and underlying weathered Mancos Shale to the west. Borehole data from wells 1057, 1058, and 1059 show that the siltstone bed dips eastward beneath the saturated terrace alluvium south and southeast of the disposal cell, as described in Section 4.2.3, "Borehole Stratigraphic and Structural Results," and shown on Figure 4–4. Discharge at the mouth of Many Devils Wash measured in March 1999 was 0.3 gpm; because this discharge measurement was made at a period of low evapotranspiration, the total spring fed discharge into Many Devils Wash is estimated to be approximately 0.3 gpm. This discharge empties directly into the San Juan River.

4.3.1.5 Additional Washes

Three additional washes drain the terrace area west of the U.S. Highway 666 bridge. These washes have no formal name and are designated from east to west as 1st, 2nd, and 3rd Washes, respectively (Plate 1). The 1st and 2nd Washes each support minor surface water discharge that appears as spring flow near the base of the terrace alluvium. Water from these washes discharges to the distributary channel of the San Juan River west of the U.S. Highway 666 bridge. In winter 1999, the baseflow was estimated to be approximately 1.5 gpm in 1st Wash and about 0.2 gpm in 2nd Wash.

4.3.1.6 Seeps and Springs

The escarpment along the San Juan River west of the mouth of Many Devils Wash and east and west of 1st Wash contains numerous active seeps and springs that issue from the Mancos Shale. The seepage flux is minor and normally manifests itself as damp zones along the cliff face. White efflorescent crust at other locations, that are now dry, suggest that seepage along the cliff face has been more common in the past.

Spring-fed flow is also apparent at several other locations, particularly at seeps 425 and 426 where discharges totaling approximately 1 gpm have been measured by bucket and stop watch. Minor seeps (that have not been measured) flow at locations 427, 922, and 936. A spring (location 935) near the mouth of 1st Wash has a flow estimated at about 1.5 gpm. Also, a seep with comparable flow (location 786) is located under the U.S. Highway 666 bridge.

Numerous springs and ponds exist in an area north of Shiprock High School. Surface water sample location 942 was established to collect surface water chemistry data from a spring in this area, and surface locations 1063 and 1064 were established to sample water from the small ponds in this area. The ponds were apparently formed while gravel was being extracted from the terrace. Depressions created by the extraction of gravel are now ponds. The surface flows originating near locations 942, 1063, and 1064 enters the irrigation return flow ditch, which flows east northeast toward the distributary channel.

4.3.2 Ground Water

This section provides information about the occurrence and general characteristics of ground water near the Shiprock UMTRA Project site, such as sources, flow rates, flow directions, volumes stored in the ground water systems, and the results of tests performed on the aquifers. Water level data from floodplain and terrace wells are presented in CD-ROM in Appendix D.

4.3.2.1 Floodplain Alluvium

The floodplain alluvial aquifer is north of the disposal cell in the millsite floodplain area between the San Juan River and the base of the escarpment. It consists of unconsolidated medium- to coarse-grained sand, gravel, and cobbles that are in direct hydrologic communication with the San Juan River. The gravel and cobble fraction is composed of detrital material that was transported as glacial outwash derived from the San Juan Mountains. Borehole evidence indicates that the sandy gravel unit is overlain in most places by a layer of silty sand several feet thick. Both the sandy gravel and silty sand layers appear to be laterally continuous.

A simple depositional facies model provides a description of the hydrostratigraphy of the floodplain alluvial aquifer. The basal gravel (or channel gravel) was deposited as the river migrated northward from the base of the escarpment to its present position. During its migration, older alluvial sediments to the north were eroded and a new layer of coarse sediment was deposited. These processes resulted in a continuous layer of channel gravel, sand, and silt that was deposited on a scoured bedrock surface. Periodic flood events later deposited sand and silt on top of the gravels, resulting in the present alluvial stratigraphy. This depositional model is similar to the fluvial-floodplain facies model of Mackin (1937), which was later described in Leopold and others (1964). According to this model, the unstratified channel gravel is the coarsest material that moved along the stream channel. Because the channel material is uniformly coarse grained, directional and spatial contrasts in hydraulic conductivity are expected to be relatively minor.

Plate 1 shows the locations of monitor wells and well points in the floodplain alluvial aquifer. Borehole logs and water level data for the 34 wells completed in the floodplain alluvium indicate that the average saturated thickness is 12.4 ft. The hydraulic gradient in the floodplain aquifer ranges from approximately 0.002 to 0.004. Figure 4–9 is a contour map of the water table for the floodplain alluvial aquifer and the terrace system.

Monitor wells in the floodplain alluvium were installed in five drill periods: 1984, 1993, 1998, 1999, and 2000. Consequently, the longest record of water levels dates back to 1984; however, these earlier water level data are sparse. Figure 4–10 presents the hydrographs of the wells with water level records dating back to 1984. It also presents (in the bottom figure) the hydrograph for well 735 that was installed in 1993. The hydrographs contain a partial-duration plot of river stage and show that the aquifer responds to fluctuations in San Juan River levels. Figure 4–11 contains plots of continuous ground water elevations collected with automatic data recorders.

Boundaries of the ground water flow system may be described as time-varying head where the alluvium contacts the San Juan River and as head-dependent flux where the alluvium contacts the base of the escarpment. Surface water, originating as discharge from flowing well 648, enters the floodplain alluvial aquifer near the mouth of Bob Lee Wash. The point of discharge for this water has varied over the years. As shown on Figures 3–4 through 3–7, it flowed northeast in 1962 and by the late 1970s or early 1980s was channeled eastward to Bob Lee Wash. A water user is presently channeling all of the outflow from the ditch into a small pond (Plate 1). The pond leaks considerably, and infiltration from the pond discharges onto the floodplain just west of the mouth of Bob Lee Wash. Some of the outflow water infiltrates the terrace material east of the small pond and flows into Bob Lee Wash just above sample location 662. The contribution from well 648 is the major source of water to the floodplain and dominates the hydrodynamics of the floodplain.

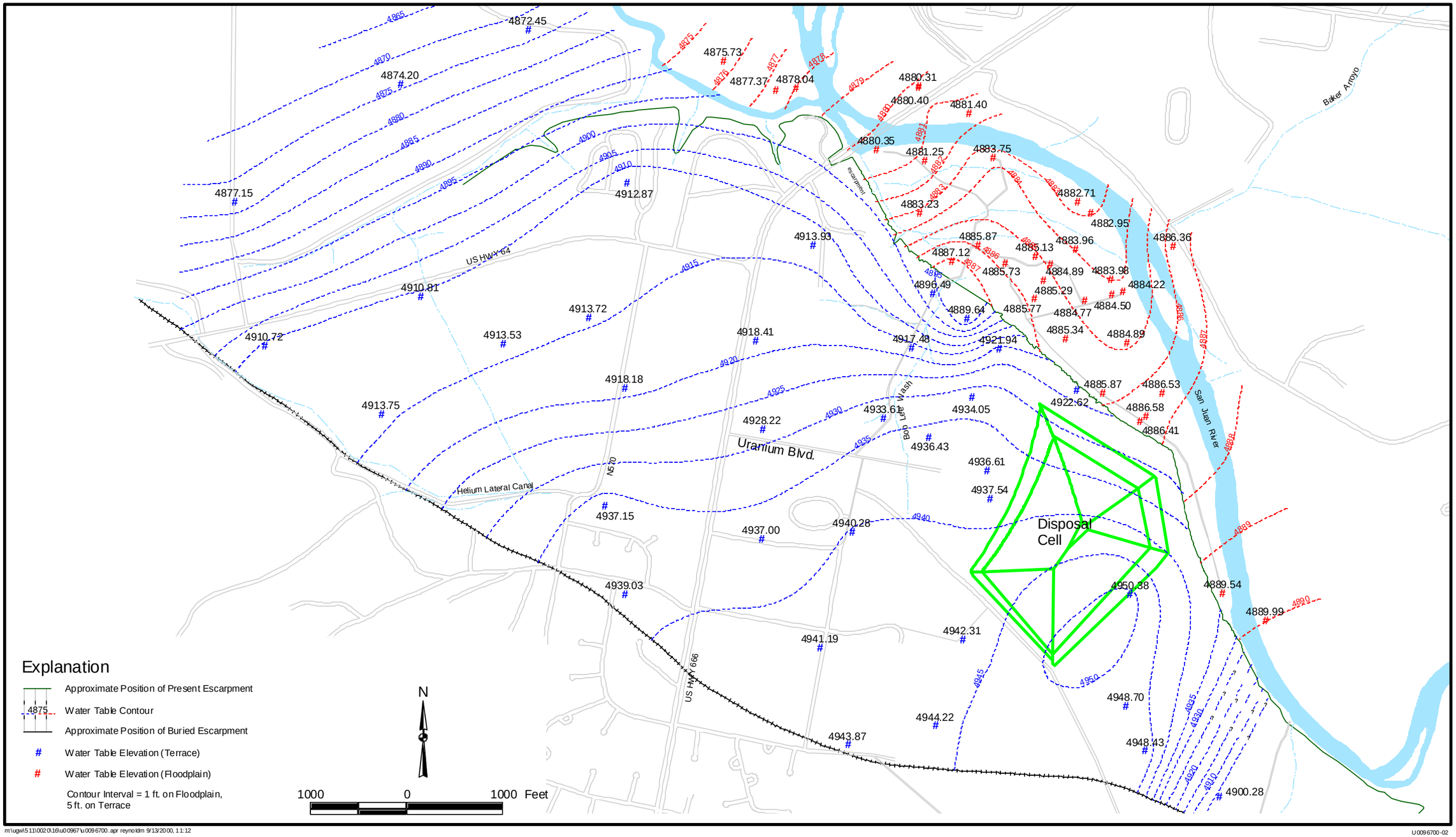

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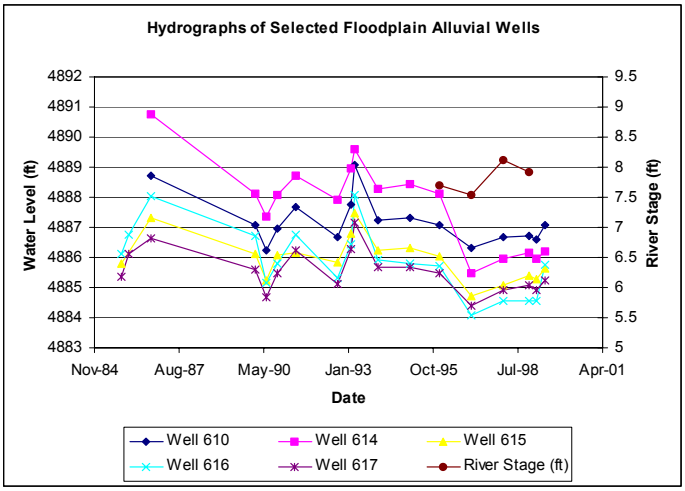
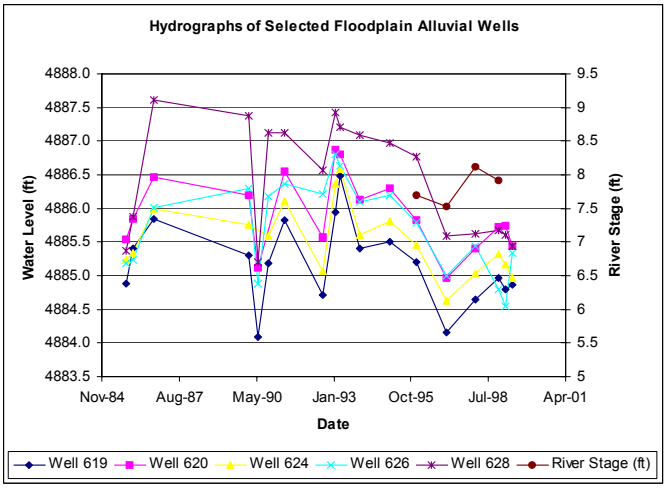
Figure 4-9. Approximate March 1999 Contours of Equipotential Surface for Both Floodplain Alluvial Aquifer and the Terrace Ground Water System, Shiprock, New Mexico, UMTRA Site





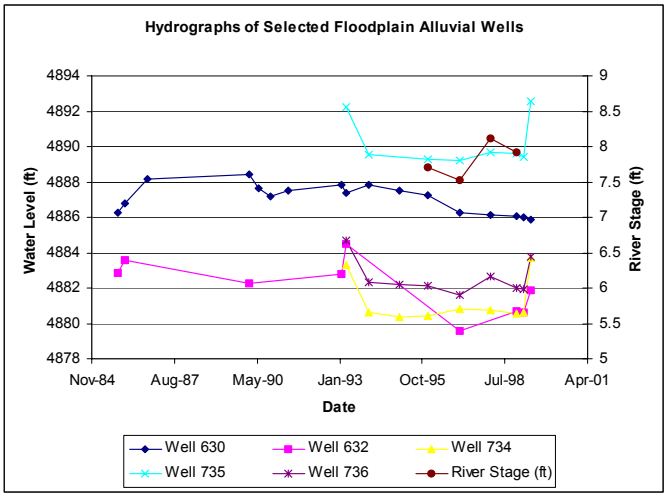


Figure 4–10. Hydrographs of Selected Floodplain Wells, Shiprock Site

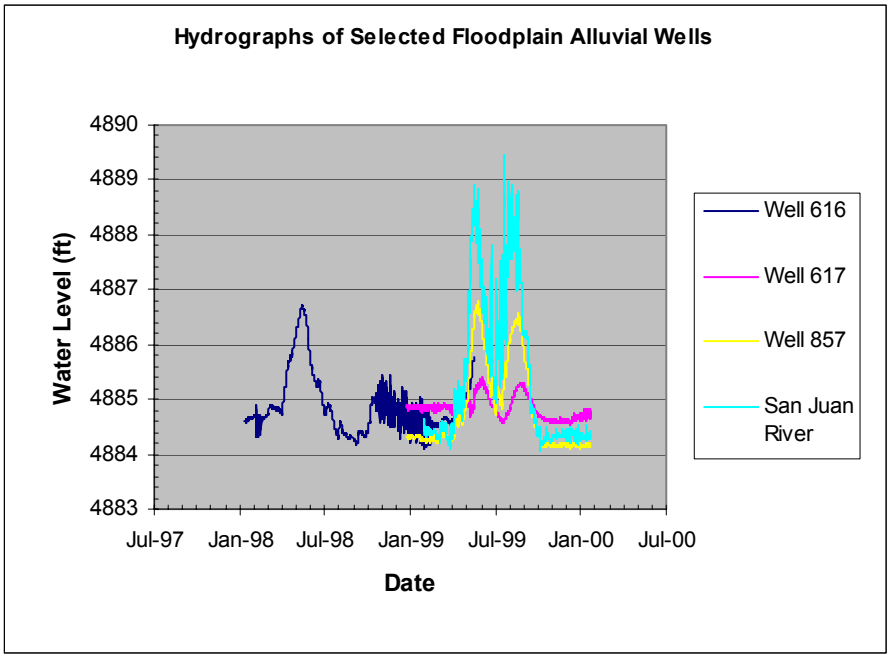


Figure 4–11. Hydrographs of Selected Floodplain Wells, Shiprock Site

The floodplain is also recharged with San Juan River water and infiltration of precipitation and runoff. Recharge from the river enters the floodplain alluvial aquifer along its southeastern end (or panhandle), and discharge exits the aquifer along its northern edge. Ground water in the floodplain alluvium presently supports the growth of phreatophytic vegetation, especially along the abandoned distributary channels that exist there. Before well 648 was drilled, and before milling operations, the floodplain alluvial aquifer is believed to have been entirely recharged by the San Juan River and by infiltration of precipitation and runoff. The floodplain itself was sparsely vegetated because overbank flows scoured the land surface in most years during spring runoff (see the 1935 aerial photograph, Figure 3–1).

Ground water discharge from the terrace ground water flow system has been observed in the form of springs and seeps along the face of the escarpment. Preferred pathways for ground water migration in the Mancos Shale are believed to be the zones where most of the ground water discharge occurs. It is likely that more zones of preferred ground water migration are in the Mancos Shale and contribute discharge to the floodplain alluvium.

Aquifer Pumping Test Results

Figure 4–2 and Figure 4–3 show the location and generalized cross sections for the aquifer pumping tests performed in the floodplain alluvium. The pumping well for the test was well 858, which was pumped at a rate of 60 gpm for 18 hours. Observation wells 859 and 612, located 13.8 ft and 30.4 ft from well 858, respectively, were monitored with electronic pressure transducers during the test. A recovery test was begun immediately after the pumping stopped.

Figure 4–12 presents the drawdown-versus-time records for the aquifer tests in the floodplain alluvium. The transmissivity measured during the pumping phase was between 1,100 and 1,400 square feet per day (ft^2/day); during the recovery test it ranged from 2,100 to 2,400 ft^2/day. The average of these data is approximately 1,800 ft^2/day. Saturated thickness in the area of the test is approximately 16 ft. Therefore, the hydraulic conductivity, defined as the transmissivity divided by initial saturated thickness, is computed to be 110 ft/day.

Water Balance

The water balance for the floodplain comprises the following components: (1) inflow from the San Juan River, (2) inflow that is recharge from precipitation and runoff, (3) inflow from well 648, (4) inflow from the terrace ground water system via the Mancos Shale, and (5) outflow to the San Juan River. Table 4–5 presents a summary of the water balance for the floodplain alluvial aquifer. The approximately 5-percent difference between estimated inflows and outflows is probably equivalent to the potential error in the water balance components. The water balance indicates that about 70 percent of the ground water in the floodplain alluvial aquifer originates as flow from artesian well 648. Discharge from the well enters the floodplain near the mouth of Bob Lee Wash and accounts for 60 percent of the ground water stored in the floodplain alluvial aquifer. Inflow from the San Juan River accounts for approximately 20 percent of the water in the aquifer, recharge from precipitation accounts for approximately 10 percent, and discharge from the terrace ground water system via the Mancos Shale accounts for approximately 20 percent. Outflow from the aquifer occurs mainly as discharge to the San Juan River. Figure 4–13 illustrates the locations of the various flow components of the water balance.

Table 4–5. Water Balance for the Floodplain Alluvial Aquifer at the Shiprock Site

Flow Component	Inflow (ft^3/day)	Outflow (ft^3/day)
1: Inflow from San Juan River	3,600	0
2: Inflow of Recharge	2,600	0
3: Inflow from Well 648	12,320	0
4: Inflow from Terrace Ground Water System via Mancos Shale	3,600	
5: Outflow to San Juan River	0	19,400
Total	20,300	19,400

Evapotranspiration occurs in the floodplain alluvial aquifer, as evidenced by the wetland area near the mouth of Bob Lee Wash and the abundant phreatophytic, salt cedar vegetation. This component exists during the growing season (April through October) and is virtually absent during the remainder of the year. Evapotranspiration is not quantified explicitly in the water balance because it is accounted for implicitly in the recharge term.

Component 1: Inflow from the San Juan River

Inflow from the San Juan River is estimated graphically using the water table contour map (Figure 4–13) in conjunction with Darcy's law. The map shows that the easternmost section of the aquifer is dominated by inflow from the San Juan River. At its widest point, the southern section of the aquifer is approximately 900 ft wide. The transmissivity (T) of the alluvial aquifer

is approximately 2,000 ft^2/day (MACTEC–ERS 1999b). The water table map indicates that the hydraulic gradient is approximately 0.002.

Volumetric inflow (Q_{in}) from the San Juan River is

$$Q_{in} = (2{,}000\ ft^2/day) \times (900\ ft) \times (0.002) = 3{,}600 \text{ cubic feet per day } (ft^3/day).$$

Component 2: Inflow due to Recharge of Precipitation and Runoff

Annual precipitation in the Shiprock area is approximately 7 in. It is assumed that inflow due to precipitation and runoff accounts for approximately 30 percent of the total. The surface area of the floodplain alluvial aquifer is 124 acres (5,401,440 ft^2). Therefore, the volumetric recharge to the aquifer is 2,600 ft^3/day. No explicit measurements of natural recharge are available for the site.

Component 3: Inflow from Well 648

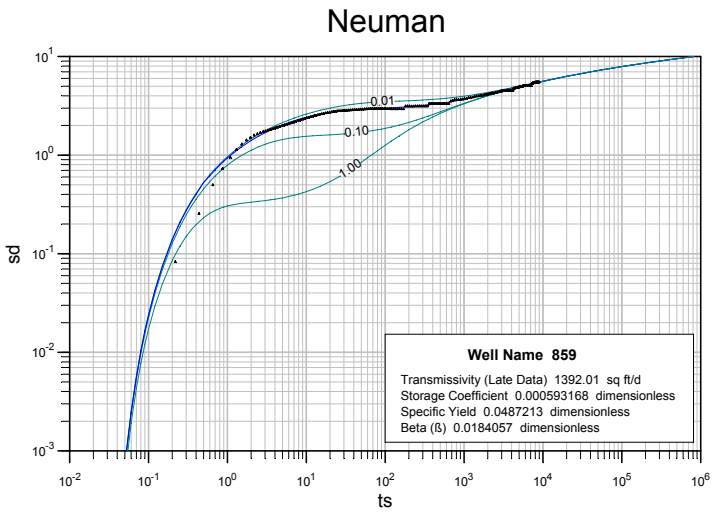
Discharge from well 648 was measured as 64 gpm (12,320 ft^3/day). It is assumed that transit losses are negligible and that essentially all the flow from well 648 discharges to the floodplain near the mouth of Bob Lee Wash. This occurs in spite of the fact that a water user is presently storing water in a small pond that is filled by flow from well 648 along its outflow ditch.

Component 4: Inflow from the Terrace Ground Water System via Mancos Shale

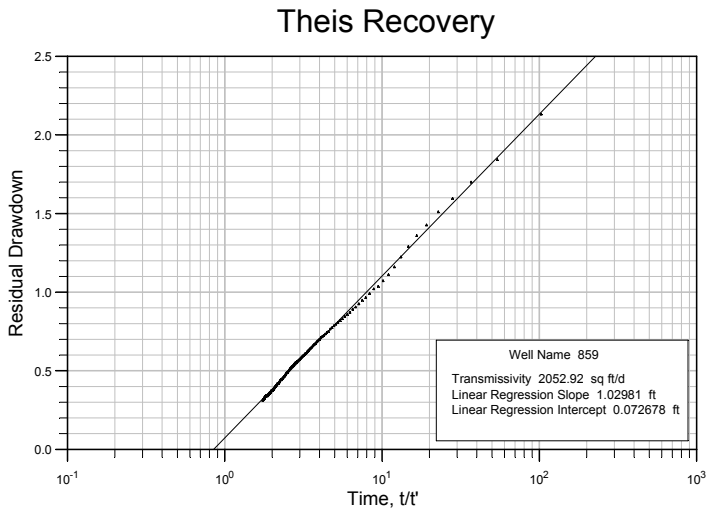
The presence of this component is inferred on the basis of numerical flow modeling. Seeps and springs are present at places along the edge of the escarpment; however, the portion of the flow hidden from view is believed to be primarily responsible for the elevated contaminant concentrations in the floodplain alluvial aquifer. The numerical modeling results indicate that approximately 3,600 ft^3/day of ground water is being contributed to the floodplain aquifer through the Mancos Shale. This flow component carries with it drainage of residual moisture and contamination from the disposal cell.

Component 5: Outflow to the San Juan River

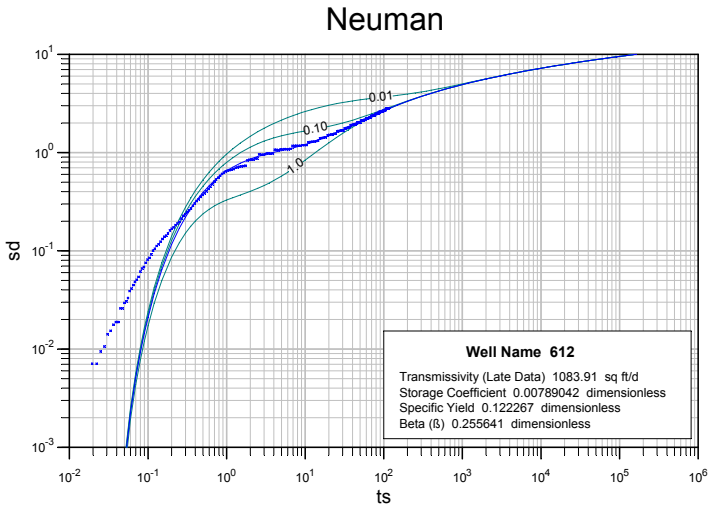
Outflow to the San Juan River is the primary mode of discharge from the floodplain alluvial aquifer. Outflow is estimated graphically from the water table map in combination with Darcy's law. A schematic depiction of flow components for the alluvial aquifer illustrates the discharge to the San Juan River (Figure 4–13). Summing the individual discharge components from the aquifer results in a total estimated discharge to the San Juan River of 19,400 ft^3/day.



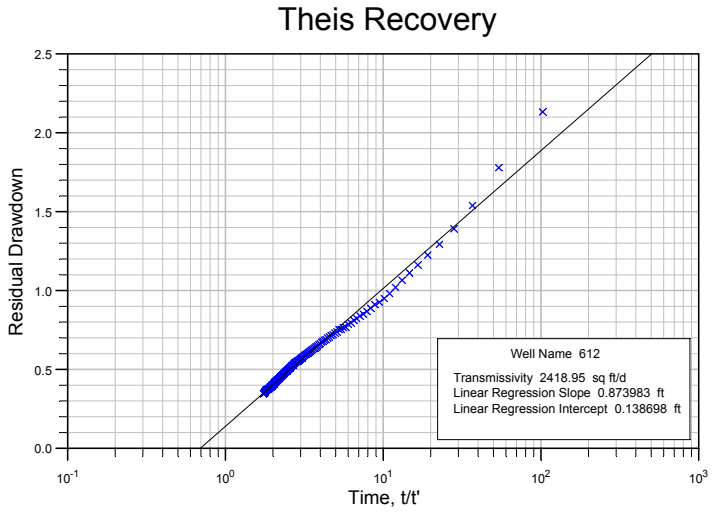
(a) Drawdown in Relation to Time for Observation Well 859



(b) Residual Drawdown in Relation to Dimensionless Time for Observation Well 859



(c) Drawdown in Relation to Time for Observation Well 612



(d) Residual Drawdown in Relation to Dimensionless Time for Observation Well 612

Figure 4–12. Aquifer Test Data Collected While Pumping From Well 858 at 60 gpm

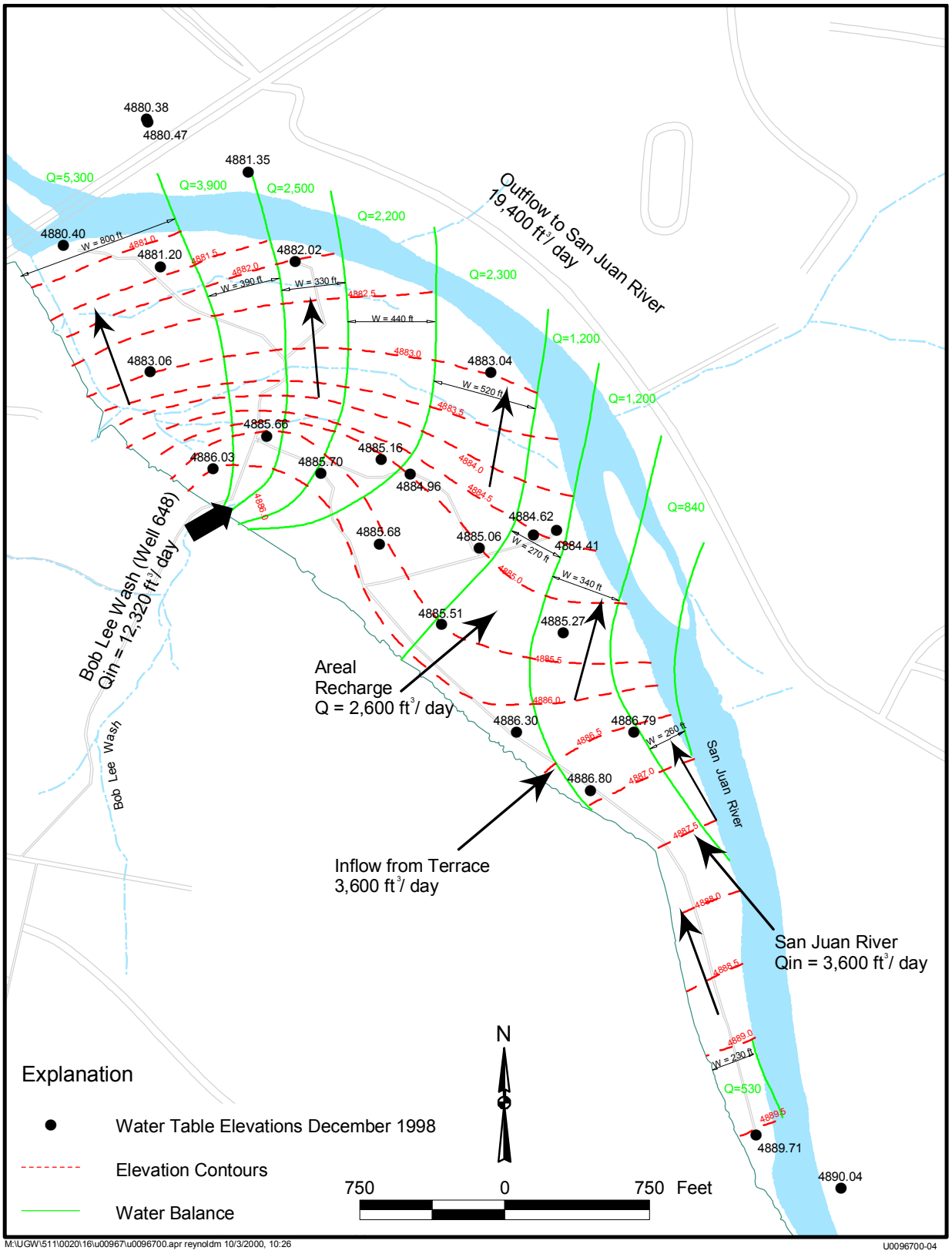


Figure 4–13. Generalized Depiction of Flow Components for Alluvial Aquifer, Shiprock Site

Volume of Water in the Floodplain Alluvial Aquifer

The volume of water stored in the alluvial aquifer is estimated by multiplying the average saturated thickness (12.4 ft) by the surface area of the aquifer (5,401,440 square feet [ft^2]) and by the assumed porosity of the alluvium (0.30). The result, expressed to three significant figures, is 20.1 million cubic feet (ft^3) (150 million gallons).

4.3.2.2 Terrace Ground Water System

Aerial photography from 1935 (Figure 3–1) of the Shiprock millsite area prior to existence of the mill shows that the terrace region was extremely arid. There were no visible sources of natural recharge, no evidence of seepage along the escarpment, and no cottonwood trees or other vegetation in areas that would have supported them under natural circumstances if ground water discharge was available. Because the photos were taken before the existence of flowing well 648, no perennial surface water was evident in Bob Lee Wash. The Helium Lateral irrigation canal, south of the San Juan River and west of the disposal cell, was also absent; consequently, the only potential source of water for a terrace aquifer south of the San Juan River was infiltration of precipitation. The original mantle of eolian silt covering the terrace area is believed to have been instrumental in restricting recharge and favoring the generation of runoff, especially just south of the disposal cell, where the silt attains a thickness of almost 30 ft. Drilling data from the 1998 to 2000 period indicated that the loess is dry even in present conditions when anthropogenic water is present in the underlying gravel. The terrace gravel unit likely received little to no recharge under pristine conditions and is hypothesized to have been unsaturated.

In contrast to the 1935 observation, more recent aerial photographs and field observations indicate that during the time of milling operations at the site, large quantities of water were being pumped onto the terrace to process the uranium ore. Evaporation ponds and raffinate ponds near the mill were full of water, flowing well 648 was discharging ground water from the Morrison Formation, irrigation water was being conveyed to the terrace west of the disposal area, and discharge was visible in seeps along the escarpment and in the ephemeral washes. Figure 3–4 indicates that human activities on the terrace by 1962 had in large measure created the sources of water that are now part of the terrace ground water system. In addition to these obvious sources of ground water, there are probably additional sources that are hidden from view and difficult to quantify. These include leaking water lines, domestic septic systems, and infiltration from leaking sewerage lines.

To further evaluate the possibility that ground water in the terrace is an anthropogenic ground water system, an analog site with comparable geologic and hydrologic features was studied on an adjacent terrace about 1 to 2 mi east-southeast of the disposal cell (see Plates 1 and 2). Test wells 800 through 803 were drilled on the analog terrace site. No water was found either in the terrace gravel section or in the upper part of the Mancos Shale in these test wells. This evidence further supports, but does not prove, the hypothesis that the terrace near the disposal cell was dry prior to milling, irrigation, and other human activities.

Water Level Measurements

Figure 4–14 shows the results of continuous water level monitoring in selected terrace wells. Uranium milling at the site began in 1954 and ended in 1968. Because the mill was only in operation for 14 years, and 20 years elapsed before ground water measurements began, the

decline in the assumed ground water mound was not captured with the ground water measurements performed for the UMTRA Surface Project. The hydrographs in Figure 4–14 reveal slight fluctuations in water levels with time. The fluctuations appear to be seasonal responses to recharge of precipitation.

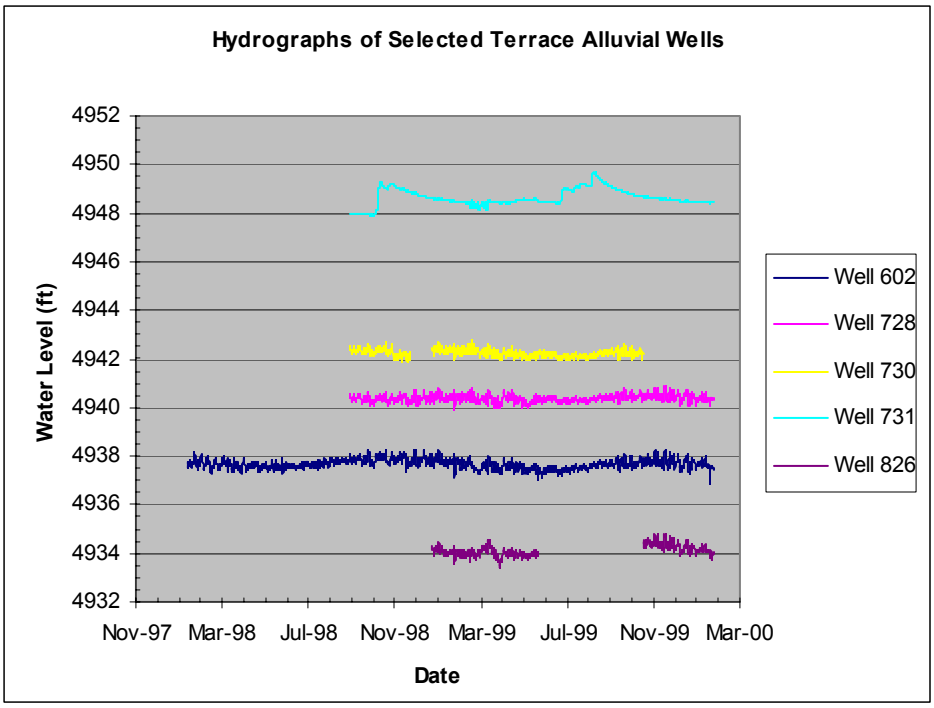
Figure 4–9 presents a water table map for the terrace ground water system based on the most recent (February 2000) water level measurements at the site. The principal source of ground water on the terrace south of the disposal cell appears to be the disposal cell itself. Discharge from the disposal cell appears to be directed toward Bob Lee Wash and the escarpment. Water stored in the terrace system south of the disposal cell appears to occupy a buried ancestral river channel, which eroded a swale in the Mancos Shale, and flows toward the northwest along the axis of the channel. The gentle hydraulic gradient in the area south of the disposal cell may be a reflection of the gentle slope of the bedrock surface (Figure 4–7). Figure 4–15 presents a map of the saturated thickness in the alluvial portion of the terrace ground water system. The map shows that the thickest area of saturation south of the disposal cell is along the axis of the ancestral river channel. Elsewhere to the south and immediately west of the disposal cell, saturation in the alluvial material is generally less than 2 ft thick, or the saturation occurs below in the weathered Mancos Shale. West of U.S. Highway 666, the saturated thickness in the alluvial material increases rapidly and reflects the irrigation water added to the system through the Helium Lateral Canal.

Source and Volume of Mill-Related Ground Water

No records were found that would indicate the exact amount of water usage during milling. The only reference that was located indicates that in the uranium circuit “approximately 270 gpm of pregnant solution are contacted with an average of 27 gpm of organic” (Merritt 1971). This reference suggests that water usage was at least 270 gpm. Merritt further states (p. 422) that the treatment rate was about 300 tons of ore per day. The approximate water balance for the terrace system during the time of milling can be reconstructed to estimate the volume of mill-related water that may be present in the terrace ground water flow system. The RAP for the Shiprock site (DOE 1985) indicates that the surface area of evaporation ponds at the site was about 20 acres.

From these data it is possible to estimate a water balance for the disposal cell during milling: The infiltration rate into the ground = (feed rate to the ponds) – (evaporation rate) – (runoff rate to floodplain alluvium). Data required to complete this estimate are:

- Water flow to evaporation ponds (270 gpm).
- Approximate pan evaporation rate for the area (70 in. per year) (Stone and others 1983). This evaporation rate, adjusted using an average pan-evaporation coefficient of 0.70 (Dunne and Leopold 1978), results in a pond evaporation rate of approximately 49 in. per year. No additional correction was made for the dissolved salt concentration in the pond.
- Surface area of evaporation ponds (20 acres).



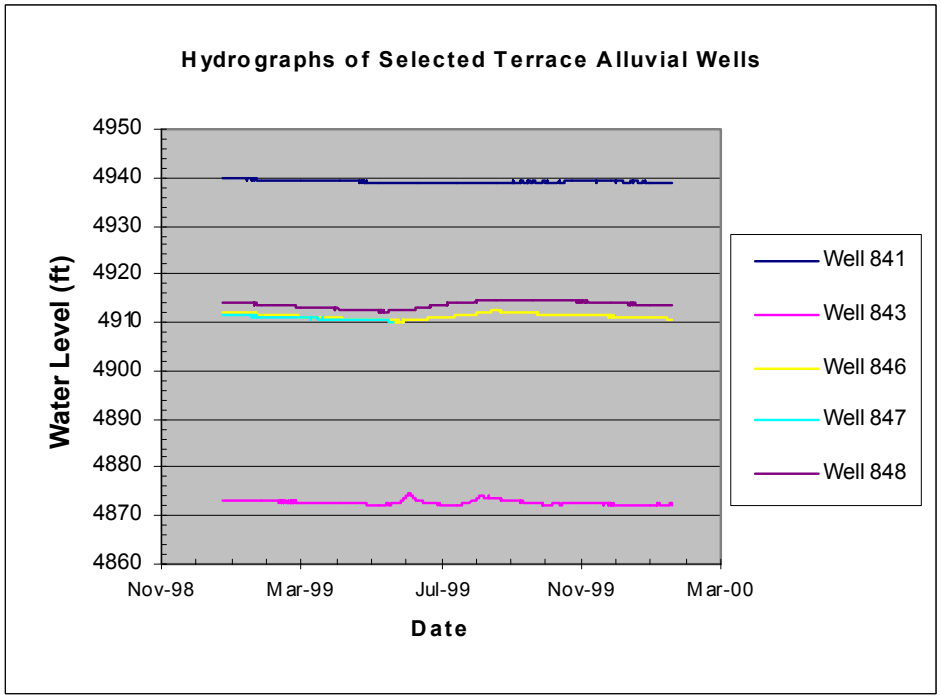


Figure 4–14. Hydrographs of Selected Terrace Alluvial Wells at the Shiprock Site

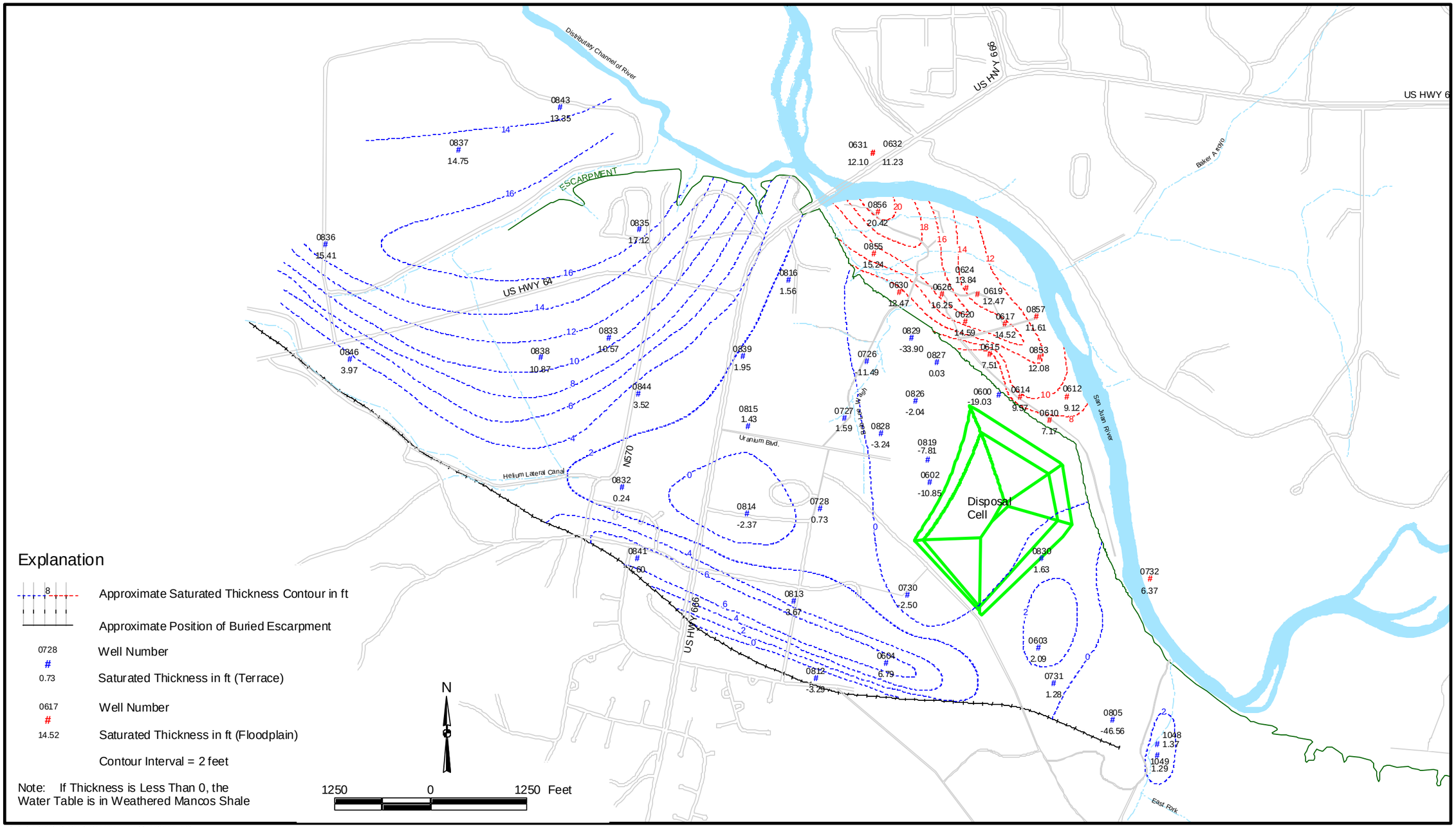


Figure 4–15. Saturated Thickness in Alluvial Material for Floodplain Alluvial Aquifer and Terrace Ground Water System, Shiprock Site

The feed rate to the ponds can be estimated to be 270 gpm × (1,440 minutes per day) × (365 days per year) = 142 × 10^6 gallons per year.

The evaporation rate can be estimated to be 49 in. per year × (1 ft per 12 in.) × (43,560 ft^2 per acre) × (20 acres) × (7.48 gallons per ft^3) = 26.6 × 10^6 gallons per year.

Runoff to the floodplain alluvium is assumed to be equal to the sum of all discharge components from the terrace alluvium. In November 1960, these were measured to be 177.7 gpm (U.S. Department of Health, Education, and Welfare 1962). Therefore, the runoff rate to the floodplain alluvium is estimated to be 177.7 gpm × (1,440 minutes per day) × (365 days per year) = 93.4 × 10^6 gallons per year.

Thus, the annual infiltration rate into the terrace ground water from milling activities is estimated to be (142 × 10^6 gallons per year) – (26.6 × 10^6 gallons per year) – (93.4 × 10^6 gallons per year) = 22.0 × 10^6 gallons per year.

Because the mill operated for 14 years, the cumulative volume of water infiltrated into the terrace alluvium could have been approximately 308 × 10^6 gallons.

Aquifer Volume

The contour map of saturated thickness in alluvial material (Figure 4–15) was used to estimate the volume of water stored in the terrace ground water system south of the disposal cell. Table 4–6 presents a summary of the estimated volume of ground water in the buried ancestral river channel south of the disposal cell. The calculation is based on an assumed porosity of 0.30 in the terrace alluvium. On the basis of this assumption, the minimum volume of ground water in the ancestral river channel alluvial material south of the disposal cell is approximately 38 × 10^6 gallons.

Table 4–6. Estimate of the Minimum Volume of Ground Water in Alluvial Material in the Buried Ancestral River Channel Section of the Terrace Ground Water System South of the Disposal Cell

Contour	Surface Area (ft^2)	Volume of Solid (ft^3)	Volume of Liquid[a] (ft^3)	Volume of Liquid (gallons)
2	4,755,241	9,510,482	2,853,145	21,341,525
4	2,404,921	4,809,842	1,442,953	10,793,288
6	1,217,369	2,434,738	730,421	5,463,549
Total	**8,377,531**	**16,755,062**	**5,026,519**	**37,598,362**

[a]Volume of liquid obtained by multiplying volume of solid by the assumed porosity of 0.3.

Packer Test Results

Table 4−7 presents a summary of the packer test results. The results indicate that the hydraulic conductivity of the Mancos Shale bedrock is low, and that the bedrock appears to be stratified in terms of its hydraulic conductivity. The upper 10 to 30 ft of the bedrock are weathered.

Table 4–7. Summary of Packer Test Results, Shiprock Site

Borehole	Depth Interval (feet below land surface)	Hydraulic Conductivity[a] (cm/s)
820	45–50	J 2.6×10^{-7}
	55–60	J 2.5×10^{-7}
	70–75	J 2.6×10^{-7}
	80–85	J 1.2×10^{-7}
	85–90	J 2.6×10^{-7}
	95–100	J 2.6×10^{-7}
	110–115	J 1.4×10^{-7}
	120–125	J 2.6×10^{-7}
823	55–60	J 2.6×10^{-7}
	65–70	J 2.6×10^{-7}
	77–82	5.8×10^{-7}
	95–100	4.1×10^{-6}
	104–109	J 1.8×10^{-7}
	114–119	J 7.3×10^{-8}
860	30–35	6.0×10^{-4}
	35–40	J 5.2×10^{-7}
	45–50	J 7.7×10^{-7}
	55–60	J 5.2×10^{-7}
	60–65	J 3.9×10^{-7}
862	20–25	1.9×10^{-3}
	34–39	4.7×10^{-6}
	41–46	6.2×10^{-6}
	50–55	3.8×10^{-5}
	55–60	J 1.6×10^{-7}

[a]J represents the quantitation limit for the test.

The weathered section of the formation has hydraulic conductivities in the range of 1×10^{-4} to 1×10^{-6} centimeters per second (cm/s); consequently, it is capable of storing and transmitting limited quantities of ground water. The bedrock below the uppermost section appears to be much less weathered, even though field observations of the core samples indicate significant subhorizontal bedding-plane partings at depth. Perhaps the release of the overburden pressure during core recovery make these partings appear more pronounced. Hydraulic conductivity of the unweathered shale appears to be less than 1×10^{-7} cm/s.

Aquifer Pumping Test Results

The pumping tests performed in the terrace ground water system were designed to test the two different stratigraphic sections of the flow system: the terrace alluvial gravel and the weathered Mancos Shale bedrock. Two tests were conducted: the first was at control well 818 and the second was at well 817.

The pumping rate at control well 818 was 1.86 gpm for 24 hours. A recovery test began immediately after the withdrawal test. The observation well for this test was well 604, which is located 18.9 ft from well 818. Figure 4–2 and Figure 4–3 show the location of these wells and a general cross section of the test site, respectively. Observation well 604 is screened mostly in the upper part of the Mancos Shale. However, the sand filter extends into the overlying terrace alluvium, and the well responds to pumping at well 818. The transmissivity determined for

well 604 is about 220 ft^2/day. Because the saturated thickness of the terrace alluvium is about 10 ft near well 604, the hydraulic conductivity of the terrace alluvium at that location is about 22 ft/day. The recovery test in control well 818 indicated a transmissivity of approximately 85 ft^2/day and, on the basis of a 10-ft saturated thickness, a corresponding hydraulic conductivity of 8.5 ft/day. The average of the hydraulic conductivity measurements is approximately 15 ft/day. Perhaps a more representative transmissivity could be obtained if the observation wells were better coupled to the aquifer. Figure 4–16 presents the results of the pumping test for well 818. Test details are presented in a calculation se (MACTEC–ERS 1999b).

The pumping rate at control well 817 was 0.25 gpm for 24 hours. A recovery test began immediately after the conclusion of the withdrawal test. The observation well for this test was well 602, which is located 15.8 ft from well 817. Figure 4–2 and Figure 4–3 show the location of these wells and a general cross section at the test site, respectively. Observation well 602 was instrumented during the initial step tests, but there was no measurable drawdown. Consequently, the only useful data provided from this test were the recovery data from pumping well 817. These data indicate that the transmissivity at this location is about 3.5 ft^2/day. The low transmissivity at well 817 is not surprising considering that the well is screened entirely within the Mancos Shale. On the basis of a minimum of 10 ft of saturated thickness in this section of weathered Mancos Shale, the hydraulic conductivity is computed to be 0.35 ft/day. This value agrees with the highest hydraulic conductivities obtained with packer tests during the core drilling on this project. Figure 4–16 presents the results of the pumping test for well 817. Additional test details are presented in a calculation se (MACTEC–ERS 1999b).

The terrace alluvium near the 818/604 well pair is sufficiently conductive that water can flow readily to a well. Similarly, the weathered Mancos Shale near well pair 817/602 yields small quantities of water to a well. Because the well yields at both locations exceed 150 gallons per day, the terrace alluvium is sufficiently permeable to be classified as an aquifer by UMTRA Project standards (40 CFR 192.11).

Hydrostratigraphic Controls

The terrace alluvial ground water system is topographically elevated above the floodplain alluvial aquifer. The primary control on the separation of these two flow systems is hydrostratigraphic or the low hydraulic conductivity of the Mancos Shale that underlies both gravel systems. Ground water in the terrace ground water system flows to the northwest along the buried ancestral river alluvial channel and to the north in the weathered Mancos Shale. A minor southeast component of ground water flow may also exist along the top of the siltstone bed in the weathered Mancos Shale. The dip of the siltstone bed is approximately 1 degree to the east. The ground water discharge into Many Devils Wash where the siltstone bed is exposed is approximately 0.3 gpm.

The hydrogeologic relationships of this ground water pathway toward Many Devils Wash were investigated during the spring 2000 drilling project. Wells 1057, 1058, and 1059 were drilled into the Mancos Shale and screened just above the siltstone bed. Each of these wells contains ground water in the Mancos Shale and above the siltstone bed, indicating, that the siltstone bed exerts hydrostratigraphic control on the terrace flow system and is responsible for ground water discharge in Many Devils Wash.

The average hydraulic conductivity of the weathered Mancos Shale in this area can be estimated from the following factors: (1) the dip of the siltstone bed, (2) the measured amount of flow in Many Devils Wash, and (3) the length of the wash that receives seepage from the west. As mentioned, the flow is 0.3 gpm and the dip of the bed is about 1 degree. The length of the wash where the discharge occurs is about 700 ft. The average thickness of the wet zone is not known precisely but is probably between 1 and 3 ft, so assume 2 ft. From Darcy's law:

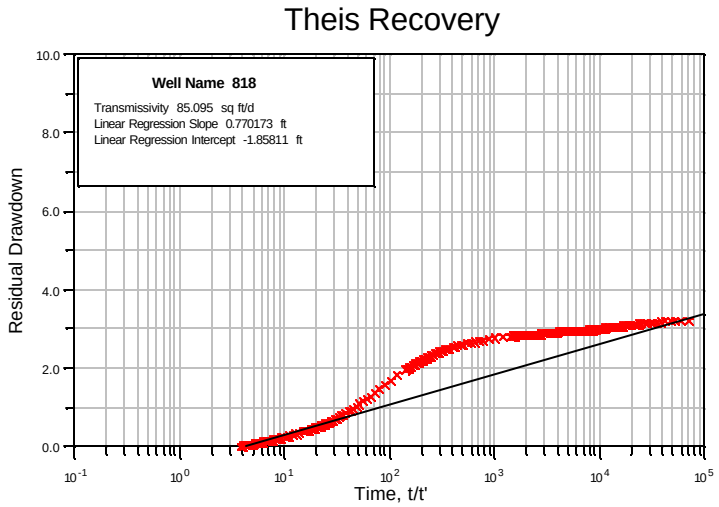
$K = (Q)/ (dh/\text{day}l)\ A = [(0.3\ \text{gal/min})\ (1{,}440\ \text{min/day})\ (\text{ft}^3/7.48\ \text{gal})]/[\tan (1°)\ (700\ \text{ft})\ (2\ \text{ft})]$
$K = 2\ \text{ft/day} = 7 \times 10^{-4}\ \text{cm/s}$

This estimated hydraulic conductivity value is computed crudely but is not unreasonable for the weathered Mancos Shale. It also compares favorably with the range of hydraulic conductivity values of 6.0×10^{-4} and 1.9×10^{-3} cm/s obtained from packer tests of the weathered Mancos Shale.

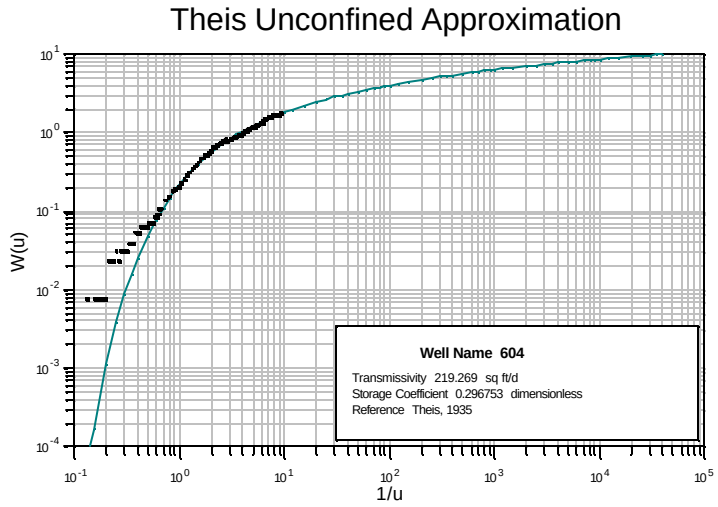
Terrace and Floodplain Alluvium Interactions

Three new well nest pairs-(1) 820, 821, 822, and 860, 861; (2) 823, 824, 825, and 862, 863, 1062, and (3) 1002,1003,1004, and 1000,1001-were drilled to evaluate the hydraulic interconnection between the terrace system and the floodplain alluvium. These three well pairs are illustrated in cross sections F–F', G–G', and H–H', respectively, on Plate 4. Measurements of hydraulic head at well nests (1) and (2) indicate that the hydraulic gradient is predominantly vertical, and the horizontal components of gradient are practically absent. These findings suggest that transfer of water from terrace system to the floodplain alluvium, if it exists, occurs in localized zones of preferred flow rather than as a large-scale phenomenon.

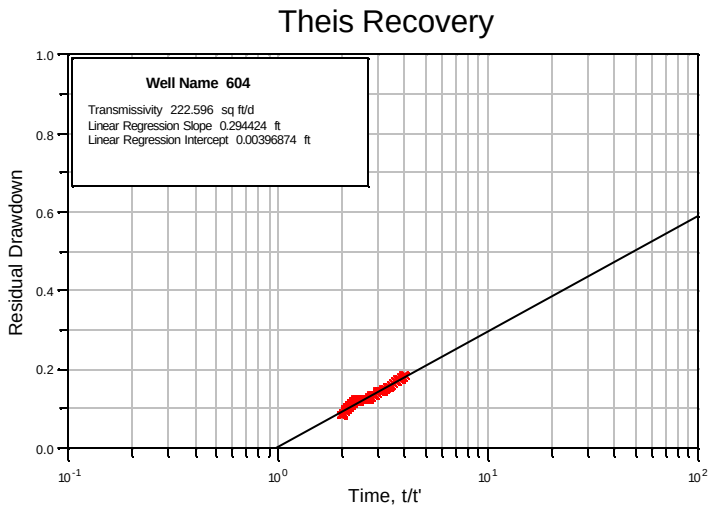
As described in Section 4.4, "Geochemistry," elevated concentrations of constituents in the floodplain alluvium near the base of the escarpment strongly suggest that a contaminant source feeds the floodplain alluvium from the terrace. The manner in which the ground water is transferred to the floodplain is hypothesized to be one or more of the following: (1) the water is transported preferentially through localized horizontal layers of higher conductivity, possibly thin bentonite beds, and are hidden from view because they enter the floodplain below the ground surface; (2) the water is transported along the axes of drainages that were filled in during the remediation and are also hidden from view; or (3) the water is transported along vertical fractures or joints in the Mancos Shale that are difficult to intersect with vertical boreholes. Any combination of these factors may also be present.



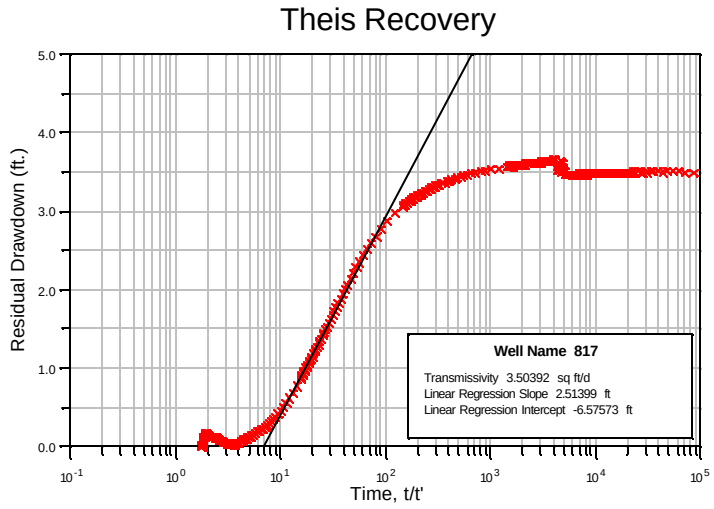
(a) Residual Drawdown in Relation to Dimensionless Time at Pumping Well 818



(b) Drawdown in Relation to Time at Observation Well 604



(c) Residual Drawdown in Relation to Dimensionless Time for Observation Well 604



(d) Residual Drawdown in Relation to Dimensionless Time at Pumping Well 817

Figure 4–16. Aquifer Pumping Test Data for Pumping Well 818 Discharge of 1.86 gpm and Pumping Well 817 Discharge of 0.25 gpm

Terrace Water Balance

Table 4–8 lists the various flow components of the water balance.

Table 4–8. Preliminary Water Balance for the Terrace Ground Water System at the Shiprock Site

Flow Component	Inflow (ft^3/yr)	Outflow (ft^3/yr)
1: Areal Infiltration of Precipitation and Runoff	2,620,000	
2: Infiltration of Water from the NECA Gravel Pit	< 39,000	
3: Drainage of Residual Moisture from the Disposal Cell	568,000	
4: Infiltration of Irrigation Water	4,150,000	
5: Leakage from the Water Supply and Sewer Lines	Unknown	
6: Discharge off the Escarpment		632,000
7: Discharge to Many Devils Wash		21,000
8: Discharge to the Floodplain through Mancos Shale		1,324,000
9: Discharge to the San Juan River		5,403,000
Total (rounded)	**7,380,000**	**7,380,000**

Component 1: Areal Infiltration of Precipitation and Runoff

Infiltration of precipitation and runoff occurs throughout the terrace area. The areal component of recharge refers to all nonirrigated portions of the project area, excluding the disposal cell. Under natural conditions, such as those that existed before 1935, the terrace gravel was mantled with a gently sloping silt layer and a drainage pattern that channeled the runoff to the ephemeral washes, such as Bob Lee Wash and Many Devils Wash. Consequently, the terrace gravel received little to no recharge. Modeling studies suggest that south of the disposal cell the terrace gravel system would have remained dry at recharge equal to or less than 9 percent of average annual precipitation.

It is assumed that infiltration of runoff accounts for no more than 7 percent of average annual precipitation. The total surface area circumscribed by the areal recharge rate is approximately 64×10^6 ft^2. When multiplied by the infiltration rate (0.041 ft/yr), the volume is estimated to be at least 2.62×10^6 cubic feet per year (ft^3/yr).

Component 2: Infiltration of Water from the NECA Gravel Pit

Water is drawn from the San Juan River and used in the NECA gravel pit primarily for dust control. It is applied at the crusher and results in about 1-percent moisture content by weight. During the past year, the gravel pit created approximately 121,000 tons of aggregate and used 290,000 gallons (1,210 tons) of water (Jonathan James, 1999 personal communication) according to the following schedule:

October 1998	85,000 gallons
November 1998	35,000 gallons
December 1998	35,000 gallons
January 1999	20,000 gallons
February 1999	60,000 gallons
March 1999	55,000 gallons

It is assumed that a small percentage of the water applied to the aggregate leaked into the terrace gravel material and weathered Mancos Shale. However, it is not believed to constitute an important fraction of the terrace water balance because the volume of water is low (less than 39,000 ft^3/yr).

Component 3: Drainage of Residual Moisture from the Disposal Cell

The drainage of residual moisture from the disposal cell was estimated in the RAP (DOE 1985), and no additional investigation of the disposal cell or numerical modeling of infiltration through the cover was performed. The numbers provided at that time were assumed to represent an upper limit of drainage through the cell. The calculation presented in the RAP stated that the infiltration through the cover is 0.04 in. per year. It also stated that the area of the disposal cell is 72 acres (3.14×10^6 ft^2). The annual flow through the cover was estimated as

$$0.04 \text{ in. per yr } (3.14 \times 10^6 \text{ ft}^2)\ (1 \text{ ft}/12 \text{ in.}) = 10{,}500 \text{ ft}^3/\text{yr}$$

On the basis of numerical modeling of the terrace area, it is now believed that drainage from the disposal cell accounts for as much as 5.9 in. per year, or approximately a one-hundred fold increase over the original estimate. However, this rate may not apply over the entire footprint of the cell and for modeling purposes is assumed to be approximately 26.5 acres. The annual drainage of residual moisture is reestimated to be

$$5.9 \text{ in. per yr } (26.5 \text{ acres})(43{,}560 \text{ ft}^2 \text{ per acre})(1\text{ft}/12 \text{ in.}) = 5.68 \times 10^5 \text{ ft}^3/\text{yr}$$

Because leachate from the disposal cell is assumed to contain significantly higher chemical concentrations than other sources of recharge, it may be an important source of chemical contamination in the terrace alluvium.

Component 4: Infiltration of Irrigation Water

During the months of April through October, water may be present in the irrigation canal system west of the disposal site and west of U.S. Highway 666. The water is conveyed to the Helium Lateral Canal through a siphon that originates along the Hogback Canal north of the San Juan River near the water treatment plant. Total flow through the siphon to the high point of the canal is 7 to 10 cfs, (Marlin Saggboy, personal communication, August 1999) depending upon the head at the siphon inlet; therefore, the average flow is assumed to be 8.5 cfs. Almost all the flow in the canal is used along its 5-mi length. Seepage losses along the canal are unknown and detailed measurements along the canal system are not taken. It is assumed that irrigation accounts for almost all the water used. The surface area of irrigated land west of U.S. Highway 666 and north of U.S. Highway 64 (where most of the irrigation occurs) is approximately 260 acres. The annual recharge rate of 4.4 in. per year is estimated from modeling studies because no recharge estimates were made for this project. The recharge rate consists of net irrigation and precipitation minus evapotranspiration and runoff and amounts to

$$4.4 \text{ in. per yr } (1\text{ft}/12 \text{ in.})\ (260 \text{ acres})\ (43{,}560 \text{ ft}^2 \text{ per acre}) = 4.15 \times 10^6 \text{ ft}^3/\text{yr}$$

Component 5: Leakage from the Water Supply and Sewer Lines

Leaking water supply lines and sewer lines are another potential source of water to the terrace alluvium but cannot be accounted for precisely. The locations of these potential sources are unknown and cannot be determined at this time and are not accounted for explicitly either in this water balance or in the flow model for the project area.

Component 6: Discharge off the Escarpment

Discharge off the escarpment includes ground water discharge to Bob Lee Wash, to the seeps and springs along the escarpment, and to the other washes and gulches west of the U.S. Highway 666 bridge. Table 4–9 lists the visible discharges from the various seeps. Cumulatively, they amount to about 9 gpm. On an annual basis this seepage flux may be 632,000 ft^3 or more. Other locations of discharge are likely present below the ground surface of the floodplain and, judging from ground water contamination, are inferred to exist near wells 735, 613, and 614.

Table 4–9. Visible Ground Water Discharge Along the Escarpment

Seepage Location	Estimated Flow (gpm)
Seep 425	0.5
Seep 426	1.0
Seep 922	<0.5
1st Wash	1.5
2nd Wash	0.2
Bob Lee Wash	1
Seeps near 936 area	2
Seeps 200 to 400 ft west of U.S. Highway 666 Bridge	1
Seep 786	1
Total	9

Component 7: Discharge to Many Devils Wash

This component of discharge is listed separately because it is a terrace-flow component that flows toward Many Devils Wash. As previously described in the "Hydrostratigraphic Controls" subsection, ground water has been observed in wells screened just above the siltstone bed in the Mancos Shale. The wintertime discharge at the mouth of Many Devils Wash is assumed to equal the ground water discharge along the wash. The measured discharge is 0.3 gpm (21,000 ft^3/yr).

Component 8: Discharge to Floodplain through Mancos Shale

This discharge component is hidden from view and cannot be measured directly. It is believed to exist, however, because the ground water at the base of the escarpment contains high concentrations of nitrate and uranium. These contaminants are believed to be discharging to the floodplain along preferred horizontal pathways in the Mancos Shale. The flux of this flow path may be as much as 1,324,000 ft^3/yr.

Component 9: Discharge to the San Juan River

This final component cannot be measured with a flow meter; therefore, it is estimated the difference between inflow and estimated outflow. The results indicate this component to be approximately 5,403,000 ft^3/yr.

4.4 Geochemistry

DOE collected ground water, surface water, soil, and sediment samples from the floodplain and the terrace from September 1985 to April 2000. Data from analyses of these samples are extensive; a summary of recent surface and ground water sample analyses from 1997 to April 2000 is presented in Appendix B. Comprehensive analytical results of surface water (Appendix C), ground water (Appendix D), and sediment, soil, and salt crust samples (Appendix E), are included in this report on a compact disk. Data used to assess current surface and ground water quality were mainly from the most recent sampling round conducted in February 2000 and from the first sampling of 16 new monitor wells in April 2000.

Table 4–10 lists the 11 constituents identified in the 1994 Baseline Risk Assessment (BLRA) as being contaminants of potential concern (COPCs) for human health or ecological risk in the floodplain area (DOE 1994). These constituents, along with major ions and field parameters have been routinely analyzed in ground water in both the floodplain and terrace locations. Most of these same constituents have also been analyzed in surface water, soils, and sediments associated with the Shiprock site. In addition, other studies were conducted for molybdenum, vanadium, Kjeldahl nitrogen, Ra-226/Ra-228, Th-230/Th-232, and U-234/U-238. An investigation of organic contamination in terrace ground water was based on historical knowledge that organic chemicals were used in the milling process at the site. The updated BLRA in the SOWP, Rev. 1 (DOE 1999g), eliminated some of the original COPCs on the basis of various criteria; human health and ecological COPCs were further refined in this document (see Section 6.0).

This section focuses on the results of sampling conducted since 1998. The discussion on surface water and ground water chemistry is confined mainly to major ions and the contaminants of concern (COCs) or COPCs identified in the SOWP, Rev. 1, as well as special studies conducted as part of characterization activities. Soil and sediment data were collected and analyzed before screening of contaminants in the BLRA update and included constituents that are not currently considered to be COCs. Those data are presented for informational purposes and completeness. Analytical results for all constituents are contained in the Appendices C, D, and E.

4.4.1 Surface Water Chemistry

Figure 4–1 shows surface water locations sampled in February 2000. Locations sampled at other times can be found on Plate 1.

4.4.1.1 San Juan River

Surface water from the floodplain drains into the adjacent San Juan River. Two river locations upgradient of the millsite floodplain (898 and 888) were sampled to provide river-water quality data representing background. Location 888 is about 700 ft downstream from the confluence with the Chaco River (Plate 1). Because the sampling in June 1999 represents a high-flow stage of the San Juan River and the sampling in February 2000 represents a low-flow stage, averages

were calculated to represent background. A summary of background water quality data from the June 1999 and February 2000 samplings is presented in Table 4–11. Higher nitrate, sulfate, chloride, and sodium concentrations in samples from location 888 than from location 898 were probably due to the influence of the Chaco River entering the San Juan River. Uranium concentrations were also higher in samples from location 888 than at location 898 but were close to the analytical detection limit. Location 898 is used to represent San Juan River water quality immediately upgradient of the millsite floodplain.

Table 4–10. Constituents Evaluated in the Geochemistry Section

Source Document	Floodplain	Terrace
BLRA (DOE 1994)	COPCs are antimony, arsenic, cadmium, magnesium, manganese, nitrate, selenium, sodium, strontium, sulfate, and uranium Contaminants of ecological concern are For the San Juan River: antimony, arsenic, magnesium, sodium, strontium, sulfate, and Th-230. For San Juan River sediments: arsenic, manganese, Ra-226, strontium, and uranium. For surface water and sediments from pools: manganese, nitrate, selenium, strontium, and uranium.	COPCs are uranium, sulfate, and nitrate.
SOWP, Rev. 1 (DOE 1999g)	COCs are manganese, nitrate, selenium, sulfate, and uranium.	COPCs are ammonium, manganese, nitrate, sulfate, selenium, and uranium.
SOWP, Rev. 2[a] (this document)	COCs for human health risk are manganese, nitrate, selenium, sulfate, and uranium. COPCs for ecological risk are: ammonium, manganese, nitrate, selenium, strontium, sulfate, and uranium.	COPCs for human health risk are ammonium, manganese, nitrate, selenium, sulfate, and uranium. COPCs for ecological risk are ammonium, nitrate, selenium, strontium, sulfate, and uranium.
Other constituents analyzed as part of special studies on terrace ground water were vanadium, molybdenum, Kjeldahl nitrogen, Ra-226/Ra-228, Th-230/Th-232, and U-234/U-238. Floodplain samples were also analyzed for vanadium and molybdenum.		

[a]See Sections 6.1 and 6.2 for a discussion of the selection of COCs and COPCs.

Table 4–11. Background Concentrations in the San Juan River (upgradient)

Location	pH	EC (µS/cm)	Ca (mg/L)	Cd (mg/L)	Cl (mg/L)	Fe (mg/L)	K (mg/L)	Mn (mg/L)	Mg (mg/L)	Mo (mg/L)
888	7.96	610	55.1	< 0.001	14.5	0.027	2.57	0.17	15.5	0.0043
898	7.95	501	48.4	< 0.001	10.2	0.027	1.95	0.008	9.8	0.0021

Location	Na (mg/L)	NH_4 (mg/L)	NO_3 (mg/L)	Ra-226 (pCi/L)	Ra-228 (pCi/L)	Sb (mg/L)	Se (mg/L)	SO_4 (mg/L)	Sr (mg/L)	TDS (mg/L)	U (mg/L)	V (mg/L)
888	47.7	0.027	1.37	0.12	0.35	< 0.001	0.001	172	0.70	411	0.002	0.0006
898	31.3	n/a	1.52	0.09	0.08	< 0.001	0.001	120	0.58	314	0.001	0.0012

Data: Average of June 1999 and Februrary 2000
Notes: EC = Electrical conductivity; µS/cm = microsiemens per centimeter; pCi/L = picocuries per liter; and TDS = Total dissolved solids; n/a not analyzed

Figure 4–17 shows a Piper diagram for the March 1999 samples of San Juan River water. The chemical signature of location 888 is different from that of the other locations, indicating that the

quality of river water at that location may be influenced by the Chaco River. Data from the most recent sampling (February 2000) indicate that concentrations at location 940 (Figure 4–1) are higher than at other locations along the floodplain. Location 940 is where the contaminant plume, represented by uranium concentrations shown on Figure 4–18, intersects the San Juan River and suggests millsite influence. River water samples collected during the low flow (February 2000) were taken near the riverbank in slow-flowing parts of the river where solids concentrations are higher than in the main, swift-flowing channel. Uranium concentrations are higher in samples from the San Juan River on site and downgradient than in samples collected upgradient of the millsite floodplain (Figure 4–19). The uranium concentration at location 940 was 0.0469 mg/L during the February 2000 sampling; uranium concentrations from all other locations except were near the instrument detection limit where analytical uncertainty is greatest. Uranium concentrations in samples collected downgradient of the millsite at location 893 were less than 0.002 mg/L.

On average, the pH of the San Juan River was 7.9. Concentrations of all constituents varied seasonally. Sulfate, uranium, nitrate, and TDS concentrations (Figure 4–19) as well as pH and chloride (not shown in Figure 4–19) were higher in samples from the February 2000 sampling than the June 1999 sampling. This variation may be due to the high river flow during the June 1999 sampling and the low river flow during the February 2000 sampling.

4.4.1.2 Floodplain

Table 4–12 is a summary of surface water data for selected ground water COCs and COPCs for samples collected in February 2000. Samples collected along the San Juan River are included with floodplain samples in the summary. The samples with the highest concentrations came from locations on the floodplain itself (658, 655, and 894; see Figure 4–1) and from the distributary channel of the San Juan River. The background floodplain concentration is an average of June 1999 and February 2000 data at sample location 898 for San Juan River water.

While ammonia ranges up to 1.02 mg/L, that high value is for a single location (location 658 on the floodplain near the escarpment). All of the other floodplain surface water locations were one or two orders of magnitude less than that particular sample. Therefore ammonia is not considered to be a COC for the floodplain.

4.4.1.3 Terrace

Surface water on the terrace includes water in a stream fed by artesian well 648, Bob Lee Wash, Many Devils Wash, 1st and 2nd Washes, irrigation return flow ditch, and old gravel pits north of the high school. High nitrate concentrations in samples from Many Devils Wash (up to 3,520 mg/L) and high uranium concentrations in samples from Bob Lee Wash (up to 1.71 mg/L) indicate millsite contamination. Most of the surface water associated with the terrace is considered to be a surface expression of terrace ground water. Interactions between terrace surface water and the terrace ground water system are discussed in Section 4.4.2.2.

Another location of surface water occurrence (not listed in Table 4–12) is the NECA pond. Concentrations of COPCs in a sample from the NECA pond (location 849) were less than concentrations in the San Juan River upgradient of the millsite. Uranium and nitrate concentrations in the sample from the pond were below detection limits.

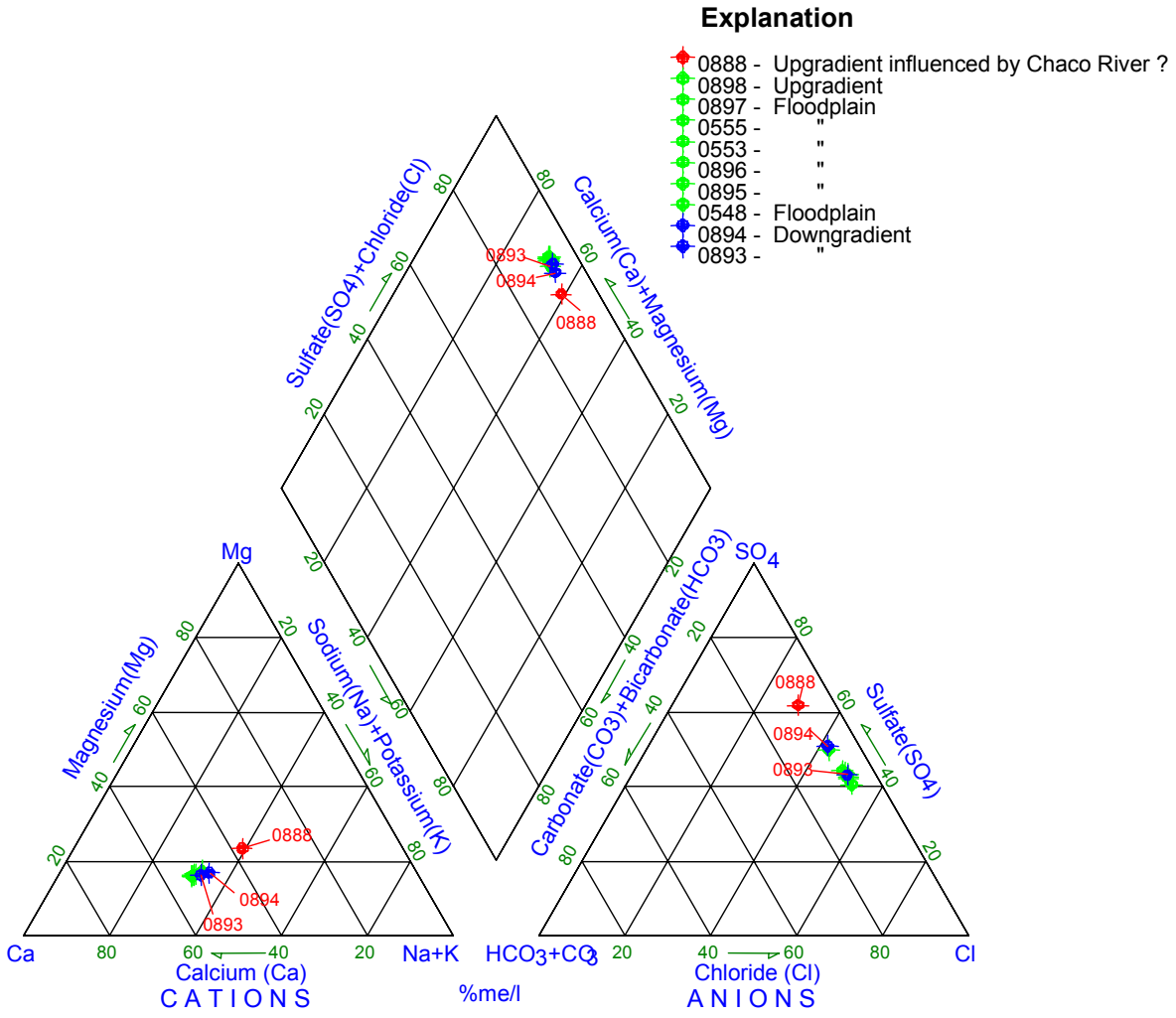


Figure 4–17. Piper Diagram of San Juan River Water (March 1999 data)

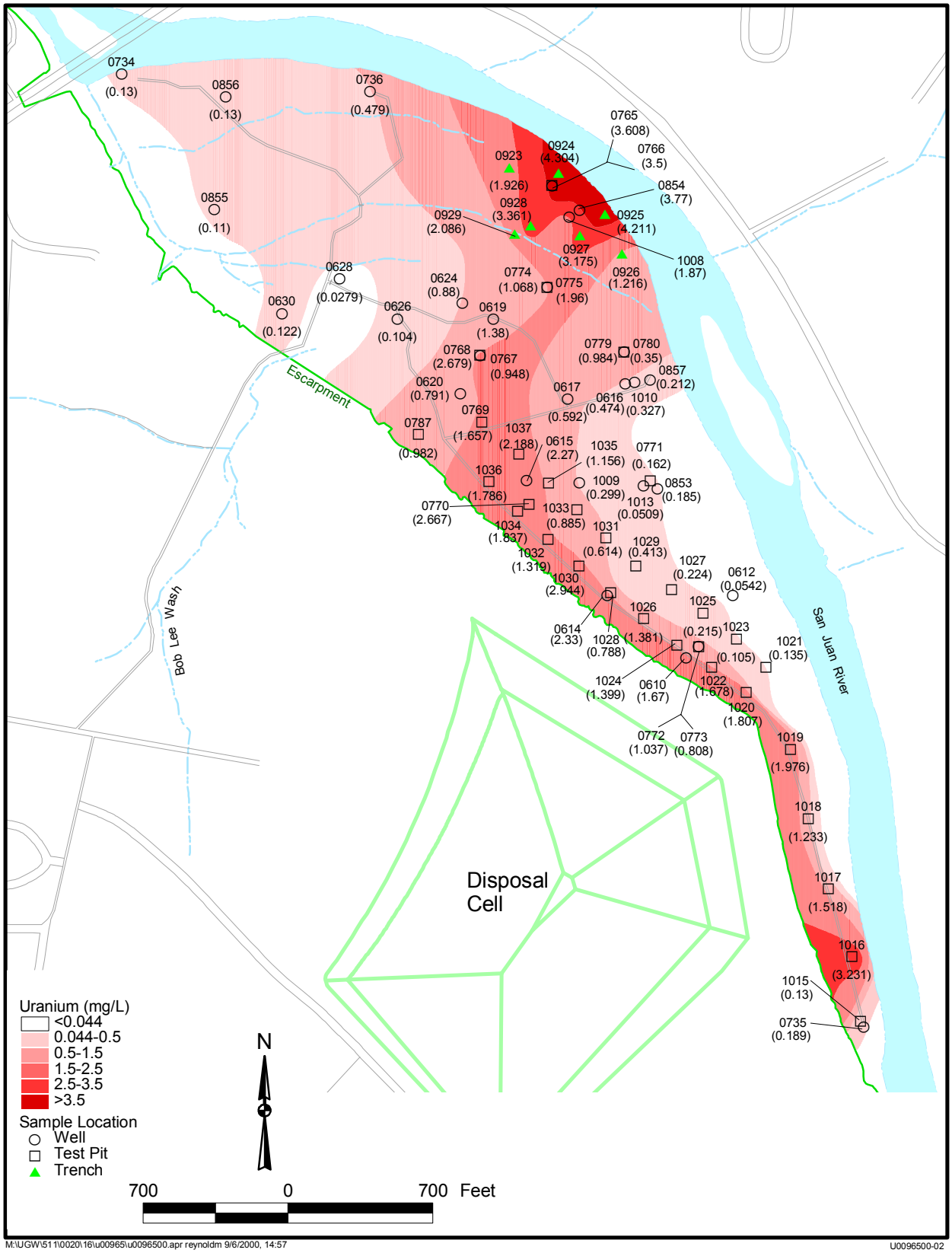


Figure 4–18. Uranium Concentrations in the Millsite Floodplain Ground Water

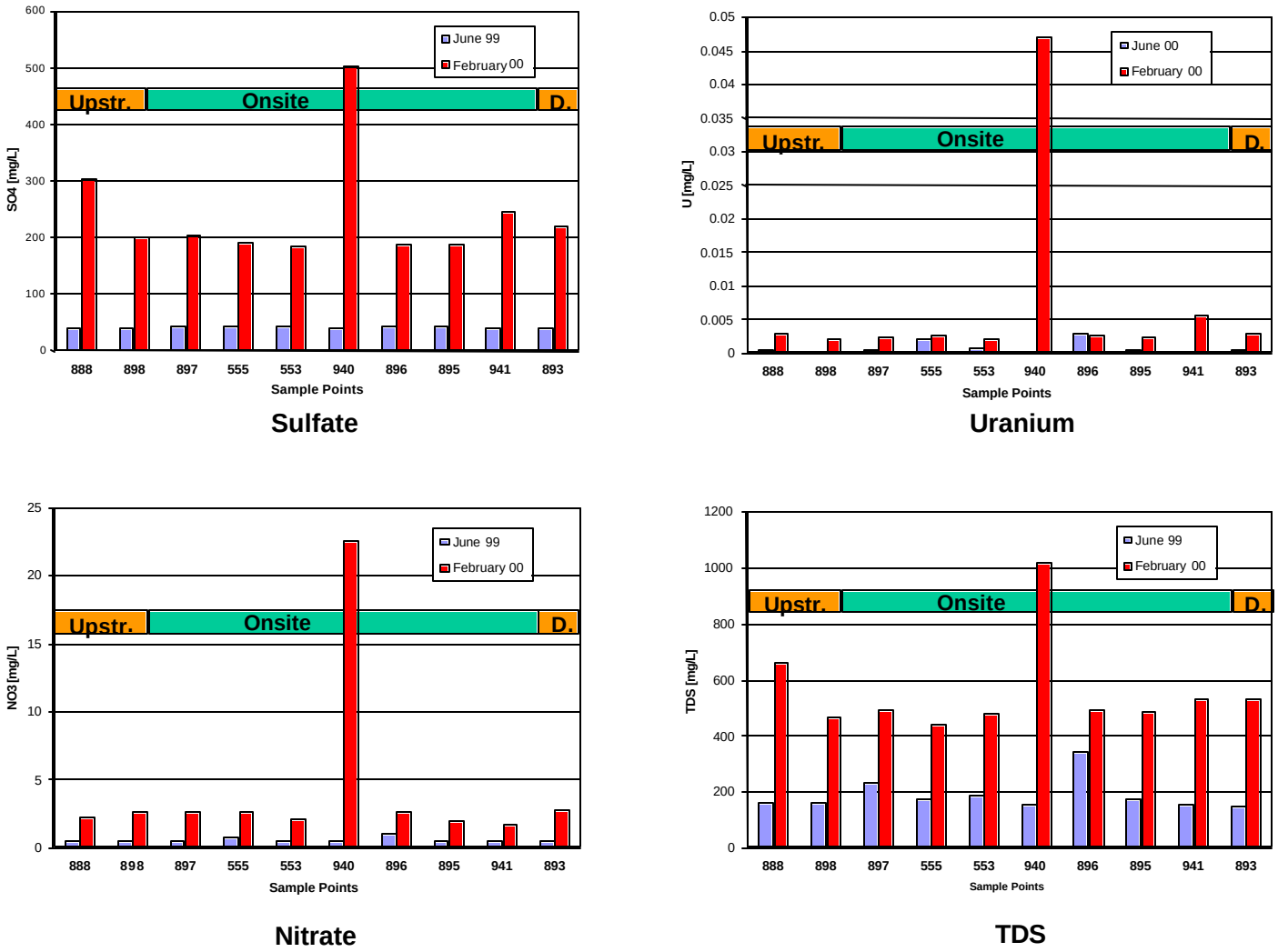


Figure 4–19. Spatial Distribution of Concentrations in San Juan River Water (June 1999 and February 2000 data)

Table 4–12. Summary of Surface Water Chemistry for the Shiprock Site

COC or COPC	Location	FOD[a]	Background[b]	Range	Mean	UCL95[c]
Ammonium (mg/l)	San Juan River	10/10	0.013	0.022-0.173	0.054	0.08
	Other floodplain	6/6		0.032-1.02	0.24	0.55
	Escarpment Seeps	5/5	n./a.	0.0213-0.847	0.201	0.0448
	Other terrace	11/11		0.0425-0.220	0.0844	0.1157
Manganese (mg/L)	San Juan River	10/10	0.001	0.0046-0.050	0.017	0.03
	Other floodplain	6/6		0.0421-0.697	0.397	0.633
	Escarpment Seeps	4/5	n./a.	0.0006-0.065	0.0208	0.0448
	Other terrace	11/11		0.0018-0.0568	0.0135	0.0232
Nitrate (mg/L)	San Juan River	10/10	1.52	1.74-22.50	4.30	8.27
	Other floodplain	6/6		5.63-203	0.40	0.63
	Escarpment Seeps	5/5	n./a.	129-515	265	132
	Other terrace	11/11		1.02-3520	657	1414
Selenium (mg/L)	San Juan River	10/10	0.001	0.0006-0.0012	0.00095	0.0011
	Other floodplain	6/6		0.0119-0.152	0.076	0.120
	Escarpment Seeps	5/5	n./a.	0.0446-0.428	0.18	0.31
	Other terrace	9/11		<0.001-2.32	0.45	0.94
Strontium (mg/L)	San Juan River	10/10	1.35	0.86-1.09	0.90	0.94
	Other floodplain	6/6		5.18-14.8	10.9	14.4
	Escarpment Seeps	5/5	n./a.	5.70-10.3	7.84	9.65
	Other terrace	11/11		4.18-13.5	8.48	10.3
Sulfate (mg/L)	San Juan River	10/10	120	182-504	229	290
	Other floodplain	6/6		2160-4200	3208	3885
	Escarpment Seeps	5/5	n./a.	2640-5670	3918	4906
	Other terrace	11/11		1670-20100	5392	9234
Uranium (mg/L)	San Juan River	10/10	0.002	0.0020-0.0469	0.007	0.02
	Other floodplain	6/6		0.0393-0.112	0.078	0.103
	Escarpment Seeps	5/5	n./.a	0.0433-0.330	0.17	0.31
	Other terrace	10/11		<0.0001-1.71	0.21	0.50

Data: February 2000
[a]FOD: Frequency of Detection
[b]Background floodplain concentration is an average for samples collected in June 1999 and February 2000 from the San Juan River upgradient of the mill site floodplain.
[c]95% upper confidence level on the mean

San Juan River locations: 546, 548, 553, 555, 893, 895, 896, 897, 940, 941
Other Floodplain locations: 655, 657, 658, 887, 894, 939
Escarpment Seeps: 425, 426, 786, 935, 936
Other terrace locations: 662, 884, 885, 886, 889, 933, 934, 942, 1263, 1264, 1265

4.4.2 Ground Water Chemistry

4.4.2.1 Floodplain

The background concentration is defined as the concentration in portions of the aquifer that are unaffected by milling activity. The background quality of ground water in the floodplain was determined from analyses of samples from three monitor wells (850, 851, and 852) at an upstream floodplain location that is lithologically similar to the millsite floodplain. The average concentrations in samples collected from these three wells in the last two samplings (June 1999 and February 2000) were used to represent background water quality (Table 4–13). Table 4–13

also provides concentration ranges, frequency of detection, means, and 95 percent upper confidence level on the mean.

Table 4–13. Ground Water Data Summary

COC or COPC	Location	FOD[a]	Background[b]	Range	Mean	UCL95[c]
Ammonium (mg/L)	Floodplain	32/32	0.045	0.009–70.38	13.14	20.92
	Terrace	16/28	n./a.	<0.0066–1,740	25	147
Manganese (mg/L)	Floodplain	32/32	1.24	0.0014–10.4	3.20	4.04
	Terrace	24/28	n./a.	<0.0006–31.4	2.54	5.16
Nitrate (mg/L)	Floodplain	32/32	0.12	0.01–3,480	593	943
	Terrace	28/28	n./a.	0.01–8,790	1,618	2,413
Selenium (mg/L)	Floodplain	28/32	<0.001	<0.0002–1.04	0.084	0.153
	Terrace	25/28	n./a.	<0.0006–6.52	0.836	1.38
Strontium (mg/L)	Floodplain	32/32	2.26	0.51–20.1	7.82	9.50
	Terrace	28/28	n./a.	2.75–18.3	8.14	9.53
Sulfate (mg/L)	Floodplain	32/32	1,432	138–25,300	6,533	8,731
	Terrace	28/28	n./a.	1,300–17,800	7,359	9,431
Uranium (mg/L)	Floodplain	32/32	0.007	0.0025–3.77	0.756	1.109
	Terrace	28/28	n./a.	0.0021–3.08	0.247	0.463

[a]FOD: Frequency of Detection
[b]Background floodplain concentrations: wells 850, 851, and 852; average of concentrations of June 1999 and February 2000 samplings
[c]95% upper confidence level on the mean

Floodplain locations—610, 612, 614, 615, 616, 617, 619, 620, 624, 626, 628, 630, 631, 632, 732, 733, 734, 735, 736, 766, 768, 773, 775, 779, 782, 783, 784, 853, 854, 855, 856, and 857.
Terrace Locations—same as original table

Areal Extent of COCs

The spatial distributions of COCs in the floodplain are shown on plume maps (Figure 4–18, and Figure 4–20 through Figure 4–23). The most recent data (February and April 2000) were used to prepare the maps. The river and the escarpment were used as geochemical boundaries for the floodplain system. During the drilling and test pit sampling, ground water samples were collected and analyzed in a mobile laboratory to define plume areas. These data were used to guide the drilling programs according to the principles of Expedited Site Characterization (ESC). In certain parts of the floodplain, monitor well sample data were supplemented with data from ground water samples from trenches dug by backhoe and analyzed using the ESC process (ASTM 1996).

The different contaminant plumes shown in Figures 4–20 through 4–23 have very similar configurations in terms of contaminant distribution. In the southeastern portion of the floodplain adjacent to the San Juan River, contaminant concentrations are low, illustrating the diluting effect that the river has as it recharges the area. The effect of recharge from the river in the southeast in combination with recharge from artesian well 648 in the northwestern portion of the floodplain may account for the movement of the centroids of the plumes in a northerly direction. For more discussion on ground water flow and contaminant transport, see Section 4.5.

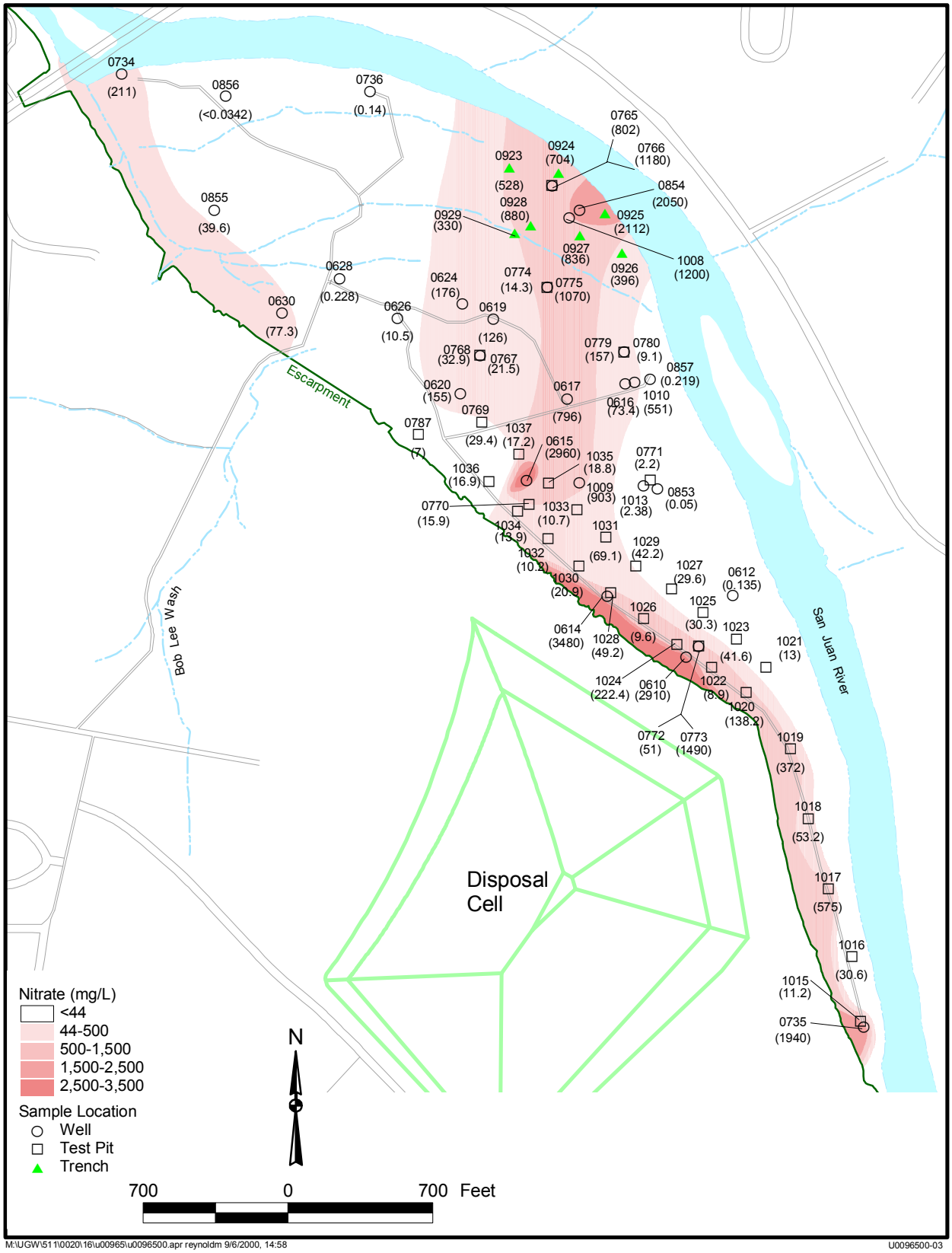


Figure 4–20. Nitrate Concentrations in Millsite Floodplain Ground Water

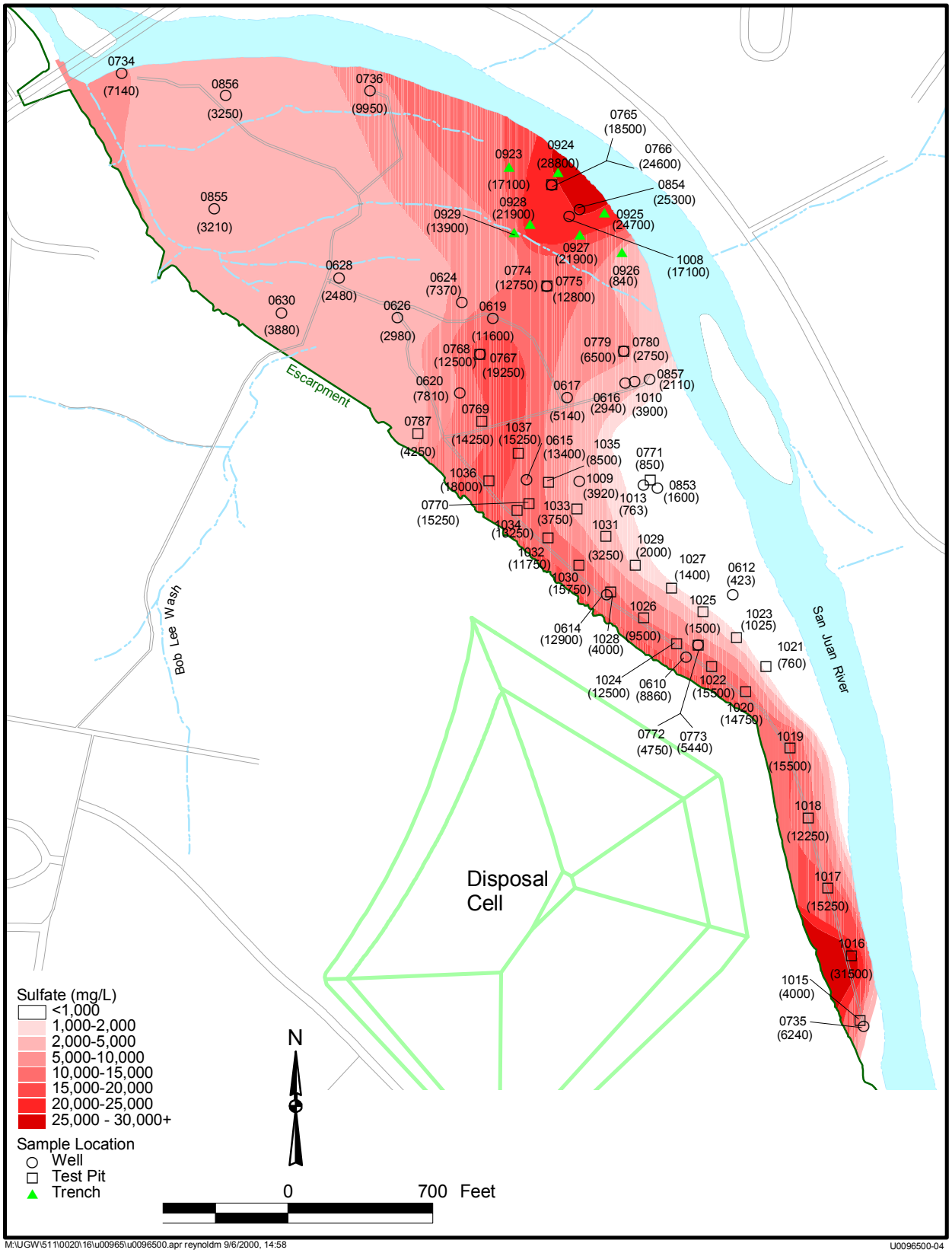


Figure 4–21. Sulfate Concentrations in Millsite Floodplain Ground Water

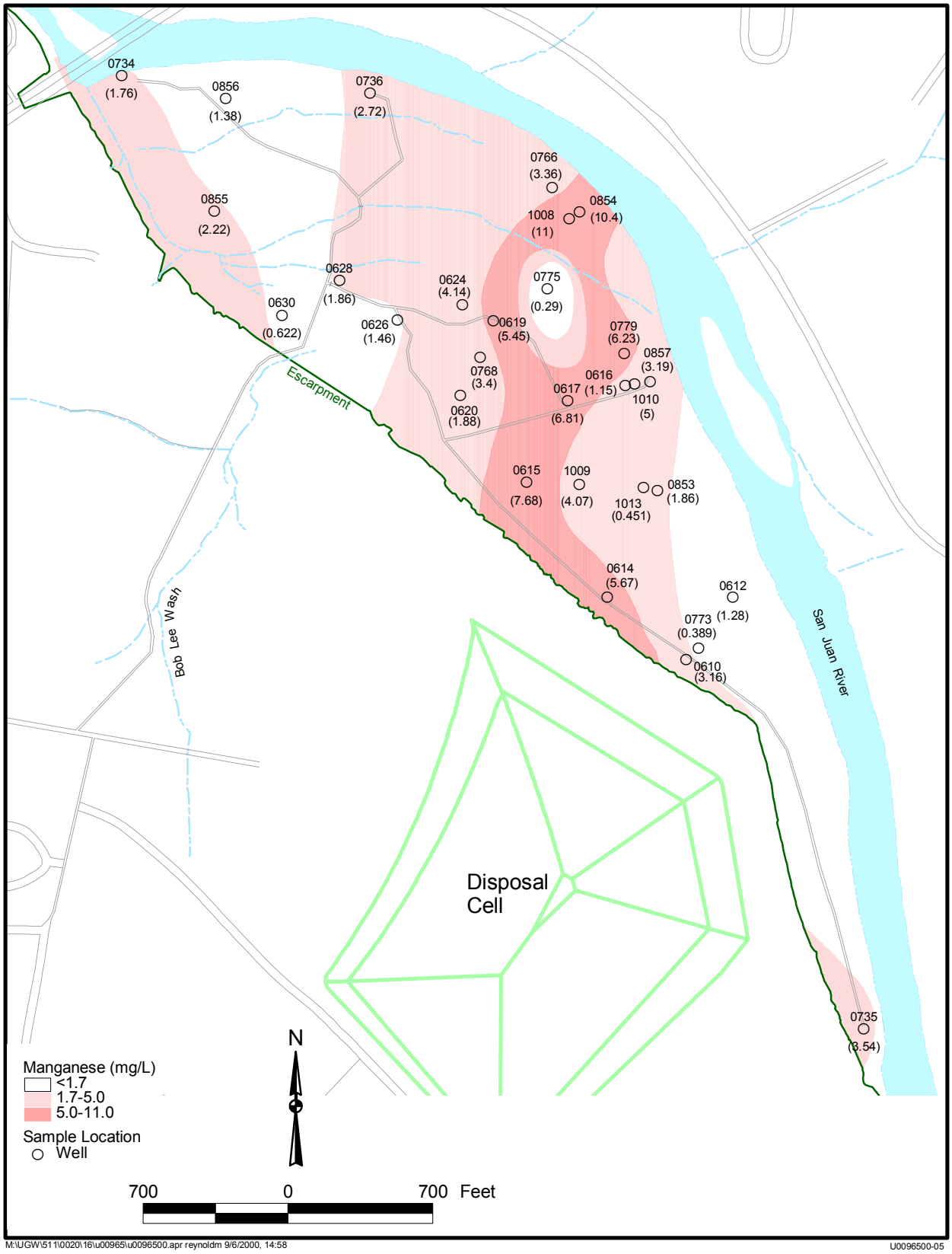


Figure 4–22. Manganese Concentrations in Millsite Floodplain Ground Water

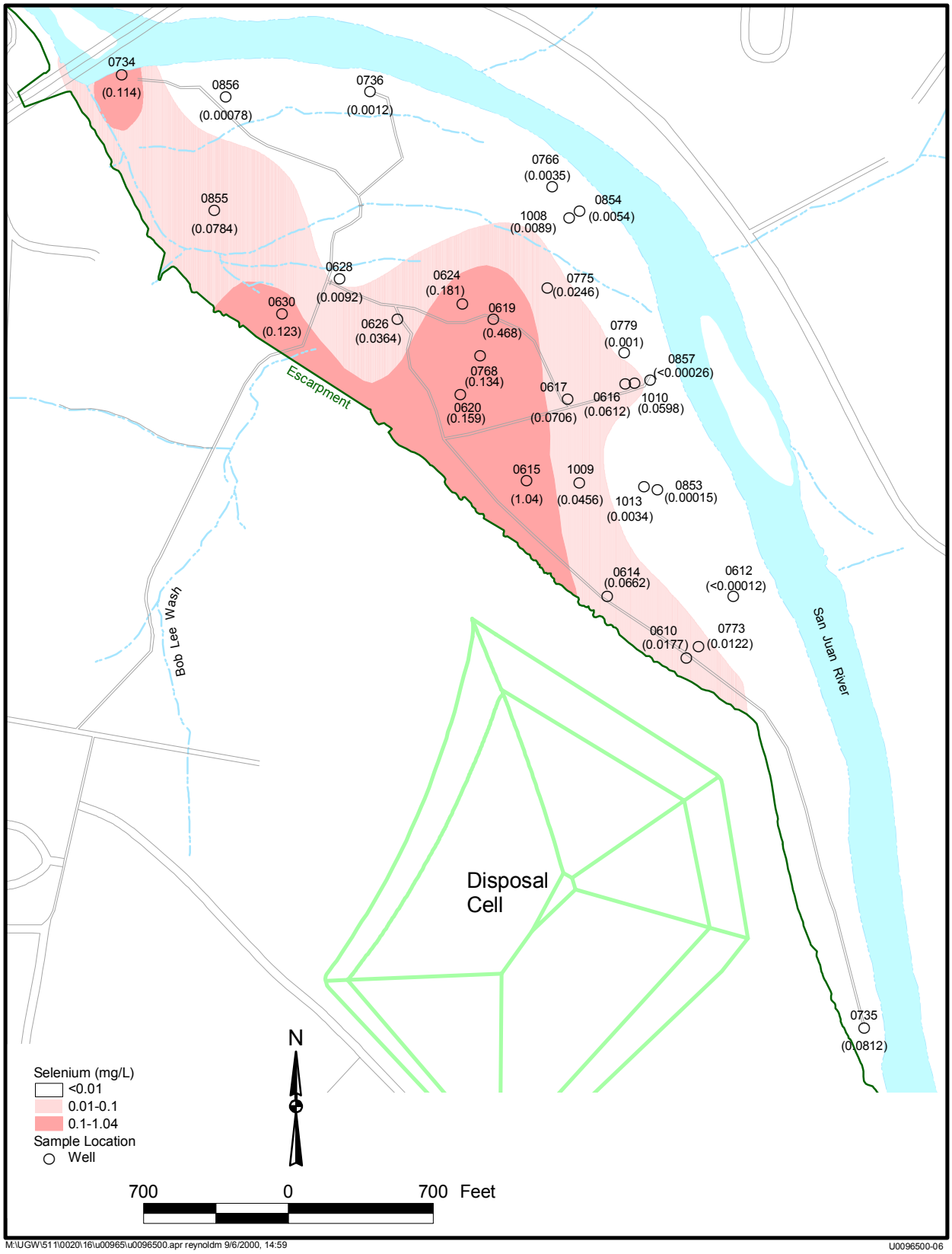


Figure 4–23. Selenium Concentrations in Millsite Floodplain Ground Water

To demonstrate the movement of the uranium, nitrate, and sulfate plume in the central portion of the floodplain during the last 12 years, data from samples from selected wells with long sampling histories were used to create contour maps (Figure 4–24 through Figure 4–26). In addition to the plume maps shown in this section, graduated symbol maps for all analytes based on constituent concentrations in samples from shallow wells on the terrace and the floodplain are presented in Appendix F.

High concentrations of uranium, nitrate, and sulfate have been flushed in the northwest part by Bob Lee Wash (Figure 4–24 through Figure 4–26). After the surface reclamation was completed in 1986, the plume centroids for these three contaminants migrated from the central portion of the floodplain to an area near the escarpment. Since 1993, the centroids have stagnated at this position. However, the highest uranium and sulfate concentrations (3.77 and 25,300 mg/L, respectively) in February 2000 were in samples from well 854, which is far from the escarpment and close to the San Juan River (Figure 4–18 and Figure 4–21, respectively).

Time series for uranium, nitrate, sulfate, and TDS concentrations in samples from three wells selected to represent the southern, central, and northern portions of the floodplain are shown on Figure 4–27. The uranium concentrations in samples from the central portion of the floodplain (well 619) decreased from 3.0 mg/L in 1985 to 0.9 mg/L in 1992 and then increased again to 1.4 mg/L in 2000. In samples from the same well, sulfate concentrations decreased from about 19,000 mg/L in 1985 to about 12,000 mg/L in 2000. Nitrate concentrations in samples from well 619 are currently as high as 126 mg/L but have remained below 400 mg/L for the past 9 years.

The uranium concentrations in samples from the northern portion of the floodplain (well 736) decreased from 1.3 mg/L in 1993 to 0.4 mg/L in 2000 (Figure 4–27). Sulfate concentrations in samples from the same well varied between 10,000 and 15,000 mg/L within the last 5 years but seem to have decreased since 1998. Nitrate concentrations in samples from well 736 were low, ranging from 0.3 to 2 mg/L.

High concentrations of uranium, nitrate, and sulfate were measured in samples from wells close to the escarpment (southern floodplain) in 2000. Uranium concentrations in samples from well 608 (near the escarpment) were as high as 3.7 mg/L after the surface remediation was completed in 1986 but decreased within the last 10 years (Figure 4–27). Uranium concentrations in samples from well 608 are relatively constant and recently averaged 2 mg/L.

Time series for selected wells at the base of the escarpment and well 600, on the terrace just above the escarpment, are presented in Figure 4–28. Uranium concentrations in ground water samples from the weathered Mancos Shale in the terrace at the north corner of the disposal cell (well 600) have been relatively constant since 1988, ranging from 1.0 to 1.5 mg/L. Well 614 is on the floodplain close to the escarpment just north of well 600. In the same period of time, uranium concentrations in samples from well 614 increased from 0.8 to 2.3 mg/L. Concentrations in samples from well 614 also increased for nitrate, sulfate, and TDS (Figure 4–28). Samples from the four wells (608, 610, 614, and 615) completed in the floodplain alluvium had similar concentrations (Figure 4–28). The increase in uranium concentrations in samples from well 614 suggest that there is an increasing contribution from the terrace.

To assess whether ground water contamination near the escarpment is the result of residual contamination on the floodplain, soil and ground water sampling at the base of the escarpment was conducted. Soil and ground water samples were collected at 24 locations on a grid. Uranium concentrations in ground water ranged from 0.13 to 3.23 mg/L at locations closest to the escarpment (Figure 4–29). Locations about 200 ft north of the escarpment contained less than 1 mg/L uranium, except locations 1035 and 1037. The uranium concentrations in soil were slightly elevated above background. The elevated uranium concentrations in ground water are consistent with partitioning between the contaminated ground water and soils; it is not the result of residual contamination on the floodplain. Detailed results of the soil and ground water sampling are provided in Section 4.4.3.

In April 2000, the filled drainages along the edge of the escarpment were investigated to evaluate if the drainages are a pathway for ground water to travel from the terrace to the floodplain. Well 1007 was installed on the terrace east of the disposal cell in a filled drainage, and well 1011 was installed in the filled drainage near well 827. Uranium concentrations are below the MCL in well 1007 and slightly above the MCL in well 1011. Sulfate, nitrate, ammonium, selenium, and manganese concentrations in these wells are relatively low compared to ground water concentrations in the immediately adjacent floodplain. Therefore, the filled drainages are not believed to be a primary pathway for contamination to reach the floodplain.

In June 1999, water was discovered in two neutron hydroprobe ports on top of the disposal cell. The ports are plugged at the bottom and should not be in contact with tailings water unless they are corroded. Recent analyses of water samples (Table 4–14) from the two hydroprobes showed low nitrate and uranium concentrations. One sample had a high sulfate concentration. The low concentrations of uranium indicate that the water in the ports was not in contact with tailings material. The elevated sulfate concentrations could result from seepage of water through the disposal cell cover.

Table 4–14. Analysis of Water from the Neutron Hydroprobe Ports on Top of the Disposal Cell

Sample ID	Nitrate (mg/L)	Sulfate (mg/L)	Uranium (mg/L)	Gross Alpha (pCi/L)	Gross Beta (pCi/L)
NDF401	2.67	14,800	0.0417	<264.8	376.7
NDF402	47.2	2,650	0.031	637.2	1,445

The composition of ground water from the terrace and the floodplain is illustrated in a Piper diagram on Figure 4–30. Wells 600 and 824 represent terrace ground water from the Mancos Shale. Ground water samples from wells 600 and 824 were collected from depths of 60 ft and 200 ft, respectively. The last two samplings of well 824 are displayed in the figure because of the unusual composition of the water. The wells marked with a blue symbol represent ground water from the floodplain close to the escarpment. The yellow symbols show the signature of ground water in the southeast portion of the floodplain, which is flushed by the San Juan River. Deep ground water from the Mancos Shale in well 824 has a different signature in all three diagrams than the other ground waters. It contains relatively higher concentrations of bicarbonate, sodium, and potassium, whereas the water from the floodplain contains relatively higher concentrations of sulfate, calcium, and magnesium. Low permeability of the Mancos Shale causes a long residence time for deep ground water. The water in well 824 seems to be influenced by

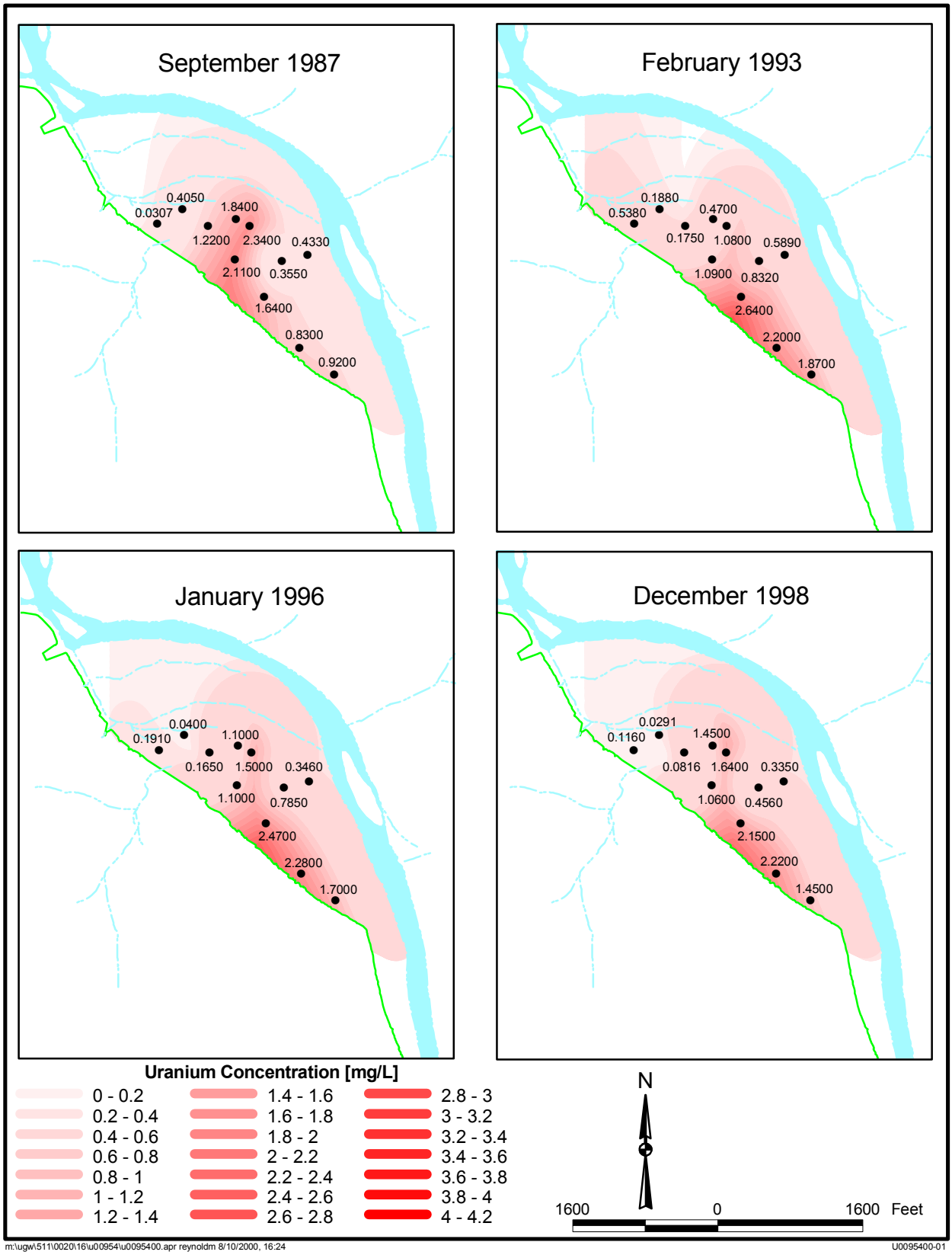


Figure 4–24. Uranium Concentrations Over Time in the Millsite Floodplain Ground Water

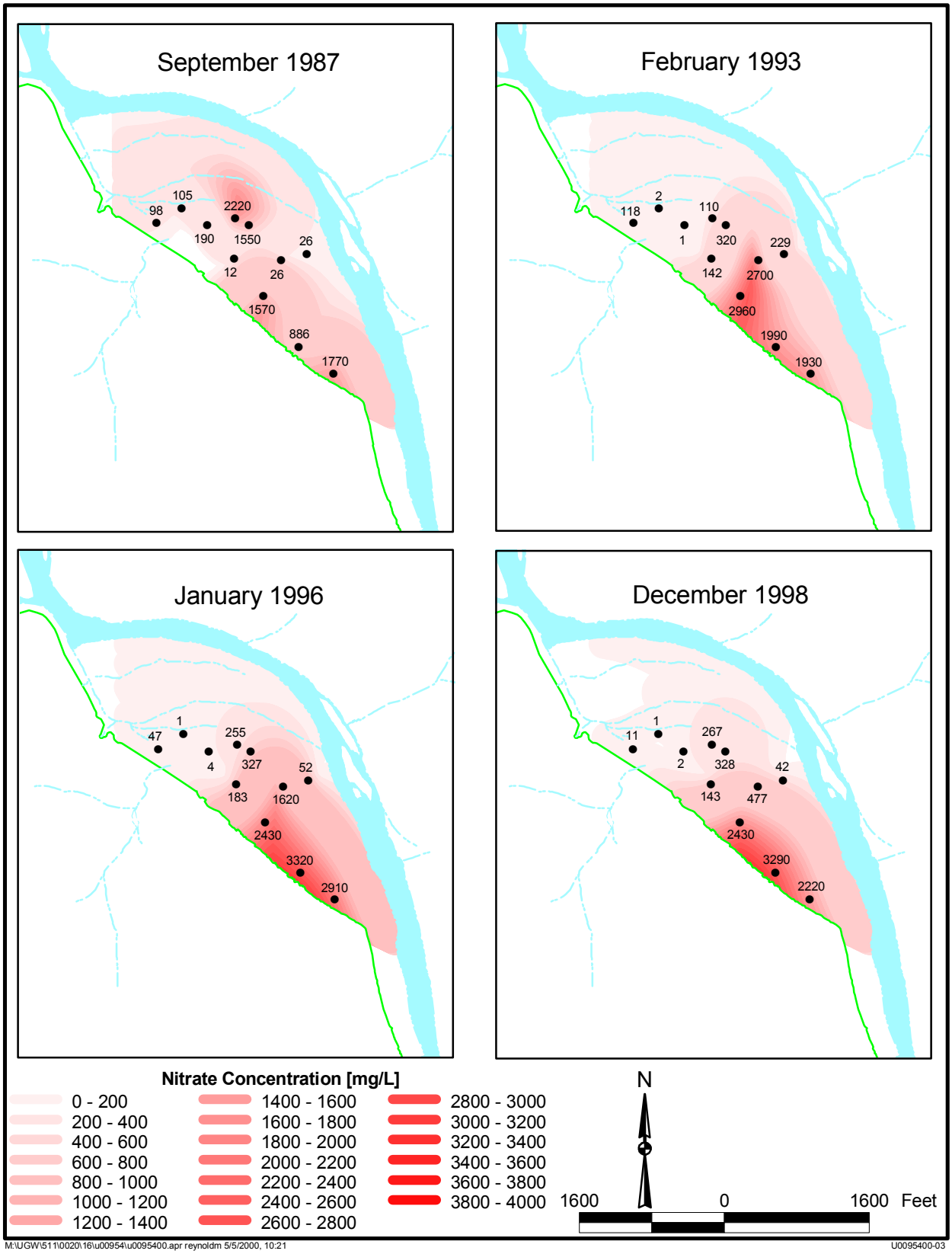


Figure 4–25. Nitrate Concentrations Over Time in the Millsite Floodplain Ground Water

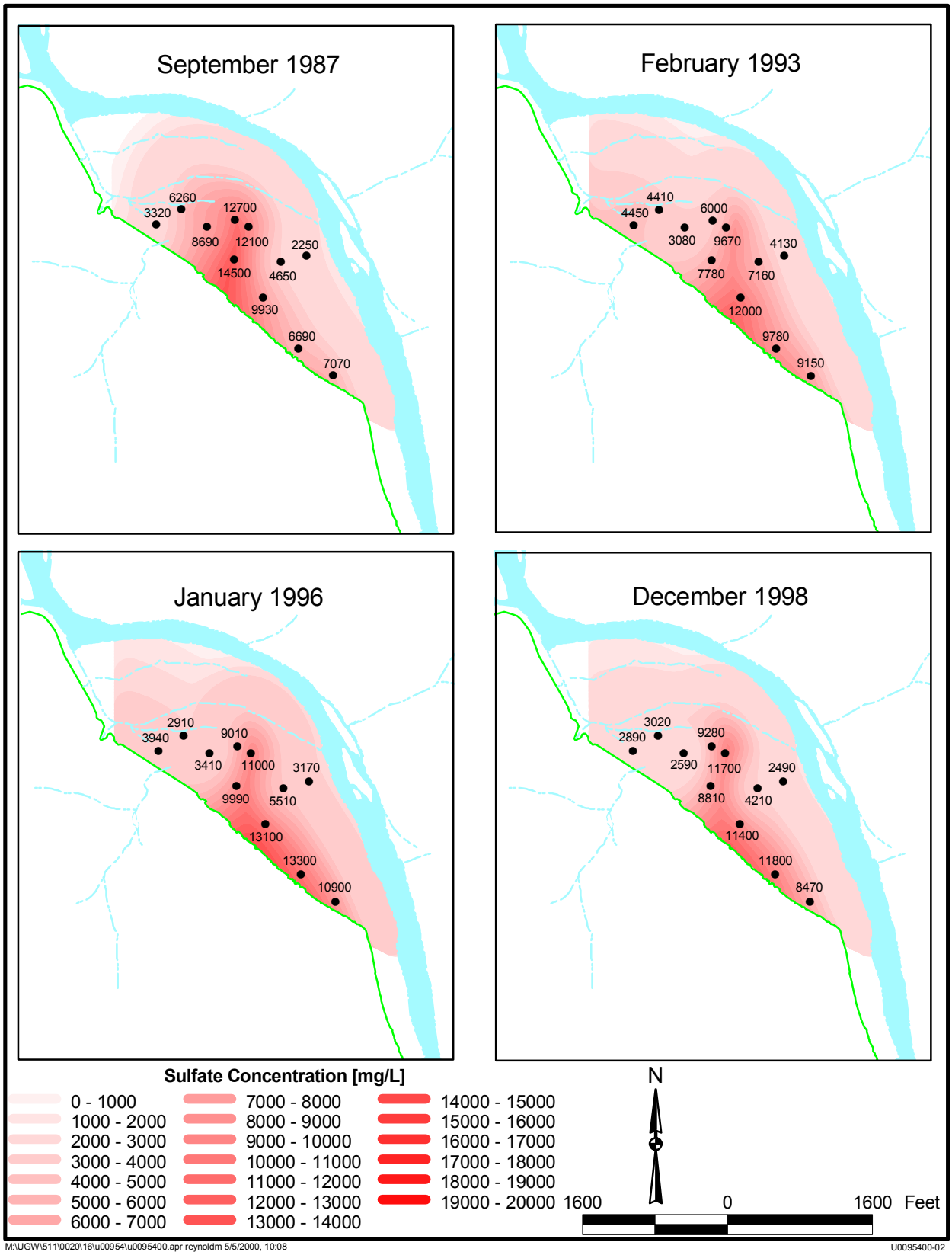


Figure 4–26. Sulfate Concentrations Over Time in the Millsite Floodplain Ground Water

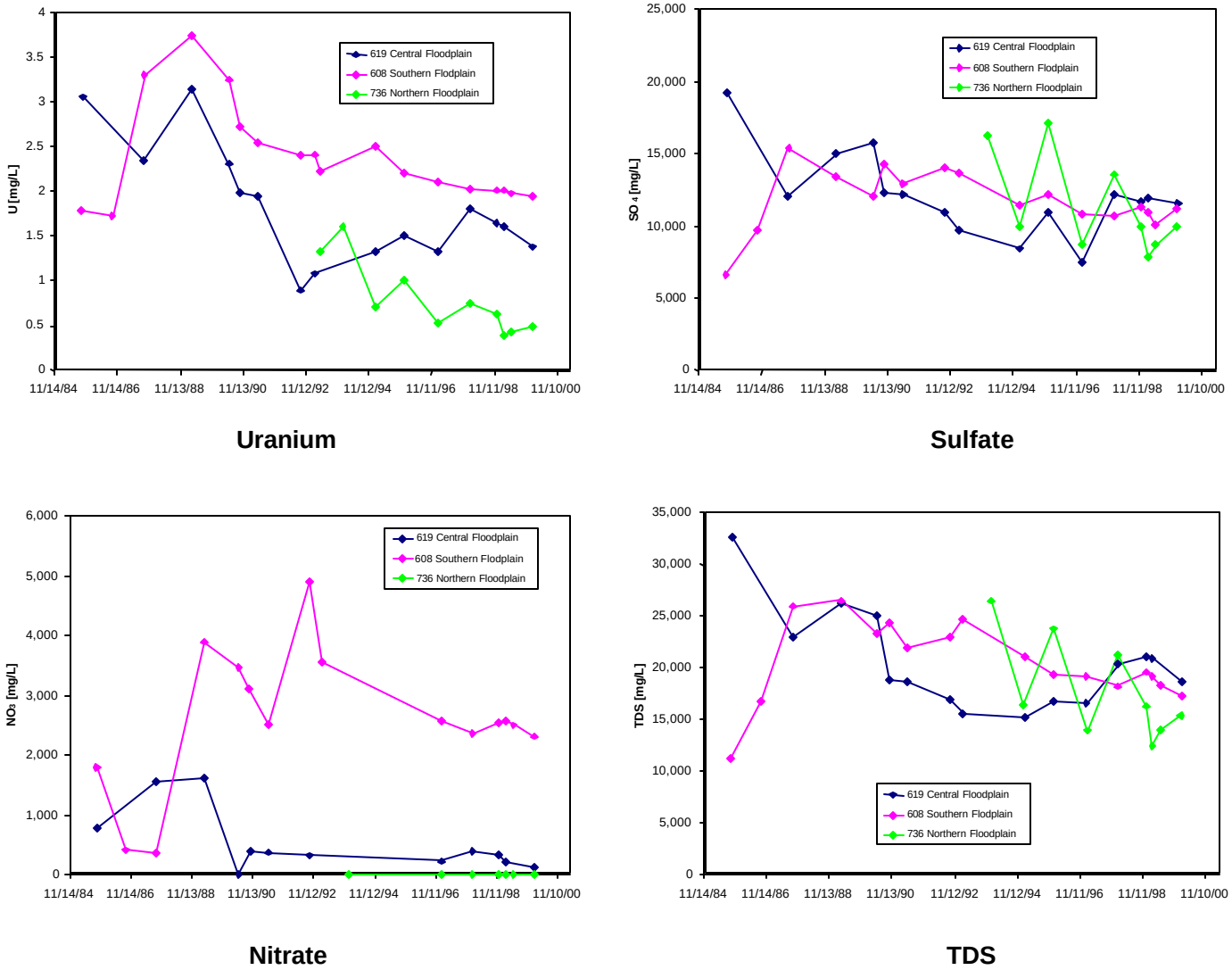


Figure 4–27. Concentrations of Uranium, Sulfate, Nitrate, and Total Dissolved Solids Over Time in the Millsite Floodplain Ground Water

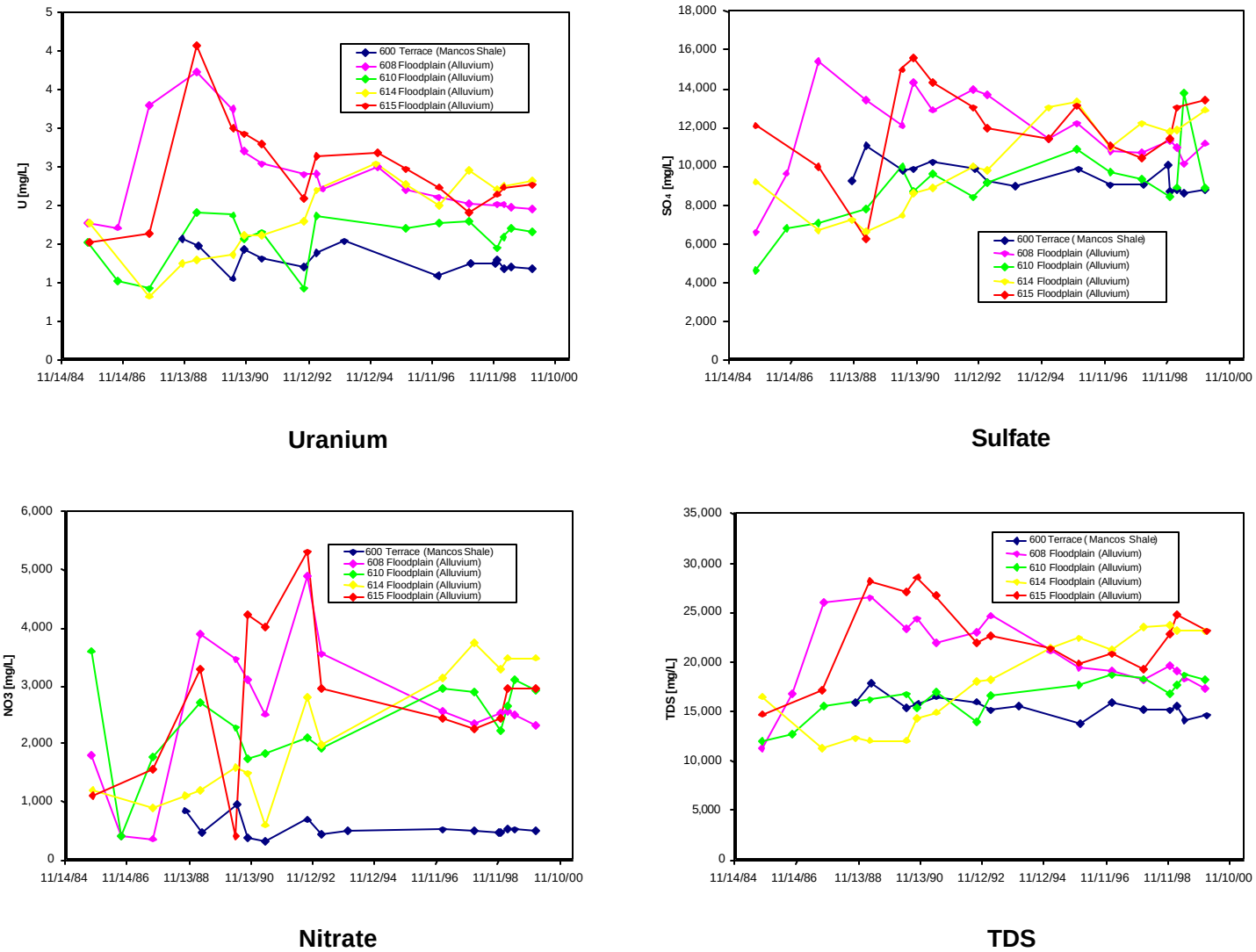


Figure 4–28. Concentrations of Uranium, Sulfate, Nitrate, and Total Dissolved Solids Over Time in Ground Water from the Base of the Escarpment on the Millsite Floodplain and Ground Water from the Adjacent Terrace

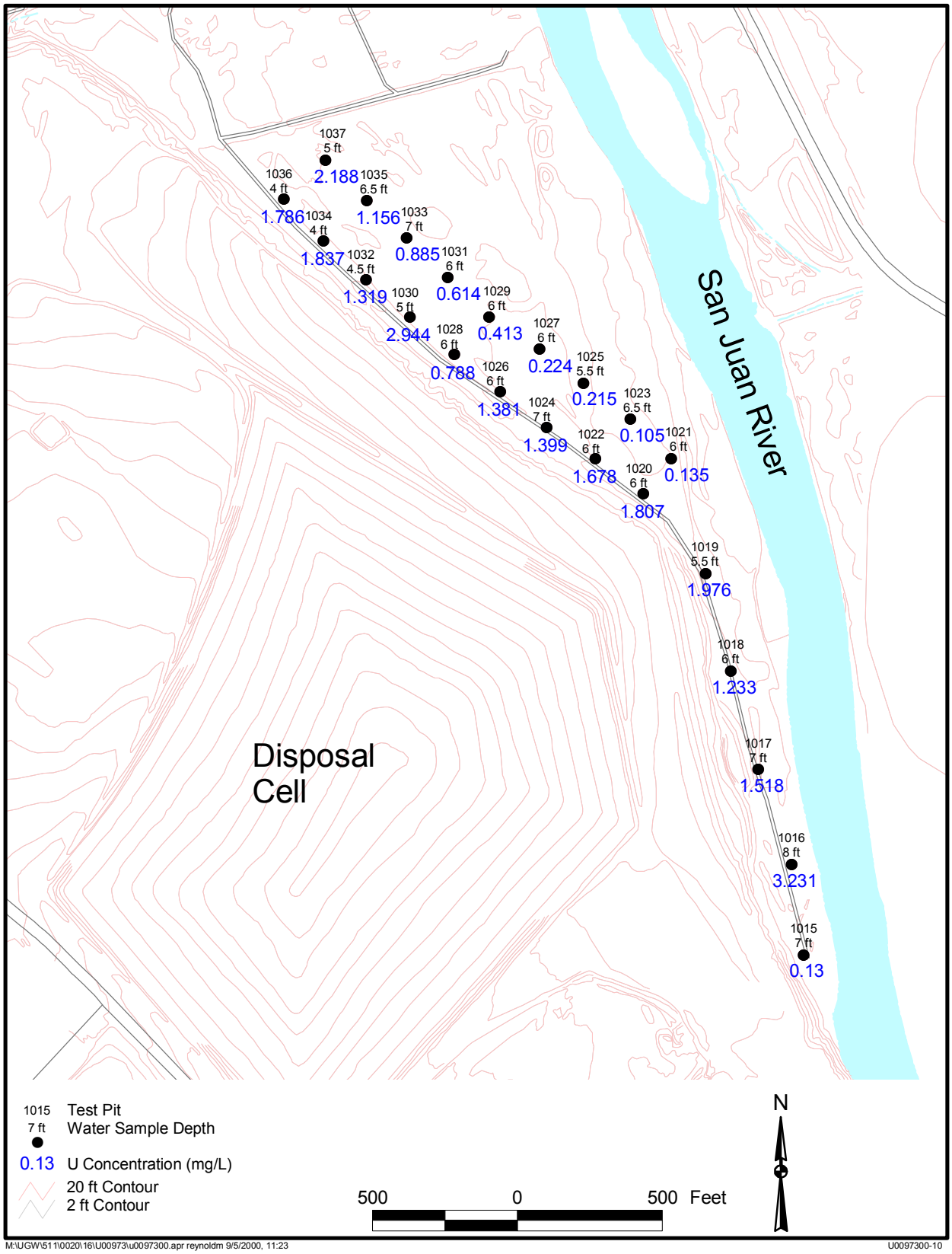
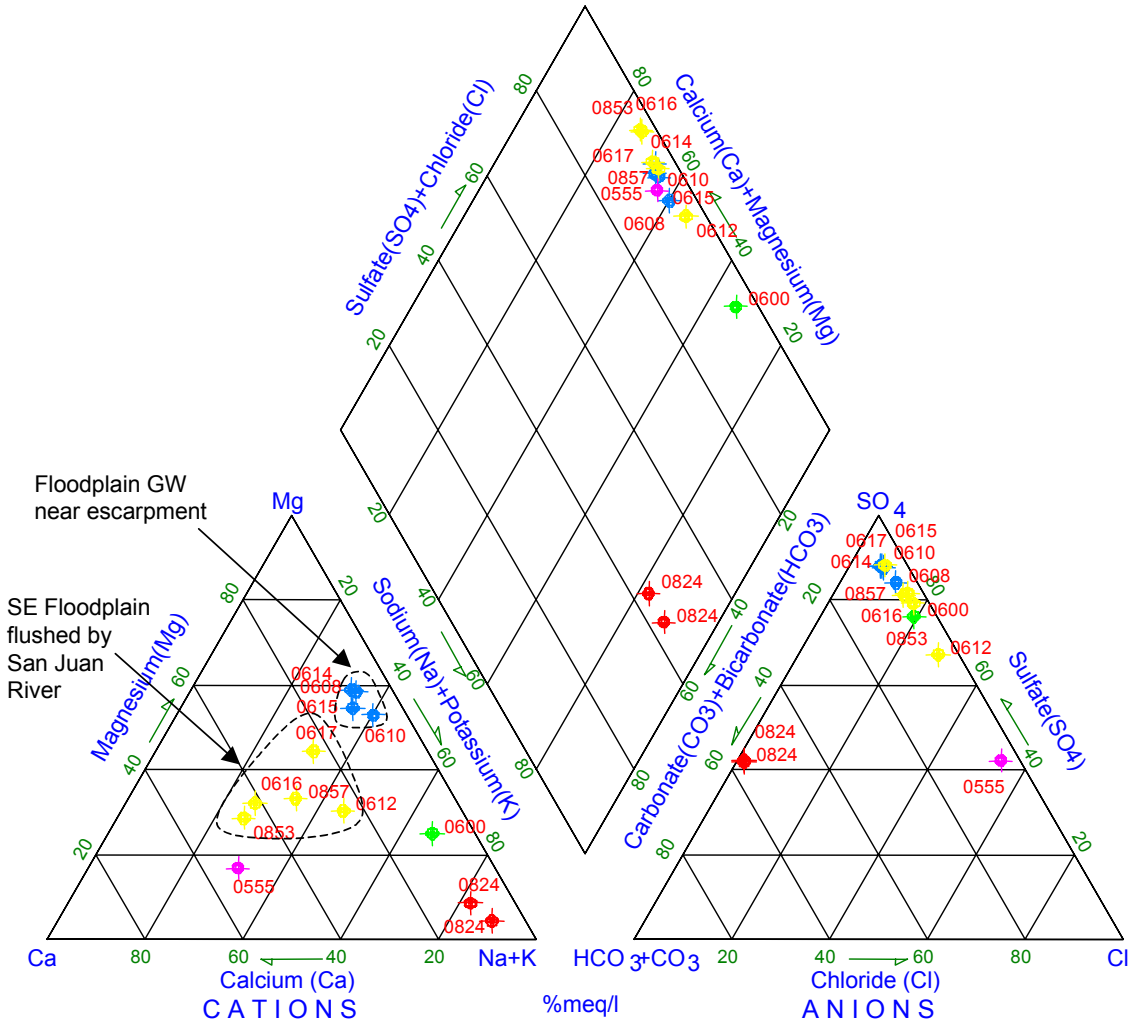


Figure 4–29. Uranium Concentrations in the Millsite Floodplain Ground Water from Test Pit Samples



Explanation

- 0608 - Floodplain GW
- 0610 - Floodplain GW
- 0614 - Floodplain GW close to the escarpment
- 0615 - Floodplain GW close to the escarpment
- 0600 - Terrace GW (Mancos) - 60 ft. depth
- 0824 - Terrace GW (Mancos) - 200 ft. depth
- 0612 - SE Floodplain flushed by San Juan River
- 0616 - SE Floodplain flushed by San Juan River
- 0617 - SE Floodplain flushed by San Juan River
- 0853 - SE Floodplain flushed by San Juan River
- 0857 - SE Floodplain flushed by San Juan River
- 0555 - San Juan River

Figure 4–30. Piper Diagram of Millsite Floodplain and Terrace Ground Water and San Juan River Water

interaction with the Mancos Shale. The saturation index for calcite is 0.01 in water from well 600 and 0.29 in water from well 824, suggesting that these waters are oversaturated with calcite. For gypsum, the water in well 600 has a saturation index of –0.02, and the water from well 824 has a saturation index of -0.37.

San Juan River water has an intermediate ratio of sulfate and chloride with low content of carbonate and bicarbonate. A mixture of ground water from the floodplain close to the escarpment (blue symbol) with San Juan River water (red symbol) could produce the composition of the southeast floodplain ground water (yellow symbol) (Figure 4–30). The cation triangle also shows that ground water from the upper, weathered portion of the Mancos Shale (well 600) could be a mixture of alluvial ground water from the floodplain near the escarpment (blue symbol) and ground water from deeper in the Mancos Shale (well 824). Nitrate and uranium concentrations in the ground water in well 600 are lower than in the floodplain, whereas sulfate and TDS concentrations are almost as high as in the floodplain.

Some of the highest floodplain contaminant concentrations occur close to the escarpment, suggesting that a continuing source is present in the terrace. Major-ion chemistry in the deep Mancos Shale (0824) close to the escarpment is different from that in the floodplain sediments (blue symbols), suggesting that the deep Mancos Shale is not a pathway to the floodplain alluvium (Figure 4–30). High contaminant concentrations in samples from well 614 (Figure 4–28) could also be caused by an area of stagnant ground water within the floodplain.

Distribution of Vanadium and Molybdenum

In recent years, vanadium had not been a routine analyte for site sampling, though it had been detected in some earlier sampling rounds. Analysis of vanadium was included in floodplain ground water samples collected during the June 1999 sampling. All samples were at or below the instrument detection limit. Analysis of molybdenum was performed on floodplain ground water samples collected in June 1999 and February 2000. Plume maps were not prepared because of the very low concentrations of these constituents (refer to “spot plots” in Appendix F). Concentrations of vanadium are all below or at the detection limit. Molybdenum concentrations in alluvial wells are all well below the UMTRA standard of 0.1 mg/L. The only appreciable molybdenum was detected in wells completed deep in the Mancos Shale. The source of the deep molybdenum is unknown but is not believed to be site-related.

Vertical Extent of Contamination

The vertical extent of contamination was monitored in samples from nested wells 820 through 822 and 615, 860, and 861 shown on cross section F–F' in Figure 4–31; wells 823 through 825 and 608, 862, and 863 shown on cross section G–G' in Figure 4–32; and wells 1000, 1001, and 614, and 1002 through 1004 shown on cross section H–H' in Figure 4–33. Plate 3 shows the location of cross sections F–F', G–G', and H–H'. No water or just a small amount of water occurred in most of the deep wells completed in Mancos Shale. No samples could be taken at wells 821, 822, and 861 on cross section F–F' and well 825 on cross section G–G'. In most cases, concentrations of uranium, sulfate, and nitrate in samples decrease with depth. Because of the limited amount of water in the nested wells, it was assumed that cross sections F–F' and G–G' were not located close to a potential pathway in the Mancos Shale. Wells on cross section H–H' are screened in more shallow zones. The uranium concentrations in wells 1002 and 1003 are below the detection limit; nitrate, sulfate, and ammonium concentrations in these wells decrease

with depth and cannot explain the high concentrations in ground water on the floodplain close to the escarpment. Shallow well 1004, screened in weathered Mancos Shale, had a uranium concentration of 0.104 mg/L.

Surface Water Influence on Floodplain Ground Water

Water from artesian well 648, drilled in 1961, flows eastward in an outflow ditch, which until the fall of 1999 drained into Bob Lee Wash. For the last 10 to 15 years, this flow has created a wetland area where Bob Lee Wash drains into the floodplain. The continuous flow of water has flushed the northwest portion of the floodplain. An analysis of water sampled from well 648 is shown in Table 4–15. Figure 4–34 shows a Piper diagram for the of the artesian well water, Bob Lee Wash area ground water, and ground water in the northwest and southeast parts of the floodplain. Ground water in the southeast portion of the floodplain is influenced by San Juan River water. It contains relatively more calcium and magnesium, whereas water from the artesian well contains relatively more sodium plus potassium. Much of the ground water in the northwest portion of the floodplain is derived from mixing of surface water flowing northward down Bob Lee Wash (artesian well 648 water) and ground water in the Bob Lee Wash area, as indicated by the Piper diagram.

Table 4–15. Water Quality of Samples from Artesian Well 648 (June 1998 sampling)

Alkalinity as $CaCO_3$ (mg/L)	Ca (mg/L)	Cd (mg/L)	Cl (mg/L)	Fe (mg/L)	K (mg/L)	Mg (mg/L)	Mn (mg/L)	Na (mg/L)
59	110	0.001	52.2	0.106	7.82	13.5	0.0886	836

NH_4 (mg/L)	NO_3 (mg/L)	pH	Ra-226 (pCi/L)	Ra-228 (pCi/L)	SO_4 (mg/L)	Sr (mg/L)	TDS (mg/L)	U (mg/L)
0.569	0.0285	7.8	0.58	0.83	2,000	12.1	3,100	0.001

Figure 4–35 presents time series for uranium, sulfate, nitrate, and TDS concentrations in artesian well water, shallow ground water in the Bob Lee Wash area, and ground water in the northwest part of the floodplain. Uranium, sulfate, and TDS concentrations in the northwest part of the floodplain decrease over time. Concentrations of uranium, sulfate, nitrate, and TDS are lower in the artesian well water samples than in Bob Lee Wash area ground water or floodplain ground water samples. Sulfate concentrations in the northwest part of the floodplain will not decrease lower than 2,000 mg/L as long as the artesian well water continues flush the floodplain. Although nitrate concentrations in the artesian well samples are lower than 0.1 mg/L, the samples of shallow ground water in the Bob Lee Wash area show slightly increasing concentrations over time, probably because of the addition of nitrate from the millsite.

4.4.2.2 Terrace

Areal Extent of COPCs

Since September 1998, numerous wells were drilled on the terrace to better define the areal extent of contamination. Terrace background ground water quality could not be determined because no water was present in any of the wells drilled for background (wells 800 through 803). Almost all wells on the terrace are either screened in the alluvium or the weathered Mancos Shale and represent the quality of the shallow ground water system. Therefore, no

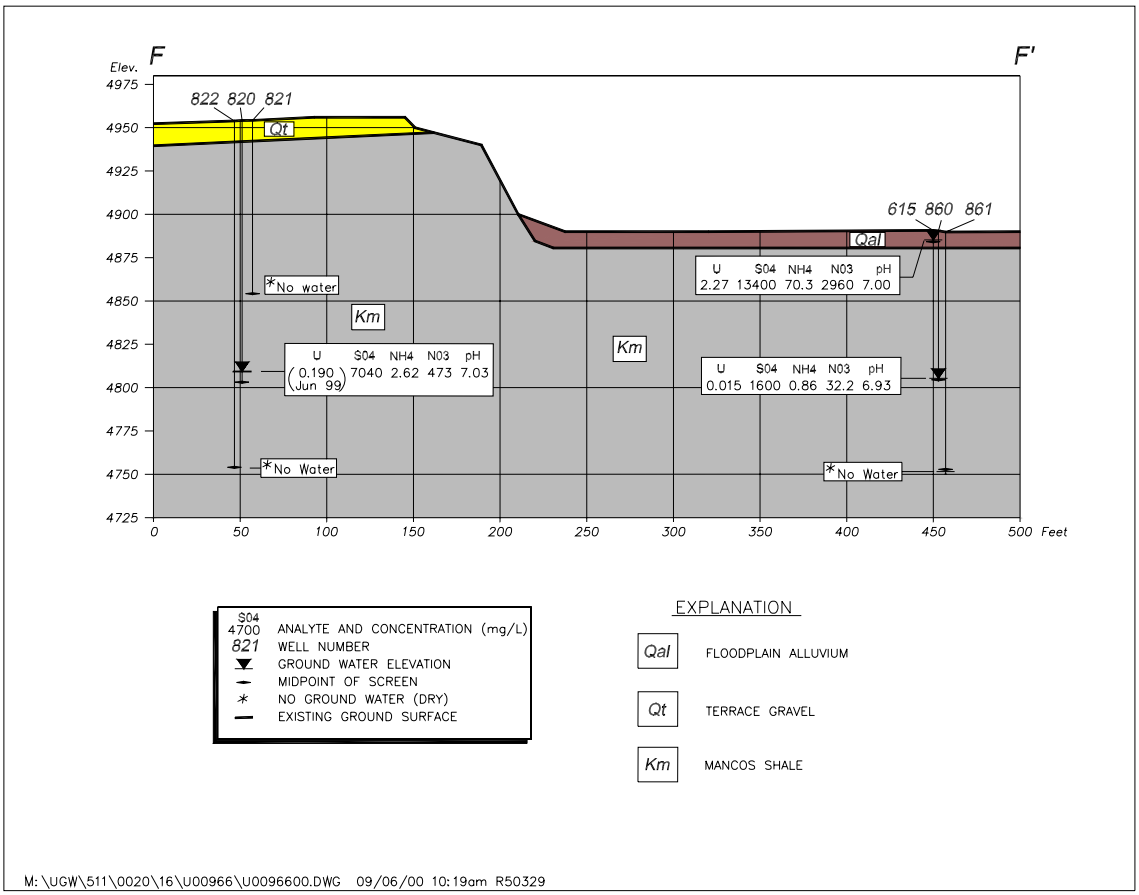


Figure 4–31. Cross Section F–F' Showing Distribution of U, SO_4, NH_4, NO_3, and pH in Ground Water. February 2000 Sampling; Samples not Analyzed in February 2000 are Indicated by an Alternate Sampling Date. Location of F–F' is shown on Plate 3.

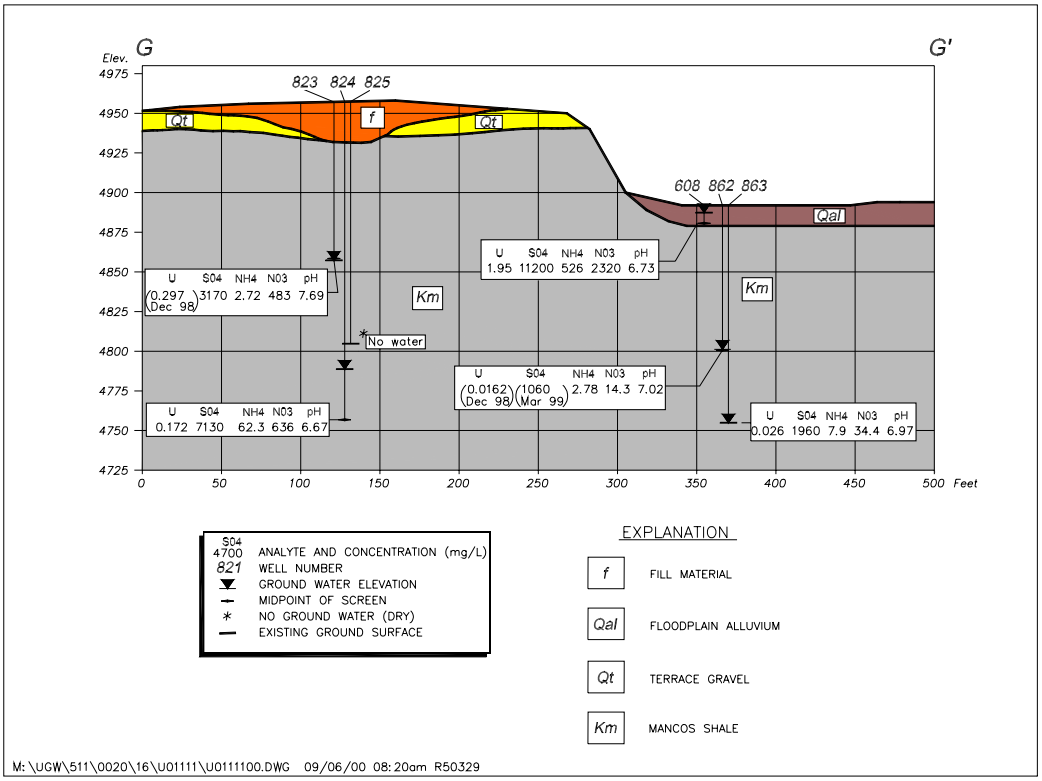


Figure 4–32. Cross Section G–G' Showing Distribution of U, SO_4, NH_4, NO_3, and pH in Ground Water. February 2000 Sampling; Samples not Analyzed in February 2000 are Indicated by an Alternate Sampling Date. Location of G–G' is shown on Plate 3.

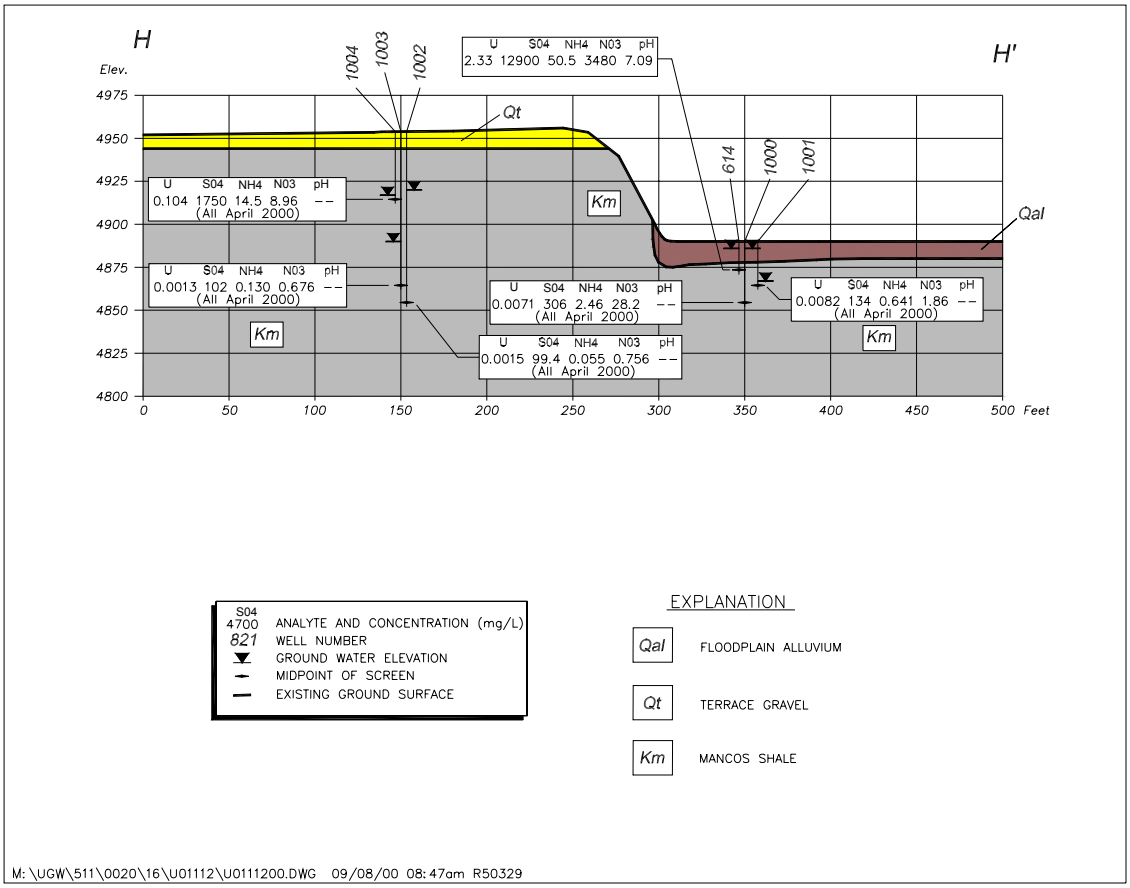


Figure 4–33. Cross Section H–H' Showing Distribution of U, SO_4, NH_4, NO_3, and pH in Ground Water. February 2000 Sampling; Samples not Analyzed in February 2000 are Indicated by an Alternate Sampling Date. Location of H–H' is shown on Plate 3.

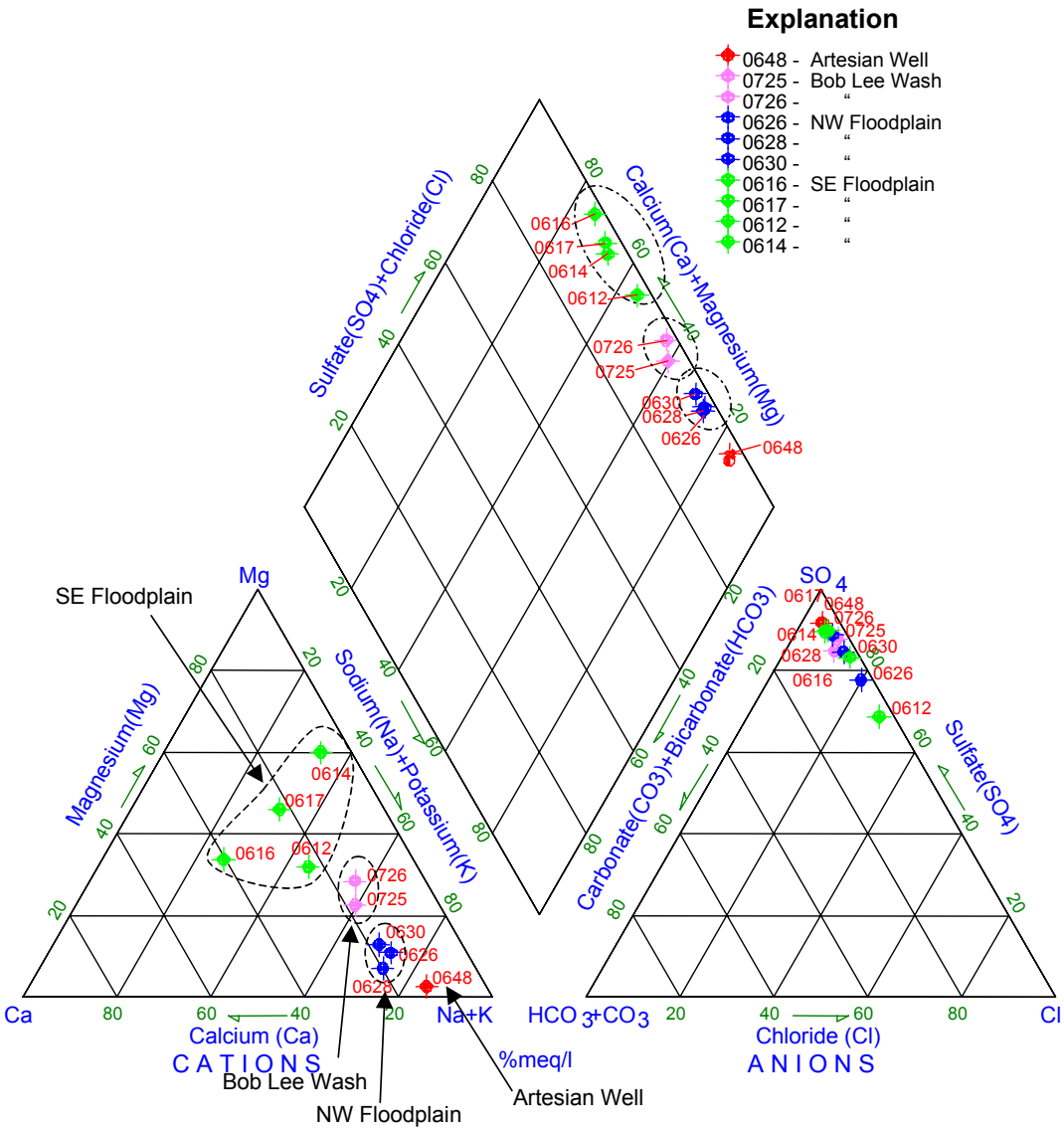


Figure 4–34. Piper Diagram of Artesian Well 648 Water, Shallow Ground Water in the Bob Lee Wash Area, and Ground Water in the Southeast and Northwest Portions of the Floodplain

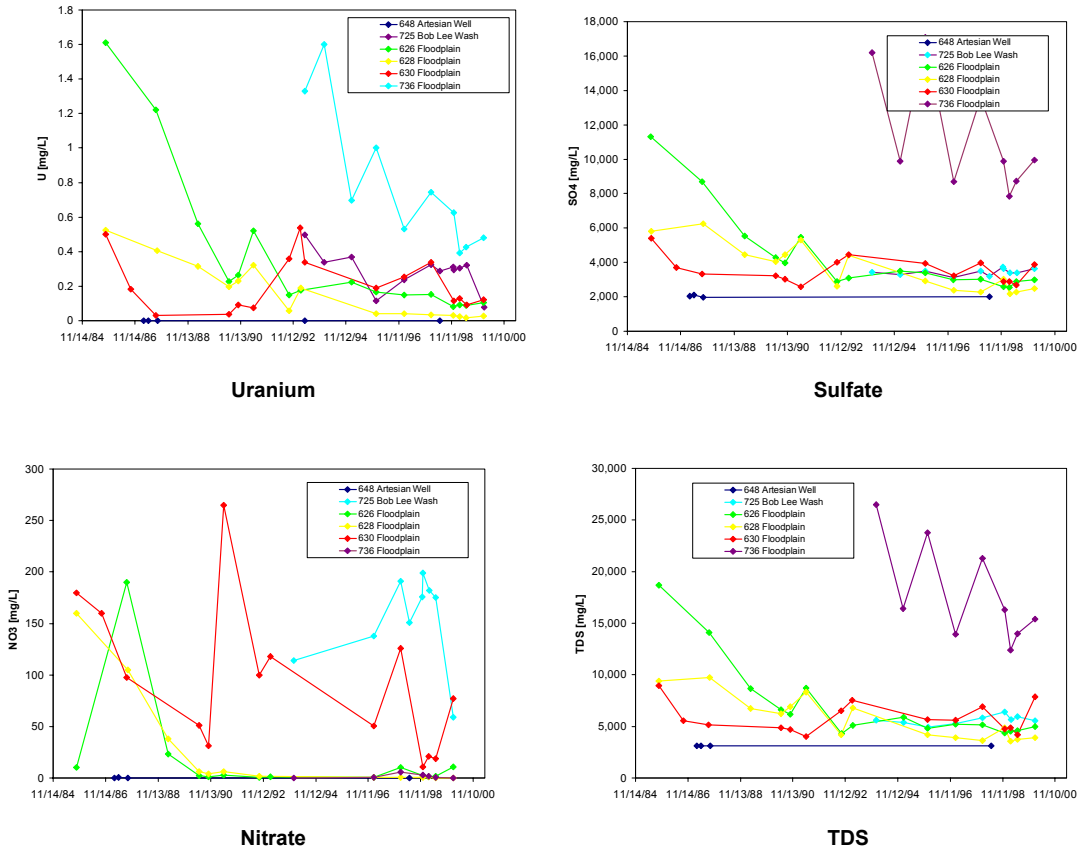


Figure 4–35. Changes in Concentrations of U, SO_4, NO_3, and TDS Over Time for Artesian Well 648 Water, Shallow Ground Water in the Bob Lee Wash Area, and Ground Water in the Northwest Portion of the Floodplain

contamination plume map for the ground water in the unweathered Mancos Shale could be created. Isotopic and other data strongly suggest that some contamination in the irrigated area west of U.S. Highway 666 is not millsite-related. In particular, some (if not most) of the uranium, selenium, and sulfate is probably derived from leaching of Mancos Shale. See Section 4.4.8 for further discussion.

The highest uranium concentration (3.08 mg/L) in the terrace system was detected in samples from well 826, which is near the former mill buildings and ore storage area (Figure 4–36). Ground water samples from wells 819 and 602, which were completed to depths of 31 ft and 96 ft, respectively, in the weathered Mancos Shale contained 1.39 and 0.726 mg/L uranium, respectively. Uranium concentrations in samples from well 602 have decreased slightly during the last 11 years, from about 1.4 to 0.7 mg/L (Figure 4–37). The southern extent of the uranium plume in the terrace alluvium is at the buried escarpment. Samples from alluvial wells 603 and 731 southeast of the disposal cell have uranium concentrations below the MCL. Uranium concentrations in samples from well 603 have not exceeded the MCL since 1990. A sample from well 830, which is completed in weathered Mancos Shale, had a uranium concentration of 0.0051 mg/L. Samples from wells 1048 and 1049 (installed in December 1999) and wells 1057 and 1059 (installed in March and April 2000) have uranium concentrations exceeding the MCL and show that the plume extends southeastward, just north of the buried escarpment, to Many Devils Wash (Figure 4–36).

Nitrate and ammonium complexes were used during the milling process as ion exchange strippers to concentrate uranium (Merritt 1971). Probably the most important source of nitrate in the terrace ground water has been from the oxidation of ammonia that was used during the milling process to adjust the pH of the slurry. Fluids leaked from the poorly lined raffinate ponds, as noted by the U.S. Department of Health, Education, and Welfare (1962). Ground water was analyzed for nitrate and ammonium but not nitrite during routine UMTRA sampling. Some field samples of floodplain and terrace ground water collected by personnel with the NABIR Program contained nitrite concentrations that were less than 5 mg/L. One exception was a sample from well 819 that had about 14 mg/L nitrite.

Nitrate concentrations exceeded the MCL in most samples of terrace system ground water. A notable exception is the far northwest part of the site area (Figure 4–38). The highest nitrate concentration (8,790 mg/L) in the terrace system was detected in a sample from well 813, which is about 1,700 ft southwest of the disposal cell. The nitrate plume coincides with a buried ancestral river channel on the terrace south of the disposal cell. High concentrations extend west of U.S. Highway 666, where a sample of ground water from well 841 contained 1,990 mg/L.

Since 1990, the concentrations of nitrate in samples from well 603, southeast of the disposal cell, have increased significantly and are still increasing. Although well 813 samples had the highest nitrate concentrations (8,790 mg/L), the sum of nitrate and ammonium concentrations is highest in well 603 samples (10,890 mg/L, expressed as nitrate). Forty-five percent of the nitrogen in well 603 has been oxidized to nitrate. If all the ammonium is oxidized, the nitrate concentrations could increase to 10,890 mg/L at well 603. It is not apparent why the combined ammonium and nitrate concentrations in samples from well 731, which is just south of well 603, were much lower (1,490 mg/L, expressed as nitrate). It is possible that activity at the adjacent NECA gravel pit (excavating and washing of gravel) has affected the geochemical conditions at well 603 and oxidized the ammonium.

Ammonium concentrations in the terrace ground water are shown in Figure 4–39. Concentrations are highest south of the disposal cell at wells 603 and 1057, and another area of elevated concentrations is immediately west and north of the disposal cell.

Sulfate was used in the form of sulfuric acid in the milling process. The spatial distribution of sulfate in the shallow terrace ground water system has three maxima (Figure 4–40). Highest sulfate concentrations in ground water at the terrace are 17,800 mg/L at well 1049 in the Many Devils Wash area. The second maximum is in the millsite area around wells 602 and 819. These wells are completed at depths of 96 and 31 ft, respectively, and have sulfate concentrations of 17,400 mg/L and 13,400 mg/L, respectively. The third maximum in the terrace system is in samples from wells 812 and 815, with sulfate concentrations of 15,600 mg/L and 15,300 mg/L, respectively. As with the nitrate plume, the sulfate plume coincides with the ancestral river channel south of the disposal cell. The similarity in the extent of nitrate and sulfate contamination is also observed west of U.S. Highway 666. Sulfate concentrations in samples from wells 833, 844, 832, and 841, in a north to south trend, are higher than samples from wells immediately to the west.

Concentrations of manganese, selenium, vanadium, and molybdenum in the terrace ground water are shown in Figure 4–41 through Figure 4–44, respectively. Manganese concentrations are highest generally around the north, west, and south sides of the disposal cell (Figure 4–41). An isolated high concentration of manganese in ground water is in well 1060 near the south edge of the terrace system. Selenium concentrations in terrace ground water are highest southwest of the disposal cell in wells 812, 814, 841, and 832 situated in the area of the buried ancestral river channel where a bedrock swale has formed a sump area (Figure 4–42). Selenium concentrations are also high at wells 1048 and 1049 adjacent to Many Devils Wash. Vanadium analyses were performed on terrace ground water samples collected in June 1999 and for three wells (603, 730, and 830) in February 2000. Results of the analyses are shown on Figure 4–43. Vanadium concentrations in all but a few of the samples were below the detection limit. The only wells in which vanadium concentrations greater than 0.1 mg/L were detected are immediately south and southeast of the disposal cell (wells 603, 730, and 830). Well 730 is the only well with concentrations that exceed a human-health risk-based benchmark of 0.33 mg/L, which was established as an ACL at the Rifle, Colorado, UMTRA site (DOE 1999j). Well 730 is in the area of former raffinate ponds where vanadium liquor from milling was disposed. Concentrations decreased rapidly with distance from the disposal cell. The isolated occurrence of vanadium and low frequency of detection are sufficient to exclude it as a COPC.

Terrace ground water samples were collected for molybdenum analysis in June 1999 and February 2000. The objective was to determine if molybdenum concentrations in irrigation-related water were different than those in millsite-related water. Results of molybdenum sampling from February 2000 are shown on Figure 4–44. The irrigated area and millsite-related area show no distinct difference in concentrations, and the distribution of molybdenum shows no well-defined pattern. All concentrations, however, are well below the MCL of 0.1 mg/L. The only exception is at well 824 just northeast of the disposal cell in a filled drainage where an elevated concentration of 0.856 mg/L occurs (Plate 1). Well 824 is not shown on Figure 4–44 because the screen is set in deep Mancos Shale and thus is not part of the terrace system ground water. Because of the low concentrations of molybdenum for the vast majority of samples, it is not considered to be a COPC.

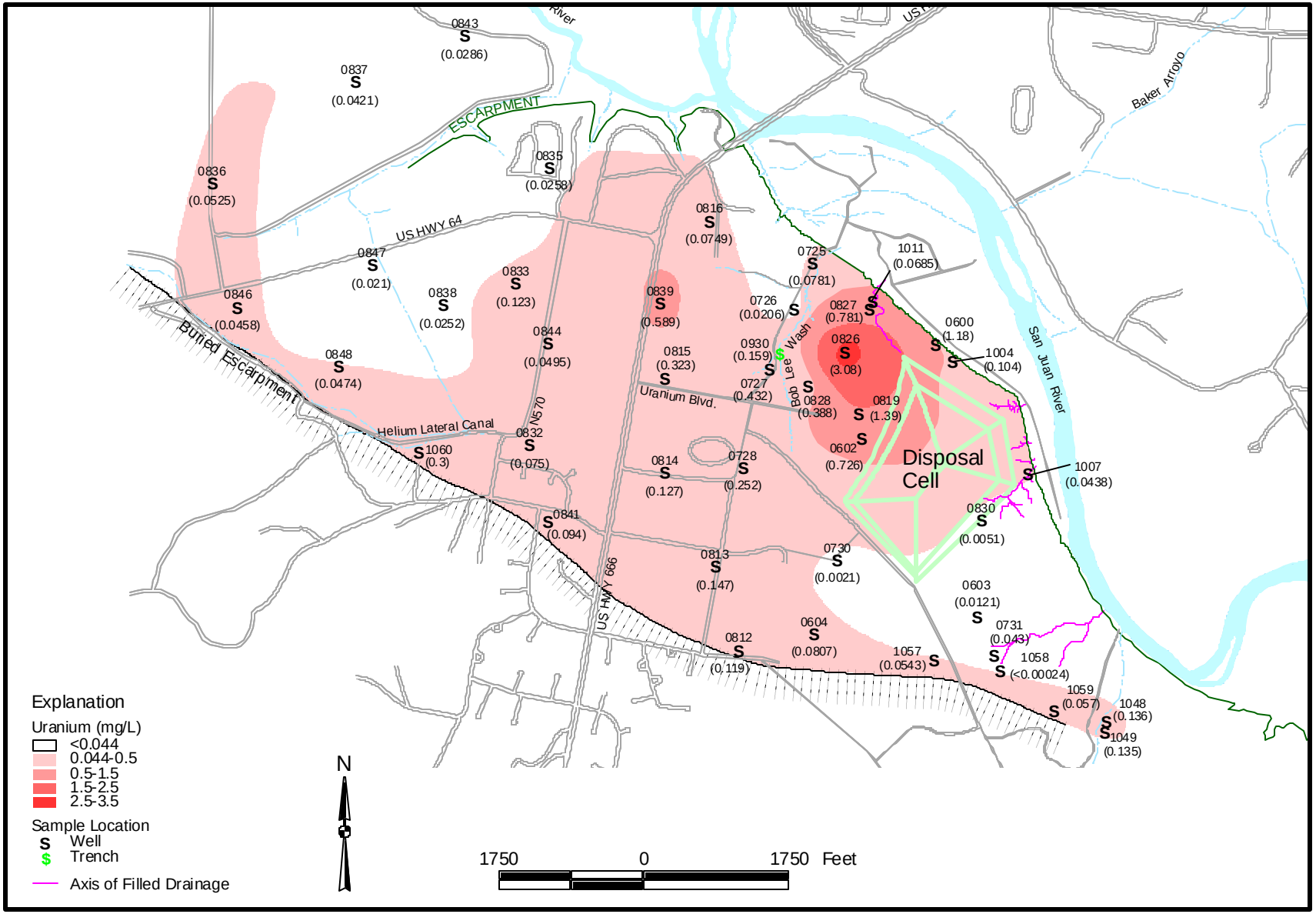


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Figure 4–36. Uranium Concentrations in the Terrace System Ground Water

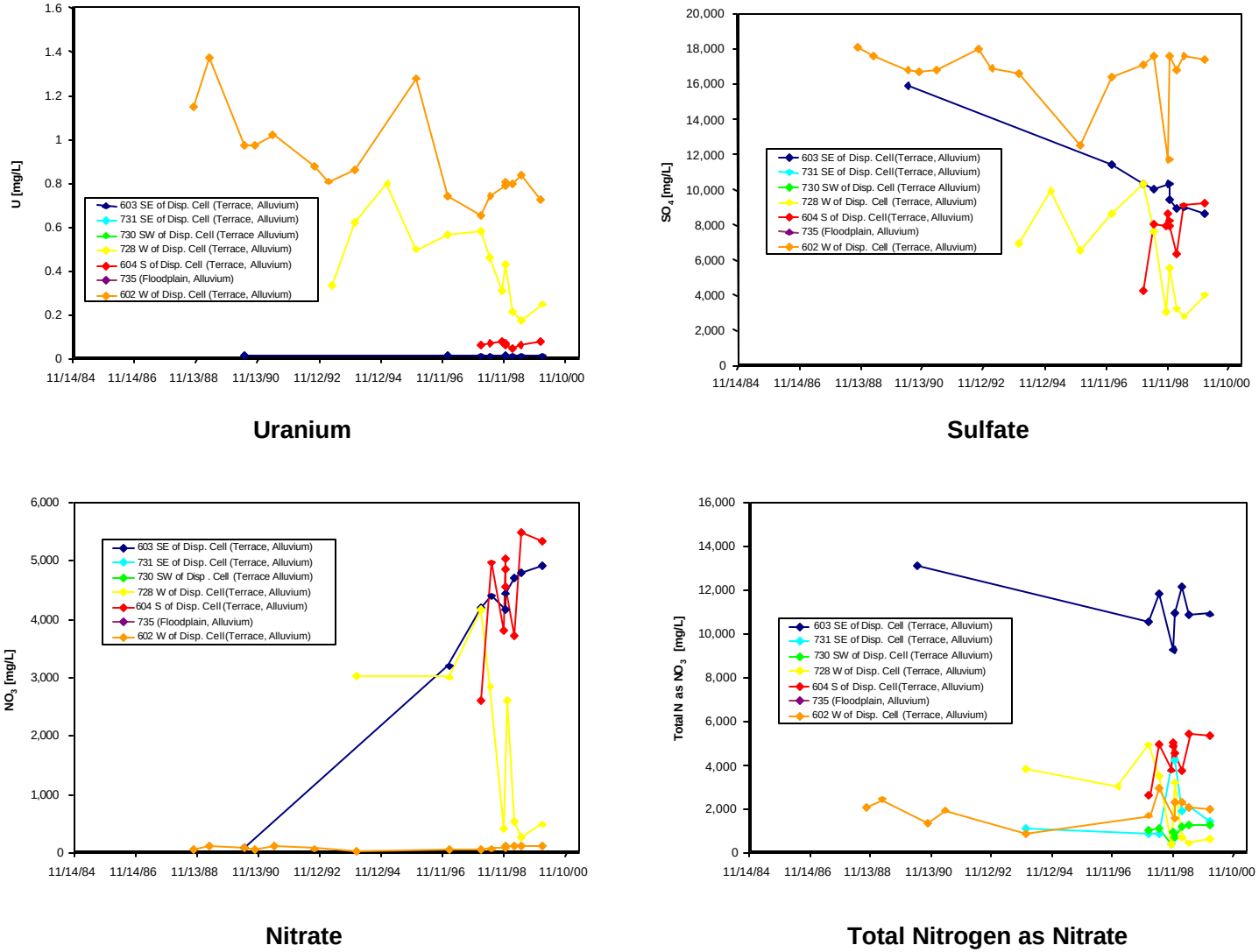

Figure 4–37. Changes in Concentrations of U, SO_4, NO_3, and Total N Over Time for Ground Water from Selected Terrace System Wells and from Floodplain Well 735

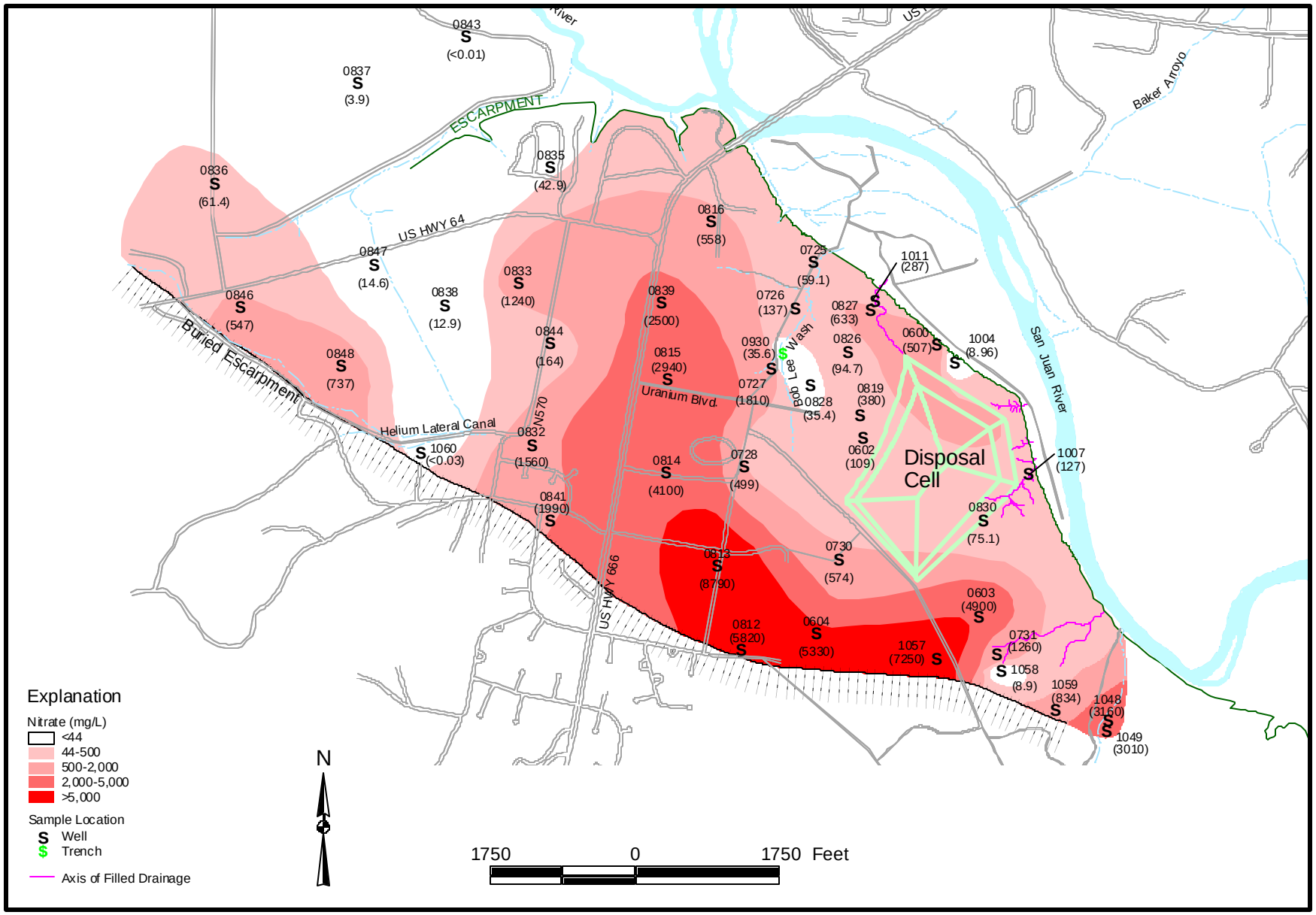

Figure 4–38. Nitrate Concentrations in the Terrace System Ground Water

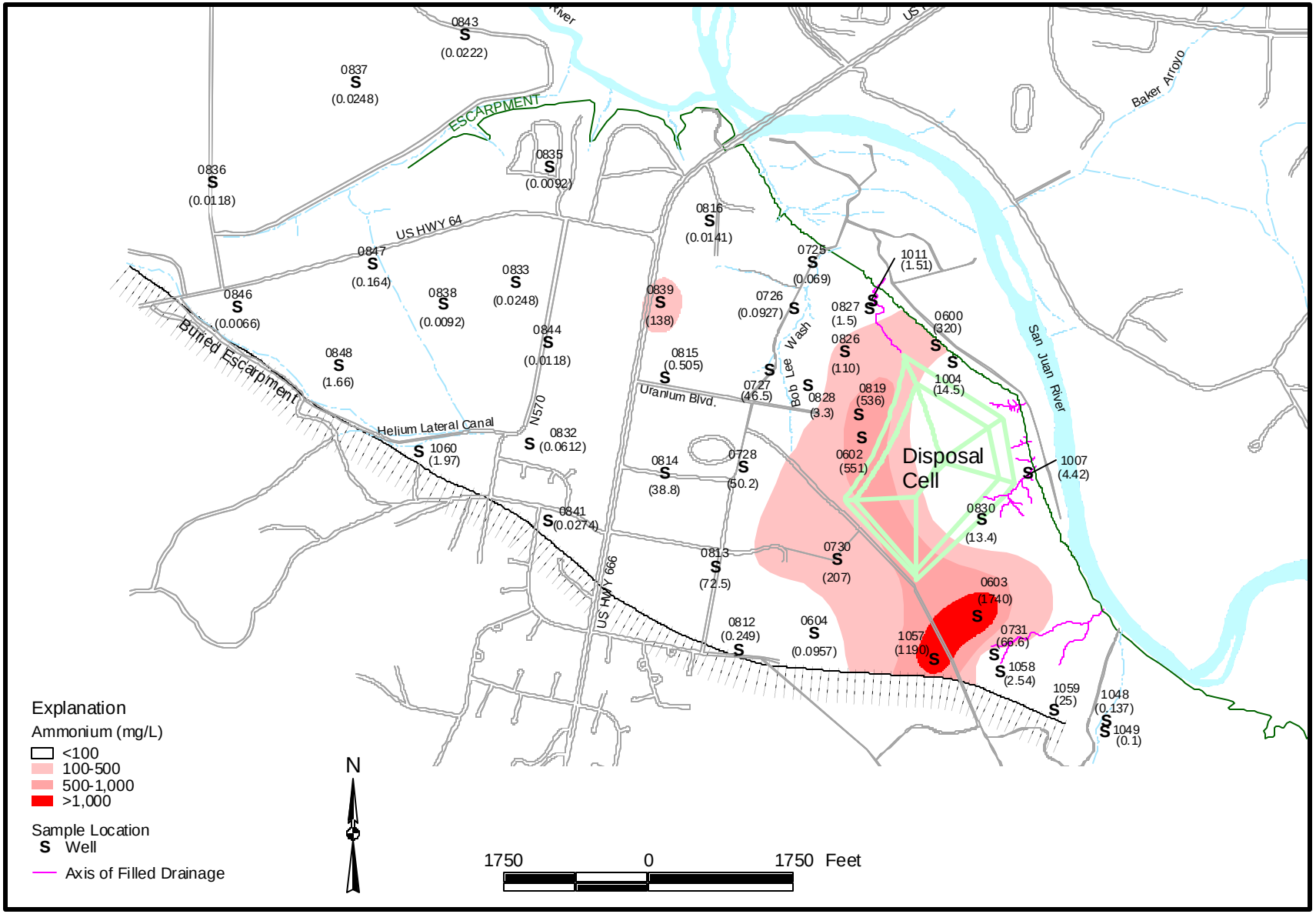

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Figure 4–39. Ammonium Concentrations in the Terrace System Ground Water

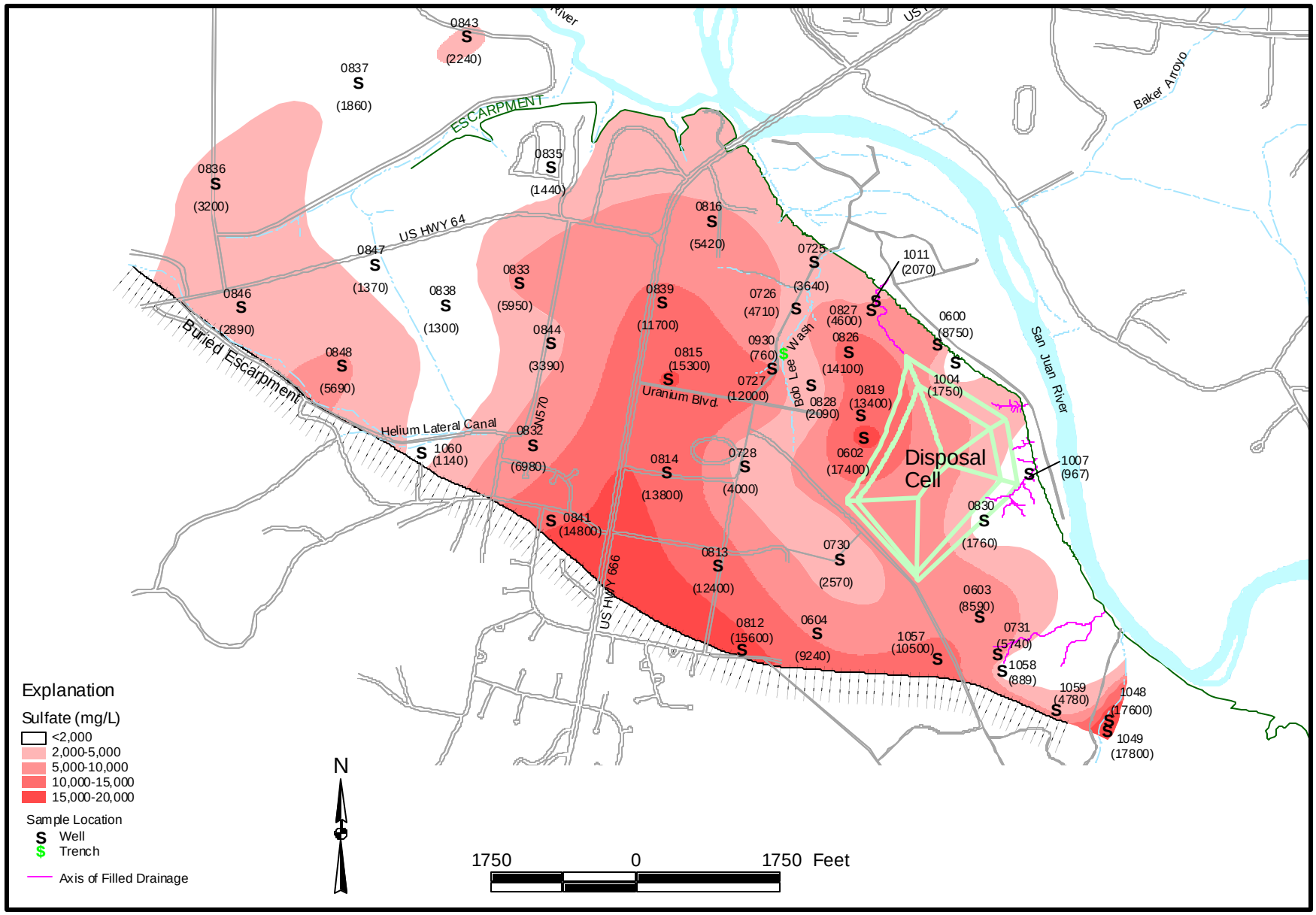

Figure 4–40. Sulfate Concentrations in the Terrace System Ground Water

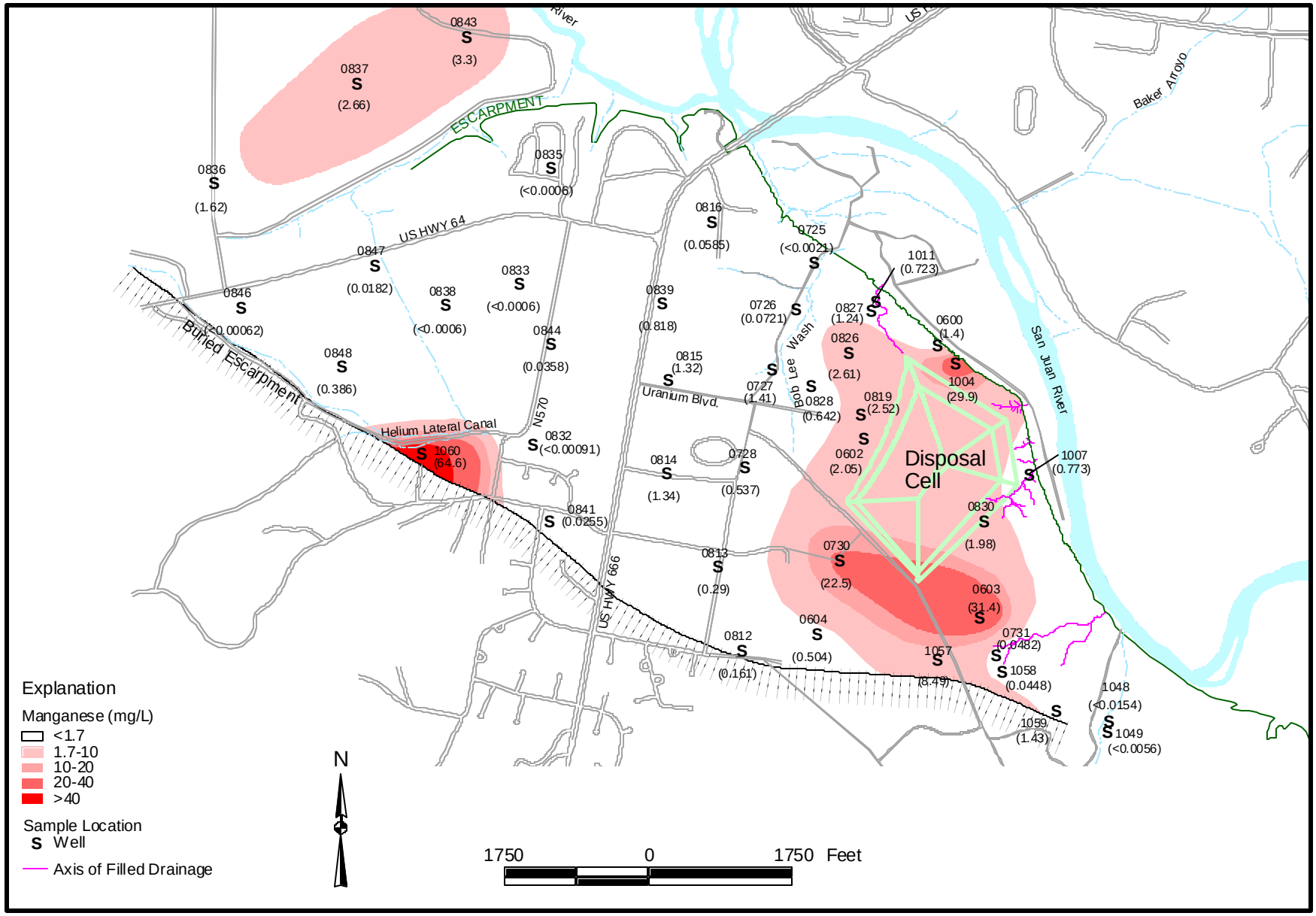

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Figure 4–41. Manganese Concentrations in the Terrace System Ground Water

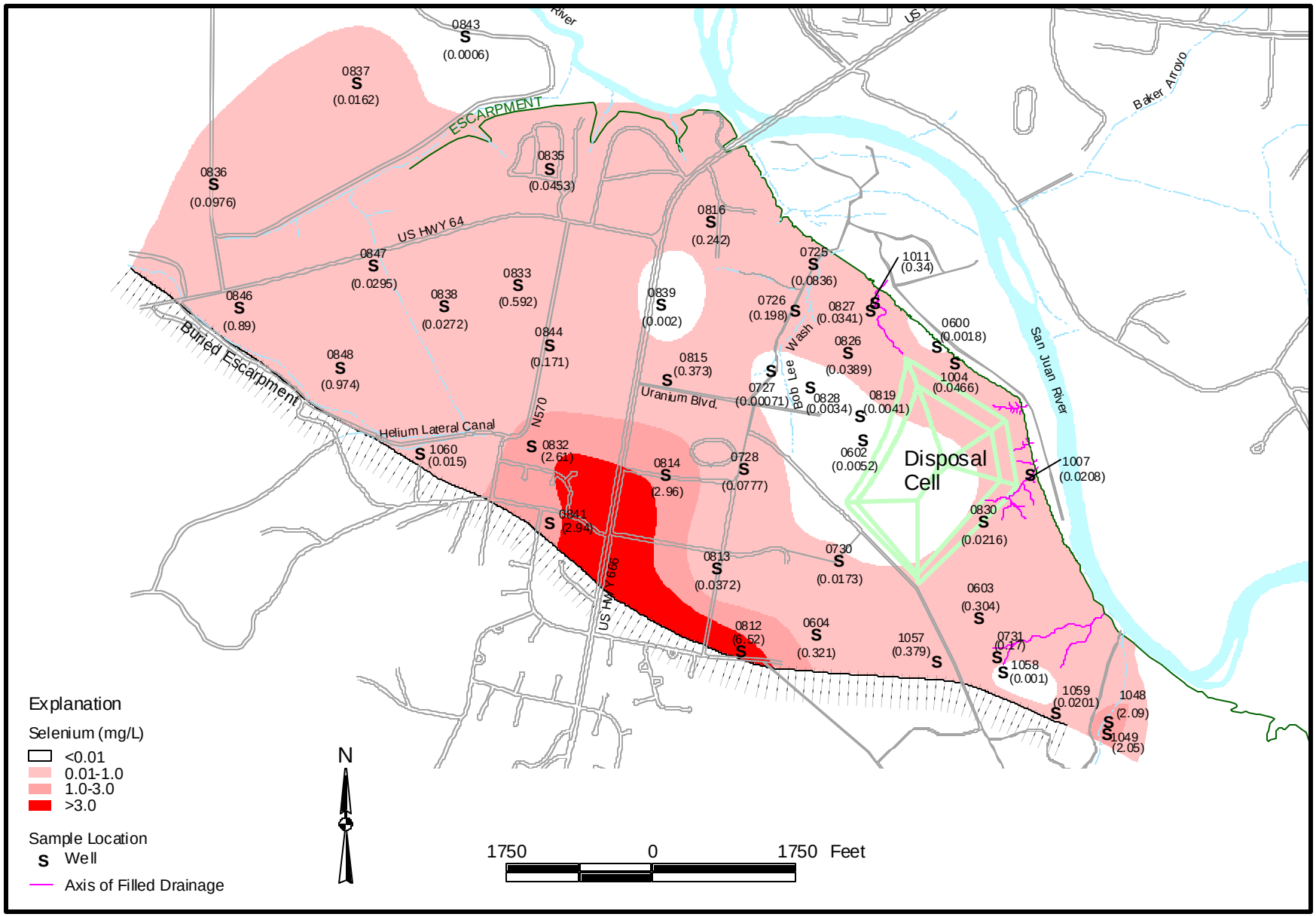

Figure 4–42. Selenium Concentrations in the Terrace System Ground Water

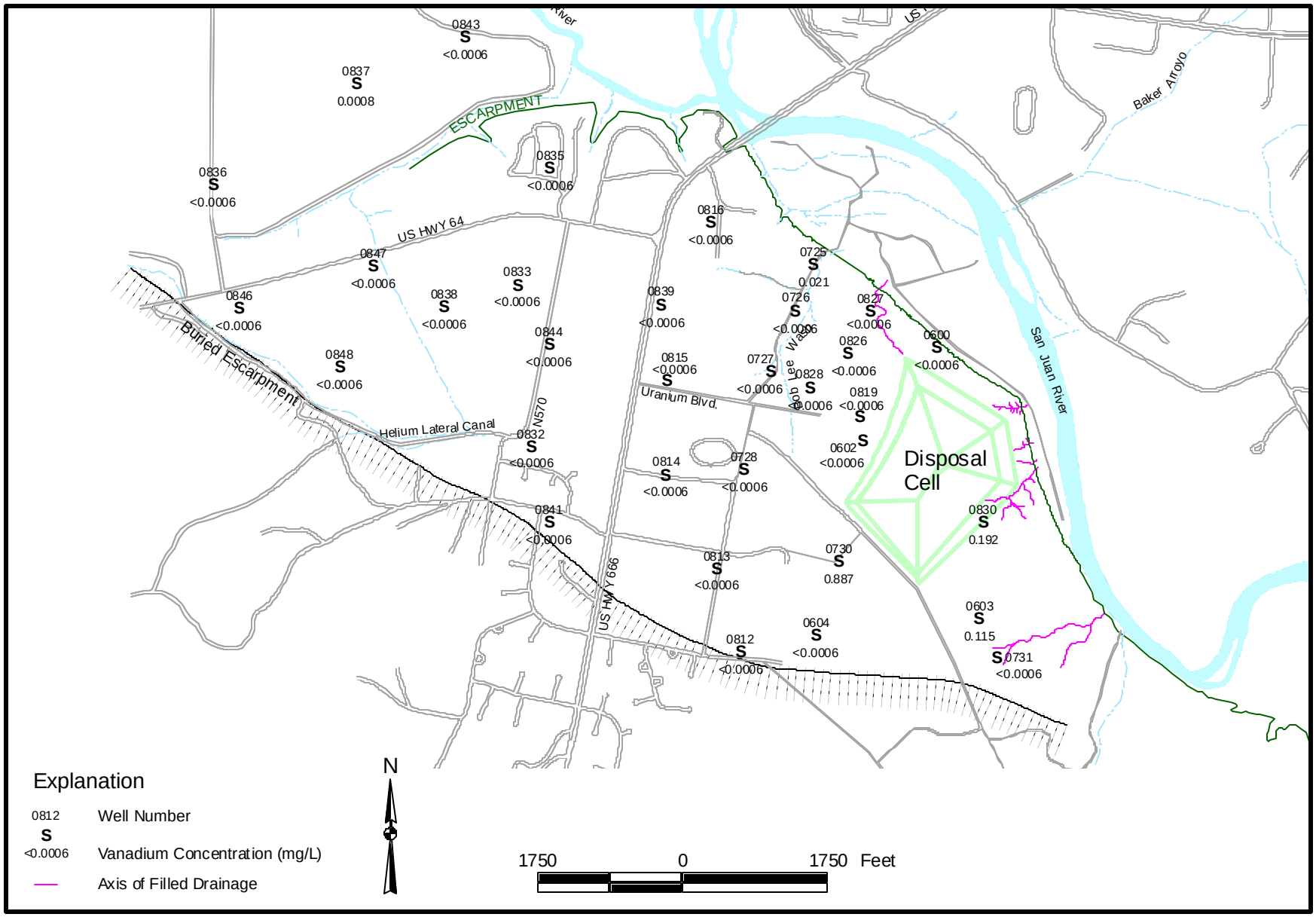

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Figure 4–43. Vanadium Concentrations in the Terrace System Ground Water

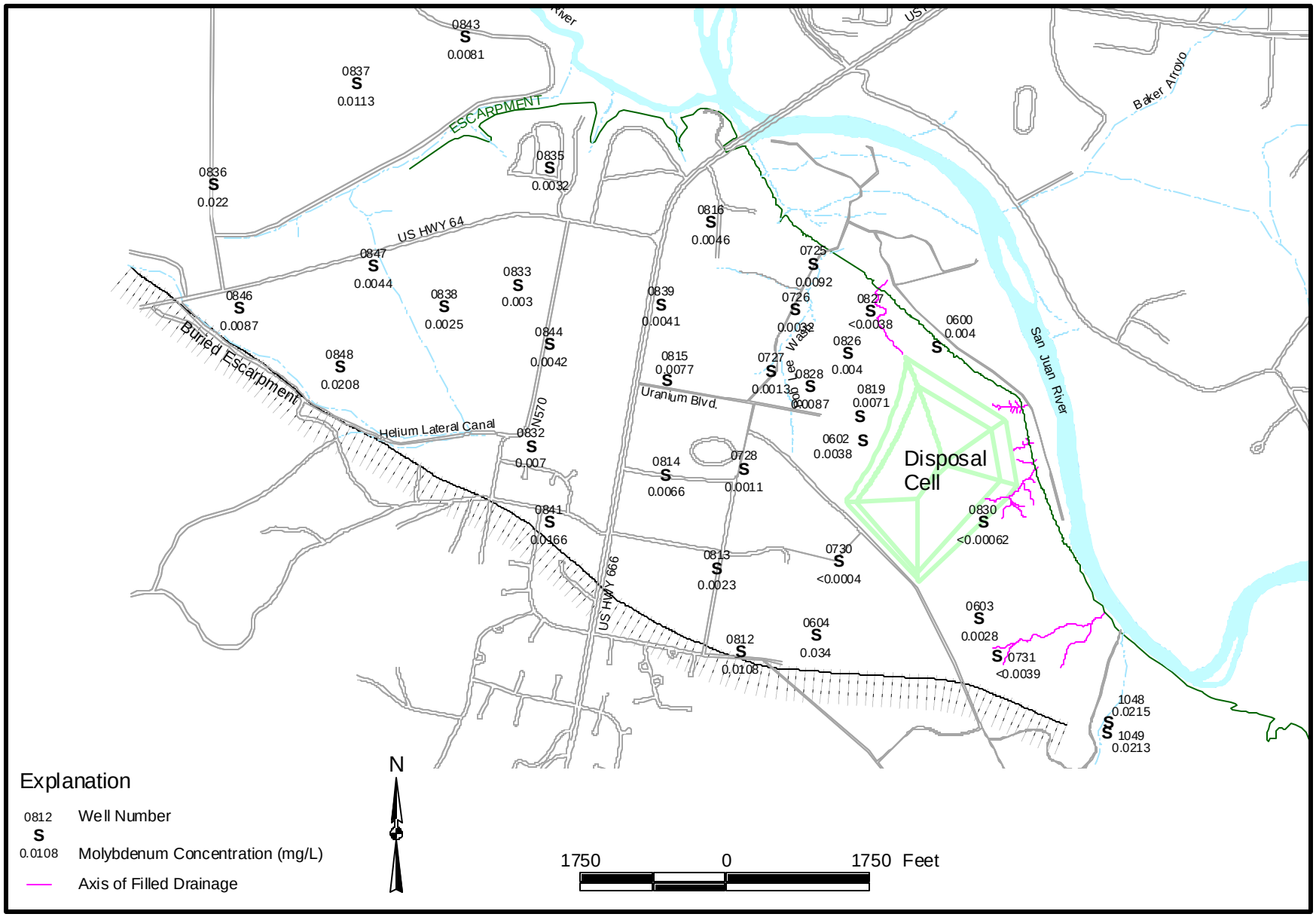


Figure 4–44. Molybdenum Concentrations in the Terrace System Ground Water

Terrace ground water has two main areas of contamination. Ground water near the former mill buildings and ore storage area has high concentrations of uranium and sulfate. The highest uranium concentrations are in the alluvial part of the system, whereas the sulfate contamination is deeper (about 100 ft) in the weathered Mancos Shale, suggesting that uranium is retained more than sulfate in the shallow part of the ground water system. The extent of the sulfate and nitrate contamination south of the disposal cell suggests that processing water from the former raffinate ponds is the source. Oxidation of ammonium in ground water near well 603 has caused increasing nitrate concentrations.

Chemical Character of Terrace Ground Water Near the NECA Gravel Pit

Samples collected southeast of the disposal cell close to the NECA gravel pit have high concentrations of sulfate and nitrate. The extent of high concentrations farther north toward the escarpment is difficult to determine because most of the terrace alluvium has been removed by the gravel operation. The only two monitor wells in that area are the dry wells 804 and 805, which were completed to depths of 70 and 50 ft, respectively, in Mancos Shale. Seepage water has been observed at the escarpment in the area of salt deposit sample 922, but this water has not been sampled. Contaminated water likely migrates through the weathered Mancos Shale beneath the gravel pit toward the San Juan River and Many Devils Wash.

Origin of Surface Water in Many Devils Wash

High nitrate concentrations in samples from locations 886 and 889 (2,930 and 3,520 mg/L, respectively) suggest that the surface water in Many Devils Wash represents ground water from the terrace system that has migrated eastward (Plate 1). A Piper diagram (Figure 4–45) illustrates the composition of escarpment seep water (seeps 425 and 426), the surface water at Many Devils Wash, and selected ground water locations on the terrace. Water at seep 426 plots approximately in the same area as the ground water from wells 725 and 600. The chemical signature of surface water from Many Devils Wash (locations 886 and 889) is different from that of the ground water samples. The Many Devils Wash water was expected to be similar to the water from wells 603 and 731, which are between Many Devils Wash and the disposal cell; instead, a plot of the water from those wells is closer to that of water from well 827 and seep 425. Also, the contaminant chemistry of wells 1057 through 1059 shows little relationship to surface water in Many Devils Wash. These wells are just west of the wash along a postulated east-flowing ground water pathway that supplies water to Many Devils Wash. Because ground water chemistry of the Many Devils Wash surface water does not resemble that of ground water to the west, it can be assumed that the Many Devils Wash surface water has incorporated significant chemical character from the Mancos Shale.

Chemical Character of Terrace Ground Water in Wells Near the High School

Terrace wells 847 and 848, south of U.S. Highway 64 on the Shiprock High School property, were drilled for irrigation purposes by a local company to estimated depths of 92.5 ft and 145 ft, respectively. The lengths of the well screens are unknown. The ground water chemistry is much different in these two wells, as indicated by their separation on a Piper diagram (Figure 4–46). Ground water in well 847 has a chemical signature similar to water in well 838. Ground water from wells 836/846 and 844/833 plot in similar locations for the cation composition. Ground water from well 848 has a composition intermediate between water from wells 832 and 841. However, well 848 has nitrate concentrations wells below either 832 or 841, indicating that

simple mixing cannot account for its complete chemical composition. Because the completion information for well 848 is not known, it may be that the wells are influenced by alluvial ground water, or that ground water in the Mancos Shale at a depth of 145 ft has naturally high concentrations of sulfate and nitrate.

Special Study of Kjeldahl Nitrogen

Historically, high concentrations of dissolved nitrogen (as nitrate and ammonium) were present in ground water, particularly in the terrace system. Much of the nitrate is likely to have been derived by oxidation of ammonium used in the milling process. Some nitrate and ammonium may, however, be contributed to the ground water from natural sources such as rock weathering or from anthropogenic sources such as septic systems. Septic system waste is likely to contribute organically bound nitrogen in addition to nitrate and ammonium. Therefore, a limited study was conducted during the December 1998 water sampling to determine the concentrations of organically bound nitrogen in the terrace ground water.

Analysis of water using the Kjeldahl digestion method provides a measure of the combined concentrations of organically bound nitrogen and ammonium nitrogen. Therefore, the difference between Kjeldahl-N and ammonium-N is the concentration of organically bound nitrogen.

Ground water from 14 terrace wells was analyzed. Kjeldahl nitrogen concentrations ranged from 0.13 to 1,010 mg/L (Table 4–16). In eight samples (samples with negative values in column 6 of Table 4–16), the NH_4-N concentration exceeded Kjeldahl-N, indicating that the Kjeldahl method does not include all of the ammonium. However, there was a reasonable correlation between Kjeldahl-N and NH_4-N concentrations. For example, the sample from well 603 had the highest Kjeldahl-N concentration (1,010 mg/L) and also had the highest NH_4-N concentration (1,470 mg/L). The similarities between the Kjeldahl-N and NH_4-N concentrations suggest that there is little organically bound nitrogen dissolved in these ground water samples.

Kjeldahl-N and NH_4-N concentrations show little correlation with NO_3-N concentrations (Table 4–16). For example, water from well 812 had low concentrations of Kjeldahl-N (0.13 mg/L) and NH_4-N (0.37 mg/L) but had a high concentration (1,362 mg/L) of NO_3-N.

Table 4–16. Concentrations in mg/L of Kjeldahl Nitrogen, Nitrate (as N), and Ammonium (as N), Sampled in December 1998.

Well	Sample Date	Kj-N	NO_3-N	NH_4-N	Kj–NH_4 as N[a]
602	12/07/98	705	28	490	214.99
603	12/07/98	1,010	1,005	1,470	-460.04
604	12/07/98	0.275	1,030	0.08	0.19
728	12/08/98	65.4	587	141	-75.38
731	12/08/98	6.92	926	37	-30.26
812	12/08/98	0.13	1,362	0.37	-0.24
813	12/07/98	23.7	1,766	52	-28.49
814	12/09/98	10.8	910	15	-3.82
819	12/09/98	254	28	253	1.22
830	12/08/98	8.43	19	11	-2.54
835	06/04/99	<0.142	6.23	<0.005	na
836	06/03/99	0.646	13.07	0.01	0.63
841	12/08/98	1.42	524	1.77	-0.35
846	06/04/99	<0.576	117	0.01	na

[a]Kjeldahl-N minus NH_4-N

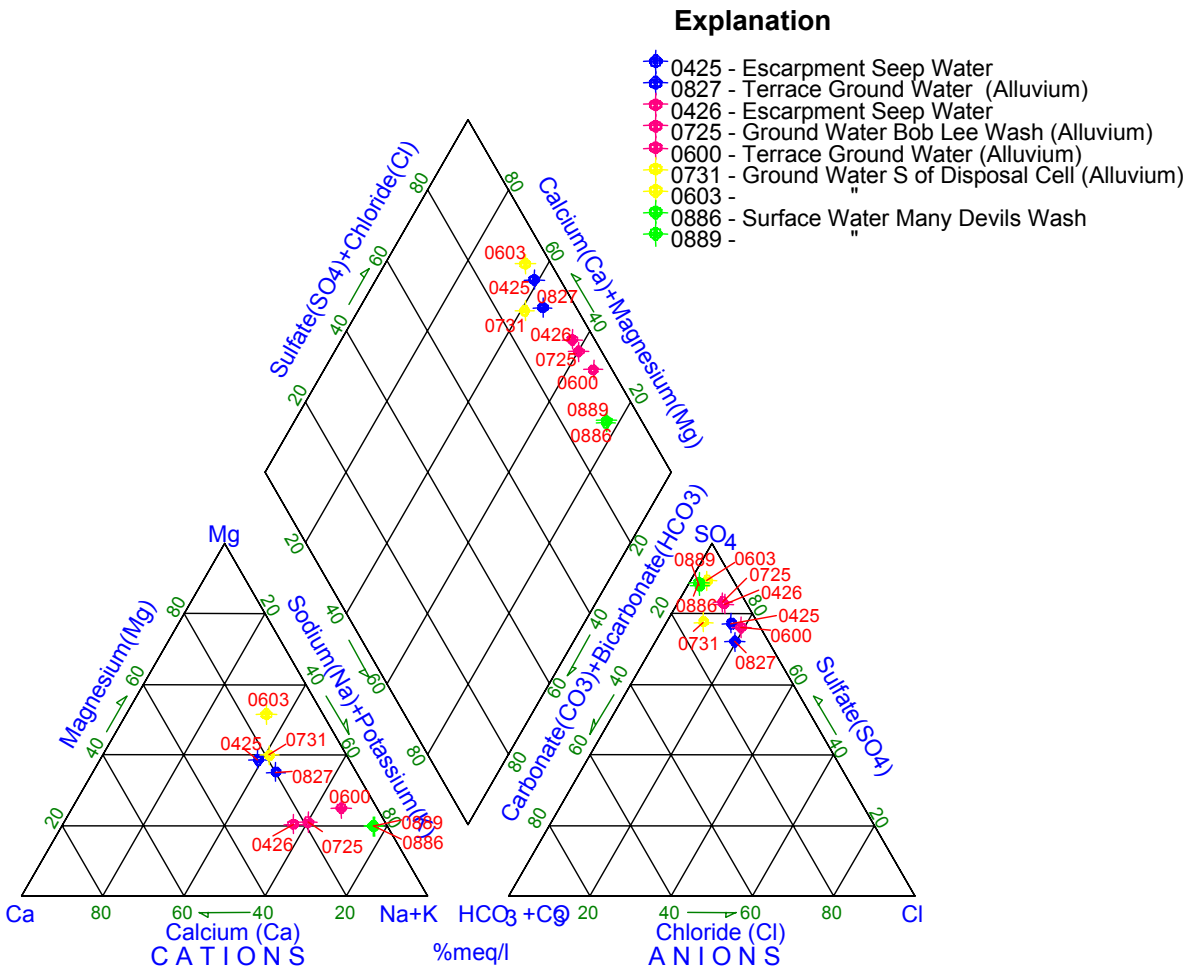


Figure 4–45. Piper Diagram Comparing Composition of Escarpment Seep Water to Terrace System Ground Water and Many Devils Wash Surface Water

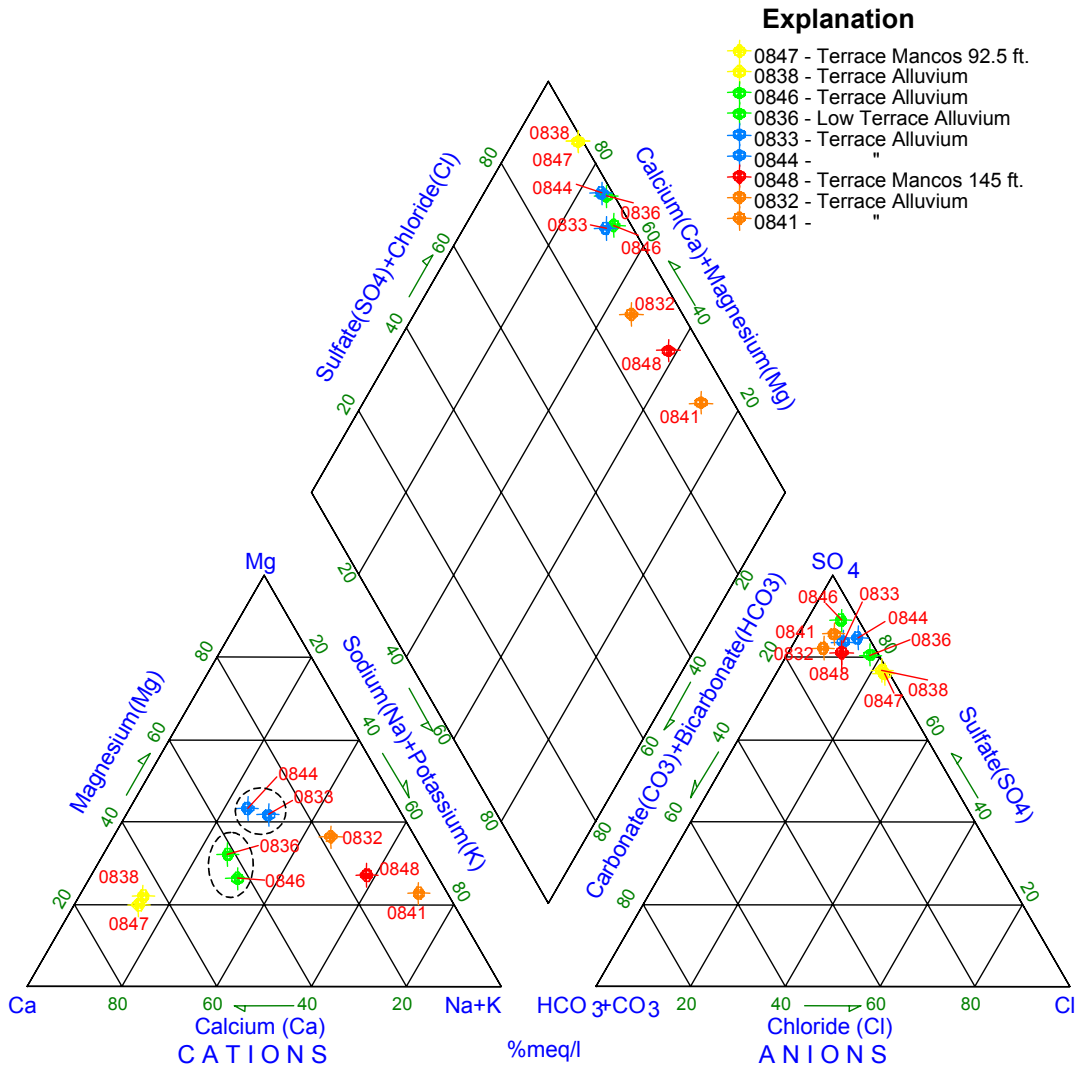


Figure 4–46. Piper Diagram Showing Composition of Water in High School Wells Compared to Other Terrace System Ground Water

Special Study of Radium/Thorium Isotopes

Historically, Ra-226 and Ra-228 combined concentrations in ground water from several terrace wells have exceeded the UMTRA MCL of 5 picocuries per liter (pCi/L) (Appendix F). Ra-226 is in the decay chain of U-238, so its presence in the ground water is due to its high concentrations in the uranium ores. Because Ra-228 is in the Th-232 decay chain, it is not expected to occur in elevated concentrations in the ores. Curiously, the radiometric concentrations of Ra-228 often exceeded those of Ra-226 in the terrace ground water. Relevant portions of the U-238 and Th-232 decay chains are shown below ($T_{1/2}$ = half life in years).

U-238 ---------- Th-230----------Ra-226
$T_{1/2} = 4.51 \times 10^9$ 8×10^4 1,600

Th-232---------- Ra-228----------Th-228
$T_{1/2} = 1.41 \times 10^{10}$ 6.7 1.913

The high ratios of Ra-228 to Ra-226 prompted a limited investigation of the isotopic decay chains during the February 2000 ground water sampling. The main objective of the study was to determine if the dissolved Ra-228 was supported by dissolved Th-232. Samples from four terrace wells (600, 602, 727, and 815) that had high concentrations of Ra-226 were used in the study. Ra-226 concentrations ranged from 0.92 to 4.73 pCi/L, and Ra-228 concentrations ranged from 3.06 to 11.1 pCi/L (Table 4–17). The Ra-228/Ra-226 ratio exceeded unity in all but one sample (Table 4–17).

Table 4–17. Concentrations of Radium and Thorium Isotopes, February 2000 Sampling (filtered samples)

Well	Ra-226 pCi/L	Ra-228 pCi/L	Th-228 pCi/L	Th-230 pCi/L	Th-232 pCi/L	Ra-228/Ra-226
600	0.92	4.01	<0.4	0.21	<0.13	4.36
602	4.73	11.1	<1.44	0.81	<0.32	2.35
727	3.05	5.85	0.52	0.24	<0.1	1.92
815	3.62	3.06	<0.3	0.37	<0.06	0.85

Note: Both filtered and unfiltered samples were analyzed. Only minor differences were observed between these, so only filtered data are shown.

Because Ra-228 has a relatively short half life (6.7 years), Ra-228 deposited in the tailings would have decreased substantially in the 30 years or so since the mill operated. Thus, the Ra-228 must be supported by a parent isotope. Dissolved Th-232 concentrations were all less than 0.32 pCi/L and, therefore, do not support the much higher concentrations of Ra-228.

Thorium is fairly immobile in ground water due largely to the low solubility of thorium oxide minerals. Thus, Th-232 is apparently present in the solid fraction (no data are available for the solid fraction) and is producing Ra-228, which is more soluble and continues to enter the ground water. The tendency for thorium to remain in the solid phase is also demonstrated by the low dissolved concentrations of Th-230, which do not support the dissolved Ra-226 (Table 4–17).

4.4.2.3 Organic Constituents

Organic compounds were used during the milling process for solvent extraction of uranium. Water samples collected from 16 wells (602, 603, 604, 728, 731, 812, 813, 814, 819, 824, 826, 827, 828, 829, 830, and 841), the NECA pond (849), seep 425, and a municipal water supply tap (299) were analyzed for volatile organic compounds by gas chromatography/mass spectroscopy (GCMS). A water sample from location 849, the NECA pond, was also analyzed for semivolatile organic compounds. The water supply sample (299) was collected from a hydrant near the Burger King restaurant, just northwest of well 816.

Concentrations of most volatile organics were less than detection limits in all samples. Acetone, bromodichloromethane, chlorodibromomethane, chloroform, silane, and silanol were detected in the municipal water supply sample and were not considered to be signatures of mill-related contamination. Some of these are known to be common laboratory contaminants. A few other alkanes were detected in ground water, but were only slightly above their detection limits and are not considered to be signatures of millsite contamination.

Alkenes and cycloalkenes were detected in some ground water samples and the NECA pond sample. Alkenes and cycloalkenes may be derivatives of the fuel oil used in the solvent extraction or may be from non-mill-related sources. All alkenes and cycloalkenes detected in the samples are listed in Table 4–18. The sample from the NECA pond had 67 micrograms per liter (µg/L) of 1,6,7-trimethyl-naphthalene and 12 µg/L of toluene. All alkene and cycloalkene concentrations in the wells were less than 7 µg/L. The concentrations are near the detection limits and are considered estimated values, as indicated by the "J" flag in Table 4–18. Because the concentrations are low and there is no obvious correlation with the disposal cell, these are not considered to be signatures of millsite contamination.

Some peaks were observed on the gas chromatograms that have not been identified (Table 4–18). Estimated concentrations of these peaks range up to 440 µg/L. Several high concentrations of unidentified chemicals were present in the NECA pond sample (849). The significant concentrations of unidentified chemicals in the NECA pond sample are probably polyaromatic hydrocarbons derived from highly weathered solvents. These solvents probably entered the NECA pond during its recent use and are not related to the milling operation. The several unidentified GCMS peaks observed in ground water samples from wells 728, 813, 819, and 824 are likely from the same compound; a volatile, nonhydrocarbon compound with low mass. Because these four wells are widely separated and not spatially related to the disposal cell, the concentrations of this volatile organic compound are not considered to be related to the milling operation.

4.4.3 Contaminants in Soils and Sediments

Two laboratory studies presented in this section address one of the data quality objectives defined in the Work Plan (DOE 1998d): "Characterize soils as a source of continuing contamination." The results can also be used in the assessment of human health and ecological risk of exposure to the soils and sediments.

Table 4–18. Alkenes, Cycloalkenes, and Unidentified Compounds Detected in the NECA Pond (849) and Terrace Wells

Constituent	Location	Sampling Date	Concentration (µg/L)
Benzene	604 824 829 849	12/7/98 3/6/99 3/8/99 11/7/98	1 J 7 5 4 J
Toluene	604 849	12/7/98 11/7/98	3 J 12
m-Xylene	819 824 849	3/6/99 3/6/99 11/7/98	2 J 3 J 4 J
o-Xylene	849	11/7/98	2 J
Chlorobenzene	826	3/4/99	2 J
Naphthalene, 1,2-dihydro-2-*m*	849	11/7/98	5 JN[a]
Naphthalene, 1,6,7-trimethyl-	849	11/7/98	67 JN
Hexene, 5,5-dimethyl-, (*Z*)-2-	819	3/6/99	6 JN
Unknown	728 813 819 824 849	12/8/98 12/7/98 3/6/99 3/6/99 11/7/98	440 J 170 J 320 J 31J 120 J[a]
Unknown hydrocarbon	849	11/7/98	76 J[a]
Unknown cycloalkane	849	11/7/98	22 J
Unknown PAH[a]	849	11/7/98	28 J 150 J

[a] Highest value of several peaks
J = estimated value
N = spiked sample recovery is not within limits
PAH = polycyclic aromatic hydrocarbon

In the first study (soil and sediment study 1), 26 samples were collected in late 1998 and early 1999. In the second study (soil and sediment study 2), 58 samples were collected at 34 locations in October and December 1999. Background information is presented in Section 4.4.3.1 and methods applicable to both studies are presented in Sections 4.4.3.2 and 4.4.3.3, respectively. Results and discussion of study 1 are given in Section 4.4.3.2, and a more complete description is provided in DOE 1999c. For study 2, results and discussion are given in Section 4.4.3.3, and a more complete description is in DOE 2000b.

4.4.3.1 Background

The contaminant chemistry of soils and sediments is needed to determine if the soils will release contamination to ground water. Some of the contaminants are incorporated in recalcitrant mineral grains. An example is the naturally occurring uranium in apatite, zircon, or monazite. Uranium is tightly bound in these minerals and will not be released to ground water. Some portions of the constituents are loosely bound by processes such as adsorption, absorption, chelation, incorporation in soluble minerals, or dissolution in immobile pore fluids. This loosely bound portion is the portion of interest for environmental work.

The concentration of a constituent in a soil or sediment is determined by digesting the sample, separating the liquid phase by centrifuge or filtration, analyzing constituent concentrations in the

liquid phase, and then calculating the concentrations in the solid phase. It is not necessary or desirable to have the tightly bound species digested. The most suitable digestion methods are those that remove only the loosely bound contaminants, because those contaminants have the highest potential for contaminating ground water and for being accessible to biota.

The many liquid media that can be used to digest samples range from deionized water to strong acids combined with hot fluxing agents. Some digestion agents are designed to selectively remove specific mineral phases. For example, a mixture of sodium citrate, sodium dithionite, and sodium bicarbonate is frequently used to selectively remove ferric oxyhydroxide minerals. These types of solutions, however, are not completely selective, in that some forms of contamination, such as adsorbed portions, are also released during digestion. The digestion method of choice may also be specific to the constituent of interest. For example, a low pH solution would be used to desorb cations, whereas a high pH solution would be used to desorb anions.

Numerous digestions with different solutions would be needed for complete characterization of the constituents in a soil or sediment, particularly at the Shiprock site, where a variety of constituents are of interest. This project was intended to provide a screening-level assessment of the accessible contamination in the soils and sediments. For this purpose, a 5-percent solution of HCl was used. This acidic solution should release the adsorbed cations and dissolve carbonate minerals. Although anions adsorb more strongly at low pH, they should also be released because the acid will dissolve most of the amorphous oxyhydroxide adsorbent phases. Five-percent HCl will not dissolve most silicate minerals (an exception is that it will partially dissolve chlorite), which is desirable because the constituents in silicate minerals are not readily available to ground water. By using HCl instead of nitric or sulfuric acid, the problem of analysis for nitrate and sulfate is avoided. Therefore, while not perfect, the 5-percent HCl digestion was considered a reasonable choice for this project.

All soils and sediments in nature contain some amount of the inorganic constituents used to process ore at the Shiprock mill. In addition, the solid-phase concentrations do not reflect the aqueous concentrations that result when water passes through the soils or sediments because the aqueous concentrations depend on such factors as flow rate and major-ion chemistry. To help interpret the soil and sediment data, samples were collected from background areas (areas that could not have been affected by the milling operation but that have similar lithology). Comparison of background samples that were digested in the same manner as the on-site samples helped to determine if the on-site samples contained releasable mill-related contaminants.

4.4.3.2 Soil and Sediment Study 1: Distribution of Contaminants at Widely Distributed Locations

Samples for the first soil and sediment study were collected from December 1998 to April 1999. Analytical work was performed from January through May of 1999. At the time the sampling and analysis was performed, the BLRA for the Shiprock site had not been updated and the revised list of COCs determined. Therefore, several constituents were analyzed that are not currently considered to be COCs. Results for those constituents are presented here for informational purposes.

Soil samples were collected with a shovel or a scoop. Sediment samples from the San Juan River and streams were collected by dipping a container into the bottom sediments near the shoreline. The choice of sampling locations was biased toward those samples that were more likely to

contain high levels of contamination, based on sample coloration or high radiometric measurements.

The samples were placed in aluminum pie plates, open to the air, until they were visibly dry (about 5 days). Dried samples were sieved to less than 2 millimeters (mm). The sieving removed only a small portion of the samples. Two grams of each sample was agitated with 100 milliliters (mL) of 5-percent HCl, end-over-end, for 4 hours. The samples were centrifuged, decanted, and leached again with 5-percent HCl. They were then filtered through a 0.45-micrometer (µm) filter and submitted to the GJO Analytical Chemistry Laboratory for analysis of arsenic (As), cadmium (Cd), sodium (Na), magnesium (Mg), manganese (Mn), antimony (Sb), selenium (Se), strontium (Sr), uranium (U), ammonium (NH_4), nitrate (NO_3), and sulfate (SO_4).
Sample locations are shown in Figure 4–47. Concentrations of constituents leached from the soils and sediments are provided in Table 4–19.

Nitrate

Nitrate concentrations in the four floodplain background samples ranged from 10.7 to 23.2 milligrams per kilogram (mg/kg) and averaged 18 mg/kg (Table 4–19 and Figure 4–48). Concentrations ranged from 19.7 to 1,010 mg/kg in samples from the millsite floodplain and from 18.6 to 1,120 mg/kg in samples from Bob Lee Wash. These data suggest that the millsite floodplain and Many Devils Wash areas were contaminated by milling activities. The nitrate concentration in the sample from location 889 in Many Devils Wash was 1,300 mg/kg, which is consistent with high concentrations of nitrate in water samples from escarpment seeps (Table 4–12).

The nitrate concentration in a sediment sample from location 884 in the irrigation return flow ditch was 37.1 mg/kg, which is only about twice the average background. The relatively low concentration contrasts with the relatively high ammonium concentration in samples from this location, indicating that nitrate may be converted to ammonium because of the reducing conditions. Nitrate concentrations in the five on-site and downgradient San Juan River sediment samples are similar to those in samples from the two upgradient locations, suggesting that the sediments have not been contaminated by millsite effluents (Table 4–19).

Sulfate

Sulfate concentrations in the four floodplain background samples ranged from 256 to 7,460 mg/kg and averaged 4,072 mg/kg (Table 4–19 and Figure 4–49). Concentrations in the millsite floodplain samples ranged from 2,960 to 42,300 mg/kg. These data suggest that samples from the millsite floodplain have higher sulfate concentrations that are related to the milling activities. Sulfate concentrations ranged from 6,500 to 50,200 mg/kg in samples from Bob Lee Wash, seep 425, and Many Devils Wash. All areas characterized by high concentrations of sulfate are also characterized by high concentrations of white salt deposits, which is probably the source of most of the sulfate.

Sulfate concentrations in the San Juan River sediment samples from the five on-site and downgradient locations are similar to those in samples from the two upgradient locations, suggesting that the sediments have not been contaminated by millsite effluents (Table 4–19).

Table 4–19. Concentrations of Constituents (mg/kg) in Soils and Sediments (5-percent HCI leach)

Location No.	Sample Date	Location	As	Cd	Mg	Mn	Na	NH_4	NO_3	Sb	Se	SO_4	Sr	U
884	12/10/98	NW-884	2.4	0.81	5,510	121	504	49.1	37.1	0.32	1.2	9,320	203	2.5
887	12/10/98	NW-887	0.69	<0.1	509	171	193	<0.11	22.2	0.11	<0.2	2,130	24.4	0.2
880	3/15/99	BLW-880	1.72	0.35	9,260	216	2,720	14.7	1,120	<0.1	<0.2	16,400	136	7.92
900	3/15/99	BLW-900	1.48	0.47	11,000	262	3,710	13	840	<0.1	0.57	50,200	407	40.2
902	3/15/99	BLW-902	1.12	0.35	5,500	168	1,230	9.63	18.6	<0.1	<0.2	6,500	75.5	10.3
425	4/7/99	ESC-425	2.06	1.17	10,800	249	989	25.9	144	0.38	<0.2	21,100	349	6.41
865	1/12/99	FP-865	0.75	0.16	328	110	105	9.2	19.7	0.22	<0.2	2,960	9.3	0.23
866	1/12/99	FP-866	2.3	0.42	3,940	384	5,020	16.1	83.8	0.22	<0.2	23,000	113	3.3
867	1/12/99	FP-867	2.2	0.48	2,790	379	11,200	8.4	48.1	0.21	<0.2	39,900	91.7	2.8
868	1/12/99	FP-868	4.2	1	4,720	723	8,630	9.7	637	0.29	<0.2	42,300	190	7.9
869	1/12/99	FP-869	2.2	0.4	3,020	149	2,970	16.1	22	0.12	0.49	26,800	89.8	35.6
870	1/12/99	FP-870	1.5	0.3	2,070	236	119	5.2	20.8	0.12	<0.2	8,700	51.6	8.4
891	1/12/99	FP-891	0.95	<0.1	1,480	120	1,080	4.1	37	<0.1	0.25	6,650	63.1	3.2
892	1/12/99	FP-892	1.8	0.29	4,550	229	8,190	7.8	1,010	<0.1	1.9	32,000	136	14.7
871	1/13/99	FPBG-871	0.74	0.11	605	146	132	8.7	10.7	<0.1	<0.2	7,220	28.6	0.49
872	1/13/99	FPBG-872	0.47	<0.1	156	94.1	42.5	5	15.5	0.19	<0.2	7,460	7.2	0.18
873	1/13/99	FPBG-873	0.7	<0.1	320	128	64.9	5.7	21.9	<0.1	<0.2	256	14.6	0.28
874	1/13/99	FPBG-874	0.94	0.12	1,010	207	315	8.9	23.2	<0.1	<0.2	1,350	40.3	0.62
889	4/6/99	MDW-889	1.05	0.26	11,900	114	3,660	11.7	1,300	0.18	0.44	19,600	184	0.86
888	12/9/98	SJR-888-U	0.58	<0.1	445	176	170	<0.11	14.2	0.16	<0.2	1,950	36.2	0.16
898	12/9/98	SJR-898-U	0.78	0.21	640	161	241	1	26.2	<0.1	<0.2	1,910	30.1	0.21
893	12/10/98	SJR-893	1	<0.1	646	209	293	1.8	15.2	<0.1	<0.2	1,990	44.6	0.22
894	12/10/98	SJR-894	0.88	<0.1	541	229	581	0.5	14.5	<0.1	<0.2	2,660	35.1	0.17
895	12/10/98	SJR-895	0.79	<0.1	541	176	160	0.22	10	<0.1	<0.2	1,800	33.7	0.18
896	12/10/98	SJR-896	0.92	<0.1	683	214	294	0.76	18.2	<0.1	<0.2	1,730	45	0.25
897	12/9/98	SJR-897	0.99	0.14	654	195	182	1.3	39	0.17	<0.2	1,780	41.1	0.2

Note: NW = Northwest area
BLW = Bob Lee Wash
ESC = Escarpment
FP = Floodplain
FPBG = Floodplain background
MDW = Many Devils Wash
SJR = San Juan River
U (in location) = upgradient

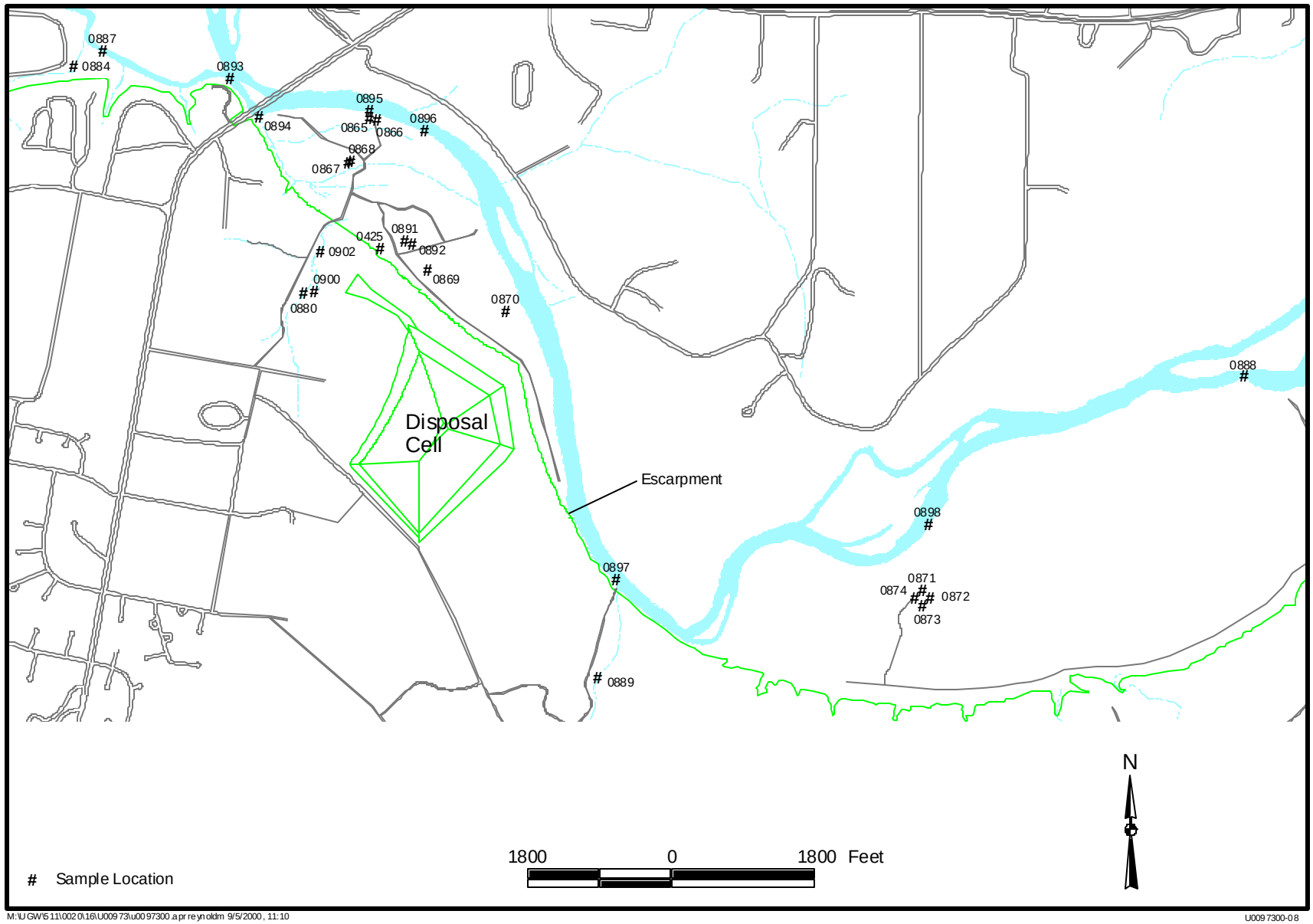


Figure 4–47. Soil and River Sediment Sample Locations

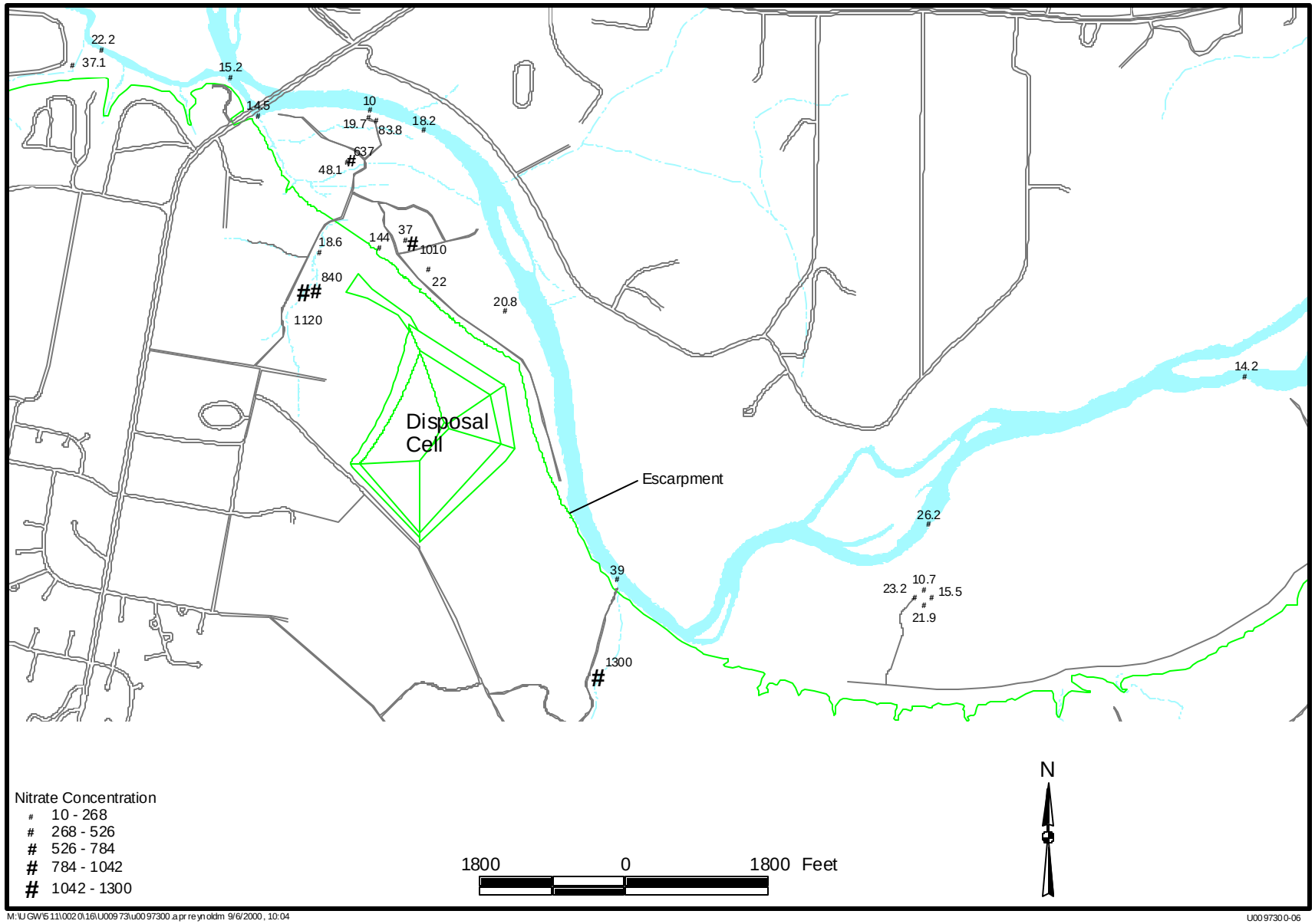


Figure 4–48. Nitrate (mg/kg) in Soil and River Sediment

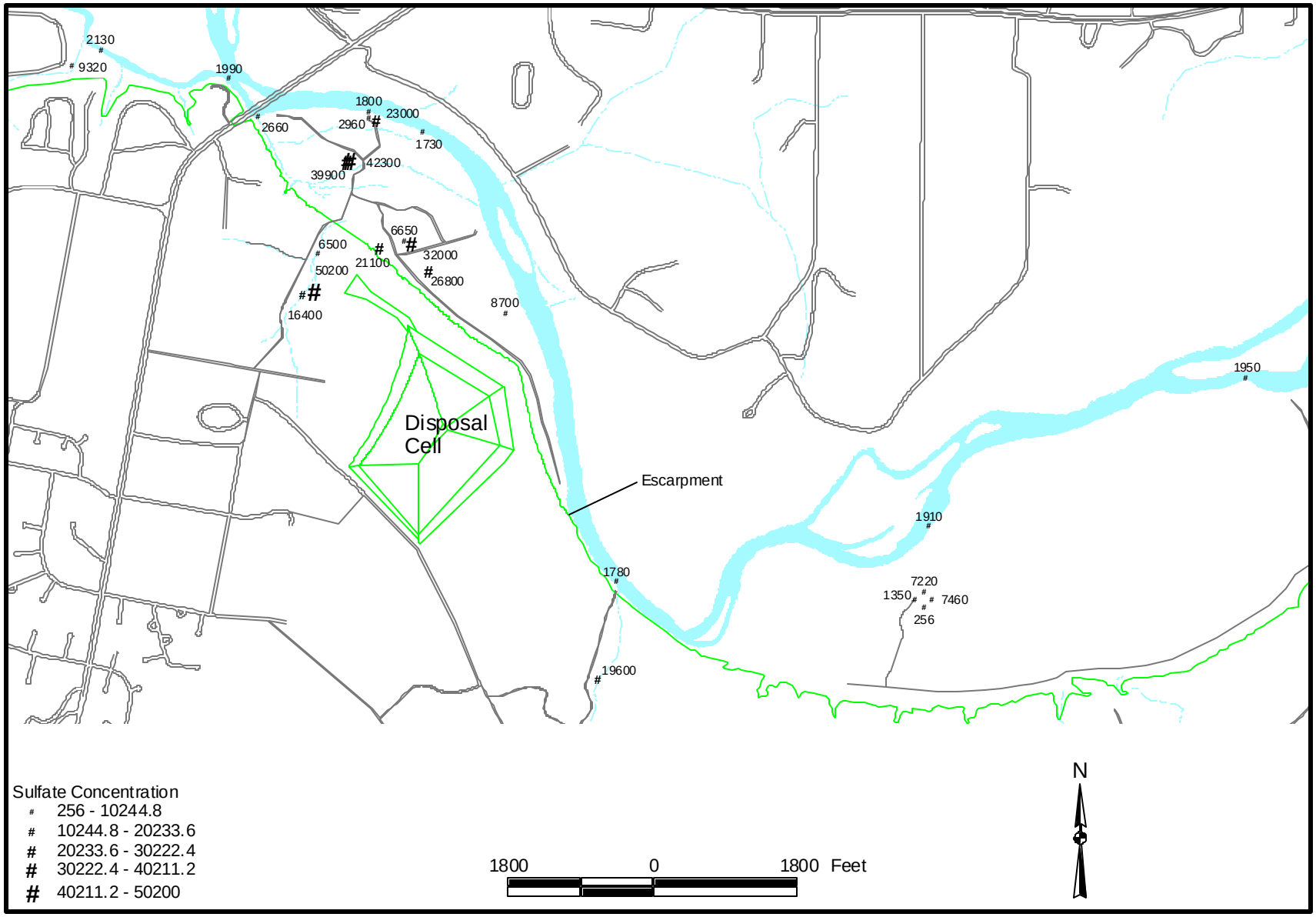


Figure 4–49. Sulfate (mg/kg) in Soil and River Sediment

Uranium

Uranium concentrations in the four floodplain background samples ranged from 0.18 to 0.62 mg/kg and averaged 0.39 mg/kg (Table 4–19 and Figure 4–50). Concentrations ranged from 0.23 to 35.6 mg/kg in samples from the floodplain and from 6.41 to 40.2 mg/kg in samples from Bob Lee Wash and seep 425. These data suggest contamination related to the milling activities. The uranium concentration in the sample from location 889 in Many Devils Wash was 0.86 mg/kg, which is only about twice the average background concentration. This relatively low uranium concentration contrasts with the high concentration of nitrate at the same location.

The three floodplain samples that had the highest uranium concentrations (35.6, 8.4, and 14.7 mg/kg) were collected from locations 869, 870, and 892, respectively, and also had elevated gamma activity. The sample (869) with the highest uranium concentration (35.6 mg/kg) was collected from sandy material around monitor well 615. This may be windblown tailings that were not completely removed during the surface remediation.

The sample collected from the sediment in the irrigation return flow ditch at location 884 had 2.5 mg/kg of uranium, which is about 6 times the average background. This relatively high value suggests that the reducing environment caused by decaying organic material has resulted in some accumulated of uranium, which is readily fixed under reducing conditions.

Uranium concentrations in the San Juan River sediment samples from the five on-site and downgradient locations are similar to those in the samples from the two upgradient locations, suggesting that the sediments have not been contaminated by millsite effluents (Table 4–19).

Other Constituents

Ammonium-Ammonium concentrations in the four floodplain background samples averaged 7.1 mg/kg (Table 4–19). Most of the samples collected from the millsite floodplain had concentrations similar to background. Two locations on the floodplain had a concentration of 16.1 mg/kg, which is more than twice the average background but is probably still within the range of natural concentrations. The sample from location 884 had the highest concentration (49.1 mg/L) of ammonium. This sample, collected underwater from an irrigation return-flow ditch, contained abundant organic matter. The high ammonium concentration may be a result of fertilizers used in the upstream agricultural fields or may have been released from decaying organic matter. Ammonium concentrations in samples collected in Bob Lee Wash (880, 900, and 902 with concentrations of 14.7, 13, and 9.63 mg/kg, respectively) are slightly above the average background value of 7.1 mg/kg but are probably within the range of uncontaminated soils. The ammonium concentration in the sample collected at seep 425 was 25.9 mg/kg, which is about 3 times the average background value, indicating the possibility of a small contribution of ammonium from the millsite. Ammonium concentrations in the five on-site and downgradient sediment samples collected in the San Juan River were similar to those in the two upgradient samples, suggesting that the sediments have not been contaminated by millsite effluents.

Antimony-Many of the antimony concentrations, both at background and on-site locations, were less than the detection limit of 0.1 mg/kg (Table 4–19). The highest concentration was 0.38 mg/kg in a sediment sample from seep 425. A sediment sample from the irrigation return flow ditch (location 884) had the second highest value of 0.32 mg/kg. These values are about

twice that of background location 872 but are probably within the range of natural variation. Antimony concentrations in the five on-site and downgradient samples collected in the San Juan River were similar to those in the two upgradient samples, suggesting that the sediments have not been contaminated by millsite effluents.

Arsenic–Arsenic concentrations in the four floodplain background samples averaged 0.71 mg/kg (Table 4–19). Several of the samples collected from the millsite floodplain had concentrations similar to background. However, the sample collected at location 868 on the floodplain had an arsenic concentration of 4.2 mg/kg, which is about 6 times the average background. Several other samples from the floodplain and the sample from seep 425 had concentrations about twice the average background. These values indicate that some mill-related arsenic is present on the floodplain. A sample from location 884, the irrigation return flow ditch, had an arsenic concentration of 2.4 mg/kg, which is about 3 times the average background and suggests a possible contribution from fertilizer or accumulation in the reduced environment caused by decaying organic material. Arsenic concentrations in the five on-site and downgradient samples collected in the San Juan River were similar to those in the two upgradient samples, suggesting that the sediments have not been contaminated by millsite effluents.

Cadmium–Cadmium concentrations in all four floodplain background samples were less than 0.12 mg/kg (Table 4–19). Three samples from the floodplain had cadmium concentrations greater than 0.4 mg/kg, indicating that some mill-related cadmium may be present on the floodplain, but these values could be within the range of natural variation. Samples from Bob Lee Wash and seep 425 ranged from 0.35 to 1.17 mg/kg, indicating the possibility of mill-related contamination in those areas. The sample collected in the irrigation return-flow ditch at location 884 had a cadmium concentration of 0.81 mg/kg (about 8 times average background), which suggests a possible contribution from fertilizer or accumulation in the reduced environment caused by decaying organic material. Cadmium concentrations in the five on-site and downgradient samples collected in the San Juan River were similar to those in the two upgradient samples, suggesting that the sediments have not been contaminated by millsite effluents.

Magnesium–Magnesium concentrations in the four floodplain background samples ranged from 156 to 1,010 mg/kg and averaged 523 mg/kg (Table 4–19). Concentrations in samples from the floodplain ranged from 328 to 4,720 mg/kg. These data suggest that the floodplain has magnesium concentrations that are related to the milling activities. Alternatively, the higher concentrations could be the result of an increase in the concentration of evaporative salts in the soils. Magnesium concentrations ranged from 5,500 to 11,900 mg/kg in samples from Bob Lee Wash, seep 425, and Many Devils Wash. These areas are characterized by high concentrations of white efflorescent salt deposits, which are probably the source of some of the magnesium. The higher than background concentration of 5,510 mg/kg in a sample from location 884 in the irrigation return-flow ditch suggests an influence from fertilizers used upstream or an accumulation of salts. Magnesium concentrations in the five on-site and downgradient samples collected in the San Juan River were similar to those in the two upgradient samples, suggesting that the sediments have not been contaminated by millsite effluents.

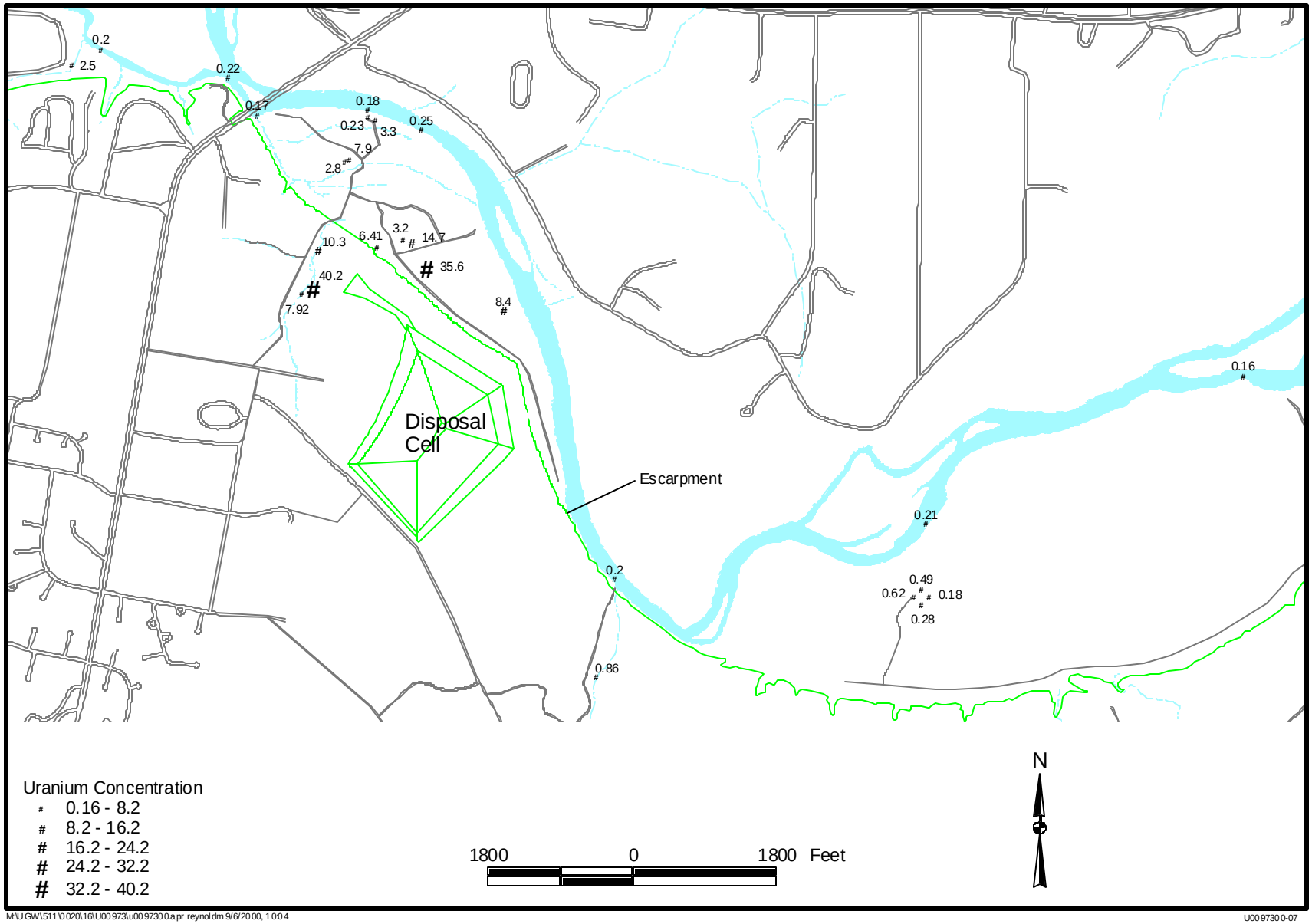


Figure 4–50. Uranium (mg/kg) in Soil and River Sediment

Manganese–The manganese concentration in the sample from the irrigation return flow ditch at location 884 is only 121 mg/kg, which is lower than the average floodplain background (Table 4–19). In contrast, this sample had anomalously high concentrations of most other COPCs. The low value could be due to the organic-rich and highly reduced conditions at this location. Manganese concentrations in the four floodplain background samples ranged from 94.1 to 207 mg/kg and averaged 144 mg/kg. Concentrations in samples from the millsite floodplain and Bob Lee Wash ranged from 110 to 723 mg/kg. These data suggest that these areas were affected by milling activities. Alternatively, the higher manganese concentration may simply reflect a slightly more oxidized environment. Manganese concentrations in the five on-site and downgradient samples collected in the San Juan River were similar to those in the two upgradient samples, suggesting that the sediments have not been contaminated by millsite effluents.

Selenium–Selenium concentrations in all four floodplain background samples were less than 0.2 mg/kg (Table 4–19). Most of the selenium concentrations in samples from the floodplain and Bob Lee Wash area were also less than 0.2 mg/kg. Two samples collected from the floodplain near the escarpment had concentrations of 0.49 and 1.9 mg/kg. One sample from the Bob Lee Wash area had a concentration of 0.57 mg/kg, and a sample from Many Devils Wash had a concentration of 0.44 mg/kg. These higher than background concentrations suggest mill-related contamination but may be within the range of natural variation. The sample collected from the irrigation return-flow ditch at location 884 had a selenium concentration of 1.2 mg/kg, which suggests a possible contribution from fertilizer or accumulation in the reduced environment caused by decaying organic material. The selenium concentrations in all San Juan River samples were less than 0.2 mg/kg, suggesting that the on-site and downgradient sediments have not been contaminated by millsite effluents.

Sodium–Sodium concentrations in the four floodplain background samples ranged from 42.5 to 315 mg/kg and averaged 139 mg/kg (Table 4–19). Concentrations in samples from the floodplain ranged from 105 to 11,200 mg/kg. These data suggest that the floodplain has sodium concentrations that are related to milling activities. Alternatively, the higher concentrations could be the result of an increase in the concentration of evaporative salts in the soils. Sodium concentrations ranged from 989 to 3,710 mg/kg in samples from Bob Lee Wash, seep 425, and Many Devils Wash. These areas are characterized by high concentrations of white efflorescent salt deposits, which are probably the source of some of the sodium. Except for one sample, the sodium concentrations in the five on-site and downgradient samples are similar to those in the two upgradient samples, suggesting that sediments have not been contaminated by millsite effluents. The sample collected near the U.S. Highway 666 bridge at location 894 had a sodium concentration of 581 mg/kg, which is about 3 times the average floodplain background concentration. Because sodium sulfate is the dominant compound in white efflorescent salt deposits that occur throughout the Shiprock region, it is likely that the elevated concentration is due to a small contribution of these salts in the sediment sample.

Strontium–Strontium concentrations in the four floodplain background samples ranged from 7.2 to 40.3 mg/kg and averaged 23 mg/kg (Table 4–19). Concentrations on the floodplain ranged from 9.3 to 190 mg/kg. These data suggest that the floodplain sediments have higher strontium concentrations that could be related to the milling activities. Alternatively, the higher concentrations could be the result of an increase in the concentration of evaporative salts in the soils. Strontium concentrations ranged from 75.5 to 407 mg/kg in samples from Bob Lee Wash, seep 425, and Many Devils Wash. These relatively high concentrations suggest a millsite

influence. These areas are characterized by high concentrations of white efflorescent salt deposits, which is probably the source of some of the strontium. The higher than background concentration of 203 mg/kg in a sample from location 884 in the irrigation return flow ditch suggests an influence from fertilizers used upstream or an accumulation of salts. Strontium concentrations in the five on-site and downgradient samples collected in the San Juan River were similar to those in the two upgradient samples, suggesting that they have not been contaminated by millsite effluents.

4.4.3.3 Soil and Sediment Study 2: Detailed Investigation at the Base of the Escarpment

Elevated concentrations of several constituents in ground water at the base of the escarpment below the disposal cell raised the question of whether residual source material was left in the sediments. An extensive investigation was done in December 1999, during which 46 soil samples were collected in backhoe pits at 23 locations on a 300-ft by 300-ft grid along the base of the escarpment. Two background samples were taken at a location close to the floodplain background wells. Sediment samples were taken 1 ft below the surface and at the water table (typically at a 4 or 5 ft depth), where one water sample per location was taken directly from the backhoe bucket. Samples were air dried and sieved (<2 mm) in the ESL. Extraction was performed using 5-percent HCl. Two grams of sediment was leached in 200 mL acid. Samples were filtered through a 0.45 µm filter and were submitted to the GJO Analytical Chemistry Laboratory for uranium, sulfate, and nitrate analyses.

The analytical results are presented in Table 4–20. Background uranium concentrations in soils of the floodplain ranged from 0.12 to 0.23 mg/kg (concentrations were as high as 0.62 mg/kg in Study 1). Soils near the escarpment had concentrations ranging from 0.43 to 3.90 mg/kg (Figure 4–51). Concentrations vary only slightly with depth.

Table 4–20. Concentration of Constituents in Soil and Sediments Adjacent to the Escarpment

		Uranium (mg/kg)	Sulfate (mg/kg)	Nitrate (mg/kg)
Background[a]	Range	0.12–0.62	256–11,700	10.7–123
	Average	0.32	5,581	34.5
Shallow[b] (1 ft)	Range	0.46–3.90	8,930–50,000	4.86–1,810
	Average	1.66	28,010	430
Deep[b] (water table)	Range	0.43–3.13	1,830–31,200	14.9–929
	Average	1.04	21,418	225

[a]Locations 1038, 871, 872, 873, and 874 [data and locations for 871, 872, 873, and 874 are in DOE (1999c)]
[b]Locations 1015 through 1037

Uranium concentrations in shallow samples (up to 1 ft depth) in soils are highest (2.85 to 3.90 mg/kg) at locations 1017, 1018, and 1019, which are on the east side of the disposal cell just below filled drainages (Figure 4–51). Shallow samples north of the disposal cell contain 0.46 to 2.50 mg/kg uranium. The average uranium concentration in samples from the second depth interval (>1 ft) is slightly lower (1.04 mg/kg) than in the shallow samples.

Figure 4–52 and Figure 4–53 shows the distribution of uranium, nitrate, and sulfate using different colors for locations that are closest to the base of the escarpment and locations that are about 250 ft northeast of the base. The bars in Figure 4–52 are sorted by location from north to south (left to right). Generally, locations closer to the escarpment have higher concentrations in soil.

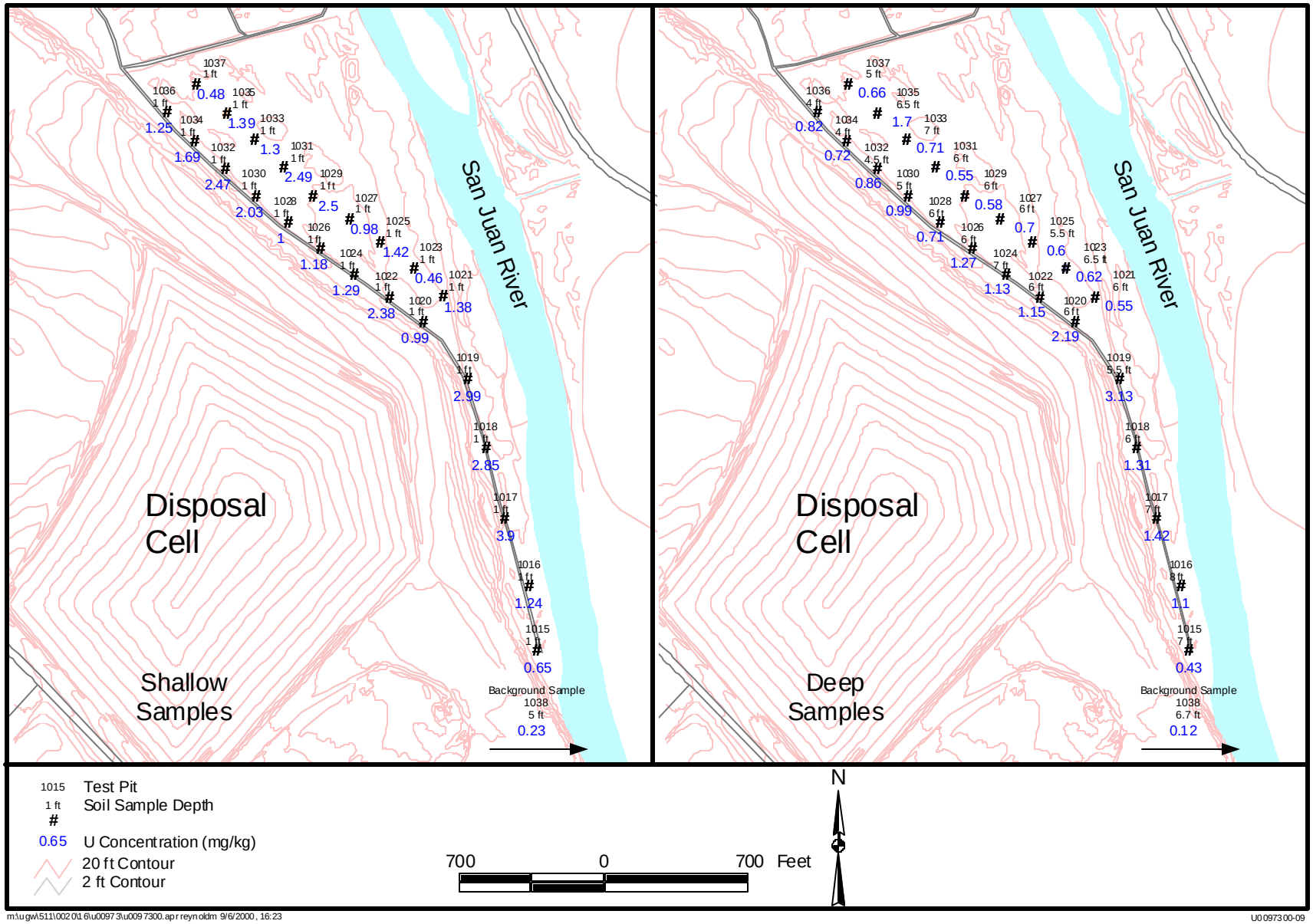


Figure 4–51. Uranium Concentrations (mg/kg) in Shallow and Deep Soil Samples on the Floodplain

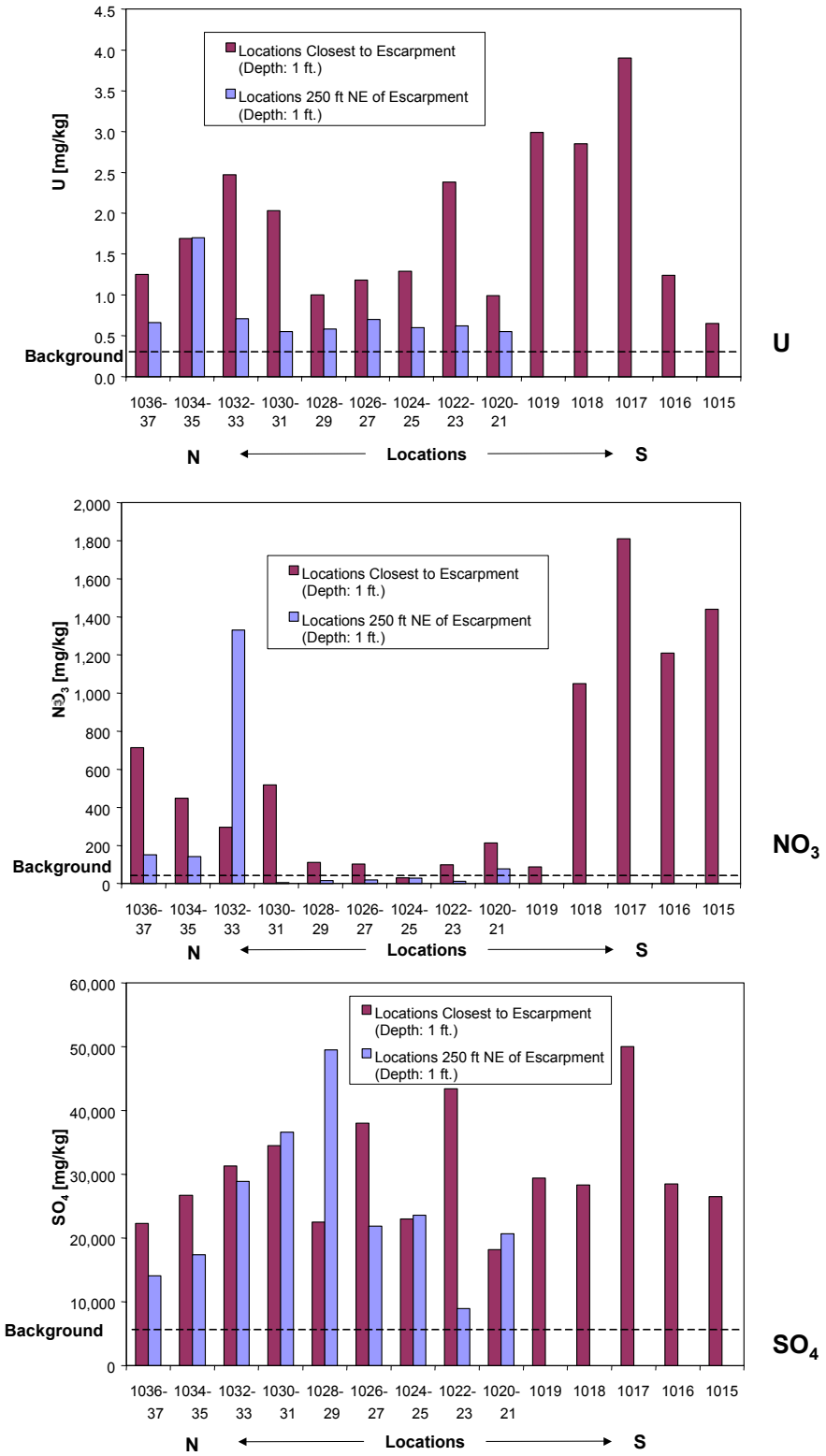


Figure 4–52. Distribution of Concentrations in Soils Related to Distance From the Escarpment—1 Foot Depth

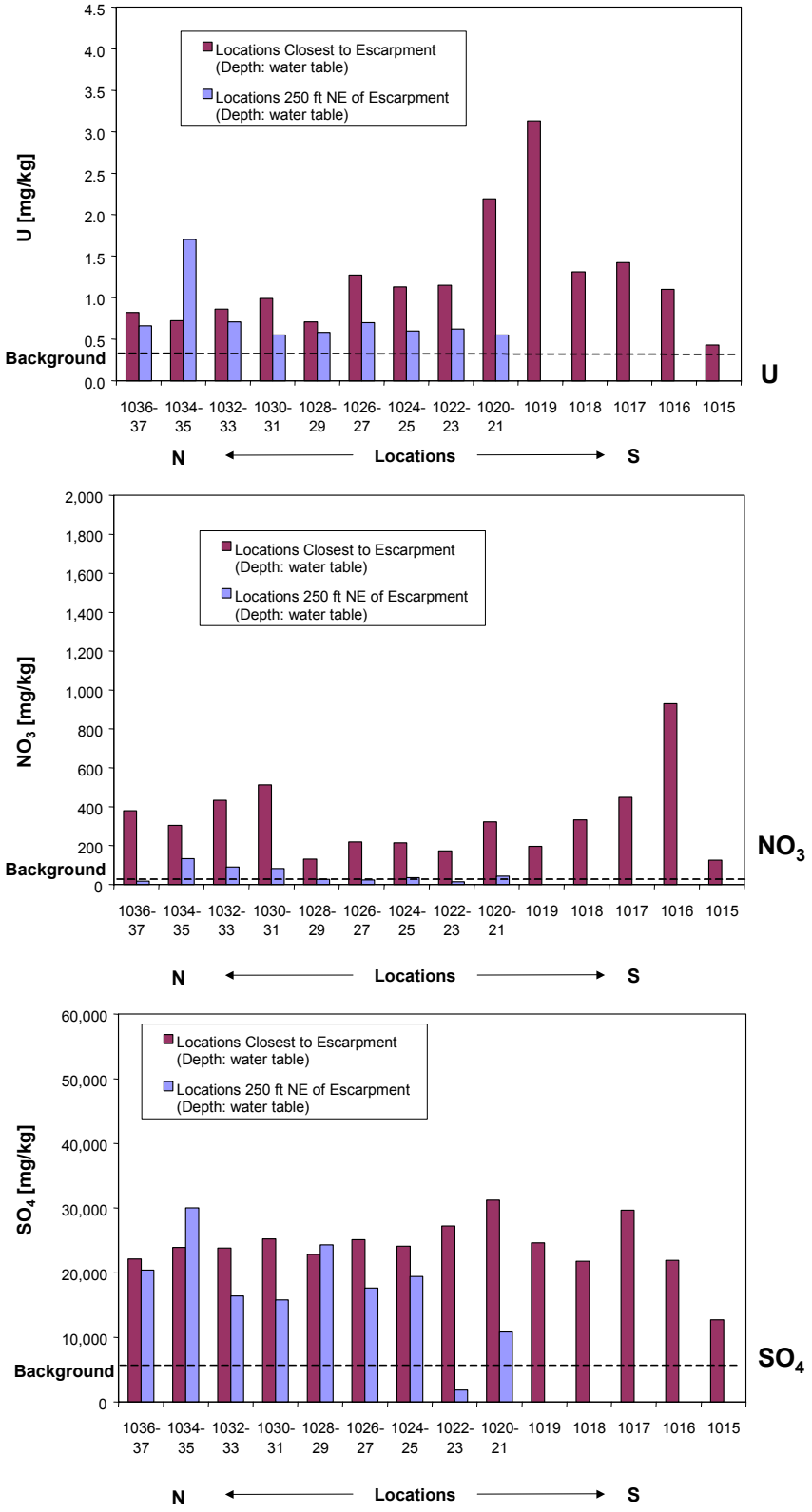


Figure 4–53. Distribution of Concentrations in Soils Related to Distance From the Escarpment—Water Table

The average background sulfate concentration in sediments was 5,581 mg/kg and ranged from 256 to 11,700 mg/kg (Table 4–20). The average sulfate concentrations were slightly higher in the shallow samples than in samples collected at the water table. The spatial distribution of sulfate in soils close to the escarpment (Figure 4–55) is similar to the distribution of uranium. The highest sulfate concentration (50,000 mg/kg) was in a sample from location 1017. Sulfate decreased gradually in shallow samples from south to north between locations 1030 and 1037 (Figure 4–52). Sulfate concentrations in soil samples collected from near the water table closest to the escarpment varied between 22,800 and 31,200 mg/kg (Figure 4–53). Unlike uranium, there was no obvious increase in sulfate concentrations closer to the escarpment (Figure 4–52). The variation in sulfate concentration is due largely to the amount of evaporite salt deposits that are included in a particular sample (DOE 1999b).

Background concentrations for nitrate ranged from 10.7 to 123 mg/kg and averaged 34.5 mg/kg (Table 4–20). Nitrate concentrations in soils were highest in samples at locations close to the escarpment (Figure 4–53 and Figure 4–54). The highest concentration was in a sample from location 1017 (1,810 mg/kg). Nitrate concentrations were also high at locations 1015, 1016, 1018, and 1033 (1,440, 1,210, 1,050, and 1,330 mg/kg, respectively). With the exception of the sample collected at location 1033, nitrate concentrations are generally higher in the soil samples collected near the escarpment than in those collected about 250 ft away (Figure 4–52 and Figure 4–53). This distribution suggests that high nitrate concentrations are in ground water entering the floodplain and are subsequently transferred to aquifer solids. Concentrations increase slightly from south to north in shallow soil samples close to the escarpment (Figure 4–52); this trend was not observed in the ground water samples.

Distribution ratios (R_d) were calculated from the measured concentrations in ground water and the concentrations in the soil (field determination). The average R_d for uranium for samples that were taken at the water table is 1.49 milliliters per gram (mL/g), and for samples taken at the surface is 2.58 mL/g (DOE 2000b). R_d values measured in the laboratory on Shiprock floodplain sediments averaged 0.64 mL/g (DOE 1999d). The similarity between laboratory values and field determinations of R_d indicates that uranium contamination in the sediments can be explained by sorption from contaminated ground water and is not from residual tailings. Contaminated processing water flowed from the terrace to the floodplain and contaminants likely sorbed to the soils.

Common uranium concentrations in subpile soils or other residual sources are 50 mg/kg and higher. Uranium concentrations of more than 80 mg/kg were measured in subpile soils at the Gunnison, Colorado, UMTRA site. The uranium concentrations in Gunnison ground water were approximately 1 mg/L. Column studies using the Gunnison soils showed that soils with a uranium content of 80 mg/kg caused uranium contamination in ground water up to 1.6 mg/L (DOE 2000a). Because the uranium concentrations in soils in the floodplain at Shiprock are significantly lower than those at Gunnison because no residual source material is present.

4.4.4 Determination of Distribution Ratios

Distribution ratios were determined to address two of the data quality objectives defined in the Work Plan (DOE 1998d): (1) “characterize contaminant sorption in the Mancos Shale below the terrace system” and (2) “characterize contaminant sorption in the floodplain alluvial aquifer.” Summaries of the methods and results are presented in the following sections. More complete details of the study are available in DOE (1999d).

The results of this study can be used to help evaluate the performance of ground water remediation methods. For example, a contaminant transport model incorporating a K_d can be used to evaluate whether natural flushing is likely to meet the ground water standards within the regulated 100-year period. The results of this study can also be used in models to help estimate the volume of ground water that will need to be pumped or passively treated to meet State and Federal ground water standards.

4.4.4.1 Background

As contaminated ground water migrates through soils and rocks, contamination is distributed between the solid and the liquid phases. This phenomenon causes the contamination to travel at a slower rate than the average ground water velocity. Chemical processes that cause this retardation can include adsorption, absorption, precipitation, diffusion into immobile porosity, and transfer to vapor phases. Generally, these processes cannot be differentiated. However, a bulk parameter (K_d) has been used with some success to model the retardation of contamination for many aquifer systems. Most numerical ground water models use the K_d concept in simulations of contaminant transport. Site-specific K_d values are approximated from R_d values that are empirically determined. A laboratory study was conducted to determine R_d values for the terrace and the floodplain systems at the Shiprock site.

R_d is defined as the concentration of a constituent on the solid fraction divided by the concentration in the aqueous phase:

$$R_d = \frac{(\text{mass of solute sorbed per unit mass of solids})}{(\text{mass of dissolved solute per volume of solution})} \tag{1}$$

R_d values are calculated from experimental data as

$$Rd = \frac{(A - B)V}{M_s B} \tag{2}$$

where

R_d = distribution ratio (mL/g)
A = initial concentration of the constituent in mg/L,
B = final concentration of the constituent (mg/L),
V = volume of solution [100 mL for this study], and
M_s = mass of soil used in grams (g).

K_d is numerically equivalent to R_d if the system is at equilibrium and R_d is constant over the range of conditions being considered. If R_d is constant over a large range of contaminant concentrations, it is said to be “linear” because a plot of aqueous concentration in relation to solid-phase concentration forms a straight line on an arithmetic plot. R_d data are often displayed on log-log concentration plots. A linear R_d (referred to as a linear isotherm because temperature is held constant) plots as a line with a slope of 1 on a log-log plot. At elevated concentrations of a constituent, R_d often varies with the aqueous concentration. In this case, the isotherm is said to be nonlinear and the migration cannot be accurately predicted using a K_d model.

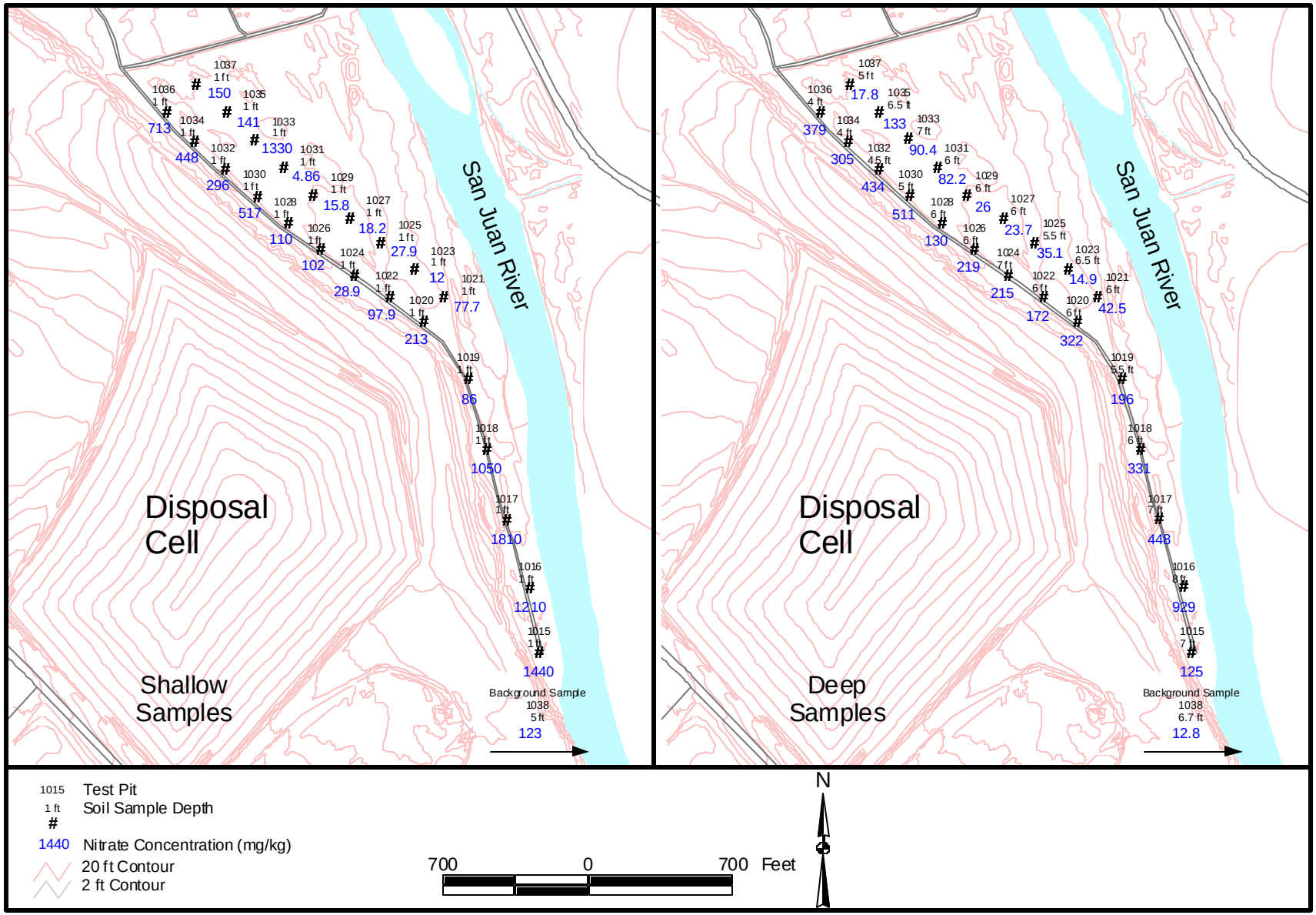


Figure 4–54. Nitrate Concentrations (mg/kg) in Shallow and Deep Soil Samples on the Floodplain

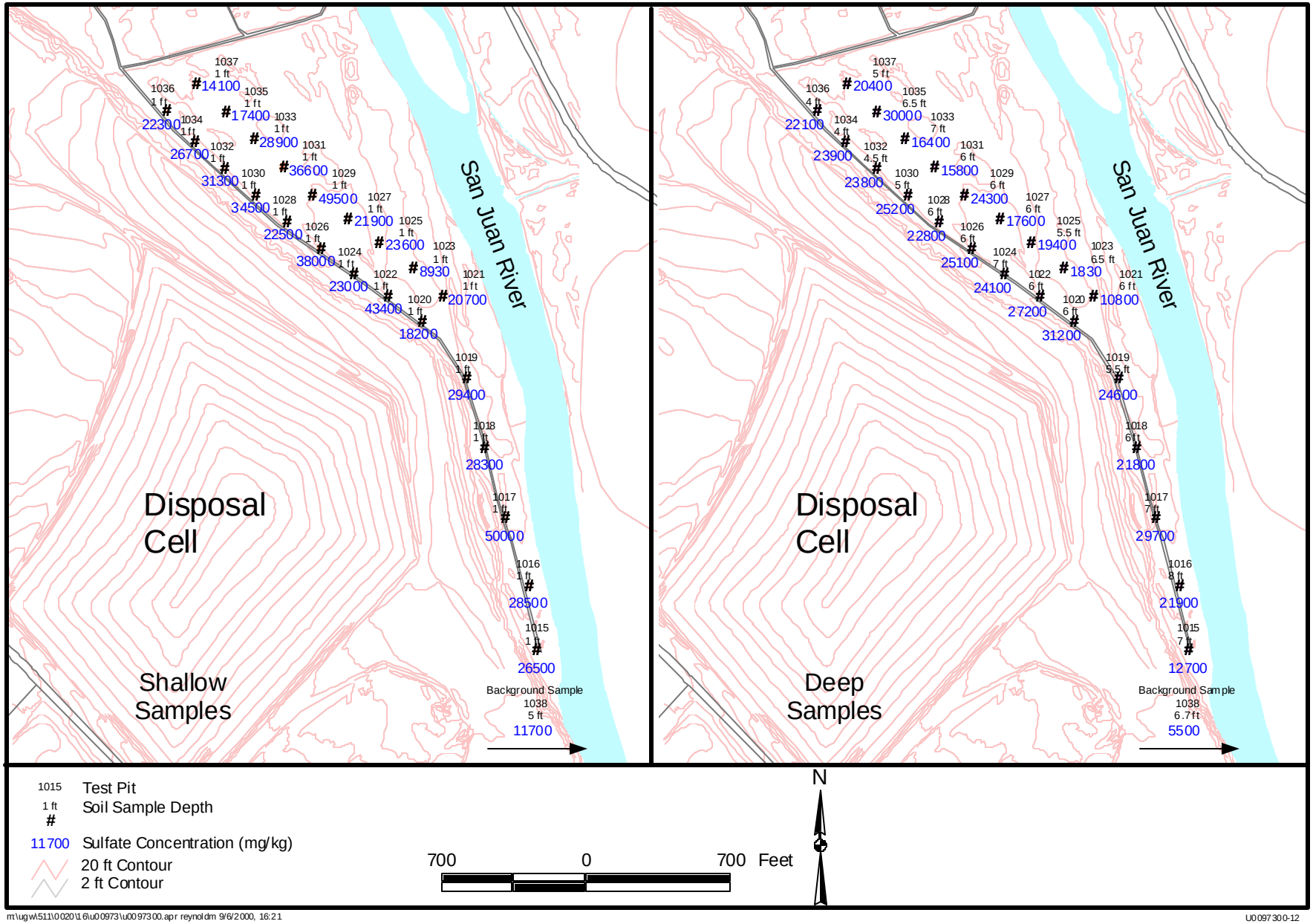


Figure 4–55. Sulfate Concentrations (mg/kg) in Shallow and Deep Soil Samples on the Floodplain

4.4.4.2 Sample Collection and Laboratory Methods

Sediment or Mancos Shale bedrock samples were obtained from two well cores in background locations on the terrace (wells 800 and 802) and from auger cuttings from three wells at background locations on the floodplain (wells 850, 851, and 852). Plates 1 and 2 show the locations of these wells. Background-area cores and cuttings were used instead of material from contaminated areas because of the difficulty in interpreting results from contaminated material.

Two samples of weathered Mancos Shale (well 800 at a 21-ft depth and well 802 at a 32-ft depth), two samples of unweathered Mancos Shale (well 800 at a 60-ft depth and well 802 at a 60-ft depth), and six samples of floodplain alluvium (well 850 at a 2-ft depth, well 850 at a 10-ft depth, well 851 at a 2-ft depth, well 851 at an 11-ft depth, well 852 at a 6-ft depth, and well 852 at a 12-ft depth) were tested. Two of the floodplain alluvial samples (well 850 at 2 ft and well 851 at 2 ft) are from the upper sand unit; all other floodplain alluvial samples are from the lower gravel unit.

R_d data were collected using ESL Procedure CB(BE–3) (DOE 1999d), which follows an American Society for Testing and Materials (ASTM) procedure for batch-type experiments (ASTM 1993). Two synthetic solutions were prepared that simulate the major-ion chemistry and pH of ground water at the site. Constituents for analysis were selected on the basis of historical data before the update of the BLRA and screening of some contaminants. Therefore, the analyte list does not match the current COC list. Contaminants that had ground water concentrations that exceeded 10 times the MCL, or twice background levels, were selected for study. Those contaminants were ammonium, cadmium, selenium, and uranium.

Five-point isotherms were determined for all four constituents for two samples of Mancos Shale (weathered and unweathered) from the terrace and for two samples of alluvial aquifer material from the floodplain. Masses of sampled material varying from 1 to 25 g were used to determine the isotherms.

4.4.4.3 Results and Discussion

Mean values of R_d for terrace weathered Mancos Shale, terrace unweathered Mancos Shale, and floodplain alluvial gravel are presented in Table 4–21. Several R_d values were significantly different from the mean values. These anomalous values are probably because of sample heterogeneity or analytical uncertainties. Table 4–22 presents mean R_d values with outliers omitted. Outliers are those values that exceeded 1 standard deviation from the mean. The mean R_d values do not change substantially by omitting the outliers; the R_d values for ammonium showed the largest changes. A grain size distribution of floodplain alluvial sediments (using the <2 mm fraction) was evaluated to correct the laboratory data to actual field values of R_d (DOE 2000a). Adjusted R_d values are slightly lower than the unadjusted laboratory values (Table 4–23).

Table 4–21. Summary of R_d Determinations

Constituent	Description	Mean R_d (mL/g)	Standard Deviation
Ammonium	Terrace - weathered Mancos Shale	4.68	6.88
Ammonium	Terrace - unweathered Mancos Shale	3.16	6.72
Ammonium	Floodplain – Qal[a]	1.39	1.85
Cadmium	Terrace – weathered Mancos Shale	213.79	86.66
Cadmium	Terrace – unweathered Mancos Shale	132.04	19.80
Cadmium	Floodplain – Qal	22.55	6.12
Selenium	Terrace – weathered Mancos Shale	68.09	40.62
Selenium	Terrace – unweathered Mancos Shale	46.63	18.22
Selenium	Floodplain – Qal	10.51	5.24
Uranium	Terrace – weathered Mancos Shale	1.13	1.15
Uranium	Terrace – unweathered Mancos Shale	1.97	0.43
Uranium	Floodplain – Qal	0.64	0.36

[a]Qal = Quaternary alluvium.

Table 4–22. Summary of R_d Determinations Omitting Outliers

Constituent	Description	Mean R_d (mL/g)	Standard Deviation	Number Omitted
Ammonium	Terrace – weathered Mancos Shale	2.08	2.91	1/6[a]
Ammonium	Terrace – unweathered Mancos Shale	0.59	2.55	1/6
Ammonium	Floodplain – Qal[b]	0.72	0.46	2/14
Cadmium	Terrace – weathered Mancos Shale	180.00	28.76	1/6
Cadmium	Terrace – unweathered Mancos Shale	135.37	8.85	2/6
Cadmium	Floodplain – Qal	21.96	2.92	5/14
Selenium	Terrace – weathered Mancos Shale	54.73	26.90	1/6
Selenium	Terrace – unweathered Mancos Shale	46.60	9.18	2/6
Selenium	Floodplain – Qal	11.44	2.45	6/14
Uranium	Terrace – weathered Mancos Shale	1.59	0.24	1/6
Uranium	Terrace – unweathered Mancos Shale	2.13	0.17	1/6
Uranium	Floodplain – Qal	0.54	0.19	3/14

Note: Outliers are those values that exceeded 1 standard deviation from the mean.
[a]1/6 = 1 of 6 points were omitted.
[b]Qal = Quaternary alluvium.

Table 4–23. R_d Values for Shiprock Floodplain, Adjusted Using Grain Size Distributions

Constituent	Laboratory R_d (<2 mm)[a] (mL/g)	Adjusted R_d (mL/g)
Ammonium	0.72	0.44
Cadmium	21.96	13.4
Selenium	11.44	6.98
Uranium	0.54	0.33

Note: R_d values are adjusted based on average fractions from Table 1 in DOE (2000a).
[a]From DOE (1999d).

Ammonium

The final concentrations of ammonium do not correlate well with the amount of solids used in the experiments. For example, the final concentration of ammonium in sample 800 from the 21-ft depth using 25 g of sample was 60,100 µg/L, whereas the final concentration with 15 g of sample was 34,100 µg/L. The lack of correlation apparently was due to the instability of the solutions with respect to ammonium. Because ammonium is volatile relative to the other contaminants in this study, it is possible that some portion was lost during vacuum filtering. Another possibility is that some ammonium was transformed to another nitrogen-bearing species, such as nitrite or nitrate. Additional tests, with careful monitoring of ammonium, nitrate, and nitrite concentrations, would be required to confirm the R_d values.

Mean R_d values for ammonium concentrations with outliers removed range from 0.59 mL/g for samples from the terrace unweathered Mancos Shale to 2.08 mL/g for samples from the terrace weathered Mancos Shale (Table 4–22). The average R_d value for the floodplain alluvium was 0.44 mL/g after correcting for grain size (Table 4–23). All five isotherm points for each of the two floodplain samples are within 10-percent error bars of the 0.2 to 1 mL/g R_d values. Most of the R_d values are relatively small (many are less than 1 mL/g), suggesting that ammonium did not partition significantly to the solid phases.

Cadmium

Corrected mean R_d values for cadmium concentrations range from 21.96 mL/g (13.4 mL/g after grain size correction) for samples from the floodplain to 180 mL/g for samples from the terrace in weathered Mancos Shale (Table 4–22). The R_d values for samples from the terrace unweathered Mancos Shale (mean of 135.37 mL/g) are similar to the values from samples from the terrace weathered Mancos Shale. The R_d values for cadmium were higher than for other contaminants measured in this study, indicating the tendency for cadmium to sorb to the solid fraction of both Mancos Shale and floodplain alluvium. Dissolved cadmium concentrations varied consistently with the amount of sediment. R_d values were nearly linear over an order-of-magnitude range in aqueous concentrations.

Selenium

Corrected mean R_d values for selenium range from 11.44 mL/g (6.98 mL/g after grain size correction) for samples from the floodplain alluvium to 54.73 mL/g for samples from the terrace in weathered Mancos Shale (Table 4–22). The R_d values for samples from the terrace unweathered Mancos Shale (mean of 46.60 mL/g) were similar to samples from the weathered Mancos Shale. The R_d values for both Mancos Shale and floodplain alluvium samples were relatively high, indicating the tendency of selenium to sorb to the solid fraction.

Plots of the dissolved concentrations compared with sediment mass for selenium showed somewhat inconsistent results. The plot for weathered Mancos Shale showed, except for the lowest mass of sediment, that the final concentrations are nearly equivalent regardless of sediment mass. This observation suggests that adsorption is not the dominant uptake mechanism. A possible explanation is that the solutions became reducing enough to precipitate a selenide mineral. Other than one point, the five values from one of the floodplain samples are within error bars of an R_d of 6 mL/g. Results of the second floodplain sample were within error bars of an R_d of 12 mL/g.

Uranium

Corrected mean R_d values for uranium range from 0.54 mL/g (0.33 mL/g after grain size correction) for samples from the floodplain alluvium to 2.13 mL/g for samples from the terrace in unweathered Mancos Shale (Table 4–22). Uranium sorption to floodplain sediments was less than to the Mancos Shale. R_d values for the floodplain samples were relatively low, indicating the tendency for uranium to remain in the aqueous phase, whereas some retardation is to be expected in the Mancos Shale samples.

Dissolved uranium concentrations decrease consistently with the decrease in mass of weathered Mancos Shale; all points are within error bars of R_d values ranging from 1.1 to 2.1 mL/g. The unweathered Mancos Shale sample showed a similar trend; all points are within error bars of R_d values ranging from 1.5 to 2.8 mL/g. All points for the floodplain samples were within the error bars for R_d values ranging from 0 to 0.7 mL/g.

4.4.5 Composition of Salt Deposits

The laboratory study presented in this section addresses one of the data quality objectives defined in the Work Plan (DOE 1998d): "characterize soils as a source of continuing contamination." Analysis of salt deposits was recommended during discussions with site stakeholders at a meeting in Tucson, Arizona, on March 4, 1999. The three objectives of this study were (1) to help characterize soils as a source of continuing contamination, (2) to provide data to help evaluate the areal extent of contaminated ground water, and (3) to provide data that will help determine the origin (mill-related or natural) of ground water contamination. This section presents a summary of the methods and results; a more complete description of the study is in DOE (1999b). Sample locations are shown on Figure 4–56.

4.4.5.1 Background

Salt deposits are common in arid environments. They appear in various forms, such as white powders that coat the ground surface, crystalline deposits on the ground surface, and fracture fillings on outcrops. The deposits are usually white, although some have a yellow tinge. Salts are deposited where ground water evaporates after contacting salt-rich sediments and are most common at locations where water has a high evaporation rate. Salts are found near seeps or in areas where capillary pressure causes ground water to migrate to the ground surface and evaporate. Salts also deposit from evaporation of surface water in closed basins. A wide variety of evaporite minerals are precipitated in the salt deposits.

Salt deposits in Bob Lee Wash, Many Devils Wash, and on the escarpment at the Shiprock site covered large portions of the ground surface. These deposits consisted of translucent white or yellow-tinted crystalline minerals that often encrusted soil or vegetation. Some salt deposits on the floodplain were crystalline, but many occurred as white powders that coated the ground. The crusts and powders were often concentrated in tire tracks (perhaps because the sediment had been compacted, causing an increase in upward capillary water movement). The areal extent of salt deposits in background areas was minor compared with the millsite area. In background areas, the salt deposits were typically observed as thin layers of white powder.

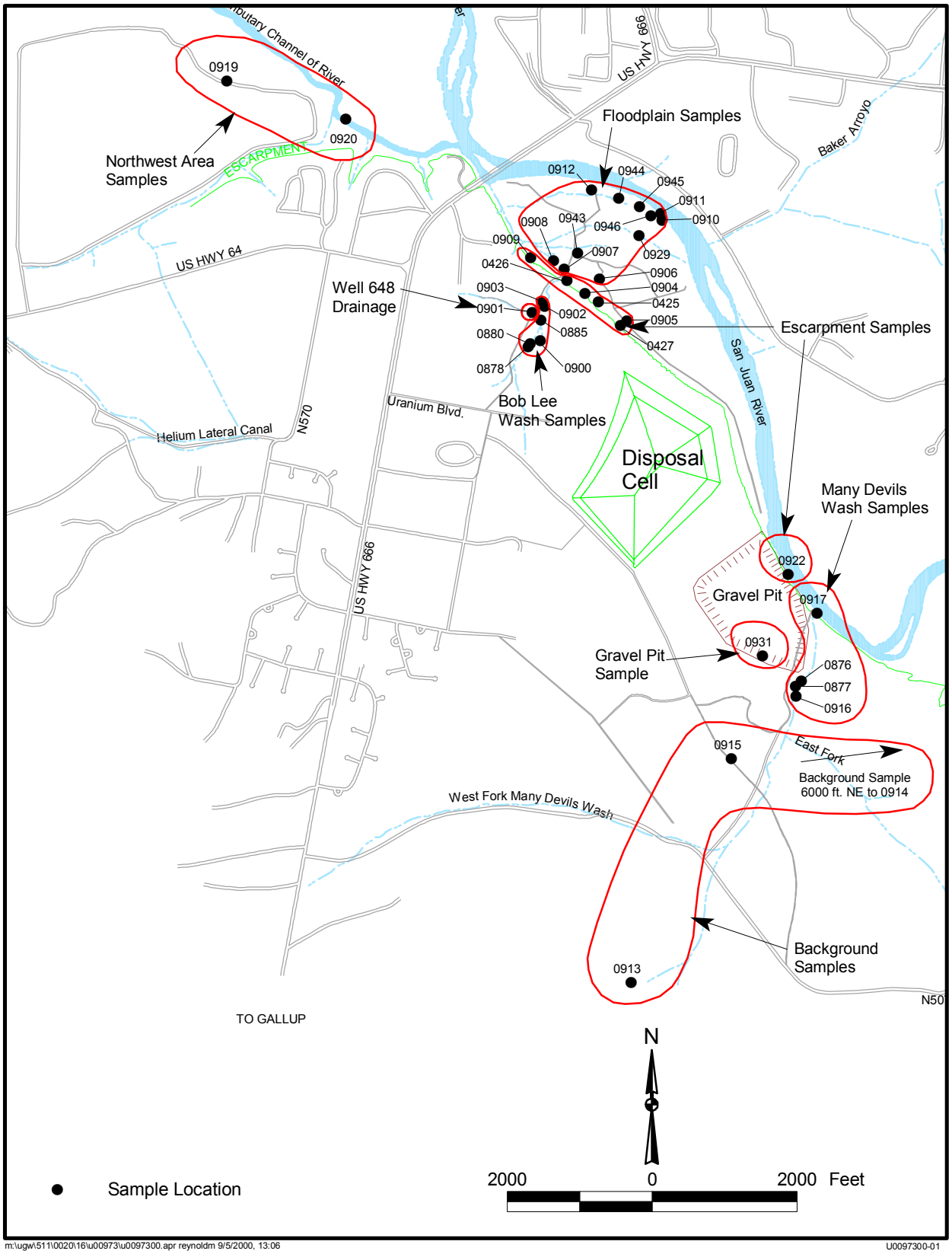


Figure 4–56. Sample Locations for Salt Deposit Study

The chemistry of the salt deposits should reflect, in part, the chemistry of the water from which they were formed. This is particularly true if the water completely evaporates and deposits its entire load of dissolved minerals. If only partial evaporation occurs, the salt deposits will be biased toward the composition of the most insoluble minerals, which are the first to precipitate.

4.4.5.2 Methods

Samples were air dried for about 5 days. Some of the samples contained large proportions of water-insoluble soil, whereas others were mostly water soluble. For those samples that had large amounts of soils, a larger quantity was used so that results would be within analytical detection limits.

Soluble salts were extracted in deionized water following ESL Procedure CB(BE–4) (DOE 1999c). Five grams of each sample was mixed with 500 mL of deionized water. If the conductivity was less than 2,000 microsiemens per centimeter (µS/cm), additional sample was added. Samples were agitated on an orbital shaker for 24 hours, then centrifuged and decanted. The supernatants were filtered through a 0.45 µm filter. The residues were oven dried at 90 ^{o}C and weighed to determine the amount of insoluble soils.

The supernatant solutions were analyzed for pH, alkalinity, and conductivity and for TDS, uranium, nitrate, and sulfate concentrations. TDS concentration was determined by weighing the residue resulting from 100 mL of solution dried at 90 ^{o}C. Supernatant solutions from 12 selected samples were analyzed for antimony, arsenic, cadmium, magnesium, manganese, nitrate, selenium, sodium, strontium, sulfate, uranium, ammonium, calcium, chloride, potassium, iron, and total inorganic carbon. Concentrations of constituents were normalized to the TDS concentration. Thus, a component with a concentration of 10,000 mg/kg (1 percent) means that this component constitutes 1 percent of the water-soluble portion of the sample.

4.4.5.3 Major Ion Composition of the Salt Deposits

The water soluble salts were dominated by sodium sulfate (Table 4–24). Sodium constituted 7.31 to 29.99 percent of the TDS. Other cations that made up significant portions of the salt deposits were calcium (to 10.09 percent) and magnesium (to 7.69 percent). Sulfate concentrations ranged from 20.17 percent (201,672 mg/kg) to 73.01 percent (730,114 mg/kg) of the TDS, excluding one sample that was calculated to have 116 percent (1,161,677 mg/kg) sulfate because of an analysis error (Table 4–25). Other anions included chloride with up to 2.18 percent (Table 4–24) and nitrate with up to 14.91 percent (Table 4–25). Trace elements (arsenic, cadmium, iron, manganese, ammonium, antimony, selenium, and uranium) constituted only 0.002 to 0.015 percent of the salts (Table 4–24). Uranium, selenium, and ammonium dominated the trace element compositions (Table 4–26).

4.4.5.4 Nitrate, Sulfate, and Uranium Concentrations of the Salt Deposits

Nitrate, sulfate, and uranium have high concentrations in ground water at the site. The concentrations of nitrate, sulfate, and uranium in the water soluble salts are listed in Table 4–25, and their areal distributions are shown on Figure 4–57, Figure 4–58, and Figure 4–59, respectively.

Table 4–24. Concentrations (%) of Major Ions in the Salt Deposit Samples [a]

Location	Area	Ca	K	Mg	Na	Sr	SO_4	Cl	NO_3	TIC	Trace[b]	Total
920	NW	2.10	0.10	7.09	14.07	0.07	60.94	1.16	0.04	0.17	0.002	86
914	BKG	4.08	0.12	4.03	18.18	0.02	61.05	1.36	0.81	0.07	0.009	90
915	BKG	10.09	0.54	0.27	18.15	0.08	64.76	0.10	0.13	0.18	0.008	94
885	BLW	5.52	0.18	4.46	13.75	0.09	57.43	2.18	2.08	0.24	0.015	86
900	BLW	0.96	0.06	1.57	26.81	0.01	62.55	2.01	1.63	0.09	0.008	96
425	ESC	5.89	0.10	7.69	7.31	0.11	57.06	0.60	0.78	0.09	0.003	80
907	FP	2.74	0.10	0.37	27.37	0.07	64.03	0.78	0.12	0.15	0.004	96
910	FP	4.00	0.09	3.44	20.32	0.07	64.00	0.39	0.43	0.14	0.006	93
876	MDW	0.63	0.04	1.39	27.27	0.02	62.50	0.77	2.56	0.08	0.005	95
877	MDW	0.60	0.04	1.72	27.15	0.02	61.14	1.08	3.28	0.05	0.005	95
917	MDW	0.64	0.01	0.39	29.99	0.01	65.73	0.25	0.76	0.04	0.004	98
901	W648	2.46	0.13	0.56	27.11	0.07	67.41	1.01	0.04	0.13	0.002	99

Notes: Samples were analyzed by the GJO Analytical Laboratory
[a]Normalized to TDS.
[b]Total percentage of trace elements from Table 4–26
TIC = total inorganic carbon.
NW = Northwest area.
BKG = background.
ESC = escarpment.
FP = floodplain.
MDW = Many Devils Wash.
W648 = artesian well 648.

The maximum nitrate concentration was 14.9 percent (149,096 mg/kg), which was in a sample from location 909 on the escarpment west of Bob Lee Wash (Figure 4–57). This sample was collected from a small ravine that drains a residential area of the terrace and may have been affected by a septic leach field in the area. The possibility of a non-mill-related source for the nitrate is supported by the relatively low uranium concentration of 0.51 mg/kg. Nitrate concentrations in excess of 1 percent (10,000 mg/kg) also occur in samples from some locations in the Bob Lee Wash, Many Devils Wash, floodplain, and escarpment areas. The sample collected at the gravel pit (931) and one of the background samples (913) also had nitrate concentrations greater than 1 percent.

The occurrence of high nitrate concentrations with relatively low uranium concentrations in samples from Many Devils Wash suggests either that nitrate has migrated farther from the millsite than uranium or that there are sources of nitrate not related to milling. A high nitrate concentration (24,357 mg/kg) in a background sample from location 913 indicates that other sources of nitrate may be present. This background sample, however, had a low proportion of soluble salts (98 percent of the sample was insoluble soil). A low percentage of soluble salt could cause the normalized value to be biased by constituents leached from the soil. Refuse dumps, septic leach fields, and leaching from Mancos Shale bedrock are possible, non-mill-related, sources of nitrate.

Sulfate was a major component in all salt deposit samples. Sulfate in ground water and surface water results from leaching of bedrock and soils in the area. Sulfate also is derived from sulfuric acid used in the uranium milling process. More than 50 percent of the TDS in most samples was sulfate, indicating the ubiquitous presence of this constituent. There were no obvious trends in the areal distribution of sulfate (Figure 4–58).

Table 4–25. Concentrations of Nitrate, Sulfate, and Uranium in Salt Deposit Samples (ESL Data)[a]

Location	Area	Recovery (%)	Insoluble Soil (%)	TDS (mg/L)	SO_4 (TDS) (mg/kg)	NO_3 (TDS) (mg/kg)	U (TDS) (mg/kg)
919	NW	97.09	93	2,580	596,512	8,527	1.09
920	NW	92.56	56	3,610	673,130	859	4.90
913	BKG	102.32	98	1,680	532,262	24,357	0.60
914	BKG	99.30	95	4,390	669,704	5,412	0.36
915	BKG	98.49	96	2,270	683,700	2,159	0.66
880	BLW	93.86	58	3,610	500,554	81,717	33.10
878	BLW	93.88	64	2,990	201,672	10,702	76.02
885	BLW	94.07	70	4,910	482,485	12,281	70.35
900	BLW	96.36	26	7,050	689,504	9,858	49.74
902	BLW	99.28	91	3,470	635,447	1,902	27.03
903	BLW	98.80	93	2,240	618,750	1,161	12.90
425	ESC	90.01	54	7,150	642,517	6,028	12.74
426	ESC	99.48	75	2,420	556,612	22,562	0.95
427	ESC	98.98	84	2,920	429,452	116,096	3.56
904A	ESC	85.89	79	2,600	569,231	2,692	0.73
904B	ESC	90.44	43	4,720	643,644	3,814	14.41
922	ESC	88.12	16	7,240	275,552	92,680	1.35
943	FP	99.76	68	3,210	523,053	822	9.72
944	FP	98.26	85	2,570	601,946	20,623	9.14
945	FP	98.78	84	2,880	457,986	1,979	2.33
946	FP	99.10	94	3,310	664,350	2,931	2.54
905	FP	95.67	71	5,020	729,880	12,530	24.84
906	FP	98.97	90	3,520	730,114	750	26.59
907	FP	99.38	72	2,780	657,194	1,424	7.84
908	FP	98.68	91	3,050	661,967	721	53.38
909	FP	96.96	80	3,320	450,602	149,096	0.51
910	FP	99.65	87	2,500	680,800	3,168	8.76
911	FP	92.17	22	4,920	662,602	5,813	13.41
912	FP	99.50	91	3,270	645,260	538	15.23
929	FP	91.56	58	3,340	1,161,677	10,000	5.33
931	GP	95.02	59	3,590	595,543	73,538	0.31
876	MDW	96.92	9	8,800	576,705	25,000	0.69
877	MDW	94.46	8	8,620	614,849	28,886	1.24
916	MDW	94.24	13	8,090	523,239	55,748	1.79
917	MDW	98.04	2	9,570	619,122	6,071	0.39
901	W648	99.68	80	4,020	714,925	871	0.95

[a](TDS) = Normalized to TDS.
NW = Northwest area.
BKG = background.
ESC = escarpment.
ESL = GJO Environmental Sciences Laboratory.
FP = floodplain.
GP = gravel pit
MDW = Many Devils Wash.
W648 = artesian well 648.

Table 4–26. Concentrations (mg/kg) of Trace Elements in the Salt Deposit Samples[a]

Sample No.	Area	As	Cd	Fe	Mn	NH_4	Sb	Se	U	Total
920	NW	<0.55	<0.28	<2.22	<0.28	12.69	<0.28	<0.55	4.74	21.58
914	BKG	<0.46	<0.23	<1.82	0.77	21.57	<0.23	66.74	<0.23	92.05
915	BKG	1.72	<0.44	<3.52	1.01	39.25	<0.44	34.93	0.62	81.94
885	BLW	<0.41	<0.20	<1.63	13.14	42.57	<0.20	12.46	82.08	152.69
900	BLW	0.33	<0.14	<1.13	<0.14	9.16	<0.14	21.13	49.50	81.69
425	ESC	<0.28	<0.14	1.43	<0.14	13.78	<0.14	3.38	11.34	30.63
907	FP	1.01	<0.36	<2.88	<0.36	22.55	<0.36	6.12	7.73	41.37
910	FP	<0.80	<0.40	<3.20	2.08	42.40	<0.40	<0.80	8.56	58.64
876	MDW	0.35	<0.11	2.26	0.16	3.50	<0.11	43.07	0.86	50.43
877	MDW	<0.23	<0.12	<0.93	<0.12	3.57	<0.12	45.13	1.28	51.48
917	MDW	0.68	<0.10	<0.84	<0.10	8.33	<0.10	28.32	0.38	38.85
901	W648	1.12	<0.25	<1.99	2.51	17.01	<0.25	<0.50	1.04	24.68

Notes: Samples were analyzed by the GJO Analytical Laboratory.
[a]Normalized to TDS.
NW = Northwest area.
BKG = background.
ESC = escarpment.
FP = floodplain.
MDW = Many Devils Wash.
W648 = artesian well 648.

Uranium concentrations measured in the GJO ESL were as high as 76.02 mg/kg in the salt deposit samples (Table 4–25). Uranium concentrations in samples from Bob Lee Wash, along the escarpment, and on the floodplain are higher than background concentrations (Figure 4–59). The uranium in these deposits was likely derived from mill effluents. Uranium concentrations in the Many Devils Wash salt deposit samples were close to background concentrations (Figure 4–59).

4.4.5.5 Constituents Other than Nitrate, Sulfate, and Uranium

Cadmium and antimony concentrations were below their detection limits (Table 4–26). Arsenic concentrations were low, and most were below detection. The highest arsenic concentration was 1.72 mg/kg, which was in a background sample from location 915 (Table 4–26). Most of the iron concentrations were less than the detection limit. The highest detectable iron concentration was 2.26 mg/kg (Table 4–26). Ammonium concentrations ranged from 3.50 to 42.57 mg/kg (Table 4–26). Although these concentrations are higher than those of many of the trace elements, they are much lower than nitrate concentrations. Ammonium concentrations in background samples are similar to concentrations in on-site samples. A sample from location 885 in the Bob Lee Wash area had a manganese concentration of 13.14 mg/kg (Table 4–26). All other samples had manganese concentrations of 2.51 mg/kg or less, and many were below the detection limit. Selenium concentrations ranged from less than 0.50 mg/kg to 66.74 mg/kg; the highest concentration was in a background sample from location 914 (Table 4–26).

The concentrations of these constituents (arsenic, cadmium, iron, manganese, ammonium, antimony, and selenium) are probably similar to concentrations in many arid salt deposits and may not be related to milling activities.

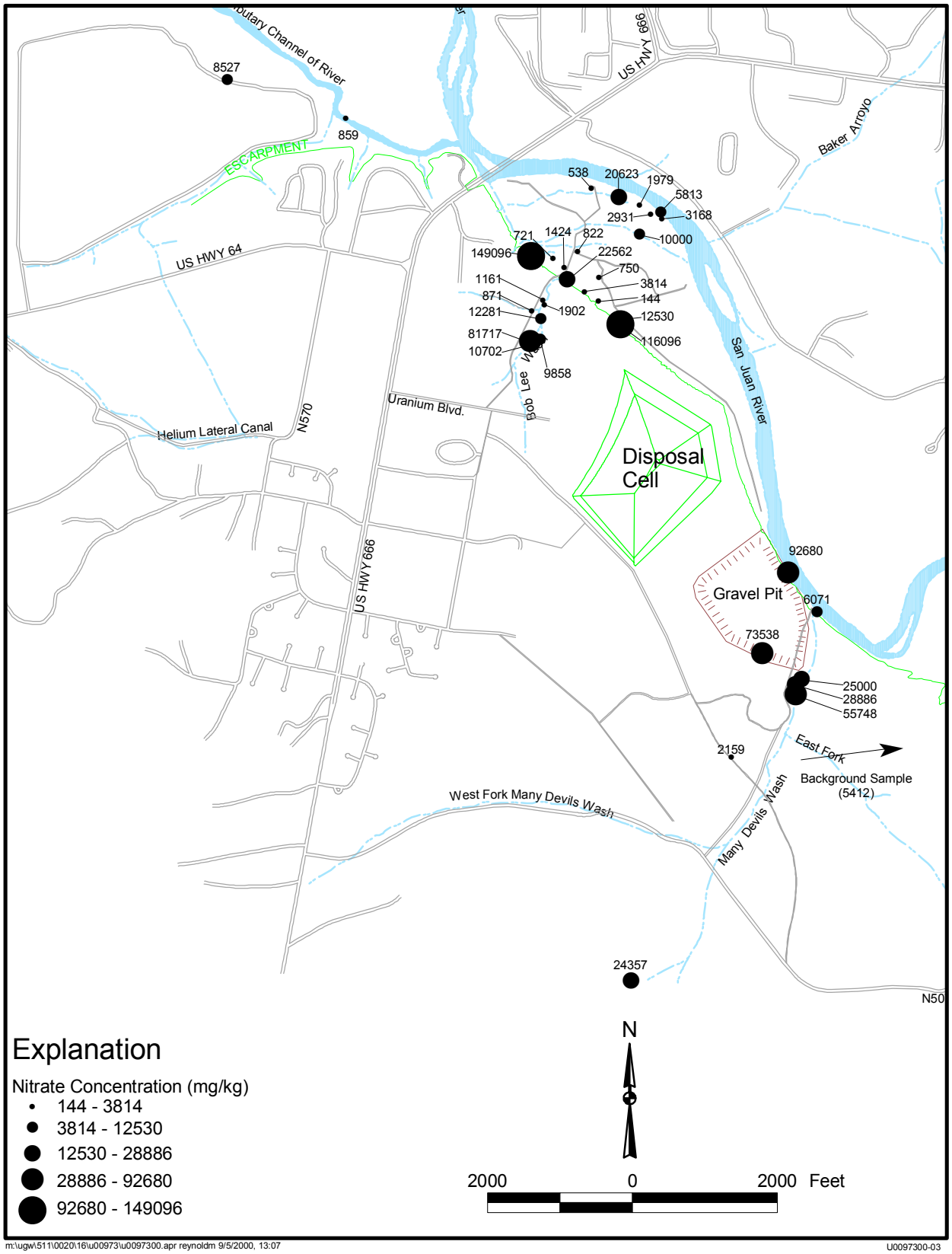


Figure 4–57. Nitrate Concentrations in Salt Deposits

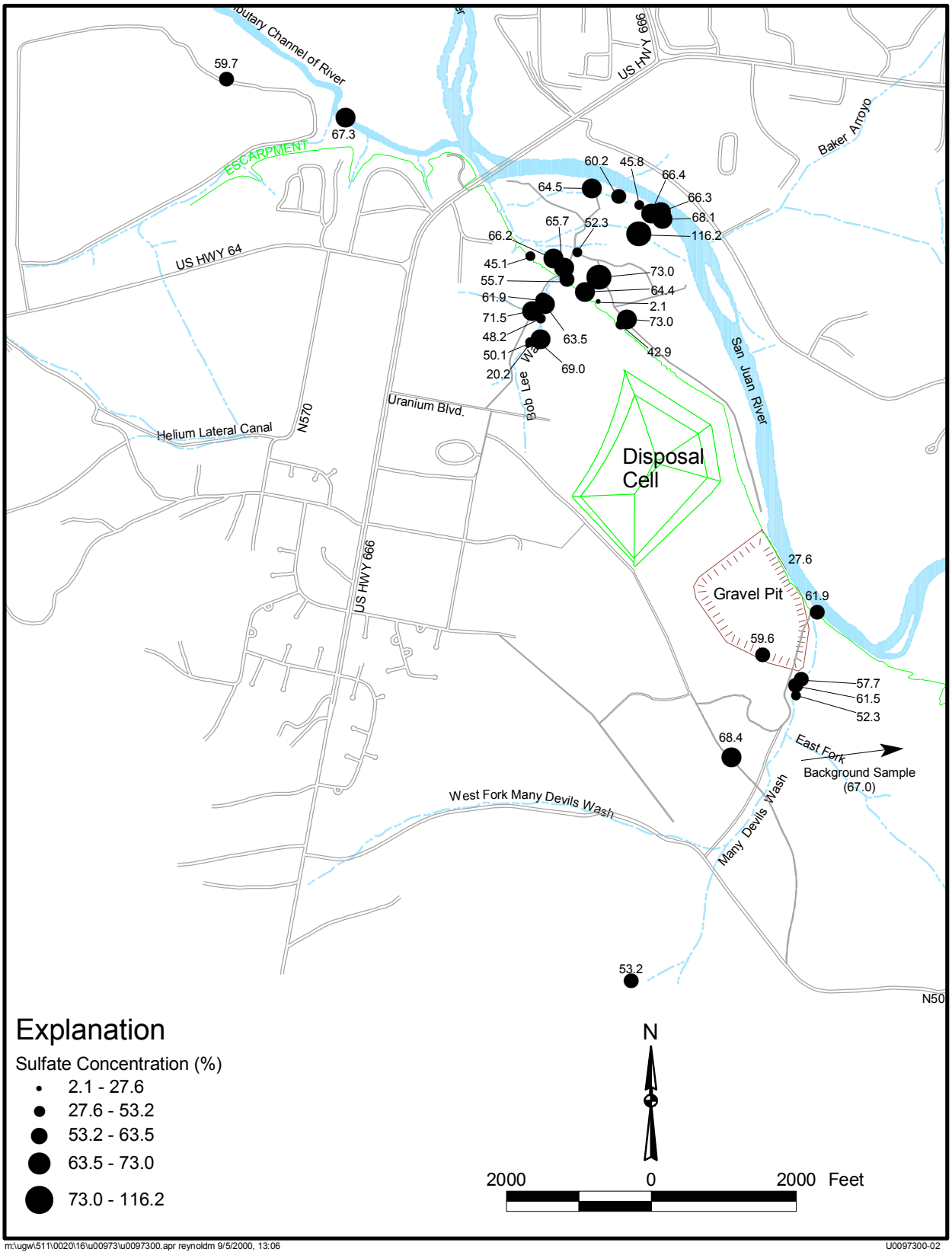


Figure 4–58. Sulfate Concentrations in Salt Deposits

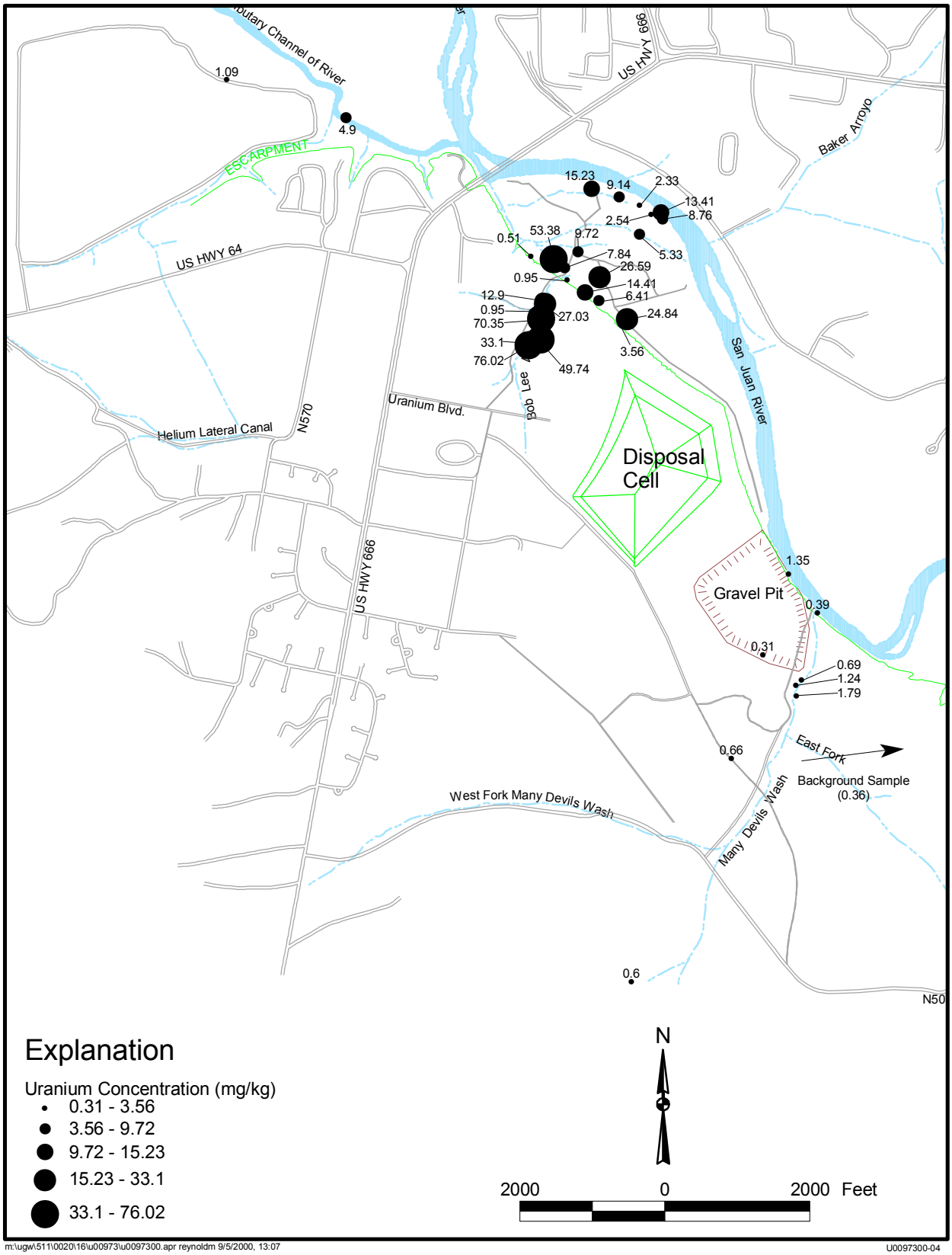


Figure 4–59. Uranium Concentrations in Salt Deposits

4.4.5.6 Mineralogy of Salt Deposits

The Mancos Shale also contains large quantities of gypsum, calcite, and dolomite; and runoff water from the shale can be saturated with respect to calcite and gypsum (Evangelou and others 1984). Larrone (1977) studied the soluble fraction in the Mancos Shale and determined that it contains 2 percent soluble minerals. These leached constituents account for the white crusts of efflorescent minerals often found in Mancos Shale terrain. In other studies of the Mancos Shale's contribution to salts in the Colorado River, sulfates of sodium, calcium, and magnesium are listed as dominant soluble species of efflorescent deposits, and species of nitrates and chlorides are also indicated (Laronne 1977; Johnson and Schumm 1982; and Schumm and Gregory 1986). Laronne (1977) also suggests that a cause and effect relationship exists for the heavy efflorescence buildups on the sides of gullies in Mancos Shale. The degree of efflorescent minerals depends on water quality of storm runoff. The chemical composition of storm runoff is a function of near-surface soluble minerals, more so than for slower precipitation events. When these rapid events occur, surface soluble fractions leach rapidly and are not readily absorbed into the relatively impermeable Mancos Shale. This can result in encrustations on the bottom and sides of ephemeral tributaries, especially in more arid regions. Thick salt encrustations exist at the Shiprock site in the lower channeled portion of Many Devils Wash.

Three samples of efflorescent minerals were collected at the Shiprock site for X-ray diffraction analysis. Two samples were within 20 ft of established efflorescence (salt crust) sample point 904 between seep locations 425 and 426; one sample was from the base of the escarpment and one sample was from the base of the talus slope. The third sample was collected just above the knickpoint (salt crust sample 916) in Many Devils Wash. Samples were placed in plastic bags, sealed for transportation, and allowed to air dry before they were ground to a fine powder. All three samples were run from 3 to 60 degrees on a Rigaku Miniflex instrument using CuK-alpha radiation and a nickel filter. A Rigaku-supplied peaks search-and-match software package was used to help identify natural phases. Results indicated the dominant efflorescence to be thenardite (Na_2SO_4) with minor amounts of blodite [$Na_2Mg(SO4)_2{\cdot}4(H_2O)$] and possible sideronatrite [$Na_2Fe(SO_4)_2(OH){\cdot}3(H_2O)$]. Other phases found in the samples were quartz and calcite.

These results are similar to previous analyses of efflorescent minerals found at the old Climax millsite in Grand Junction, Colorado (DOE 1999f). Samples were collected near seeps draining ground water from the alluvium that overlies Mancos Shale. Efflorescent samples from Grand Junction consisted of thenardite, blodite, wattevillite [$Na_2Ca(SO4)_2{\cdot}4(H_2O)$], and one sample contained halite (NaCl). The similar mineralogy of salts precipitated from ground water flowing over the Mancos Shale in both locations suggests that similar concentrations of ions are available for leaching.

4.4.6 Column Leaching of Alluvial Aquifer Sediment

The laboratory study presented in this section addresses one of the data quality objectives defined in the Work Plan: "characterize leachability conditions of alluvial material in several contaminated areas of the floodplain." The study examined the effectiveness of San Juan River water to leach uranium and other constituents from floodplain alluvial sediments. The methods and results are summarized here; a more complete description of the project is provided in DOE (1999a).

4.4.6.1 Background

Column leaching is often used to estimate the concentration of contaminants that will occur when a solution flows through contaminated sediments. Effluent concentration profiles over time can also provide information that indicates how rapidly the concentrations will decrease.

Contaminants can be present in sediment in different forms, including crystalline structure of minerals, adsorbed to mineral surfaces, and immobile pore fluids. Some of the forms of contamination are more easily released than others. Complexing agents in the leach solution enhance the release of some contaminants. Therefore, the choice of leach solution is important. An example is uranium, which desorbs more efficiently in a solution with high concentrations of dissolved carbonate. The pH and oxidation potential of the solution can also affect the leaching process.

The goal of this study was to determine the concentrations of constituents that are to be expected if San Juan River water were to flow through contaminated alluvial aquifer sediments in the floodplain. Therefore, a leaching solution consisting of the major ions in San Juan River water was used. Leaching with water of a different composition is likely to produce different concentrations in the effluent.

4.4.6.2 Methods

Alluvial aquifer sediment was sampled from six borings. Three borings (locations 854, 856, and 864) are in the contaminated portion of the millsite floodplain, and three (locations 850, 851, and 852) are in the background floodplain. Locations of these borings are shown on Plate 1. The samples were collected by driving a split-spoon tube into the alluvial sediment. In some cases the split-spoon was incapable of retrieving a suitable sample and auger cuttings were used instead. The samples from the millsite floodplain were selected from the most uranium-contaminated portion of the ground water plume. These samples are believed to be representative of those areas that are likely to release the most contamination from the alluvial sediments.

The cores consisted of partially disaggregated floodplain alluvial sediment. Splits of the cores were placed in aluminum pie pans exposed to the air until visibly dry (about 5 days). The sediments were crushed lightly by hand to increase the drying rate. The dried sediment was sieved to less than 3 mesh (6 mm). About 4 in. of the material was placed in the columns at a time and was compacted by lightly tapping the material with a rubber mallet.

This study used a procedure similar to GJO ESL standard column test procedure CB(CT-1) (DOE 1999e). Six columns (2-in. diameter and about 18 in. in height) were constructed from clear acrylic; each column contained sediment from one location. Synthetic San Juan River solution was pumped with a peristaltic pump set at 0.8 mL per minute from bottom to top through the column. The major-ion chemistry of San Juan River water collected at location 546 was synthesized from reagent-grade chemicals.

Effluent samples were collected every 12 hours. Concentrations of uranium and nitrate, pH, electrical conductivity, oxidation-reduction potential, and alkalinity were measured in the GJO ESL soon after sample collection. Samples were preserved and submitted to the GJO Analytical

Chemistry Laboratory for analysis of arsenic, cadmium, magnesium, manganese, sodium, nitrate, antimony, selenium, sulfate, strontium, uranium, and ammonium.

4.4.6.3 Results and Discussion

Data are plotted as concentration in relation to the number of pore volumes (using midpoints) that have passed through the column. A pore volume was measured as the amount of solution used to fill each sediment column.

Nitrate, Sulfate, and Uranium

Nitrate-The concentrations of nitrate in effluents from the columns that contain floodplain sediments are similar to those from columns that contain background sediments (Figure 4–60). The concentrations are much lower than nitrate concentrations observed in the ground water on the floodplain near the millsite. Apparently, nitrate is strongly partitioned into the aqueous phase and little is contained on the solid particles.

Sulfate-The sulfate concentrations in the first effluent from columns that contain sediment from borings 854, 856, and 864 were 3,200 mg/L, 576 mg/L, and 485 mg/L, respectively (Figure 4–60). These high levels decreased to about 150 mg/L after 10 pore volumes. Concentrations of sulfate in the effluents of all three columns containing background sediment were nearly constant at about 100 mg/L, which is similar to the influent concentration (121.3 mg/L). The higher concentrations of sulfate from the millsite floodplain were probably due to dissolution of sulfate salts that were deposited from the ground water as the sample was dried.

Uranium-Effluents from all three columns with alluvial aquifer sediments from a boring on the contaminated floodplain had higher uranium concentrations than those from the background borings (Figure 4–60). The first effluent from the column containing sediment from boring 854 had a uranium concentration of 0.073 mg/L. The concentration decreased rapidly and was less than the UMTRA MCL (0.044 mg/L) after about 4 pore volumes. These results suggest that some mill-related uranium contamination is in the alluvial sediments. Alternatively, some of the uranium in the samples could have been deposited from contaminated ground water as the sample dried. Uranium released during flushing with San Juan River water is likely to be slightly above the UMTRA MCL initially but should rapidly decrease to relatively low levels.

Constituents Other Than Nitrate, Sulfate, and Uranium

Ammonium-The ammonium concentration in the first sample from the column containing sediment from boring 854 was 1.97 mg/L. The concentration decreased to 0.29 mg/L after 10 pore volumes. Effluent concentrations of ammonium from all the other columns were much lower; the highest value was 0.086 mg/L from the column containing sediment from background boring 851. The highest concentration of 1.97 mg/L is relatively low compared with ammonium concentrations in ground water samples from the site.

Antimony-The highest concentrations of antimony were in leachate from the column containing sediment from background boring 850. These results are consistent with the observation that elevated concentrations of antimony are rare in the floodplain ground water. San Juan River water will probably not leach antimony from the floodplain at concentrations above background.

Arsenic-Effluents from all three columns with sediment samples from the contaminated floodplain had higher concentrations of arsenic than the background samples. The highest concentration was 0.0083 mg/L from the column containing sediment from boring 856. Although leachate concentrations from the millsite floodplain samples are higher than those in background samples, the concentrations are well below the UMTRA MCL of 0.05 mg/L. These results suggest that arsenic will not be leached from the floodplain at concentrations above the MCL.

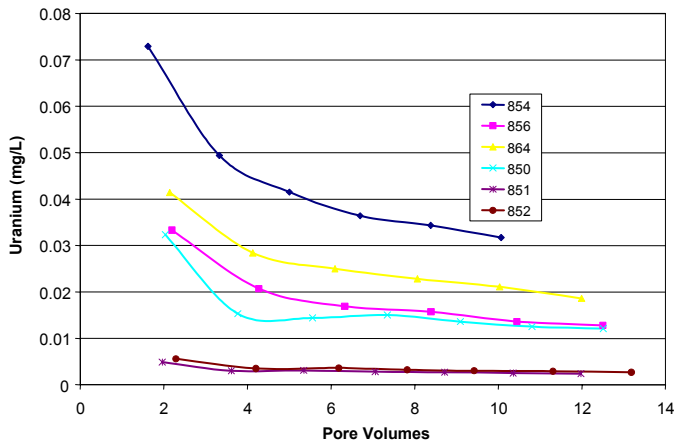
Cadmium-Concentrations of cadmium in effluents from all columns were less than the detection limit of 0.001 mg/L. These results are consistent with the relatively rare occurrences of elevated cadmium concentrations in the ground water at the millsite. San Juan River water will probably not leach cadmium from the floodplain at concentrations above the MCL (0.01 mg/L).

Magnesium-The magnesium concentrations in effluents from the three columns containing sediment from background borings were about the same as the concentration in the synthetic San Juan River water (2.99 mg/L), indicating that no magnesium was exchanged with the sediment. The first effluent sample from the column containing sediment from boring 854 had a magnesium concentration of 265 mg/L. It is likely that the magnesium concentration in this first sample is derived from the dissolution of water-soluble salts in the sample. Effluents from all the other columns had concentrations less than 50 mg/L, and most were less than 20 mg/L. The three columns with sediments from the millsite floodplain had higher concentrations than the three columns with background location sediments.

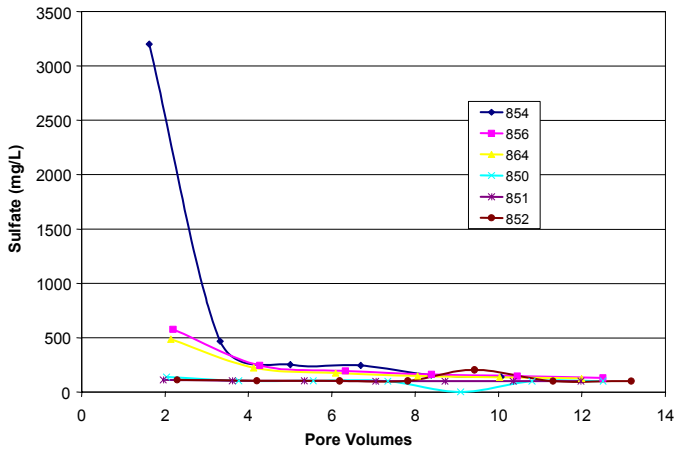
To help evaluate the significance of the magnesium concentration in the column effluents, those concentrations can be compared with concentrations in ground water from background wells and with San Juan River water. Samples from wells on the opposite side of the San Juan River from the disposal cell had magnesium concentrations ranging from 40.8 to 318 mg/L (DOE 1999a). Samples of river water at upstream locations 888 and 898 had magnesium concentrations of 32.3 and 12.2 mg/L, respectively, in March 1999. The magnesium concentrations in the column leachates are lower than those in background ground water and similar to those in the San Juan River. These results suggest that leaching of floodplain alluvial sediments with San Juan River water will not contribute a significant amount of magnesium.

Manganese-Manganese concentrations in all effluents from two of the columns containing sediments from the contaminated floodplain (borings 856 and 864) were less than 0.0135 mg/L and are lower than the concentrations in effluents from the background samples. The manganese concentration in effluent from the other column containing sediment from the floodplain (boring 854) was initially 0.552 mg/L but decreased rapidly to about 0.040 mg/L. Effluents from all three columns containing background sediments had manganese concentrations of about 0.060 mg/L. These results suggest that San Juan River water will not leach manganese appreciably from the floodplain alluvium.

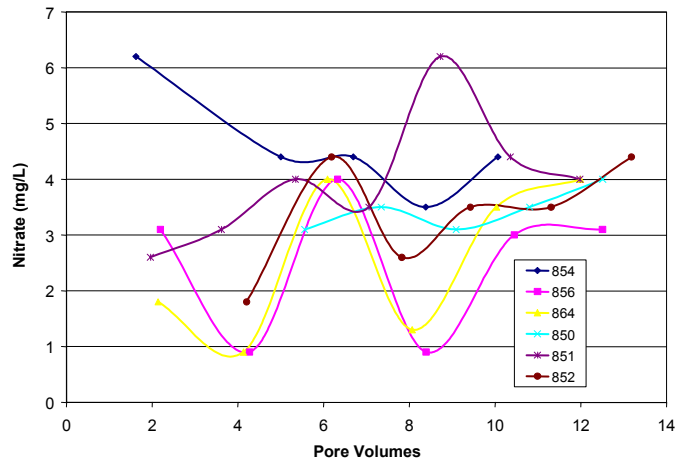
Selenium-All three columns containing alluvium from the contaminated floodplain had effluent concentrations of selenium that were less than the detection limit of 0.002 mg/L. Effluent from all three background columns had selenium concentrations of 0.007 to 0.011 mg/L initially, and the concentrations decreased rapidly to between 0.0018 to 0.003 mg/L. The Mancos Shale is known to be a source of selenium, which contaminates ground water. The higher concentrations of selenium in the effluents from the background sediment samples is probably the result of the natural leaching of Mancos Shale.



Uranium



Sulfate



Nitrate

(850, 851, and 852 = Background Floodplain; 854, 856, and 864 = Contaminated Floodplain)

Figure 4–60. Column Leaching Results

Sodium-The concentration of sodium in effluent from the column containing floodplain sediments from boring 854 was initially 516 mg/L, but the concentration decreased after the first pore volume to 54.9 mg/L. The first effluent is probably affected by the initial dissolution of soluble salts. Sodium concentrations in all other columns was about 30 mg/L, which is near the concentration (30.12 mg/L) in the synthetic San Juan River water. These results indicate that the sodium concentration may increase slightly initially, but no sustained increase in sodium concentration of the San Juan River water is likely.

Strontium-Concentrations of strontium in effluents from the three columns containing floodplain sediments (borings 854, 856, and 864) were higher (1.0 to 2.22 mg/L) initially than those in the columns containing background sediments. The concentrations in the columns containing floodplain sediments decreased to about 0.5 mg/L after several pore volumes. Effluent concentrations of strontium from the columns containing background sediments were about 0.15 mg/L initially but increased to about 0.5 mg/L after several pore volumes. These results suggest that a small amount of soluble strontium may be released from the alluvial sediment initially, but that no sustained contribution will occur. Concentrations of strontium in the San Juan River from locations 888 and 898, upgradient of the millsite, are 1.29 and 0.786 mg/L, respectively. Because strontium concentrations in the leachates are lower than the concentrations in the river, no significant contribution of strontium to San Juan River water flowing through the alluvial aquifer is likely.

4.4.7 Fate and Transport

Some constituents are readily transported by ground water, whereas others are strongly partitioned on immobile solid mineral phases. The rate at which contamination migrates and the concentration in the ground water are controlled by the biogeochemical nature of the aquifer. The biogeochemical factors that typically affect migration of constituents that are currently COCs or have historically been detected at the Shiprock site are discussed in this section.

4.4.7.1 Ammonium

Under oxidizing conditions, ammonium reacts to form nitrite (NO_2^-), nitrate (NO_3^-), or nitrogen gas (N_2). Some of the transformation reactions are catalyzed by microbiological activity. Ammonium (mainly) and nitrate complexes were used during the milling process at the Shiprock site. It is reasonable to assume that most of the nitrate concentration in the ground water is an oxidation product of ammonium. The MCL for nitrate is 44 mg/L. An equivalent would be a concentration of 12.7 mg/L ammonium.

Ammonium is a strong cation exchanger on clay minerals that are present in most aquifers. At pH values (about 9) above those in San Juan River water, it will transform to ammonia (NH_3) and is volatile. Ammonium is also a nutrient used by plants.

4.4.7.2 Antimony

Antimony is geochemically similar to arsenic (Hem 1985). Because of its low abundance in ground water (about one-tenth that of arsenic), it has not been studied in detail and little is known about its chemical mobility. In the floodplain, 29 of 36 sampled wells had antimony concentrations exceeding the detection limit of 0.0004 mg/L, whereas only 14 of 35 terrace wells sampled had antimony concentrations exceeding the detection limit. The highest antimony

concentration detected in ground water samples was only 0.0038 mg/L. In surface water, antimony was detected in 21 of 34 locations, and the highest concentration was only 0.0014 mg/L.

4.4.7.3 Arsenic

Arsenic occurs commonly in nature in two oxidation states, As^{3+} and As^{5+}. The arsenate anion ($H_2AsO_4^-$) is the dominant dissolved species under the pH conditions in the Shiprock ground water. Under strongly anaerobic conditions it can also occur with a negative oxidation state and, in the presence of sulfur, form arsenide minerals. Arsenate will form minerals with ferric iron and other metal cations, but these minerals are not likely to precipitate at the low concentrations present in the Shiprock ground water. One form of arsenic (As^{5+}) adsorbs strongly on sediment minerals such as iron oxyhydroxides, whereas As^{3+} is less adsorptive. Most of the arsenic in sediments at Shiprock is probably adsorbed.

The MCL for arsenic is 0.05 mg/L. In the floodplain, 19 of 36 sampled wells had arsenic concentrations exceeding the detection limit of 0.0004 mg/L. Only four of 35 wells sampled on the terrace had arsenic concentrations exceeding the detection limit. The highest arsenic concentration detected in ground water samples was only 0.0039 mg/L. In surface water, arsenic was detected in only 6 of 34 locations, and the highest concentration was only 0.0009 mg/L.

4.4.7.4 Cadmium

Cadmium is present in ground water as the uncomplexed cation Cd^{2+} or complexed with an anion (e.g., $CdSO_4^0$). Cadmium readily substitutes for Ca^{2+} in carbonate minerals. Coprecipitation with calcite ($[Ca,Cd]CO_3$) is the most likely mechanism for removal of cadmium from the alluvial ground water. Because the aquifer is saturated with calcite, this mechanism is likely to keep cadmium concentrations low. Cadmium can precipitate as greenockite (CdS) under sulfate-reducing conditions. Cadmium will also effectively adsorb to ferric oxyhydroxides.

Cadmium concentrations in ground water from two wells (603 and 730) immediately south of the disposal cell exceeded the MCL of 0.01 mg/L. The highest concentration was 0.0471 mg/L from well 730. Samples from 10 additional terrace wells had cadmium concentrations that exceeded the detection limit of 0.0003 mg/L. Only 10 of 36 wells sampled on the floodplain had cadmium concentrations exceeding the detection limit. No surface water sample locations had cadmium concentrations exceeding the detection limit.

4.4.7.5 Magnesium

Magnesium is present in the dissolved state as Mg^{2+} or as carbonate or hydroxide complexes. It forms minerals with carbonate such as dolomite [$CaMg(CO_3)_2$] or magnesite ($MgCO_3$) and can substitute for calcium in calcite. Magnesium is a major cation in many minerals and its concentration in ground water at Shiprock is probably controlled largely by the precipitation and dissolution of these minerals.

4.4.7.6 Manganese

Manganese mobility is related to the oxidation-reduction potential of a soil or sediment. Manganese forms oxide minerals under oxidizing conditions and is soluble under more reduced

conditions. Therefore, the more oxidized the state of a sediment, the more likely it is to have higher concentrations of manganese. Manganese occurs in the 2+ and 4+ oxidation states at the Shiprock site. In the dissolved state, it is present mainly as Mn^{2+} ion. Its redox chemistry is similar to that of iron. Manganese will also partition to sediment by substituting for calcium in calcite.

The average concentration of manganese in ground water samples from the Shiprock floodplain is 1.63 mg/L. Concentrations of manganese in samples from the millsite floodplain are variable, and many are less than background. The Mancos Shale may contribute manganese to the ground water.

4.4.7.7 Molybdenum

Molybdenum most commonly occurs in nature in the +4 or +6 oxidation states. It is most commonly transported in ground water as an anionic molybdate (MoO_4^{2-}) species. Molybdate can form a variety of polymeric species in solution. It can also form weak complexes with sodium, potassium, and calcium.

Molybdenum is generally quite soluble in ground water. At uranium milling sites it often migrates relatively far from the source areas, similar to uranium.

Molybdate sorbs readily to ferric oxyhydroxides and oxides—a dominant mechanism for retardation in most aquifers. Molybdate will combine with calcium, sodium, or iron to form metal molybdate minerals. Under reducing conditions molybdenum will combine with sulfide to form molybdenite (MoS_2), which has low solubility.

The UMTRA MCL for molybdenum is 0.1 mg/L. Molybdenum concentrations in ground water exceed the MCL only in small areas along the escarpment near the disposal cell; no surface water locations had molybdenum concentrations that exceeded the MCL. In floodplain ground water the MCL was exceeded in only two of 36 wells; the highest concentration was 0.349 mg/L at well 863. In terrace ground water, the MCL was exceeded at only one of 36 wells—a 0.856 mg/L concentration at well 824.

4.4.7.8 Nitrate

The oxidation state of nitrogen in nitrate (NO_3^-) is +5. It does not complex significantly with other ions under ground water conditions and is transported without significant interaction with the rock matrix. Under reducing conditions, nitrate can transform to nitrite, elemental nitrogen, or ammonium. The reduction is catalyzed by microbial processes. Significant denitrification is not expected to occur without a suitable organic nutritional source such as acetate. Therefore, nitrate probably transports nearly conservatively through the aquifer. Concentrations decrease by mixing with other ground water and by dispersion. In high concentrations, such as in salt deposits, nitrate can precipitate in water-soluble minerals. A small amount of nitrate can also adsorb to sediments.

The MCL for nitrate is 44 mg/L. Nitrate concentrations are very high in several areas at the Shiprock site; the highest concentration from the February 2000 sampling is 8,790 mg/L from terrace well 813. Ammonium (mainly) and nitrate complexes were used during the milling

process at the Shiprock site. It is reasonable to assume that most of the nitrate concentration in ground water is due to oxidation of ammonium.

4.4.7.9 Radium

Two radium isotopes are present in the ground water. Ra-226 is a decay product of U-238 and has a half-life of 1,600 years. Ra-228 is a decay product of Th-232 and has a half-life of 5.7 years. Radium preferentially attaches to particles, and dissolved concentrations are typically low. One of the most important reactions to fixate radium is the coprecipitation in $(Ba,Ra)SO_4$. Radium substitutes readily for barium because of its similar ionic radius. Because of the low solubility of barium sulfate, radium has not migrated far from the tailings at most uranium millsites.

The MCL for radium (Ra-226 + Ra-228) is 5 pCi/L. Concentrations in ground water samples from the floodplain and all surface water samples are below the MCL; concentrations in samples from four wells on the terrace west of the disposal cell exceed the MCL; the highest radium concentration is 15.93 pCi/L at well 602.

4.4.7.10 Selenium

Aqueous selenium occurs predominantly as selenate $(SeO_4)^{2-}$ or selenite $(SeO_3)^{2-}$; selenate is probably favored under the oxidized conditions of the alluvial aquifer. Concentrations of selenium are not high enough to precipitate selenium minerals at the Shiprock site. Selenium can substitute for sulfur in sulfur-bearing minerals and can precipitate as ferroselite ($FeSe_2$) or coprecipitate with pyrite (FeS_2) under reducing conditions. Selenate adsorbs to ferric oxyhydroxides at moderate to low pH values.

The MCL for selenium is 0.01 mg/L. Concentrations in ground water and surface water at the site exceed the MCL in extensive areas in both the floodplain and terrace. The highest concentration in floodplain ground water from the February and April 2000 samplings was 1.04 mg/L at well 615. Concentrations exceeded the MCL in samples from 18 of 43 wells in the floodplain. In the terrace the highest ground water concentrations from these samplings was 6.52 mg/L at well 812; 35 of 45 terrace wells had concentrations that exceeded the MCL. The highest selenium concentration in surface water from the February 2000 sampling was 2.32 mg/L at location 889.

The Mancos Shale has high concentrations of leachable selenium that is known to contaminate ground water. High concentrations of selenium in samples of ground water from the terrace area at the Shiprock site are related to the milling process or are derived from leaching of the Mancos Shale, or both.

4.4.7.11 Sodium

Sodium occurs in ground water as the monovalent cation Na^+ and is a major component of many minerals. It is relatively mobile in ground water but can readily exchange for other cations on clays and oxyhydroxide minerals. In arid areas, it often occurs in relatively high concentrations in ground water because of the dissolution of evaporite minerals.

There is no MCL for sodium. Concentrations vary in ground water at the Shiprock site because of the varying amounts of dissolution of salt minerals.

4.4.7.12 Strontium

Strontium is present in the dissolved state as Sr^{2+} or as carbonate or hydroxide complexes. Its chemistry is similar to that of Ca^{2+} and forms minerals with carbonate such as strontianite ($SrCO_3$); strontium can substitute for calcium in calcite. Strontium is a major cation in many minerals and its concentration in ground water at Shiprock is probably controlled by the precipitation and dissolution of these minerals.

4.4.7.13 Sulfate

In alluvial ground water, dissolved sulfur occurs mainly as the unassociated sulfate ion (SO_4^{2-}). The precipitation of gypsum ($CaSO_4$) or sodium sulfate (Na_2SO_4) can partition significant amounts of sulfate into the solid phase. The concentrations of sulfate in solution will remain high even in the presence of these minerals. Much of the concentration gradient in ground water is caused by mixing with other ground water and dispersion. Under reducing conditions brought about by microbial stimulation, sulfate can form sulfide that precipitates heavy metals and arsenic. Investigations by the NABIR Program (McKinley and Long 1999) at the Shiprock site determined that sulfide concentrations were low in ground water samples from the floodplain and the terrace.

4.4.7.14 Thorium

Thorium may be present at uranium millsites because Th-230 is a decay product of U-238. In general, thorium has a very low solubility in ground water due to the formation of thorium oxides that have very low solubilities. Thorium will become mobile only if low pH and high sulfate conditions prevail. Such conditions are rare at uranium milling sites.

There is no UMTRA MCL for thorium. Thorium isotopes 228, 230, and 232 were analyzed in ground water samples from four terrace wells just north and west of the disposal cell during the February 2000 sampling. Well 602, just west of the disposal cell in the ore-processing area of the former millsite, had the highest concentrations of each of the isotopes for the sampled wells.

4.4.7.15 Uranium

Most naturally occurring uranium is either in the uranyl (6+) or the uranous (4+) oxidation state. The uranyl form is predominant in oxidized ground water. The uranyl ion forms strong aqueous complexes with carbonate, and uranyl dicarbonate [$UO_2(CO_3)_2^{2-}$] is a dominant mobile species. Uranium adsorbs to ferric oxyhydroxide and clay minerals in soils and rocks. Under reducing conditions, uranium precipitates as uraninite (UO_2), which has a low solubility. The reduction is catalyzed by microbial activity.

The MCL for uranium is 0.044 mg/L (Table 2–1). Uranium concentrations are very high in several areas of ground water and surface water. The highest concentration in ground water from the February and April 2000 samplings was 3.77 mg/L at floodplain well 854. Concentrations exceeded the MCL in samples from 29 of 43 wells in the floodplain and in 31 of 45 wells in the terrace. The highest uranium concentration in surface water from the February 2000 sampling

was 1.71 mg/L from terrace location 885. For surface water, concentrations exceeded the MCL in six of 18 samples from the floodplain and in nine of 16 samples from the terrace.

4.4.7.16 Vanadium

Vanadium exists in nature in three oxidation states: V^{3+}, V^{4+}, and V^{5+} (Hem 1985). In most ground waters, dissolved vanadium occurs predominantly in the V^{5+}oxidation state, forming anionic complexes with oxygen and hydroxide. In some contaminated ground waters, V^{5+}sulfate complexes can be significant. At uranium millsites, V^{5+} is generally less mobile in ground water than uranium and most other contaminants.

Formation of metal vanadates such as $Fe(VO_3)_2$ or Ca $(VO_3)_2$ may control the mobility in some ground water systems. Sorption to aquifer minerals, especially ferric oxyhydroxides, is probably an important retardation mechanism. Vanadium (+5) can also combine with uranyl to form uranium-vanadium minerals such as carnotite (K UO_2 VO_4). Under reducing conditions, vanadium forms low-solubility minerals that would maintain low dissolved concentrations.

There is no UMTRA MCL for vanadium; however, high concentration of vanadium are known to be harmful to health. An ACL was established at 0.33 mg/L for the Rifle, Colorado, UMTRA site. Concentrations of vanadium are high in ground water in several small areas near the disposal cell. The highest concentration of vanadium in ground water from the June 1999 and February 2000 samplings was 0.887 mg/L in a sample from well 730, which was the only well of 35 sampled terrace wells that had vanadium concentrations exceeding 0.33 mg/L. This well is in the area of the former raffinate ponds where vanadium liquors were stored during milling. All 23 floodplain wells and all surface waters sampled had vanadium concentrations less than 0.01 mg/L.

4.4.8 Mancos Shale and Associated Ground Water Chemistry

The following information is presented to support statements made in Sections 5.0 and 7.0 that water contacting Mancos Shale can acquire relatively high concentrations of non-milling-related uranium, selenium, and sulfate. Studies show that concentrations of uranium and selenium can exceed UMTRA Project MCLs when Mancos Shale underlies river alluvium. No terrace or upland background ground water was found at the Shiprock site, but irrigation water west of U.S. Highway 666 is in contact with weathered Mancos Shale that might be expected to leach these constituents from the shale.

4.4.8.1 Geographic Distribution of Late Cretaceous Marine Shales

Dark marine shales were deposited in the Late Cretaceous Western Interior Seaway, a shallow continental sea. The seaway extended from northern Canada south to the Yucatan Peninsula in a broad swath across the central and western United States. The east part of the swath extended from the Dakotas south to New Mexico and the west part extended from Montana south to Nevada.

The sequence of marine shales and associated siltstones and sandstones has several formational names depending on the region. In the north central United States the shales are called the Pierre Shale, in the mid-continental United States they are the Niobrara Chalks, and in the Four-Corners (New Mexico, Colorado, Utah, and Arizona) region they are called the Mancos Shale

(McGookey and others 1972). Northwestern New Mexico contains extensive areas where Mancos Shale crops out; the type locality for the formation is near Mancos, Colorado, about 50 mi north of Shiprock. Recent investigations by Leckie and others (1997) and Lucas and others (1998) define the stratigraphy of the Mancos Shale in the Four-Corners area and show the continuous nature of the strata over this region. The chemistry of these and other Late Cretaceous dark marine shales (e.g., shales in the underlying Dakota Sandstone) have been the subject of numerous reports since the 1940s because of high concentrations of soluble constituents such as selenium, uranium, and sulfate that may affect humans, livestock, and wildlife.

4.4.8.2 Chemistry of Ground Water Contacting Weathered Mancos Shale

Chemical Uranium

The average crustal abundance of uranium is 1.8 mg/kg (Mason and Moore 1982), and the average concentration in shales is 4 mg/kg (Levinson 1980). Organic-rich, dark marine shales are known to carry anomalously high concentrations of a number of cations, including uranium. Levinson (1980) describes black marine shales that have uranium concentrations ranging from 3 to 1,250 mg/kg; for example, the marine Chattanooga Shale in Tennessee has an average uranium concentration of 57 mg/kg over large areas (Mickle and Mathews 1978).

Butler and others (1994) studied the potential effects on fish, other wildlife, and humans from major and trace elements leaching into surface and ground water in the Grand Valley area of western Colorado. Numerous samples of Mancos Shale, ground water, and surface water were collected and analyzed from this area underlain by Late Cretaceous marine shales. In this study, uranium content in six samples of Mancos Shale ranged from 3.7 to 11.2 mg/kg and averaged 6.2 mg/kg. Ground water passing through the Mancos Shale and across the contact between saturated Mancos Shale and surficial alluvial deposits could leach uranium from this shale. Uranium concentrations of 11 ground water samples from the Mancos Shale ranged from 0.004 to 0.450 mg/L and averaged 0.061 mg/L (Butler and others 1994). Concentrations of six ground water samples from the alluvial Cobble Aquifer, which directly overlies the Mancos Shale, ranged from 0.025 to 0.170 mg/L and averaged 0.085 mg/L.

In a study of ground water in the Grand Junction, Colorado, area, DOE analyzed ground water from seven background locations north of the Colorado River where saturated Mancos Shale contacts surficial alluvial deposits (DOE 1999f). Uranium concentrations from two sample rounds ranged from 0.023 to 0.067 mg/L and averaged 0.048 mg/L. This demonstrated that uranium values could be above the UMTRA MCL of 0.044 mg/L from leaching naturally elevated concentrations of uranium in the Mancos Shale.

Uranium Isotopes

The activity ratio (AR, U-234/U-238) for two naturally occurring isotopes of uranium, U-234 and U-238, should ideally be near unity in a closed system that maintains secular equilibrium. However, this ratio varies significantly in naturally occurring systems of ground water indicating that secular equilibrium is often not maintained (Osmond and others 1973). This radioactive disequilibria is commonly observed and has been shown to vary in watersheds as they cross strongly differing rock types (e.g., granites and carbonates) (Riotte and Chabaux 1999). It has also been suggested as a prospecting tool to locate uranium ore deposits in sandstone environments (Cowart and Osmond 1977; Meunier and others 1992).

The origin of U-234/U-238 disequilibria in natural systems is still under debate but investigators have suggested it is a function of alpha-recoil displacement of a U-234 atom (or its daughter products) from the site of its parent U-238 atom (Osmond and Cowart 1992; Sun and Semkow 1998). This displacement can be within the crystal lattice or implantation of the U-234 isotope into a neighboring solid phase (causing damage to the crystal or solid phase), or across mineral-solution boundaries. This results in the U-234 becoming more leachable (compared to U-238) and being transferred into the aqueous phase (Zielinski and others 1997). Optimal environments to support this mechanism are near-surface, unconfined aquifers that contain sparse organic matter and have measurable concentrations of dissolved oxygen and carbonate (Langmuir 1978).

Zielinski and others (1997) used ARs at a former uranium mill-processing site in Cañon City, Colorado, to determine the extent of mill-related uranium contamination in ground water near the site. This application only works if the AR near the millsite is distinctive from the AR found in surrounding ground water. In the Cañon City study, ground water from highly to moderately contaminated samples (greater than 0.100 mg/L uranium) on or directly downgradient of the site had AR values in a narrow range of 1.00 to 1.06. Ground water more distant from the site had AR values consistently greater than 1.3, which probably indicated little or no contamination. It was concluded that the milling processes do not appreciably alter the isotopic ratio of U-234 and U-238 in uranium ores and therefore, the milling-related fluids reflect the ores' isotopic signatures. In another study of a uranium-processing mill at Tuba City, Arizona, isotopic U-234 was found to be concentrated in ground water around the former millsite producing a low AR; lower contents of U-234 further away from the millsite produced higher ARs (Tso 2000).

Analyses for U-234 and U-238 were performed for a limited number of ground water samples collected at the Shiprock site during June 2000 in an effort to determine if it was possible to distinguish between naturally occurring and millsite-related uranium in ground water. As noted above, the Mancos Shale can act as a natural source of uranium and lead to background concentrations of uranium exceeding the UMTRA standard.

For ground water in the vicinity of the Shiprock site, it would be anticipated that water associated with leaching and weathering of Mancos Shale exposed at and near ground surface would be oxidized, saturated to over saturated with respect to calcium carbonate, and enriched in U-234. Leaching of newly mined uranium ore, on the other hand, would produce a leachate with ARs closer to unity. Therefore, ground water contaminated with this leachate would have U-234/U-238 ratios nearer to 1.

Figure 4–61 shows the locations of wells where ground water was sampled for uranium isotopes. Values shown in the figure are the U-234/U-238 ratios or ARs for samples collected from those locations. All of the AR values close to 1 are in locations close to or immediately downgradient of the millsite. Compare low AR values to the uranium concentration maps shown in Figure 4–18 and Figure 4–36. Figure 4–62 shows the relationship between chemical uranium and the ARs for this data set. AR values less than 1.4 are almost always located near the disposal cell or former raffinate ponds even if current uranium concentrations are low suggesting a strong contribution of uranium from ore-related sources. Two of the lowest AR values were from ground water in the most contaminated part of the floodplain (wells 768 and 854). Higher AR values (greater than 1.4) generally correspond to wells located farther from the disposal cell or

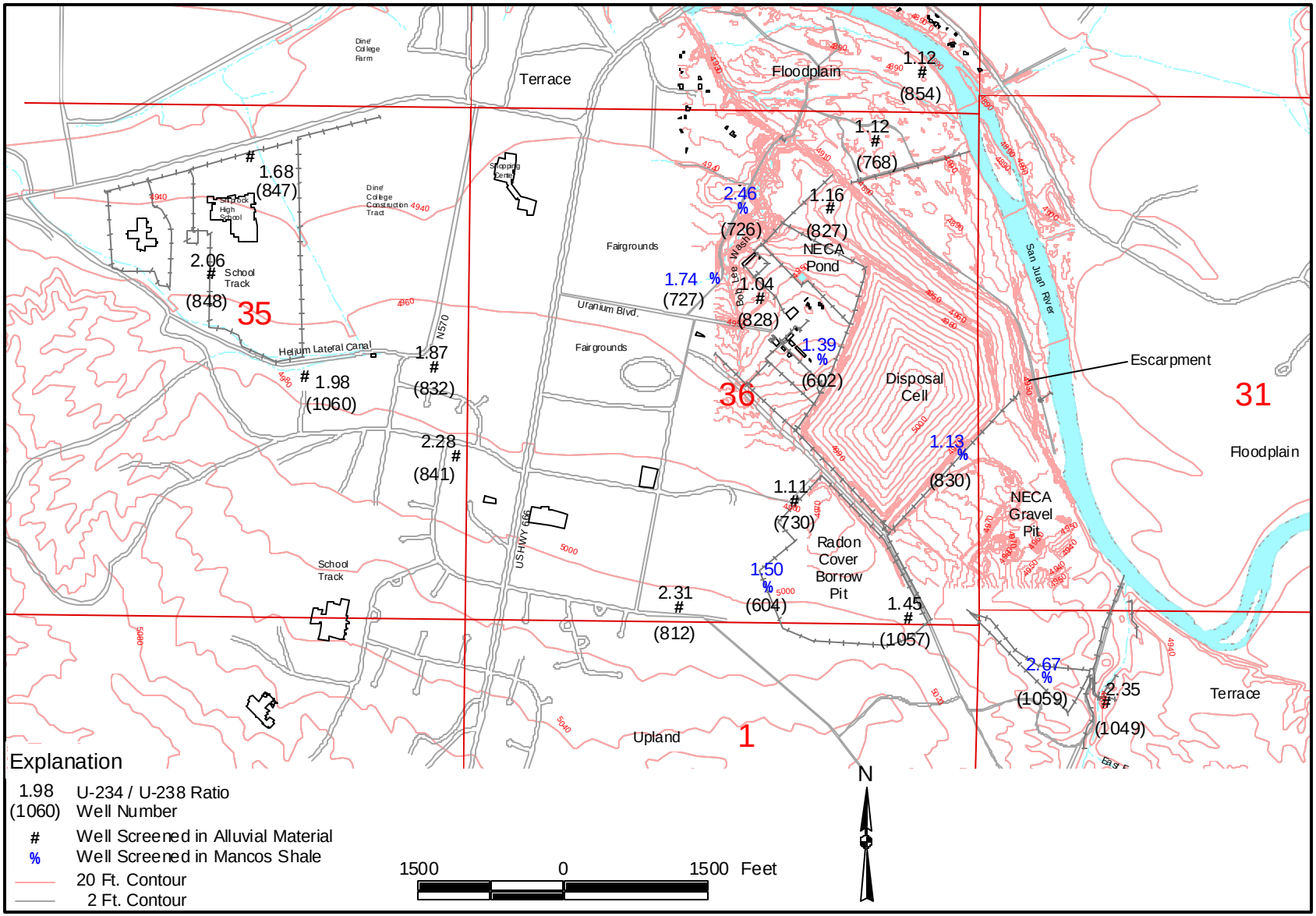


Figure 4–61. U-234/U-238 Ratio in Ground Water in Alluvium and Mancos Shale

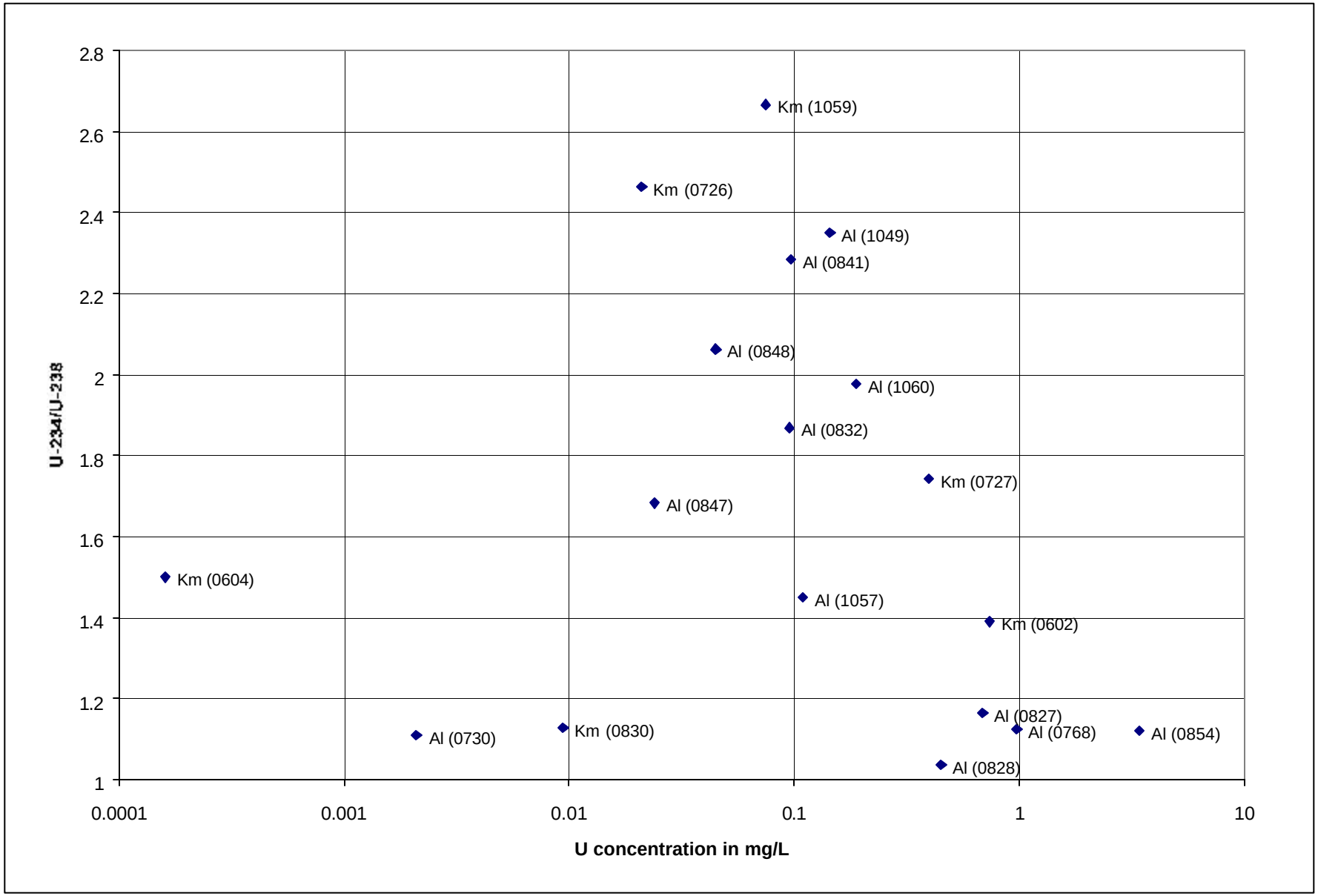


Figure 4–62. U Concentration in mg/L

former raffinate ponds. These values probably represent an isotopic signature of the weathering Mancos Shale. The three Mancos Shale wells to the west of the disposal cell (602, 726, and 727) have AR values that reflect mixing of ore-related fluids with Mancos Shale pore water. The Mancos well (830) on the southeast edge of the cell is screened across the alluvium and the Mancos and probably reflects an ore-related AR value. Wells located west of U.S. Highway 666 and near Many Devils Wash have some of the highest AR values, which probably reflect the isotopic signature of the Mancos Shale. Well 812 is located near the buried scarp of the Mancos Shale and may also reflect the isotopic signature for this bedrock; however, nitrate concentrations in this well indicate a contribution of milling-related fluids.

Isotopic ratios of U-234 to U-238 from the Shiprock site indicate that measurable variations exist between the Mancos Shale and milling-related fluids. Locations where ground water AR values are near 1 are interpreted as being associated with millsite-related contamination and higher values probably reflect the influence of the Mancos Shale pore water. Mixing of these two components certainly occurs. This suggests that most of the uranium concentration in ground water west of U.S. Highway 666 and in the Many Devils Wash area may be naturally occurring. Application of supplemental standards based on widespread ambient contamination may be appropriate for the area in the western portion of the terrace (see Section 7.0 for further discussion).

Selenium

The average abundance of selenium in shales is 0.6 mg/kg according to Turekian and Wedepohl (1961), Wiersma and Lee (1971), and Levinson (1980). Selenium is known to be concentrated in Late Cretaceous marine shales of the western United States and was studied in the 1940s to determine its deleterious health effects on cattle in the northern mid-continent states (Larkin and Byers 1941). These shales are part of the sequence deposited by the Late Cretaceous Western Interior Seaway.

Butler and others (1994) also analyzed for selenium in their study of the Grand Valley. Whole rock analyses of six Mancos Shale samples resulted in an average concentration of 1.1 mg/kg selenium, or about twice the average value for shales. Selenium values in ground water derived from Mancos Shale varied from below detection (0.002 mg/L) to 0.13 mg/L, and analyses of 13 samples averaged 0.037 mg/L. Samples of overlying alluvial ground water contained higher concentrations of selenium up to 1.3 mg/L that may have been caused by evaporative concentration from arid conditions in the Grand Valley.

In the Grand Junction study (DOE 1999f), the mean selenium content from seven background wells was 0.033 mg/L, which is above the UMTRA MCL of 0.01 mg/L but below the EPA Clean Water Act standard of 0.05 mg/L. This was cited as evidence of widespread ambient ground water contamination from natural causes, that is, the Mancos Shale.

The USGS studied the origin of selenium contamination related to the high selenium concentrations in the Imperial Valley of California and other locations downstream in the Colorado River drainage (Seiler and others 1999). High selenium values are not indigenous to those locations, and the problem is thought to result from upstream irrigation of selenium-rich bedrock. Selenium and other contaminants introduced by irrigation were studied at 26 locations in the western United States. Selenium leached from soil by irrigation water is eventually carried into the Colorado River. Therefore, the Colorado River is receiving selenium from various rock

formations in its drainage system. The report concludes that concentrations of selenium are usually greatest (1) where underlying Late Cretaceous marine sediments occur that contain high concentrations of leachable selenium and (2) where evaporation rates are highest. Both criteria are met in the Shiprock area, where maximum evaporation occurs in an area of exposed Late Cretaceous marine shale bedrock. The San Juan River in the Shiprock area was classified as seleniferous because more than 25 percent of surface samples exceeded a selenium content of 0.003 mg/L. The Grand Valley of western Colorado was classified as having irrigation-induced selenium contamination because more than 25 percent of the surface water samples have selenium contents equal to or greater than 0.005 mg/L.

Progress Report No. 19, a part of the Colorado Basin Salinity Control Program sponsored by the Department of Interior (DOI 1999) discusses the dissolved solids entering the Colorado River. Selenium is a minor constituent of this greater problem, but has impacts on certain birds and fish. The high selenium values entering the Colorado River system are ascribed to leaching of Late Cretaceous marine shales. Selenium data for the report were derived from EPA's water quality database (STORET) during the period from 1978 to 1995. Flow-weighted averages for river water from this database show that selenium concentration in the San Juan River is 0.002 mg/L. This concentration is not as high as some other tributaries of the Colorado River, such as the Gunnison River that also flows over a large area of Mancos Shale and has an average selenium concentration of 0.007 mg/L.

Sulfate

The studies cited above that evaluate contributions of total dissolved solids to the Colorado River basin also include analysis of sulfate. Again, the main contributors of sulfate are Late Cretaceous marine shales, especially the Mancos Shale in the Four-Corners area.

Evangelou and others (1984) concluded that the sulfate ion had been an early sink for precipitation of sodium, calcium, and magnesium, during and soon after deposition of the Mancos Shale, but these constituents are presently being released during erosion of the Mancos. This accounts for the large contribution of these ions to the Colorado River drainage area and the widespread formation of efflorescence where Mancos Shale is being weathered. Butler and others (1994) determined that 13 sulfate analyses of ground water from four Mancos wells in the Grand Valley ranged from 1,800 to 7,300 mg/L and averaged 3,515 mg/L. The report also showed that analyses of sulfate from six ground water samples from two wells screened in alluvial material overlying the Mancos Shale ranged from 2,500 mg/L to 3,800 mg/L and averaged 3,267 mg/L. Analyses of sulfate in ground water from all wells averaged 3,437 mg/L.

DOE (1999f) evaluated ground water sulfate concentrations from seven alluvial background wells in the Grand Valley of western Colorado that are in hydrologic contact with underlying Mancos Shale. Results from two sampling rounds showed that sulfate concentration in ground water ranged from 416 mg/L to 3,720 mg/L and averaged 2,566 mg/L. These results are comparable to the study by Butler and others (1994). Water in contact with Mancos Shale contains relatively high concentrations of sulfate.

4.4.8.3 Shiprock Wells West of U.S. Highway 666

Table 4–27 shows the results of analyses of ground water sampled since 1998 from six wells in the Shiprock site west of U.S. Highway 666 in the area affected by irrigation. Table 4–28 lists the results of analyses of ground water sampled from five wells west of U.S. Highway 666, but just outside the area affected by irrigation. The values of uranium, selenium, and sulfate in the irrigated area ground water are generally less than those for the nonirrigated area. The higher values from ground water in wells shown in Table 4–28 are considered to be the result of westward migration of millsite contamination. Ground water in the irrigated area has been flushed, and contaminant concentrations have probably decreased over time; however, wells were installed in this area only in late 1998, and ground water has not been sampled and analyzed for a sufficient time to confirm this. Contaminant concentrations in ground water in the irrigated area are mostly in the range found in Mancos Shale terrain and will probably not decrease below concentrations discussed in Section 4.4.8.2 because they are leaching from the underlying Mancos Shale.

Table 4–27. Analyses (mg/L) of Ground Water from Terrace Wells in Irrigated Area

Well	Se (min)	Se (max)	Se (avg)	SO_4 (min)	SO_4 (max)	SO_4 (avg)	U (min)	U (max)	U (avg)
836	0.0976	0.142	0.11512	2780	3200	2928	0.0525	0.0563	0.05496
837	0.0099	0.0187	0.01448	1660	1860	1754	0.0353	0.043	0.03858
838	0.0272	0.0538	0.0395	1180	1770	1497.5	0.0235	0.0335	0.0284
843	0.0006	0.0014	0.001	1720	2240	1962	0.0274	0.0314	0.02932
846	0.668	0.931	0.84475	2240	4550	3062.5	0.0405	0.0458	0.042975
847	0.0295	0.0409	0.03565	1370	1650	1517.5	0.0197	0.0275	0.025325

Table 4–28. Analyses (mg/L) from Terrace Wells in a Nonirrigated Area

Well	Se (min)	Se (max)	Se (avg)	SO_4 (min)	SO_4 (max)	SO_4 (avg)	U (min)	U (max)	U (avg)
832	0.444	2.61	1.736	3030	6980	5037.5	0.0269	0.075	0.05335
833	0.18	0.592	0.4095	3100	5950	4895	0.0743	0.128	0.107825
841	2.71	3.42	3.0225	13300	14800	14000	0.09	0.114	0.10025
844	0.155	0.208	0.17325	2670	3390	3032.5	0.0404	0.0495	0.045225
1060	0.015	0.015	0.015	1140	1140	1140	0.3	0.3	0.3

4.4.8.4 Summary

Mancos Shale or its formational equivalents were deposited in the Late Cretaceous Western Interior Seaway that extended across many of the western United States including the Four-Corners region. In general, dark marine shales such as the Mancos have elevated concentrations of uranium compared to other shales. In particular, Late Cretaceous marine shales in the United States have been studied since the 1940s to evaluate their role in causing selenium toxicity in livestock. Since about 1990, the Department of Interior has been studying Late Cretaceous

marine shales, especially the Mancos Shale, to evaluate their contribution to total dissolved salt load in the Colorado River basin. Selenium concentrations in the Colorado River where it drains across the Mancos Shale can be elevated enough to cause health problems for wildlife. Sulfate is being mobilized from Mancos Shale by weathering of gypsum, which is a major constituent of the shale. This sulfate is a major contributor to the total dissolved salt load entering the Colorado River basin. Ground water in contact with the Mancos Shale often has concentrations of selenium and uranium above UMTRA MCLs and sulfate concentration greater than 2,000 mg/L. It is likely that ground water from the Shiprock site will also have elevated concentrations of uranium, selenium, and sulfate where it has contacted Mancos Shale.

4.5 Numerical Ground Water Modeling

DOE (1999g) presented the results of a calibrated two dimensional flow-and-transport model of the floodplain aquifer that was developed to evaluate preliminary compliance strategies. Simulations were developed that assumed the source of contamination could be cut off from the floodplain aquifer; however, the interactions between the terrace and floodplain were poorly understood and provided no specifics on how the source could be cut off from the floodplain. The present modeling evaluates the interactions between the terrace and the floodplain. It simulates the resistant siltstone bed within the Mancos Shale, the drainage of residual moisture and source contamination from the disposal cell, the transport of RRM-laden water from the disposal cell, and the effects that ground water pumping could have on controlling plume migration.

The MODFLOW code (McDonald and Harbaugh 1988) was used for the flow modeling. Output from the model was used in particle tracking simulations and transport simulations. Particle tracking was accomplished using the code MODPATH (Pollock 1989), and the MT3D code (Zheng 1990) was used in the transport simulations. The pre- and post-processing software used to generate the data sets and to run the models is GWVistas Version 2.50 (ESI 1998).

4.5.1 Flow Model Construction

The Model Grid

A three-dimensional flow model was developed to simulate the interaction between the terrace flow system and the floodplain alluvial aquifer. Figure 4–63 illustrates the geographic extent of the numerical model. The horizontal dimension of the model is discretized into cell sizes measuring 100 ft by 100 ft. The cells shrink to a minimum dimension of 6 ft by 6 ft to allow the finite difference solution to converge and to provide accurate head solutions near the edge of the escarpment. This is required along the escarpment near well 600.

The model is composed of four layers of varying thickness. Figure 4–64 presents a series of schematic cross sections that illustrate the vertical discretization of the four-layer model. The lines of the schematic cross sections are shown on Figure 4–63. In the floodplain, all four layers are about 2 ft thick and represent the floodplain alluvium. On the terrace, the uppermost layer represents the terrace gravel unit, the 2nd layer represents the weathered Mancos Shale, the 3rd and 4th layers represent the unweathered Mancos Shale. In the eastern part of the terrace, the top of the third layer corresponds to the top of the resistant siltstone bed in the Mancos Shale,

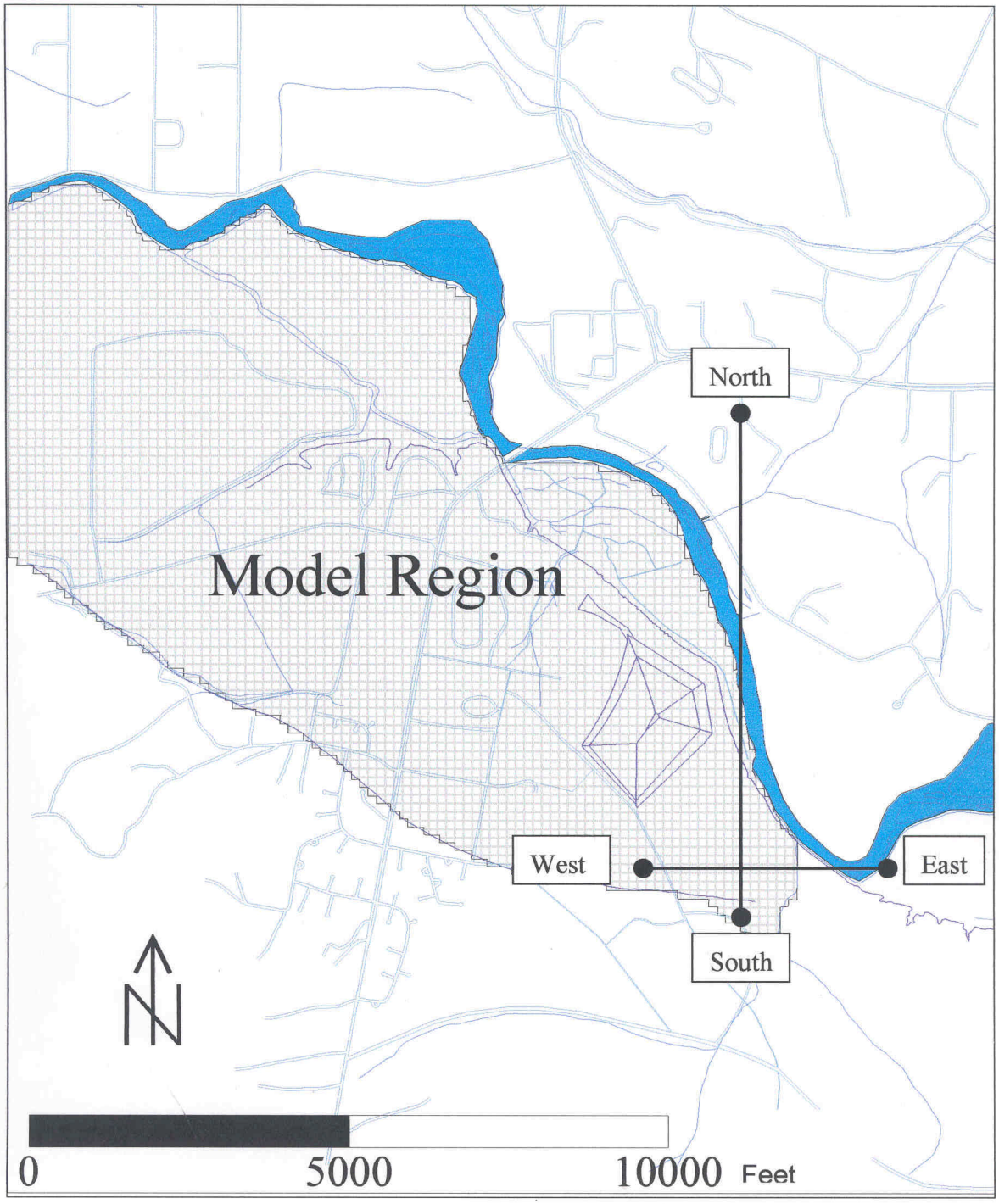
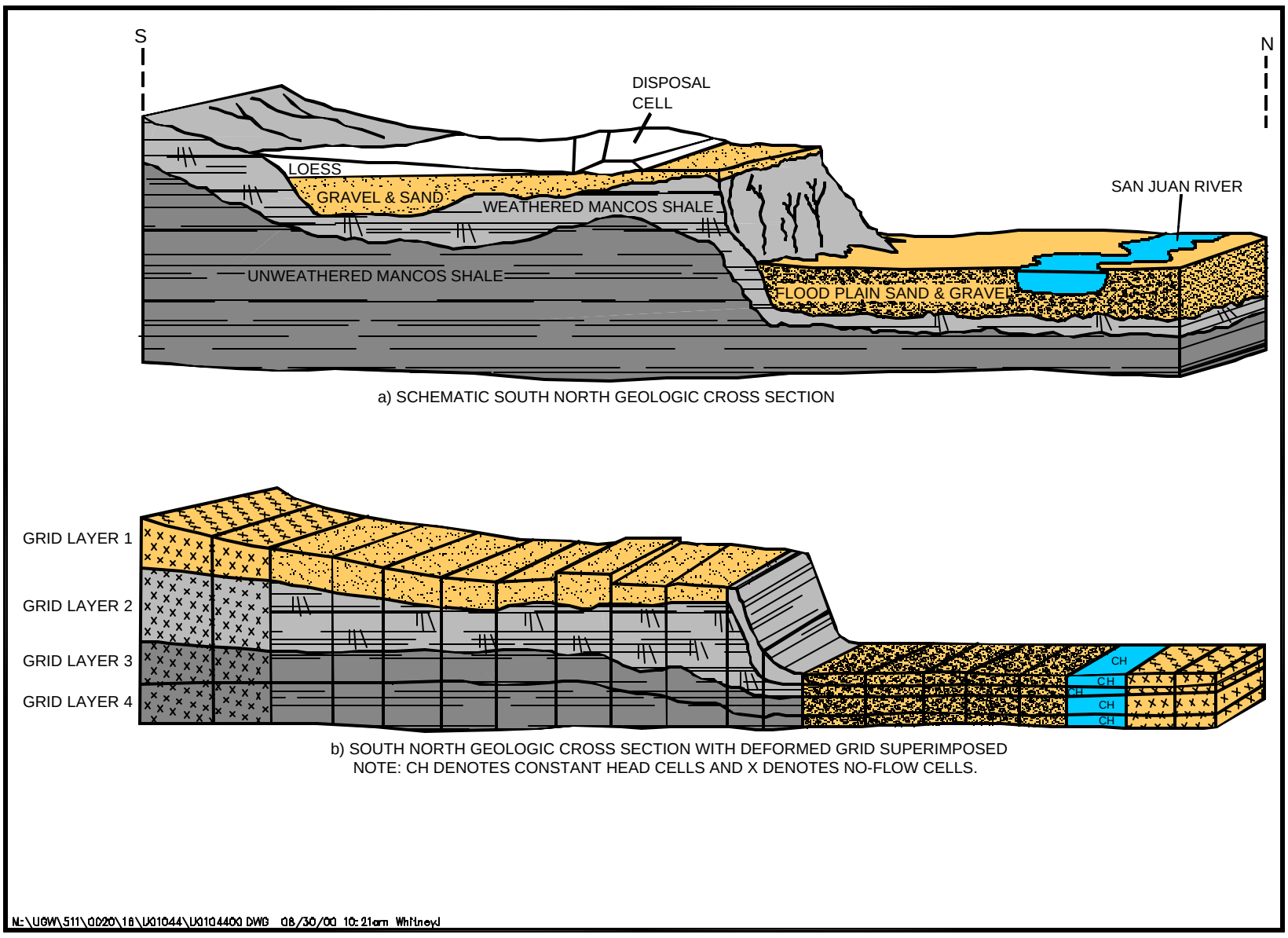


Figure 4–63. Geographic Extent of Numerical Model and Lines of Schematic Cross Sections for the Shiprock Site
(note: the grid pattern does not correspond to finite-difference cells)



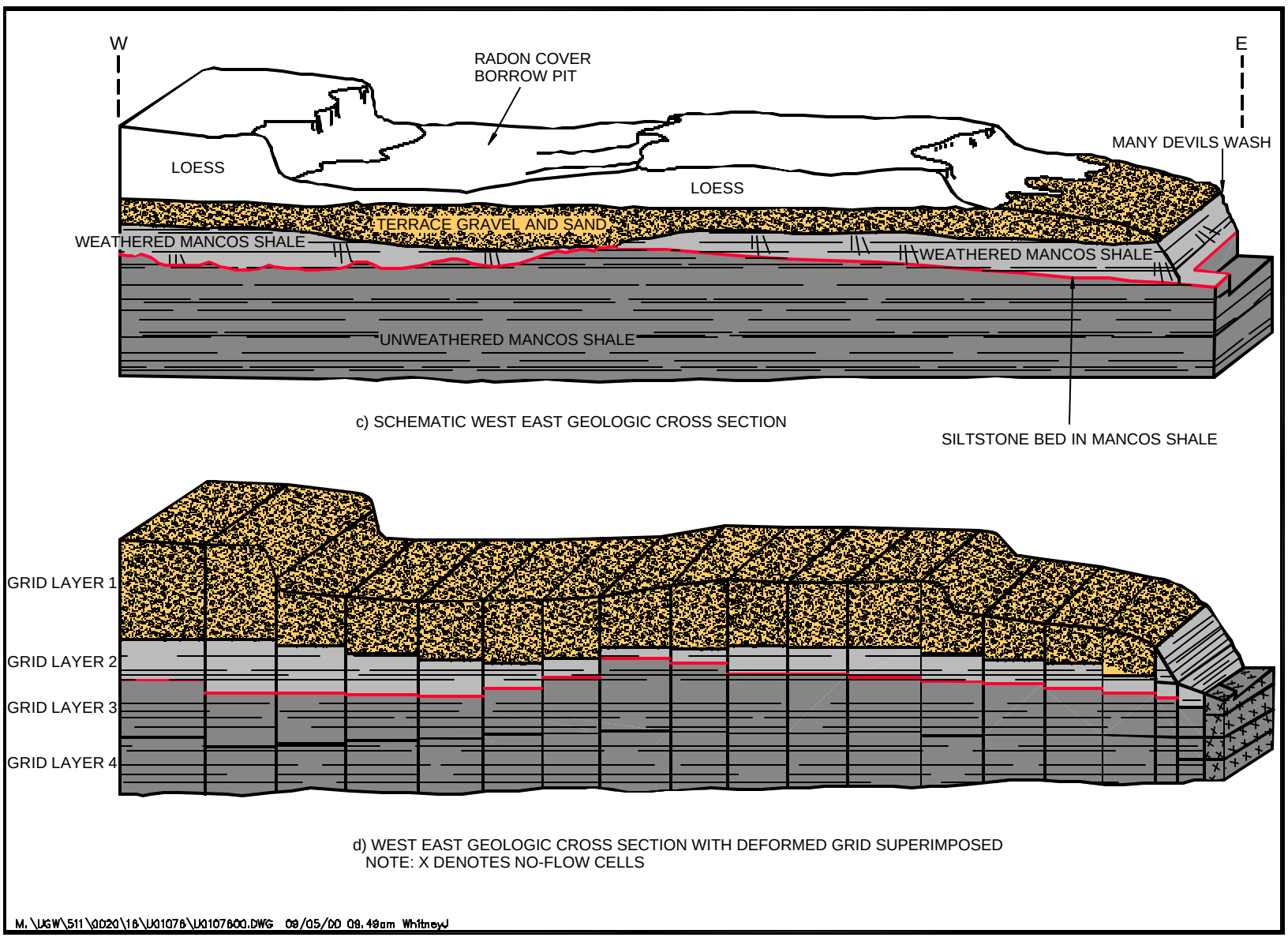


Figure 4–64. Schemes of Vertical Discretization for the Shiprock Flow and Transport Model (a) schematic geologic cross section from south to north and crossing the disposal cell, (b) south to north geologic cross section with deformed grid superimposed, (c) schematic geologic cross section from west to east , (d) west to east geologic cross section with deformed grid superimposed.
Note: the symbols x and ch denote no-flow cells and constant head cells, respectively

thereby permitting the 2nd layer, the weathered Mancos Shale, to serve as a pathway for the RRM-contaminated ground water flowing east from the terrace gravel system. The rapid change in surface elevation along the escarpment is simulated by changing the thickness of each cell. This approach, referred to as creating a deformed grid (McDonald and Harbaugh 1988), permits each layer of the model to conform to a specific portion of the hydrogeologic sequence.

Boundaries

The southern edge of the flow domain is bounded by Mancos Shale outcrops. Boreholes 806, 808, 809, 810, and 811 were drilled south of the buried terrace gravel flow system to a depth of 100 ft; no ground water was found in any of the boreholes. On the basis of this finding, the southern boundary of the flow system is treated as a no-flow boundary.

The northern edge of the model contacts the San Juan River. The water surface elevations of the San Juan River are treated as a specified head boundary by scaling the elevations off a map and assigning them to the model at the appropriate locations.

The eastern boundary of the model corresponds to the axis of Many Devils Wash. Because Many Devils Wash separates non-water-bearing units to the east from water-bearing units to the west, the eastern boundary of the flow system is treated as a no-flow boundary.

The western boundary of the model is an approximate boundary constructed just west of the road leading north to the sewage treatment plant. This boundary is treated as "no flow" because canal recharge from the south feeds the largely alluvial flow system, and the ground water flow direction is predominantly northward toward the San Juan River. Because the aquifer equipotentials are oriented perpendicular to the western model boundary (Figure 4–9), there is little flow exiting the system along it; consequently, the boundary is treated as no-flow.

Hydraulic Parameters

Table 4–29 presents a summary of the hydraulic parameters used in the numerical model. The model parameters were assigned on the basis of the aquifer testing performed during this investigation and described in Section 4.3. Hydraulic conductivity is a widely varying parameter, but in this model each hydrostratigraphic unit is assigned an average hydraulic conductivity.

Table 4–29. Summary of Hydrologic Parameters used in the Flow Model

Parameter	Floodplain Alluvium	Terrace Alluvium	Weathered Mancos Shale	Unweathered Mancos Shale	Disposal Cell
Hydraulic Conductivity (ft/day)	100	10	0.2	0.1	
Recharge (ft/day)	1.44×10^{-4}	1.44×10^{-4}			1.34×10^{-3}
Porosity *	0.30	0.30	0.06	0.01	
Bulk density (lb/ft^3)	120	120	140	150	
Uranium K_d (ft^3/lb_m)	8.6×10^{-4}	8.6×10^{-4}	2.6×10^{-3}	3.4×10^{-3}	

* Dimensionless

Recharge is a term describing the flux of water that crosses the water table and becomes part of the ground water flow system. For lack of a way to measure recharge directly, modelers have traditionally assumed a spatially uniform recharge rate across the water table equal to some

percentage of average annual precipitation and then adjusted the recharge rate during model calibration (Anderson and Woessner 1992). The approach taken for this model differed slightly because, according to the conceptual model the terrace alluvial system is hypothesized to have been dry before human activities at the site. Trial and error revealed that the terrace flow system becomes saturated south of the disposal cell when areal recharge exceeds 7 percent of average annual precipitation. Therefore, the maximum areal recharge that satisfies the initially-dry-terrace hypothesis is 7 percent of average annual precipitation.

A higher recharge rate is required for the disposal cell to account for the drainage of residual moisture through the cell. The addition of recharge through the cell is necessary to create an active flow system in the terrace alluvial gravel, because the areal recharge rate by itself is insufficient to saturate the terrace flow system. The magnitude of cell recharge has changed over the years. During milling, when the raffinate lagoons were active and the recharge flux was higher, the recharge was distributed over a larger area and included the former raffinate ponds. Figure 4–65 summarizes the hydrologic history of the site and how the recharge flux might have changed over the years.

Porosity and bulk density were selected from typical values presented in Morris and Johnson (1967). The uranium K_d was selected from the laboratory values presented in the ESL report (DOE 1999d).

Boundary Conditions

Internal boundaries in the model consist of wells and drains. The discharge of well 648 onto the floodplain is simulated with an injection well placed at the mouth of Bob Lee Wash. The pumping rate for this injection well is 10,275 ft^3/day. Evaluations of natural flushing through the aquifer would need to consider the possibility of this well being shut in and no longer discharging onto the floodplain.

Drains are another type of internal boundary condition used in the model. The drains are located along Many Devils Wash and along the escarpment to account for spring flow. The elevation of the drains is set equal to the surveyed spring elevations.

Calibration Targets

A total of 63 head-calibration and concentration-calibration targets were established by obtaining a database listing of all monitoring wells at the project area. Some of the wells are no longer active so they were excluded from the list. Other wells are located in clusters and therefore eliminated from the list in order to avoid having multiple targets in any given model cell. The 63 head targets are average heads; the concentration targets represent the latest conditions observed during routine monitoring. The target values were formatted and electronically imported into the input files.

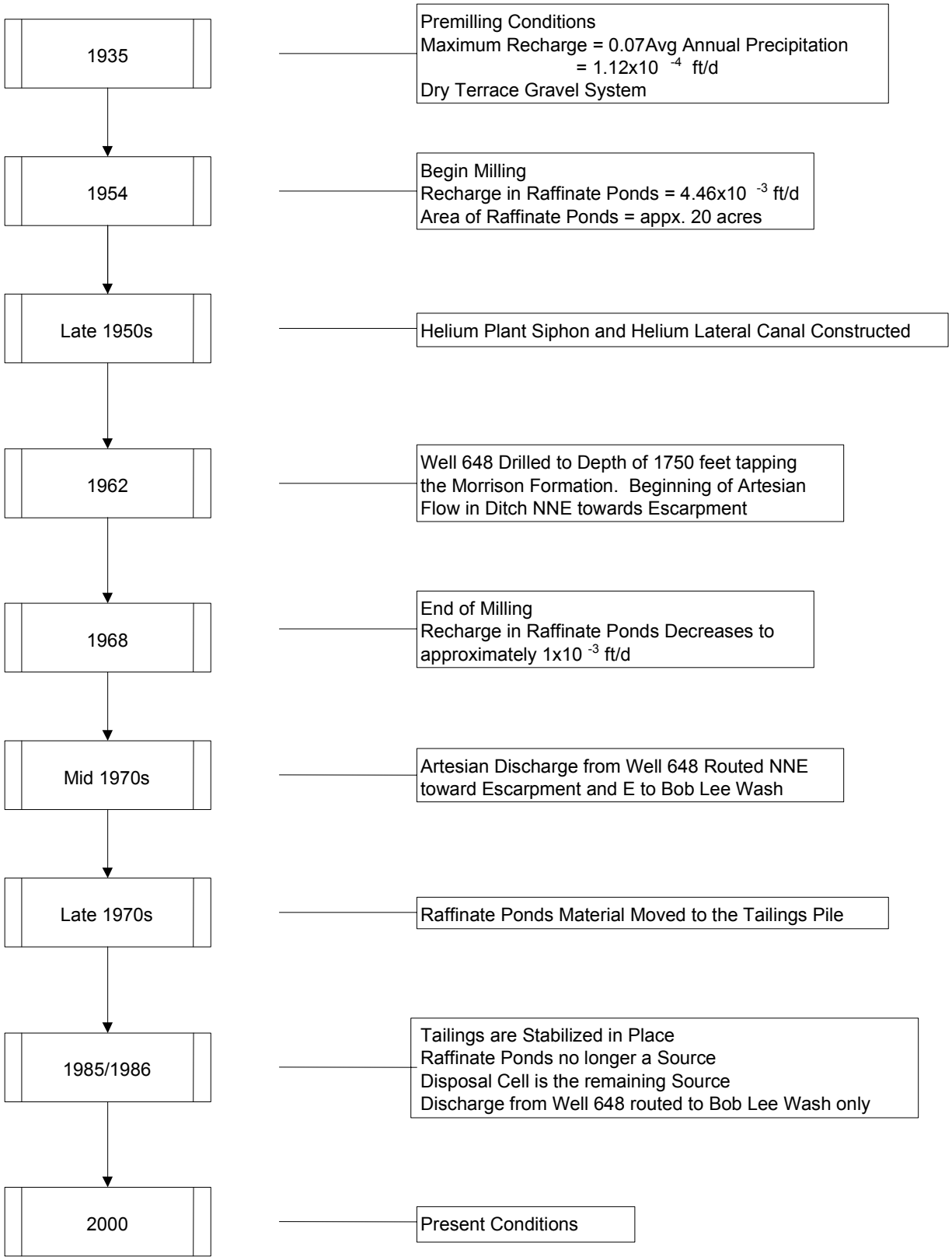


Figure 4–65. Summary of Hydrologic History of the Shiprock Site

4.5.2 Model Calibration

Calibration of the ground water flow model is the process of adjusting hydraulic parameters, boundary conditions, and initial conditions within reasonable ranges to obtain an acceptable match between observed and simulated potentials, flow rates, and concentrations. The range over which model parameters and boundary conditions may be varied is determined by data presented in the conceptual model.

Residual Analysis

The calibration of a model is evaluated through analysis of residuals. A residual is the difference between the observed measurement and the calculated measurement. Calibration may be viewed as a regression analysis designed to bring the mean of the residuals close to zero and to minimize the standard deviation of the residuals (ASTM D-5447). Model calibration objectives were established prior to beginning the modeling effort. The acceptance criteria for model calibration were that the standard deviation of the residual errors divided by the total range in head should be 10 percent or less and calibration residuals should be distributed normally about the mean in order to eliminate spatial bias from the model.

Figure 4–66 presents a map of the output from the flow model, consisting of the simulated water table and a posting of the residuals. Equipotentials on this map are for Model Layer 1, which includes the terrace gravel. Residuals for all four layers are projected onto this surface. The minimum and maximum residuals of –14 ft and 8.3 ft, respectively, occur at wells DM7 and 826. Each well is completed in Mancos Shale. Because of the low hydraulic conductivity of the Mancos Shale, the calculated water levels values for the Mancos Shale are extremely sensitive to recharge. The mean residual for the model as a whole is 0.31 ft. The mean residual is more representative of the residuals calculated for the alluvial gravel deposits on both the terrace and the floodplain.

Table 4–30 presents a summary of the residuals obtained with this input data set. The summary statistics indicate that the model is calibrated to the residual standard deviation divided by the head range of 4.1 percent. This value falls well within the calibration objective of 10 percent. A plot of the flow-model calibration data is also illustrated in Figure 4–67.

Sensitivity Analysis

Sensitivity analysis is a quantitative method of determining the effect of parameter variation on model results. Sensitivity analyses are performed during model calibration and during predictive analyses to provide data users an understanding of the level of confidence in model results. Hydraulic conductivity and recharge were selected as important parameters for sensitivity analysis because of their potential to affect the flow field near the Shiprock site. The recharge term for the disposal cell (Zone 3) represents drainage of residual moisture from the cell. The areal (Zone 1) recharge is another important parameter, because it is insufficient to saturate the terrace ground water system. Figure 4–68 summarizes the model sensitivity to Zone 1 and Zone 3 recharge.

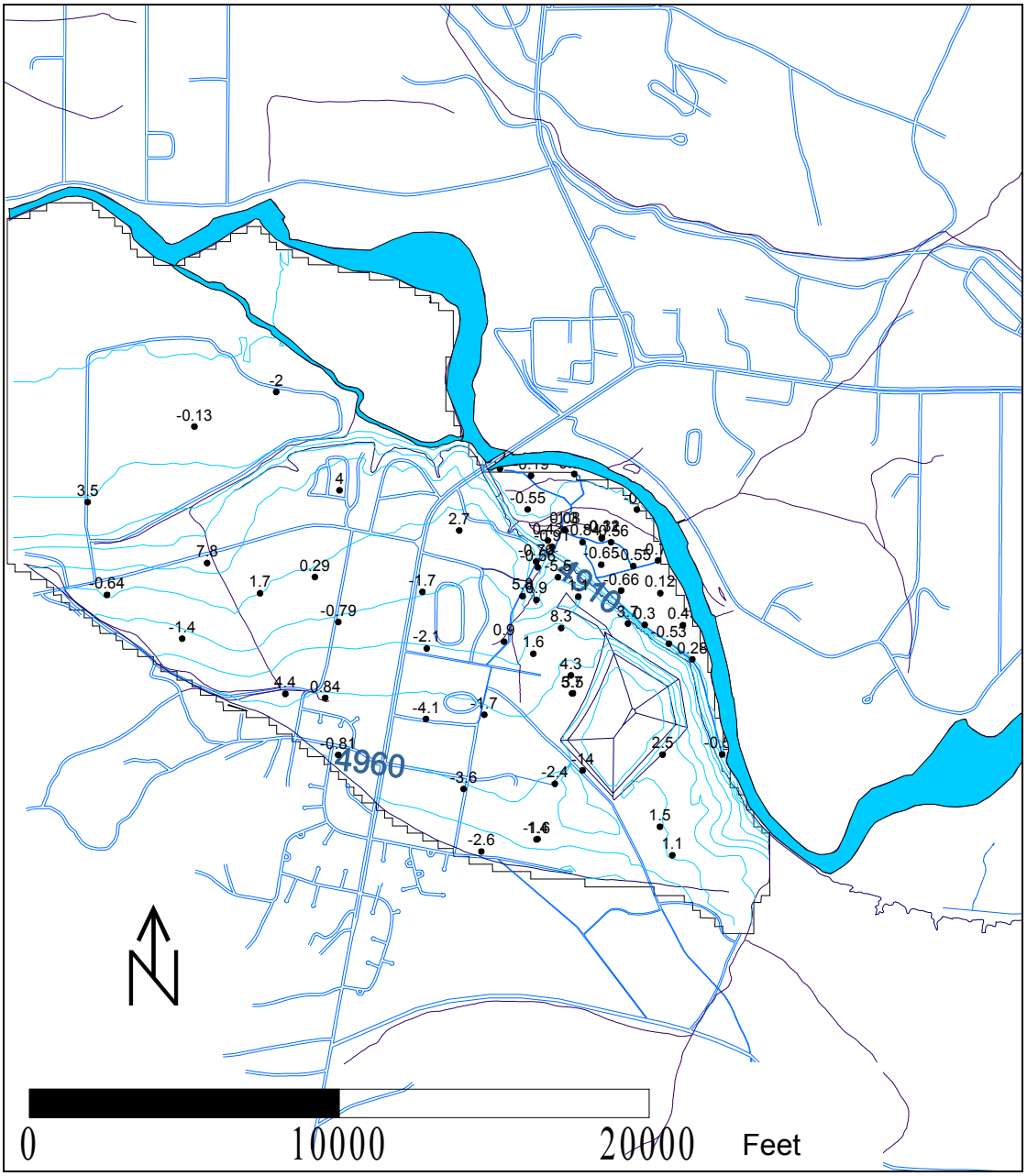


Figure 4–66. Hydraulic Head and Residuals from Calibrated Four-Layer Model of Terrace Alluvial System and Floodplain Alluvial Aquifer

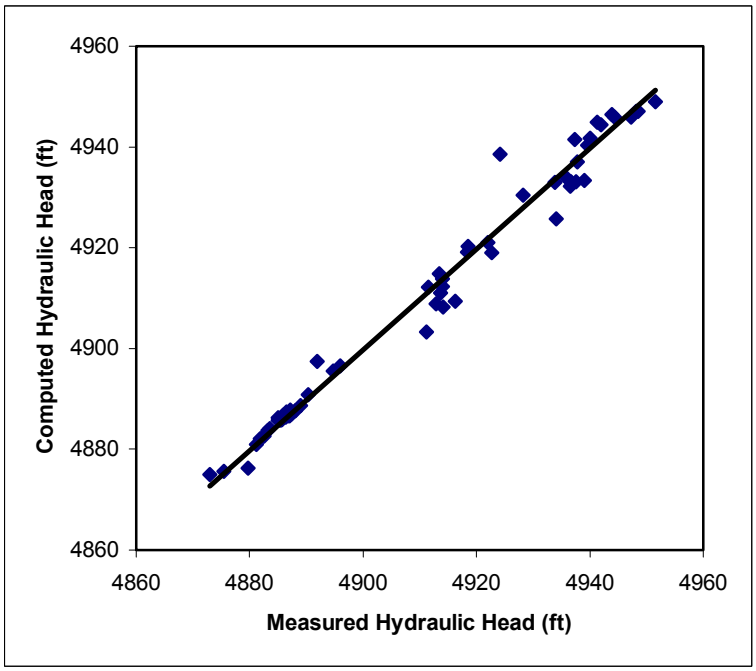


Figure 4–67. Plot of Flow-Model Calibration Data, Shiprock Site

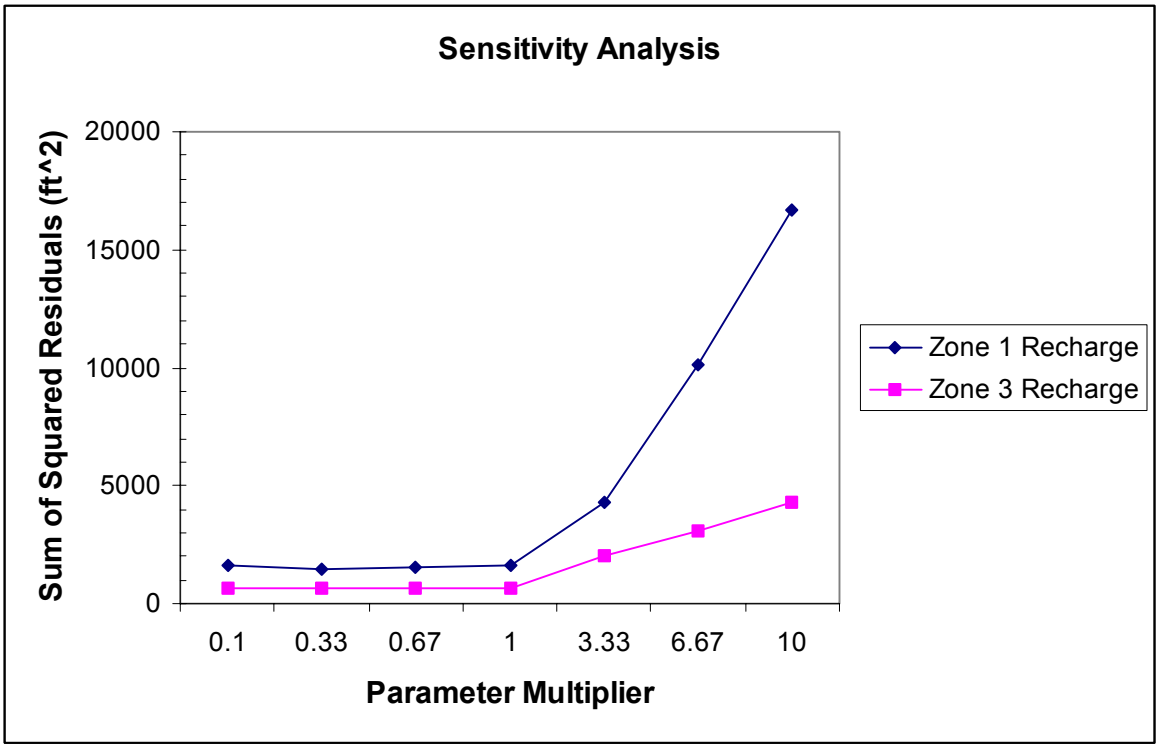


Figure 4–68. Summary of Sensitivity Analysis for Recharge Zones 1 and 3, Shiprock Site

The recharge that results from drainage of residual moisture from the disposal cell has not been estimated explicitly for the disposal cell; therefore, any estimate of this parameter is subject to a high degree of uncertainty. The RAP (DOE 1985) indicates that the maximum infiltration through the cover is 0.04 in/yr, or approximately 0.6 percent of average annual precipitation. Calibration of the flow model reveals that if the maximum infiltration rate presented in the RAP is combined with the maximum areal recharge of 7 percent of average annual precipitation, there would be little, if any, water in the terrace alluvium south of the disposal cell. The best model calibration is achieved with disposal-cell recharge set to 6.5×10^{-4} ft/day, approximately 40 percent of average annual precipitation, and an areal recharge value set equal to 9 percent of average annual precipitation. The benefit of using the parameters from the best calibration as an initial condition for transient modeling is that the heads around the cell are as close as possible to the observed conditions, and the ground water flow directions are similar to those observed in the field. However, an areal recharge value of 9 percent of average annual precipitation is only capable of partly saturating the terrace alluvium in the buried channel south of the disposal cell, and this condition is contrary to the site conceptual model.

4.5.3 Particle Tracking

Particle tracking is a form of transport modeling that uses the velocity field generated by the flow model to illustrate the direction that fluid elements and dissolved constituents will migrate. The predicted ground water velocity is dependent upon the porosity. The porosities used in this model are presented in Table 4–29.

Figure 4–69 summarizes the particle tracking simulation for the calibrated steady-state flow model. The particle tracking simulation shows that ground water originating from south of the disposal cell could migrate along the buried alluvial channel that comprises the terrace ground water system and spread northwest. It also illustrates that advection through the Mancos Shale around the disposal cell is a potential mechanism for transporting contamination to the floodplain alluvial aquifer. Another observation drawn from the particle tracking is that terrace ground water is likely to flow from the disposal cell toward Bob Lee Wash via the NECA yard and from the buried channel to Many Devils Wash and along the top of the east-dipping siltstone bed within the Mancos Shale. These flow patterns help explain the relatively high concentrations of nitrate and sulfate in Many Devils Wash and the buried channel, and also explain the presence of uranium and the relative absence of nitrate along Bob Lee Wash. Conclusions drawn from the particle tracking simulations, however, should be treated with caution because the flow directions are highly dependent upon the rate of drainage of residual moisture from the disposal cell.

4.5.4 Transport Modeling

Transport simulations for the floodplain alluvial aquifer were discussed in the SOWP, Revision 1 (DOE 1999g). Those simulations evaluated the ability of the floodplain aquifer to flush if the source could be contained. The present simulations account for the interactions between the terrace and floodplain and treat the disposal cell as continuing to drain residual moisture. The source flux is transported to the aquifer as a result of recharge through the disposal cell and is treated as a constant-flux source.

Table 4–30. Summary of Residuals and Calibration Statistics for the Calibrated Ground Water Flow Model, Shiprock Site

Well ID	Model Layer	Average Measured Head (ft)	Computed Head (ft)	Residual (ft)	Well ID	Model Layer	Average Measured Head (ft)	Computed Head (ft)	Residual (ft)
604	4	4,885.05	4,886.32	-1.27	728	1	4,940.04	4,941.77	-1.73
606	4	4,886.47	4,887.38	-0.91	730	2	4,941.93	4,944.34	-2.41
608	4	4,888.96	4,888.68	0.28	731	1	4,948.14	4,947.00	1.14
610	4	4,887.20	4,887.73	-0.53	812	2	4,943.89	4,946.48	-2.59
612	4	4,888.11	4,887.69	0.43	813	1	4,941.32	4,944.92	-3.60
613	4	4,887.39	4,887.09	0.30	814	2	4,937.31	4,941.45	-4.13
615	4	4,885.99	4,886.65	-0.66	815	1	4,928.25	4,930.38	-2.13
617	4	4,885.55	4,886.10	-0.55	816	1	4,913.67	4,911.00	2.67
619	4	4,885.18	4,885.74	-0.56	817	2	4,938.99	4,933.30	5.69
620	4	4,885.90	4,886.55	-0.65	818	1	4,944.29	4,945.88	-1.60
624	4	4,885.48	4,885.80	-0.31	819	2	4,936.56	4,932.22	4.34
625	4	4,885.72	4,885.83	-0.12	826	2	4,934.03	4,925.75	8.28
626	4	4,885.88	4,886.72	-0.84	827	1	4,922.06	4,921.00	1.06
628	4	4,886.42	4,886.34	0.08	828	2	4,935.69	4,934.05	1.64
630	4	4,887.03	4,886.60	0.43	829	2	4,891.94	4,897.41	-5.47
734	4	4,881.18	4,880.94	0.24	830	1	4,951.52	4,949.00	2.52
735	4	4,890.27	4,890.78	-0.51	832	1	4,937.84	4,937.00	0.84
736	4	4,882.60	4,882.56	0.04	833	1	4,914.05	4,913.76	0.29
853	4	4,886.78	4,886.66	0.12	835	1	4,912.85	4,908.81	4.03
854	4	4,883.60	4,884.10	-0.50	836	1	4,879.72	4,876.20	3.51
855	4	4,883.19	4,883.73	-0.55	837	1	4,875.42	4,875.55	-0.13
856	4	4,881.85	4,882.04	-0.19	838	1	4,913.96	4,912.27	1.70
857	4	4,885.02	4,885.77	-0.75	839	1	4,918.50	4,920.22	-1.72
600	1	4,922.66	4,919.00	3.66	841	1	4,939.54	4,940.35	-0.81
602	2	4,936.63	4,933.10	3.53	843	1	4,872.97	4,874.93	-1.96
603	1	4,948.52	4,947.00	1.52	844	1	4,918.37	4,919.16	-0.79
604	2	4,947.27	4,945.87	1.40	845	1	4,937.58	4,933.16	4.43
605	1	4,895.96	4,896.52	-0.56	846	1	4,911.56	4,912.20	-0.64
633	1	4,916.24	4,909.38	6.86	847	1	4,911.15	4,903.30	7.85
725	1	4,894.72	4,895.48	-0.76	848	1	4,913.46	4,914.85	-1.39
726	2	4,914.08	4,908.24	5.83	DM7	4	4,924.12	4,938.61	-14.49
727	1	4,933.90	4,933.00	0.90					

Residual Mean	0.31
Res. Std. Dev.	3.26
Sum of Squares	674.82
Abs. Res. Mean	2.09
Min. Residual	-14.49
Max. Residual	8.28
Head Range	78.55
Std/Head Range	0.04 (4.1 %)

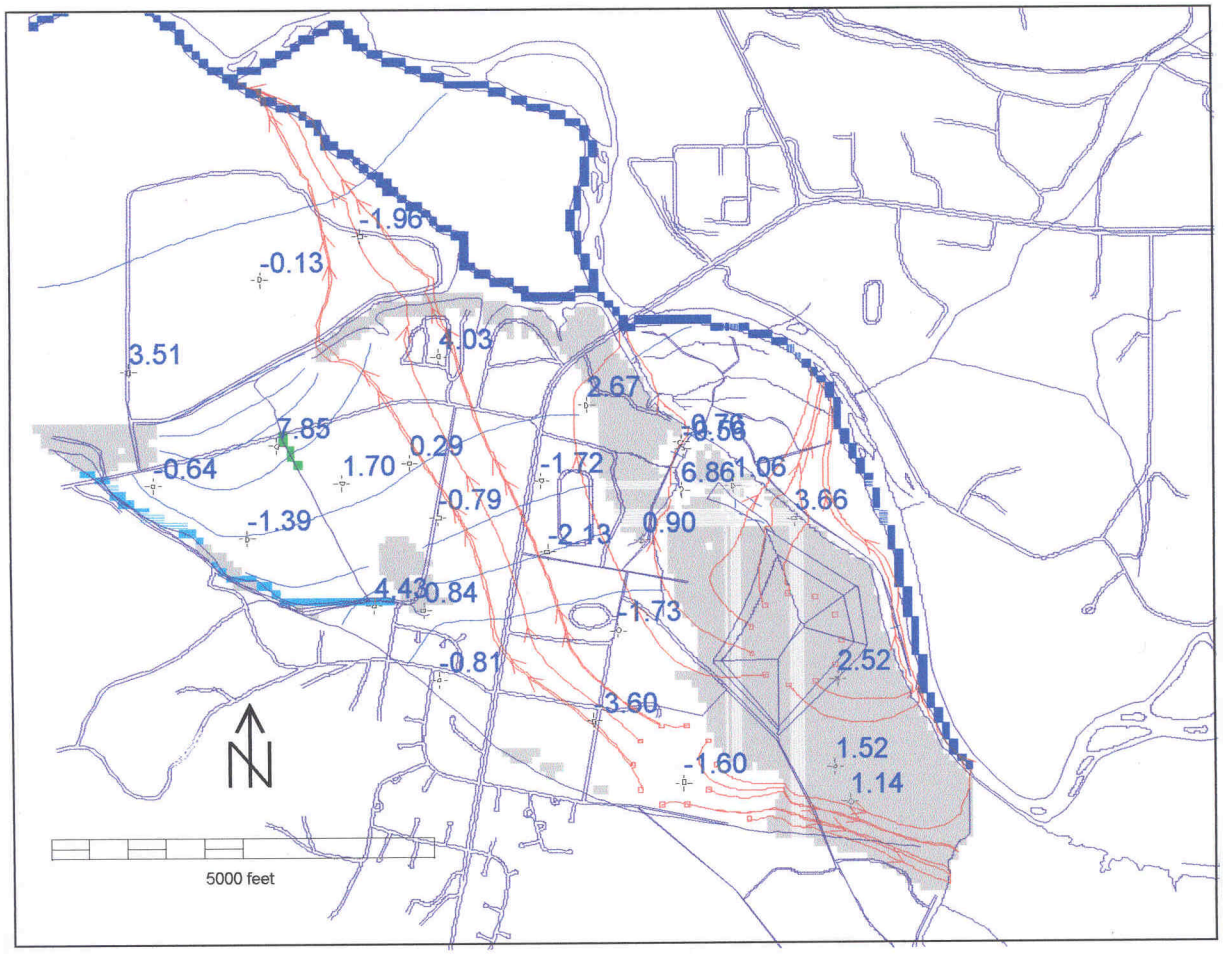


Figure 4–69. Results of Particle Tracking Simulation for the Terrace Alluvial Ground Water System and the Floodplain Alluvial Aquifer, Shiprock Site

The present contaminant distribution is simulated by creating a series of stress periods to mimic the conditions as they are hypothesized to have existed in the past. An approximation of the present distribution of contamination was thus recreated. The stress periods are approximated as follows:

1. Premilling conditions: A steady-state model is constructed with areal recharge equal to 7 percent of average annual precipitation. This condition results in a dry terrace gravel system.

2. Milling period (1954–1962): An 8-year stress period is created during which recharge through the raffinate ponds equals the rate that was estimated in the water balance. The area of the raffinate ponds is 20 acres, and the recharge rate in the area of the ponds is 4.46×10^{-3} ft/day. During the later part of this period the Helium Lateral Canal is operating and supplying water to the terrace alluvial system. Well 648 has not been constructed yet.

3. Milling period (1962–1968): A 6-year stress period is created to simulate the operation of the raffinate ponds, the Helium Lateral Canal, and the discharge of well 648 water to the floodplain alluvium. The well 648 discharge is conveyed to the floodplain through a ditch trending north-northeast from the well.

4. Post milling (1968–1978): A 10-year stress period is created to simulate the reduction in the source flux through the raffinate ponds. The Helium Lateral Canal continues to operate and well 648 continues discharging to the floodplain alluvium north-northeast of the well.

5. Post milling (1978–2000): This 22-year stress period simulates the early cleanup and relocation of the raffinate ponds to the present site of the disposal cell. The flux rate through the disposal cell area decreases to the present rate and is assumed to remain at that rate throughout the stress period. Discharge from well 648 is conveyed into Bob Lee Wash. The Helium Lateral Canal continues to function.

The final stress period creates an approximate distribution of contaminants as they exist at the site today. Simulations of remedial actions use these contaminant distributions as initial conditions for predictive modeling.

Simulations were performed to re-create the present concentrations of total nitrogen and uranium. Total nitrogen was chosen as a transport parameter for convenience to avoid modeling ammonium and nitrate separately. The rationale is that although ammonium was used during the milling process (Merritt 1971) and was predominantly present at the source, redox transformations along the flow path oxidized the ammonium to nitrate. Since the nitrogen mass does not change with a change in valence state, the total nitrogen is used as a transport parameter.

The modeling was conducted using a series of stress periods to simulate the changing hydrological conditions during the past 50 years. The output file corresponding to the final stress period is saved and used as an initial condition for additional simulations that predict the effects of pumping. Figure 4–70 depicts the nitrogen plume that corresponds to present conditions. This figure illustrates that the nitrogen has migrated to the northwest along the flowpath given by the MODPATH simulation. It has also migrated to the east toward Many Devils Wash; however, this

transport pathway is hidden from view in Figure 4–70 because the flow and transport occurs in Layer 2 of the model. The model shows Layer 1 to be dry above the transport horizon in Layer 2, and this is borne out by the field relationships.

Another observation is that the simulated floodplain alluvial aquifer contains elevated concentrations along a band that mimics the field relationships. The contamination enters the ground water flow system through the Recharge Zone 3 term, or the drainage of residual moisture from the disposal cell. This model simulation supports the field observations that there is no residual source in the floodplain that creates the elevated concentrations both along the panhandle (southeast or upgradient part) of the floodplain and north of the disposal cell. Drainage of residual fluids from the terrace system is capable of creating the observed contamination.

Table 4–31 presents a summary of the residuals for the nitrogen plume simulation. The summary statistics indicate that the model is calibrated to the residual standard deviation divided by the head range of 20 percent. This value falls above the flow-model calibration objective of 10 percent; however, calibration to observed concentrations is seldom achieved in practice. Nevertheless, a preliminary analysis of the residual mean and the maximum residuals reveals that there is a negative bias in the data that may be corrected by increasing the source area concentration. This and other adjustments may be required to improve the calibration.

The uranium plume is simulated with the source being applied at the disposal cell area; however, the concentration at the source is much lower than for the nitrogen plume. As the K_d values indicate, the uranium plume migrates much less than the nitrogen plume. The K_d describes the degree that a constituent's concentration is attenuated as it migrates through the subsurface. Figure 4–71 presents the uranium plume for Layers 1 and 2 of the model, the two layers that describe the terrace alluvium and the weathered bedrock, respectively.

The standard deviation divided by the range is approximately 20 percent for the calculated uranium plume. The positive residuals outside the area of the plume indicate that low levels of uranium have been detected in those areas but that the calculated concentrations are absent in those areas. The possibility exists that the uranium levels in those areas are part of natural background. Contributions of background uranium could be simulated by applying a low mass flux with the areal recharge term. This would have the effect of reducing the residuals in those areas. However, if the low concentrations of uranium originate at the cell, the laboratory K_d values are far too high. With a K_d of zero the uranium would have an areal extent similar to the nitrogen plume presented in Figure 4–70 and could partly explain how the low uranium concentrations migrated that far west of U.S. Highway 666.

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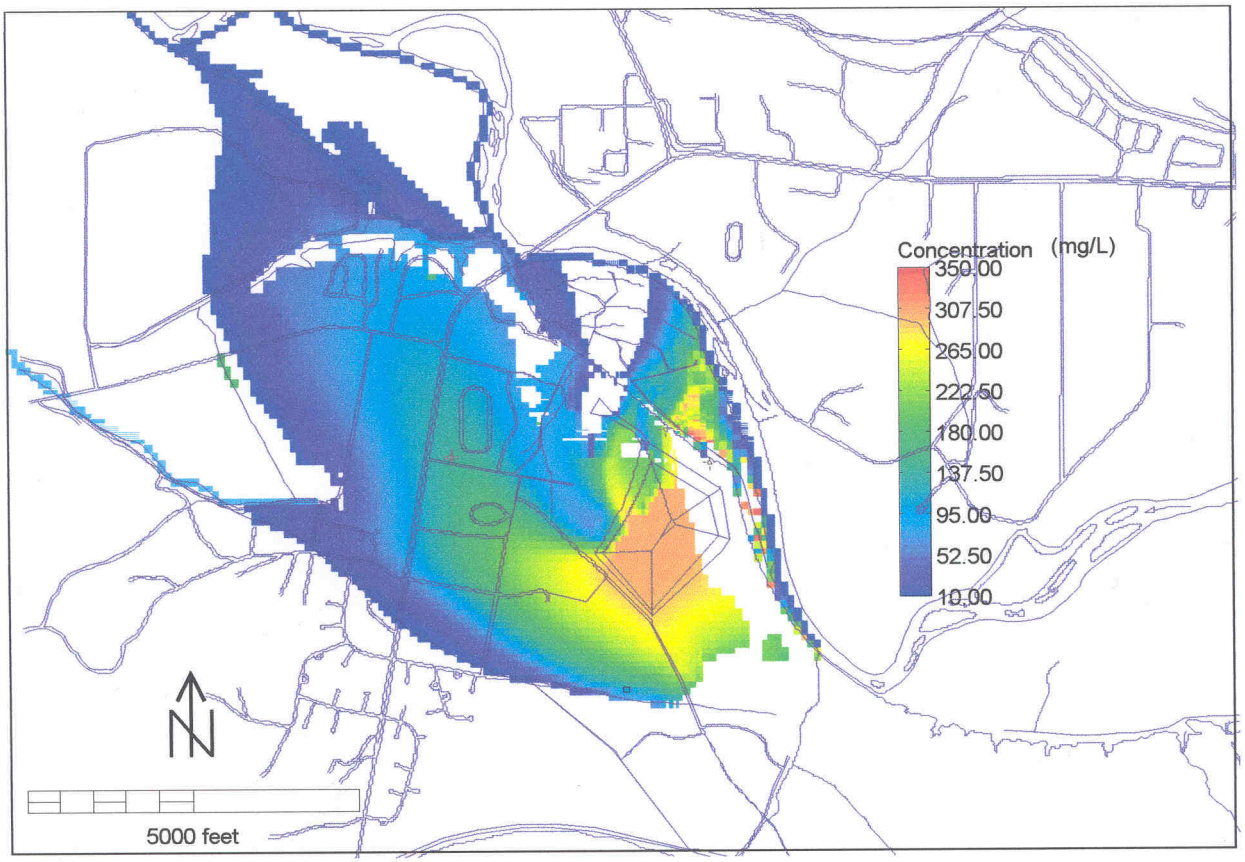


Figure 4–70. Model Simulation of Present Nitrogen Concentrations in Ground Water in Layer 1 of the MT3D Transport Model for the Shiprock Site

4.5.5 Active Remediation Simulations

Because the model is reasonably well calibrated for present conditions it is assumed that it can also provide useful and reasonable estimates of water level changes and plume migration that would result from active remediation. Projections were carried through for a 100-year simulation period and output files were written for each of 15 time steps during that period.

The remedial actions contemplated for the Shiprock site consist of active pumping to control ground water discharge to exposure areas, and no remediation of distal portions of the plume that are beyond the well field. Figure 4–72 shows the location where the active remediation would occur. The questions that must be addressed include:

- Is pumping an effective method to gain hydrologic control of the plume?
- What is a reasonable pumping rate and how many wells would be required to control the plume near the disposal cell?

Table 4–31. Summary of Residuals and Calibration Statistics for the Nitrogen Transport Model, Shiprock Site

Well	Layer	Observed	Computed	Residual	Name	Layer	Observed	Computed	Residual
606	4	24.83	8.34	16.49	728	1	501.13	187.04	314.09
608	4	406.32	844.50	-438.18	730	2	91.87	387.86	-295.99
610	4	812.64	127.38	685.27	731	1	166.59	-1.00	167.59
612	4	0.90	173.15	-172.25	812	2	962.30	18.78	943.52
613	4	67.72	313.51	-245.79	813	1	1527.09	140.40	1386.68
615	4	248.31	94.73	153.58	814	2	707.67	173.70	533.97
617	4	19.86	144.35	-124.49	815	1	449.44	137.94	311.50
619	4	178.33	41.73	136.60	816	1	6.06	80.01	-73.95
620	4	158.01	53.78	104.23	818	1	983.30	149.27	834.03
624	4	270.88	5.73	265.15	819	2	4.67	245.81	-241.14
625	4	200.90	6.08	194.82	826	2	9.28	52.01	-42.73
626	4	2.26	0.10	2.16	827	1	123.70	-1.00	124.70
628	4	36.12	0.00	36.12	828	2	19.47	121.80	-102.33
630	4	40.63	6.39	34.24	829	2	3.27	8.26	-4.99
734	4	0.23	22.41	-22.18	830	1	16.39	46.60	-30.22
735	4	532.73	28.22	504.51	832	1	54.18	18.87	35.30
736	4	39.50	0.00	39.50	833	1	61.63	59.92	1.71
853	4	0.02	275.05	-275.03	835	1	6.09	76.10	-70.01
854	4	501.13	68.22	432.91	836	1	20.77	0.25	20.52
855	4	6.09	19.39	-13.29	837	1	0.59	8.10	-7.52
856	4	0.04	15.87	-15.83	838	1	2.93	17.09	-14.15
857	4	0.03	108.74	-108.72	839	1	399.77	121.56	278.21
600	4	189.62	250.17	-60.55	841	1	347.63	20.36	327.27
602	2	12.42	215.65	-203.23	843	1	0.50	26.99	-26.49
603	1	23.93	238.42	-214.50	844	1	62.98	60.36	2.62
604	2	589.16	63.05	526.12	845	1	1.20	17.35	-16.16
605	1	36.12	70.30	-34.18	846	1	57.11	0.00	57.11
633	1	54.18	65.67	-11.49	847	1	1.29	3.97	-2.68
725	1	44.47	74.76	-30.29	848	1	1.20	0.01	1.19
726	2	5.87	87.56	-81.69	DM7	4	414.67	283.24	131.43
727	1	521.44	81.87	439.58	MW1	1	2.11	-1.00	3.11

Residual Mean: 99.54
Res. Std. Dev.: 311.75
Sum of Squares: 6438994.94
Abs. Res. Mean: 198.88
Min. Residual: -438.18
Max. Residual: 1386.68
Head Range: 1527.07
Std/Head Range: 0.20

- Will pumping create measurable declines in ground water levels in the irrigated areas west of the disposal cell?
- Will pumping dry up the seeps and springs that discharge to Bob Lee Wash and Many Devils Wash?

To address these questions it was useful to create preliminary data sets to test prospective well locations and pumping rates. One strategy for removing water from the terrace alluvium is to

position two recovery wells (Qt-01 and Qt-02) along the deepest part of the buried channel. The wide spacing between these two wells and the modest pumping rate would yield ground water for a relatively long duration. It would also recover ground water with high concentrations of nitrate and sulfate primarily. However, obtaining ground water from the thin saturated thickness of the terrace system would also be necessary to intercept the contaminated ground water that feeds Bob Lee Wash and Many Devils Wash. Ground water from these thinner sections of the terrace flow system might best be intercepted with an interceptor trench equipped with a pump to lift the ground water to the surface for treatment. Two such interceptor trenches were simulated: one to intercept the ground water that drains to Bob Lee Wash (Drain 01), and the other to intercept the ground water that drains to Many Devils Wash (Drain 02). The remaining ground water on the terrace would be allowed to flow unimpeded northwest toward discharge points near 1st, 2nd, and 3rd Washes and to locations farther west.

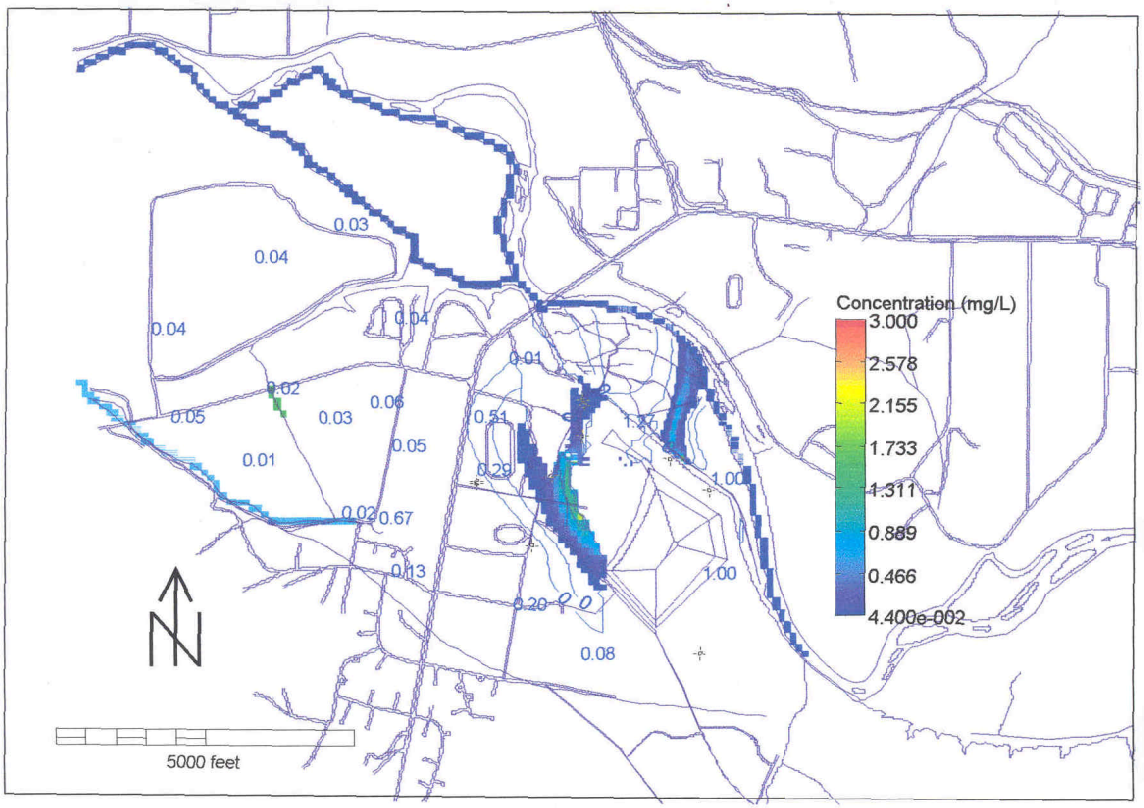
For the floodplain alluvium, wells Qal-03, Qal-04, Qal-05, and Qal-06 were tentatively located along the band of contamination that transects the floodplain alluvium from the base of escarpment north to the San Juan River. The purpose of these wells is to pump from the region with the greatest levels of ground water contamination in the floodplain and to allow the lower levels of contamination to naturally attenuate by discharging into the San Juan River. The fifth well in the floodplain, Qal-01, is designed to intercept contamination that exists at the base of the escarpment west of Bob Lee Wash.

Time Stepping

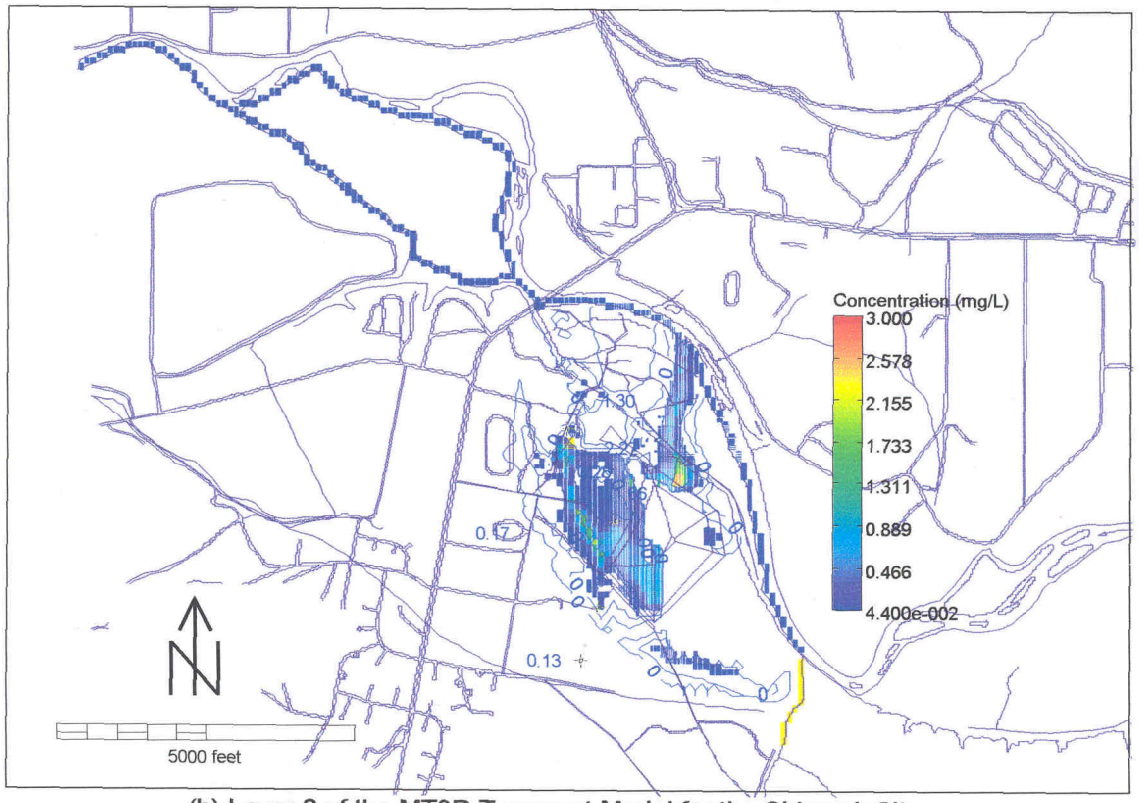
Time steps are used in flow and transport modeling when detailed information is desired at discrete time increments. Output files are written for each time step, enabling the decision maker to evaluate the questions posed in the introduction to this section. Table 4–32 presents the schedule used for the time stepping. A simulation period of 100 years and a time step multiplier of 1.2 was selected to yield ample data early in the pumping project when changes are significant, and sparser late-stage data for when conditions are stabilizing.

Table 4–32. Time Stepping Schedule for Transient Simulations of Remedial Action at the Shiprock Site

Time Step	Delta (m+1)	Total Time (d)	Total time (yr)
1	507	507	1.4
2	608	1,115	3.1
3	730	1,844	5.1
4	876	2,720	7.5
5	1,051	3,771	10.3
6	1,261	5,031	13.8
7	1,513	6,544	17.9
8	1,816	8,360	22.9
9	2,179	10,539	28.9
10	2,614	13,153	36.0
11	3,137	16,291	44.6
12	3,765	20,055	54.9
13	4,518	24,573	67.3
14	5,421	29,994	82.2
15	6,506	36,500	100.0



(a) Layer 1



(b) Layer 2 of the MT3D Transport Model for the Shiprock Site

Figure 4–71. Model Simulation of Present Uranium Concentrations in Ground Water

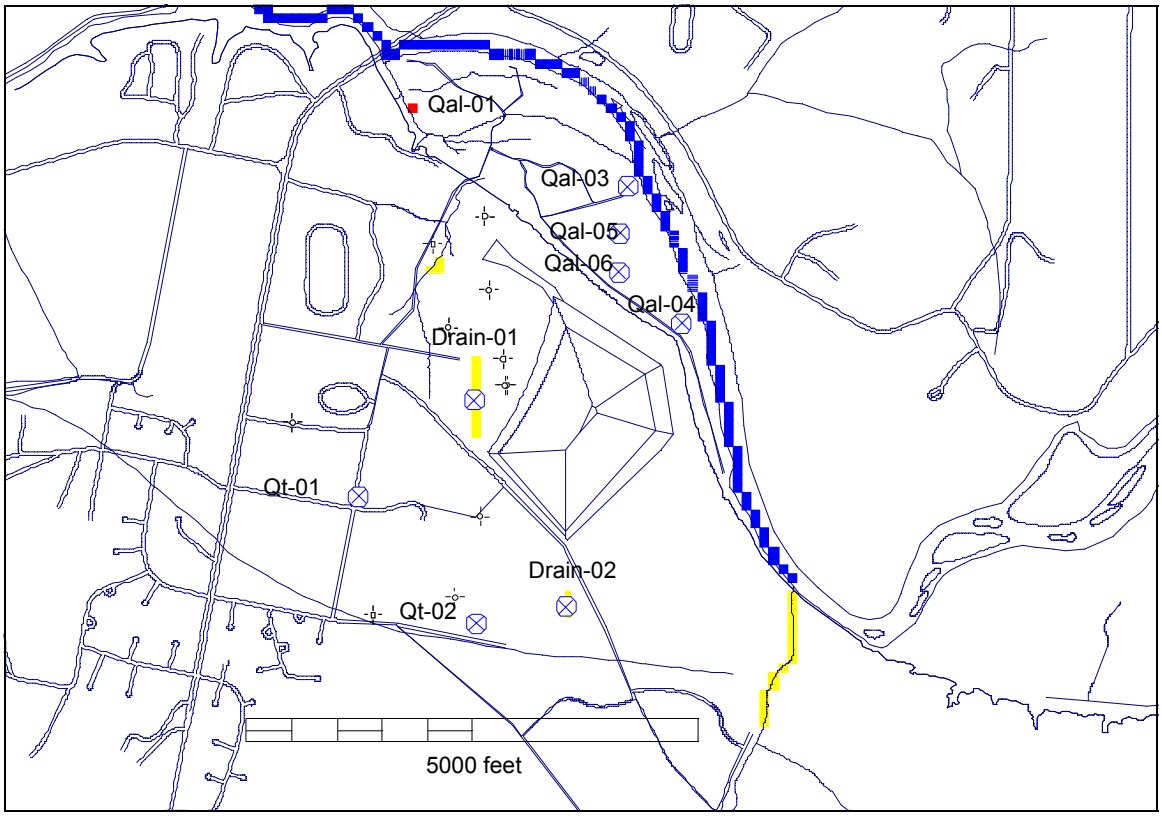


Figure 4–72. Map of Area where Remedial Actions were Simulated for the Shiprock Site

Predicted Effects of Future Withdrawals

Projections of future ground water withdrawals on the terrace and in the floodplain are shown in Table 4–33. A combination of two interceptor trenches and two recovery wells for the terrace ground water system and five extraction wells for the floodplain were used for the simulations.

Table 4–33. Summary of Withdrawal Rates for the Simulations of Active Remediation at the Shiprock Site

Well ID		Withdrawal Rate (ft^3/day)	Withdrawal Rate (gpm)
Terrace Ground Water System			
Qt-01		576	3
Qt-02		192	1
Drain-01		Variable: declines over 20 years to 600	Variable: declines over 20 years to 3
Drain-02		Variable: declines over 20 years to 200	Variable: declines over 20 years to 1
Subtotal		Stabilizes at 1540	Stabilizes at 8
Floodplain Alluvial Aquifer			
Low-Pumping Rate	Qal-01	1540	8
	Qal-03	1920	10
	Qal-04	1540	8
	Qal-05	1540	8
	Qal-06	1540	8
	Subtotal (rounded)	8000	40
High-Pumping Rate	Qal-01	3080	16
	Qal-03	3840	20
	Qal-04	3080	16
	Qal-05	3080	16
	Qal-06	3080	16
	Subtotal (rounded)	16000	80

The effectiveness of active ground water remediation was evaluated with a set of two ground water extraction scenarios, or projections. The first projection was made with a relatively low extraction rate of 40 gpm in five wells located in the floodplain. The other projection consisted of extraction rates that were double the first, or 80 gpm in the floodplain. The pumping rates on the terrace were approximately 8 gpm for both simulations.

Terrace Ground Water System

Modeling of the terrace ground water system is pursued to evaluate the time required for Many Devils Wash and Bob Lee Wash to become hydraulically isolated from (1) the buried channel south of the disposal cell and (2) the ground water that originates as drainage from the disposal cell. Of particular concern is the discharge of contaminant-laden ground water to potential receptor areas where interim actions will have occurred.

The extraction rates for wells Qt-01 and Qt-02 are set at 3 gpm and 1 gpm, respectively. Well Qt-02 is likely to encounter low-level cutoff sometime during the 4th time step. If the cell recharge rate is double the calibrated steady state value, both wells would operate into the 8th time step. The extraction of water through the interceptor trenches occurs initially at a rate of approximately 2 gpm per trench. As the water levels decline near the trenches, the flux rate into

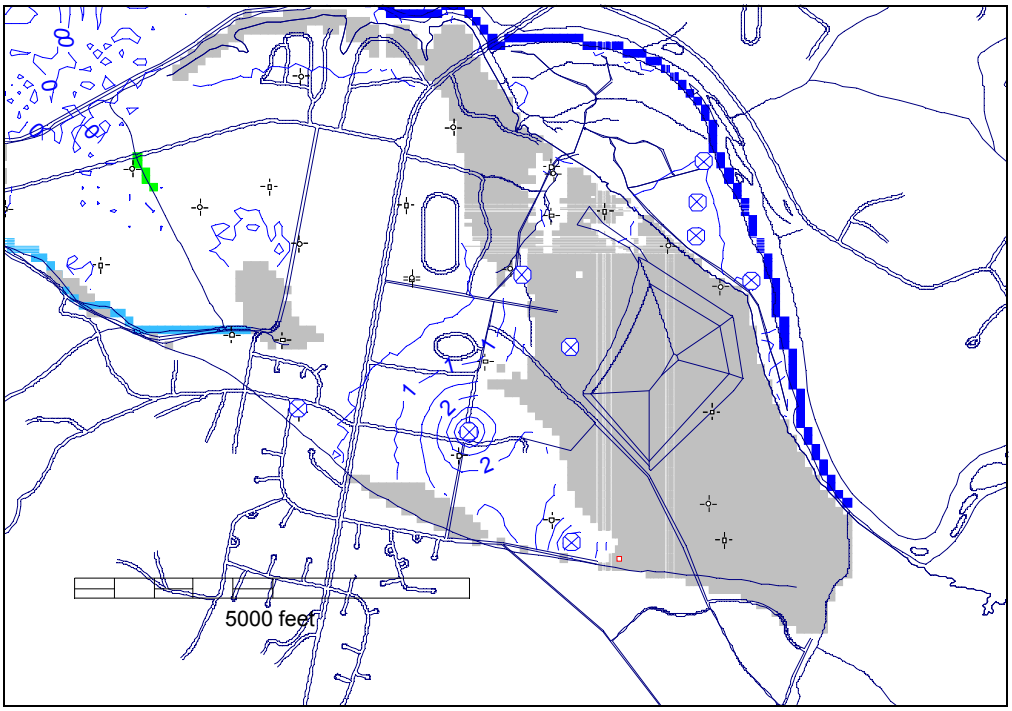
them decreases. Eventually, the water levels decline to the point where the trenches no longer intercept ground water. The trenches are assumed to be keyed into the weathered Mancos Shale bedrock. Figure 4−73 depicts the simulated water level declines near the disposal cell and shows how the terrace gravel dries out during the 4th time step, or a period ranging from approximately 5.1 to 7.5 years. This is the minimum time required for the compliance strategy to be achieved in the terrace ground water system. It is a minimum because the recharge rate on the terrace is unknown and may be larger than the rate used in the modeling.

Although the pumping would be an effective method to lower the ground water on the terrace, it alone is not capable of eliminating the spring flow at the base of the escarpment. Spring flow is curtailed by the elimination of excess recharge in the areas between the disposal cell and the seeps (Figure 4–72). The excess recharge in these areas is believed to be an important source of ground water feeding seeps 425 and 426. When ranked in order of decreasing importance, the excess recharge has two main sources: (1) disposal-cell-runoff, energy-dissipation area located northwest of the disposal cell, and (2) NECA Pond and equipment washing areas. Runoff from the cell during rain events has been observed at this area on numerous occasions and is well documented. The absence of the resistant siltstone bed in this area means that runoff from the cell readily infiltrates into the ground at this location and helps sustain the flow in seeps 425 and 426. Geochemical evidence also supports this model because the measured concentrations of nitrate and uranium are about 10 times less than those that discharge as seeps into Bob Lee Wash. Elimination of the excess recharge was simulated with modeling and revealed that the water levels in the area of the seeps would decline 10 ft or more during the 4th time step (approximately 5.1 to 7.5 years).

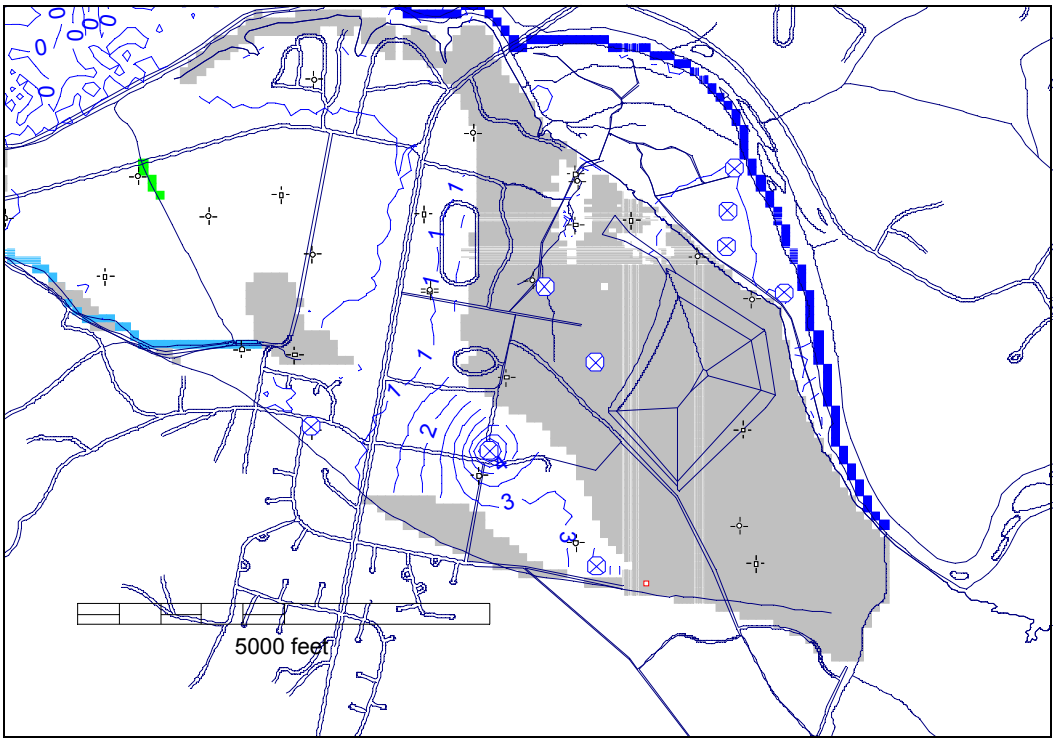
Floodplain Alluvial Aquifer

The compliance strategy for the floodplain alluvial aquifer is to employ active remediation to reduce contaminant concentrations to a point where natural flushing can take over. Higher hydraulic conductivity, relative to the terrace ground water system, and the recharging influence of the San Juan River are responsible for higher potential well yields in the floodplain alluvium.

Projections of the effectiveness of active remedial action in the floodplain alluvium were simulated with both low and high pumping rates. The lower pumping rate of 40 gpm was accomplished with four extraction wells positioned along the axis of the nitrate and uranium plumes in the floodplain and one extraction well at the base of the escarpment west of the mouth of Bob Lee Wash. Figure 4–74 and Figure 4–75 show that the lower pumping rate is incapable of reducing uranium and nitrate concentrations to below the MCL over the entire floodplain. It is capable of intercepting the contamination that would otherwise have been discharging into the San Juan River.

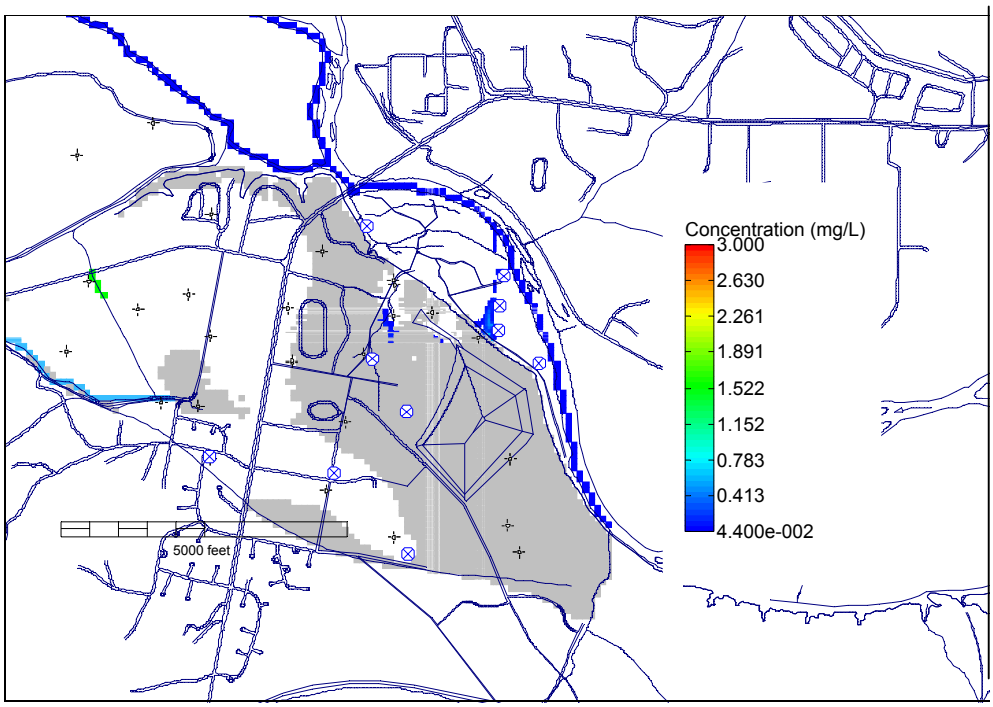


(a) 3 yrs

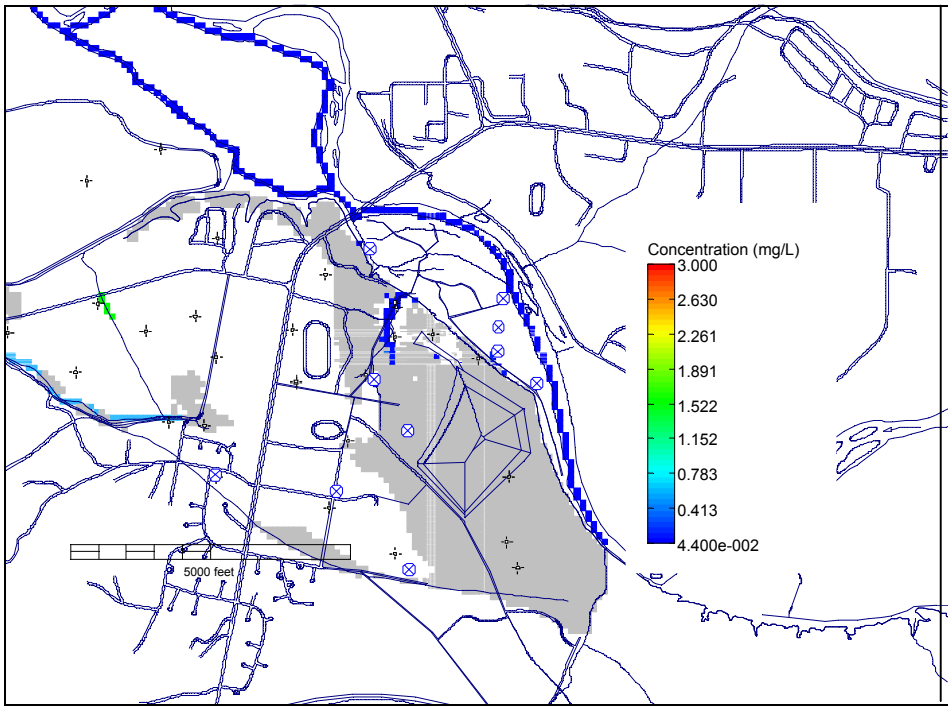


(b) 5 yrs after active remediation begins in the terrace ground water system at the Shiprock site

Figure 4–73. Simulated Decline in Water Level in Feet Near the Disposal Cell (Note: the gray areas indicate dry cells in the terrace gravel)

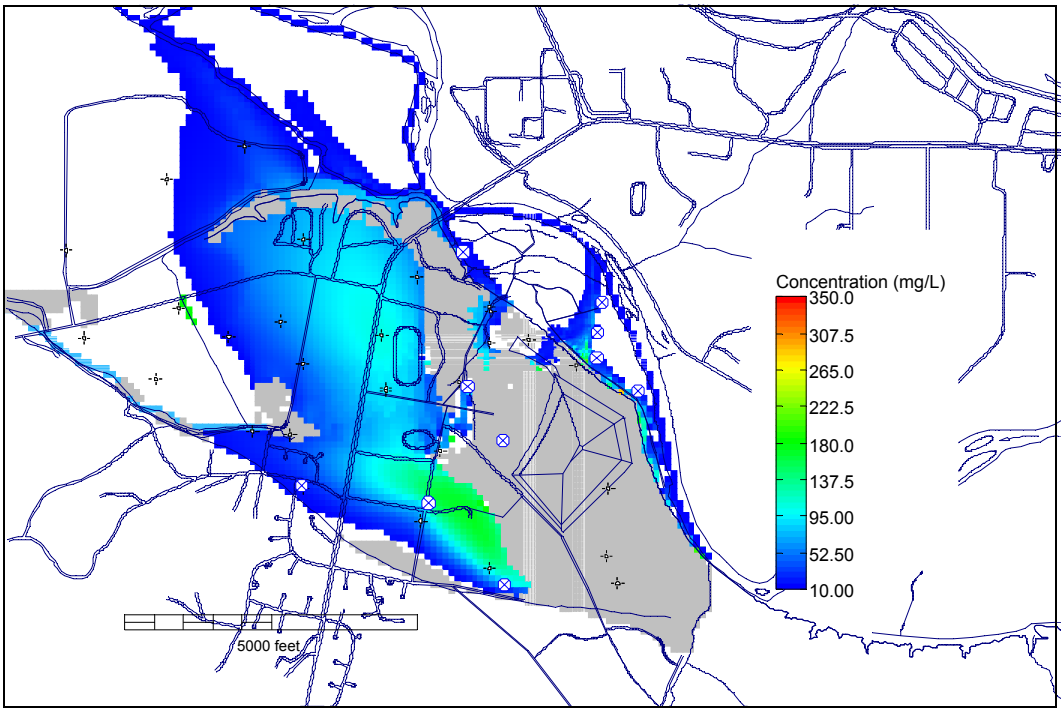


(a) 40 gpm

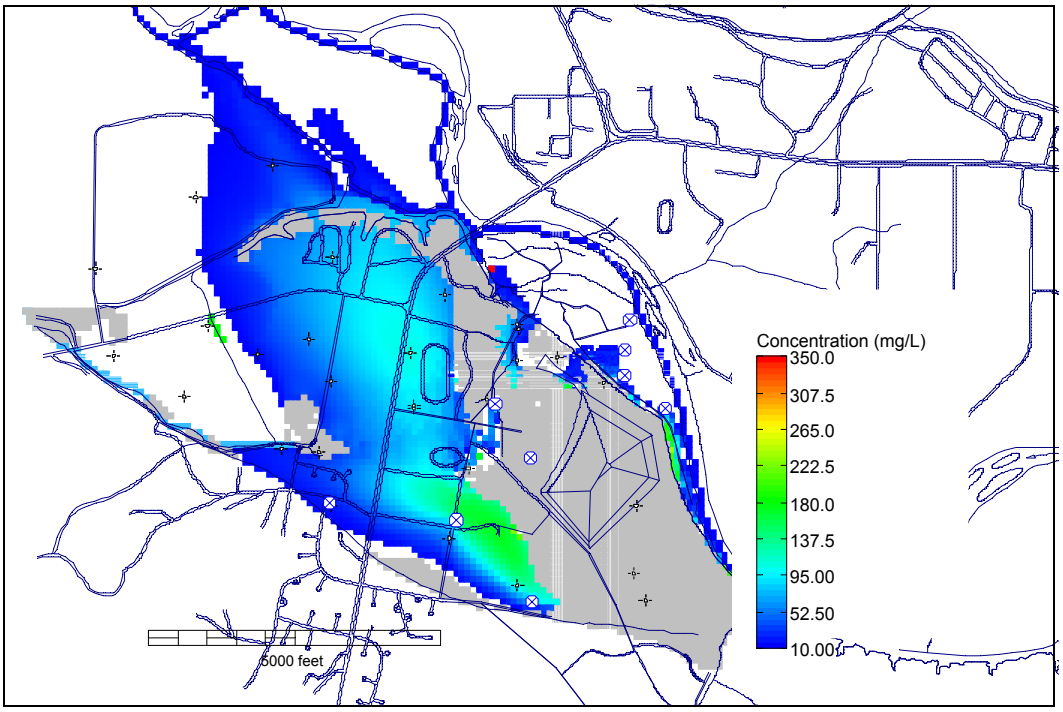


(b) 80 gpm

Figure 4–74. Projection of Simulated Uranium Plume 13.8 Years After Pumping Begins in the Floodplain Alluvial Aquifer



(a) 40 gpm



(b) 80 gpm

Figure 4–75. Projection of Simulated Nitrate Plume 13.8 Years After Pumping Begins in the Floodplain Alluvial Aquifer

The higher pumping rate of 80 gpm was simulated by doubling the pumping rate for the wells used in the first simulation. The higher pumping rate was considered because the lower pumping rate was incapable of removing all the contamination in the floodplain even after a period of 100 years. The higher pumping rate is incapable of completely eliminating all the uranium and nitrate contamination from the floodplain aquifer. The contamination that remains in the alluvial aquifer after the 6th time step, a period of 13.8 years, is slightly above the MCL and located north of the disposal cell near the base of the escarpment. The uranium and nitrate concentrations in this area are approximately twice the MCL, and nitrate concentrations exceed the MCL over a larger area than the uranium. The contamination is coming from layers 3 and 4 in the Mancos Shale beneath the disposal cell. This contamination remains in the Mancos Shale throughout the 100-year simulation period. It is believed that it remains there because the source flux remains constant throughout the simulation period. If the source flux were to decay to trace rates during active remediation, and the contamination stored in the Mancos Shale was exhausted or removed, then the floodplain extraction wells would likely remove all the contamination in the floodplain, and natural attenuation would be effective.

Selenium, with a K_d factor of 20 times that of uranium, would feed into the floodplain at a much slower rate and would not be expected to be a long-term problem from the perspective of meeting MCLs. However, the natural background flux of selenium from the Mancos Shale is believed to be sufficiently high to mask the contribution from the disposal cell. Because the natural selenium cannot be separated from mill-related selenium from the disposal cell, little benefit would be derived from modeling selenium transport. Laboratory data, prepared by the ESL, indicated that soils from background locations had higher levels of leachable selenium than did site-related samples, for which selenium was not detected (DOE 1999a).

The higher pumping rate results in an exchange of 1 pore volume in 1.4 years, or 10 pore volumes in 14 years. It is assumed that any mill-related selenium would flush from the floodplain in less than 20 years at the higher pumping rate.

4.5.6 Conclusions

The transport and remedial modeling simulations are based on the assumption of a continuing source located at the disposal cell. The source flux is assumed to be constant with time. If the source flux decreases substantially with time, as would be the case with rapid transient drainage, the active remediation would be effective in removing contamination from the floodplain aquifer. If the source flux remains constant with time, it may be difficult to permanently remove the contamination from the floodplain. A projection carried out to 100 years shows that nitrate, uranium, and sulfate would continue to feed the remediation wells in the floodplain alluvial aquifer as long as the source exists. With active remediation the MCLs could be met in the floodplain in most areas but would be difficult, if not impossible, to achieve at the base of the escarpment north of the disposal cell.

The modeling results presented in this section are preliminary, they are intended to portray the feasibility of active remediation and to identify the parameters to which the modeling predictions are sensitive. Clearly, the long-term effectiveness of active ground water remediation at the Shiprock site is dependent upon recharge through the disposal cell, that is, the long-term performance of the disposal cell. If residual drainage of moisture from the disposal cell proves to be transient, the likelihood is higher that active remedial action would be effective. If the drainage of residual moisture is constant, then remedial action will be ineffective. Ground water

monitoring and evaluation during remedial action would be necessary to assess any effects of the disposal cell. Specific monitoring would be designed for areas that are particularly sensitive to the recharge terms, such as the base of the escarpment and in the buried channel on the terrace. A specific monitoring strategy would be developed during design of the active remediation.

After reviewing SOWP Rev. 1, the Navajo Nation requested DOE to consider what effect eliminating flow from well 648 would have on ground water flow and transport. This evaluation was not performed for the present SOWP because the revised conceptual model, together with the proposed active remediation, presents an almost limitless set of initial conditions on which to base the modeling. Additionally, jurisdictional control of well 648 resides with the Navajo Water Code Administration. The Navajo Nation has not provided a clear indication of its intentions for well 648.

It would be risky for DOE to proceed with construction of a remediation project if the status of well 648 remains unresolved. The highest chances for a successful remediation would exist if DOE applied for, and was granted, control of water from well 648. Under the present scenario, flow from well 648 would be a necessary component of the active remediation. Consequently, the modeling herein assumes that well 648 will continue discharging indefinitely.

4.6 Ecological Field Investigations

The ecology of the former Shiprock millsite and surrounding areas has been further characterized to support the assessment of potential ecological risks associated with site-related contaminated ground water and to update the BLRA (DOE 1994). A defensible ecological risk assessment (ERA) will provide a sound basis for the development of a risk-based compliance strategy. In general, the goal of the continued ecological field investigations is to acquire additional data needed to evaluate potential exposure pathways and receptors at the Shiprock site.

A summary of the BLRA, including discussion of the ecological contaminants of potential concern, potential receptors, and potential adverse effects, is available in Chapter 5 of the *Work Plan for Characterization Activities at the Shiprock UMTRA Project Site* (DOE 1998d). The Work Plan also contains a summary of specific ecological data needed to update the BLRA. Sections 4.6.1 and 4.6.2 below present descriptions of ecological field activities conducted in 1998, 1999, and 2000.

The 1998 and 1999 ecological field investigations addressed the following data needs:

- Characterization of the current vegetation of the floodplain area adjacent to the millsite and nearby reference areas. This activity focused on plant communities containing phreatophytes (deep-rooted plants capable of contacting and using ground water aquifers) and wetland species potentially rooted into contaminated ground water and surface water and similar plant communities in reference areas.
- Sampling and chemical analysis of phreatophyte and wetland plant tissues in contaminated areas and in reference areas for comparison. The purpose of the plant tissue analyses is to determine whether exposure pathways may exist between ground water and biota through uptake by deep-rooted plants and, if so, to assess the potential toxicity to these plants and to animals that might ingest them.

- Sampling and chemical analysis of sediment and surface water in the wetland area at the mouth of Bob Lee Wash and in a reference wetland area for comparison. Results of the surface water and sediment analyses will be used to characterize potential risk to aquatic life in the wetlands and to wildlife and livestock receptors that might ingest the water, sediment, or plants and animals that have been exposed to these media.

The 2000 ecological field investigations addressed the following data needs:

- Sampling and chemical analysis of surface water in the San Juan River at the mouth of Many Devils Wash and along the eastern half of the floodplain, in the grassy area below seep 425 and the newly-inundated areas on the floodplain just west of the mouth of Bob Lee Wash, and at the distributary channel. Results of these analyses will be used to supplement existing data for these media at these locations for the evaluation of risk to aquatic life and to terrestrial receptors that might ingest the water or food items from the water.
- Sampling and chemical analysis of key range grasses from the floodplain area to evaluate potential risks to livestock that may graze the area in the future. Results from this sampling event are not currently available. These data will be presented in the Shiprock EA.

4.6.1 Vegetation Characterization

Plants that root into sediment contaminated with site water or are irrigated with contaminated site water are potential exposure pathways for humans and ecological receptors. The vegetation also influences recharge and discharge components of the hydrologic system. Current vegetation of the Shiprock floodplain and associated wetlands was characterized as part of the evaluations of (1) potential human health and ecological risks associated with site-related contaminated ground water and (2) the relative importance of on-site evapotranspiration as a component of the site water balance.

4.6.1.1 Methods

The vegetation of the floodplain and associated wetlands was characterized using the semiquantitative relevé technique (Bonham 1989). This technique was used to characterize the composition and relative abundance of species in plant communities by subjectively selecting representative stands, to compile a list of all species identified while walking through the stands, and then to assign the species to one of six cover classes. Plant cover was not measured precisely. Millsite floodplain and wetland stands were characterized in June 1998 and September 1999.

Vegetation was characterized both in areas influenced by the site-related contaminated ground water, the millsite floodplain and wetlands, and in reference areas. The millsite floodplain is the relatively broad plain between the escarpment north of the disposal cell and the San Juan River (Plates 1 and 2). The millsite wetlands is a poorly drained 5-acre area at the mouth of Bob Lee Wash on the floodplain.

Reference areas, or background areas, resemble the site ecologically—landform, soil, and vegetation are similar—but without the influence of millsite-related ground water contamination. Reference areas were used for baseline chemical data for the ERA (Section 4.6.2) and to help project possible successional pathways. The reference area for the millsite floodplain is a floodplain approximately 1 mi upstream from the disposal cell at the site of wells 850 through

852 (Plate 1). The reference area for the floodplain wetland is a ditch along which outflow water from artesian well 648 flows to Bob Lee Wash and onto the floodplain (Plate 1).

4.6.1.2 Results

Figure 4–76 and Table 4–34 and Table 4–35 present the results of the plant ecology characterization for areas east of U.S. Highway 666. The area to the west of U.S. Highway 666 was characterized only qualitatively because of difficulty of access. Vegetation types west of U.S. Highway 666 were very similar to those mapped on the east. The results confirm the occurrence of phreatophytic and wetland plants (plants that root in ground water) in areas with elevated ground water contamination. Cottonwoods (*Populus fremontii*), saltcedar, Russian olive (*Eleagnus angustifolia*), and greasewood (*Sarcobatus vermiculatus*) growing in the floodplain are all phreatophytes.

Spikerush (*Eleocharis palustris*), common reed (*Phragmites australis*), alkaligrass (*Puccinellia airoides*), bulrushes (*Scirpus* spp.), saltcedar (*Tamarix ramosissima*), and cattails (*Typha latifolia*) growing in the wetland area are also potentially in contact with contaminated water. All these plants may create exposure pathways.

The results of the plant ecology characterization provide input to several other aspects of the field investigation:

- The plant community map and plant characterization data, in combination with ground water data (Section 4.4), were the basis for selecting locations for chemical analysis (Section 4.6.2) as part of the ERA (Section 6.2).
- The plant community data support habitat evaluations for threatened and endangered species (Ecosphere Environmental Services 1998, 1999) and other receptors.
- The occurrence and relative abundance of certain plant species provide a measure of the magnitude of past ecological impacts, such as grazing, and the current health of the ecosystem.
- The plant community data are needed to estimate evapotranspiration as a component of recharge and discharge calculations for site water-balance modeling.

The plant community data are also needed to evaluate potential effects of remediation alternatives and future land-use alternatives.

The vegetation map for the millsite floodplain and wetlands (Figure 4–76) consists of several different plant associations. A plant association is a unit of classification that defines a particular plant community. An association generally has a consistent floristic composition, a fairly uniform appearance, and a distribution that reflects a certain mix of environmental factors that can be shown to be different from other associations. The relevé data (Table 4–34 and Table 4–35) were obtained in plant stands that were considered to be representative of an association. The tables also include relevé results for the references areas. The two-part name of a plant association generally consists of the dominant overstory and understory species.

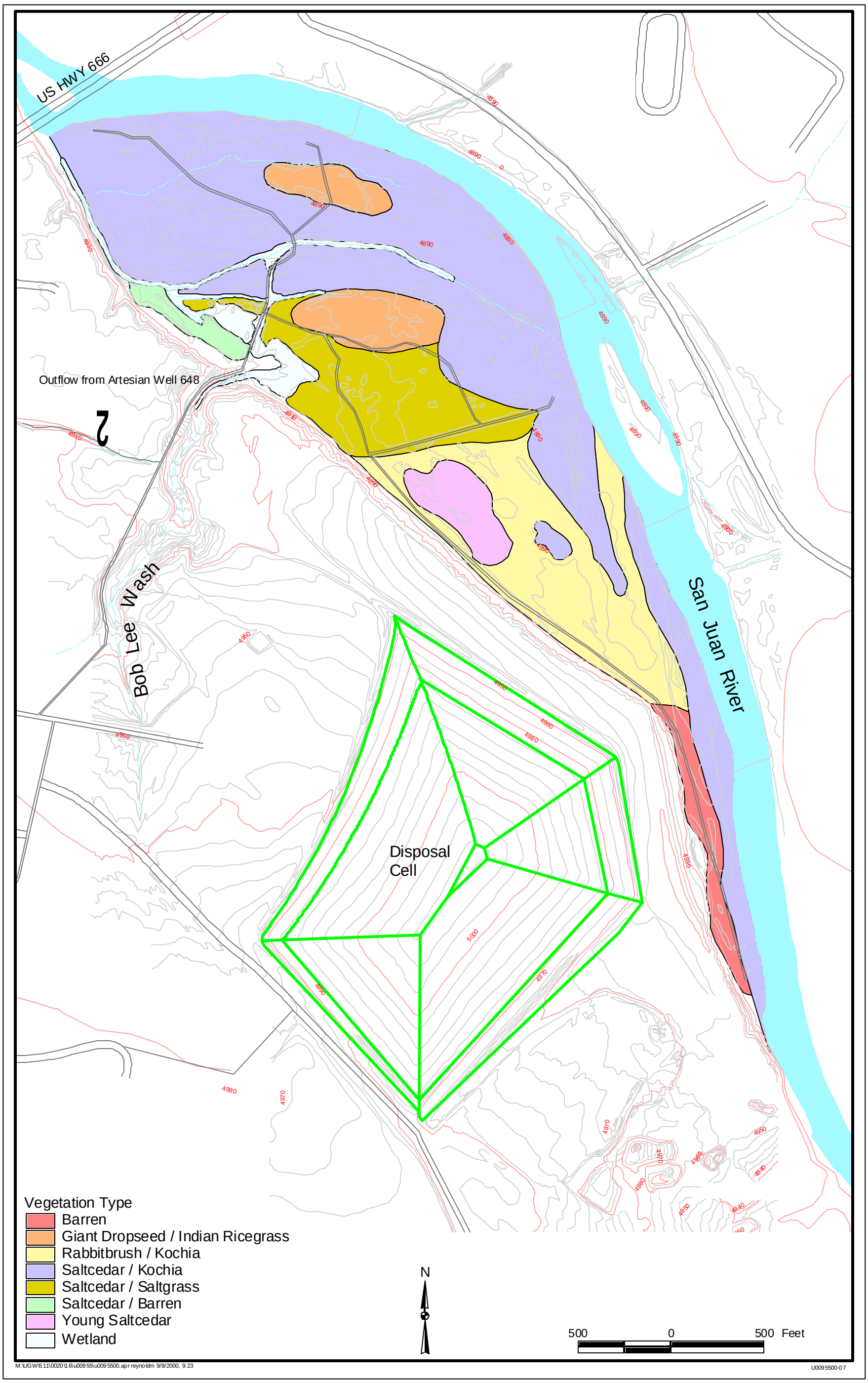


Figure 4–76. Shiprock Millsite Floodplain Vegetation Map

Table 4–34. Relevé Plant Cover for the Shiprock Millsite Floodplain and Floodplain Reference Area[a]

Taxonomic Name	Common Name	Reference Area	Saltcedar Kochia	Saltcedar Barren	Rabbitbrush Kochia	Saltcedar Saltgrass	Giant Dropseed
Atriplex canescens	Four-wing saltbush		+				
Cardaria draba	Whitetop		+				
Chrysothamnus nauseosus	Rubber rabbitbrush						+
Chrysothamnus viscidiflorus	Green rabbitbrush	+	+		2		1
Distichlis spicata	Saltgrass	2	2	3		5	1
Eleagnus angustifolia	Russian olive	2	1				
Guitterezia sarothrae	Broom snakeweed		+				+
Kochia scoparia	Kochia		2				
Kochia sp.	Kochia			2	3	1	
Lactuca serriola	Wild lettuce	+				2	
Machaeranthera canescens	Hoary aster						1
Mentzelia pumila	Blazing star	+	+		1		1
Oenothera albicaulis	Evening primrose		+				
Oryzopsis hymenoides	Indian ricegrass	1	1				2
Populus fremontii	Fremont cottonwood	2	1				
Salix exigua	Sandbar willow	+	+				
Salsola kali	Russian thistle		+		1		+
Sarcobatus vermiculatus	Greasewood		+				
Sitanion hystrix	Squirreltail					2	
Sporobolis airoides	Alkali sacaton	3		+	2		
Sporobolis gigantea	Giant dropseed	+	1		1		4
Tamarix ramosissima	Saltcedar	3	4	3	2	3	+
Xanthium strumarium	Cocklebur	+					

[a]Cover Classes: (+) <1%, (1) 1-5%, (2) 5-25%, (3) 25-50%, (4) 50-75%, and (5) 75-100%.

Table 4–35. Relevé Plant Cover for the Shiprock Millsite and Reference Wetlands[a]

Taxonomic Name	Common Name	Millsite Wetland	Reference Wetland
Cardaria draba	White top	2	1
Distichlis spicata	Inland saltgrass	2	2
Eleagnus angustifolia	Russian olive		+
Eleocharis palustris	Creeping spikerush	1	
Hordeum jubatum	Foxtail barley	1	
Kochia scoparia	Kochia		3
Lactuca serriola	Wild lettuce		2
Melilotus officionale	Yellow sweet clover	1	
Pascopyron smithii	Western wheat	+	
Polypogon monspeliensis	Rabbitsfoot grass		2
Phragmites australis	Common reed	1	
Puccinellia airoides	Alkaligrass	+	2
Rumex crispus	Curly dock	+	
Salix exigua	Sandbar willow	1	
Salsola kali	Russian thistle		1
Scirpus maritimus	Alkali bulrush	1	1
Scirpus americanus	American 4-square	+	
Scirpus acutus	Hard stem bulrush	3	1
Tamarix ramosissima	Saltcedar	2	2
Typha latifolia	Cattail	3	2

[a]Cover Classes: (+) <1%, (1) 1-5%, (2) 5-25%, (3) 25-50%, (4) 50-75%, and (5) 75-100%.

Brief descriptions of the floodplain and wetland plant associations follow:

Saltcedar/Kochia. This association is the dominant vegetation on the millsite floodplain. It consists primarily of dense stands of saltcedar, an exotic shrub or tree that has taken over most low-elevation riparian areas in the San Juan and Colorado River basins. Only a few native cottonwood and even fewer native willows remain in the millsite floodplain. Grasses and forbs form the understory of the saltcedar thickets. The relatively high abundance of kochia in the understory reflects a history of heavy grazing. This association is divided into two mapping units in Figure 4–76: Saltcedar/Kochia and Young Saltcedar.

Saltcedar/Barren. This association occupies a small area just west of the mouth of Bob Lee Wash on the west part of the floodplain. The structure of the community was different from other saltcedar associations; it consists of widely spaced, mature saltcedar interspersed with large bare patches and some inland saltgrass patches.

Rabbitbrush/Kochia. The large area of green rabbitbrush in the central portion of the millsite floodplain, with an understory dominated by kochia and Russian thistle, reflects a history of disturbance. Some remnants of alkali sacaton and giant dropseed populations were observed. Large bare patches were also observed in the association.

Saltcedar/Saltgrass. This association borders the wetland area at the mouth of Bob Lee Wash. The area is dominated by extensive mats of saltgrass. A few mature saltcedar dot the area. The presence of saltgrass indicates a water table very close to the ground surface.

Giant dropseed/Indian ricegrass. This association is found on sandy soils and often on stabilized sand dunes. The dominance of giant dropseed suggests a history of moderate grazing. Giant dropseed and its close cousins, sand dropseed and spike dropseed, tend to increase under moderate grazing where more palatable grasses have been killed. The dropseed grasses will also decrease under heavy grazing pressure.

Wetland. The 5-acre wetland on the millsite floodplain is an artifact of drainage from artesian well 648. If the well was not free flowing, the wetland would not occur. The wetland is dominated by cattails and bulrushes.

4.6.2 Sampling for Chemical Analysis

Field sampling for chemical analyses of samples was conducted in September 1998, June 1999, September 1999, March 2000, and June 2000. Data from analyses of the 1998, 1999, and March 2000 samples are presented in CD-ROM format in Appendix G. Data from the June 2000 sampling are not currently available. This sampling event focused on sampling of range grasses for the purpose of evaluating risk to livestock. Results and evaluation of these data will be presented in the EA for the Shiprock site.

This section discusses sampling methods and rationale for September 1998 through March 2000 sampling events. The results are discussed with regard to potential ecological risk in Section 6.2. The following is a list of sampling locations, types of media sampled, and sampling dates:

Location	Media	Dates
Millsite Floodplain	Vegetation	1998, 1999
Reference Floodplain	Vegetation	1998, 1999
Millsite Wetland	Surface Water	1998
	Sediment	1998
	Vegetation	1998, 1999
Reference Wetland	Surface Water	1998
	Sediment	1998, 1999
	Vegetation	1998, 1999
Disposal Cell Terrace	Vegetation	1998, 1999
Reference Terrace	Vegetation	1998, 1999
San Juan River	Surface Water	2000
Millsite Floodplain Seeps	Surface Water	2000
Distributary Channel Pool	Surface Water	2000

Appropriate reference areas for wetlands and terrace habitat were difficult to find. The ditch containing outflow from well 648 was the only wetland area in the vicinity of the Shiprock site that was not influenced by the contaminant plume. Small reference areas for greasewood, a terrace phreatophyte, were found east of the site at an elevation similar to the elevation of the disposal cell terrace and on the reference floodplain in the vicinity of wells 850 through 852 (Plate 1).

All ecological sampling locations are shown in Figure 4–77 and Plate 5. Location numbers and location abbreviations (in parentheses) for the 1998 and 1999 sampling rounds were identified in field books and chain of custody (CoC) forms as follows: (Note. Sample identification numbers 1248 through 1250 were not used. The location abbreviations in field books and CoC forms for sample locations 1280 through 1284 and 1285 through 1287 were changed from HSE to ECA and from HSW to WCA, respectively.)

Location 1236 (Seep 426)—Millsite floodplain wetland (FPW)
Location 1237 (Seep 425)—Millsite floodplain wetland (FPW)
Locations 1238 through 1243—Millsite floodplain wetland (FPW)
Location 1244—Bob Lee Wash/Millsite floodplain wetland (FPW)
Locations 1245 through 1247—Reference wetland (east of well 648) (REFW)
Locations 1251 through 1253—Repository (terrestrial) terrace (TT)
Locations 1254 through 1256—Reference terrace (RT)
Locations 1257 through 1259—Millsite floodplain (terrestrial) (FPT)
Locations 1260 through 1262—Reference floodplain around wells 850 through 852 (FPR)
Locations 1263 through 1265—Reference wetland (east of well 648—1999) (REFW)
Locations 1266 through 1273—Reference floodplain near wells 850 through 852—1999 (FPR)
Locations 1274 through 1276—Repository (terrestrial) terrace—1999 (TT)
Locations 1277 through 1279—Reference floodplain used in place of the 1998 reference terrace locations for greasewood collection—1999 (RT)
Locations 1280 through 1284—East Contaminated Area on floodplain near well 854 (ECA)
Locations 1285 through 1287—West Contaminated Area—floodplain near well 856 (WCA)
Locations 1288 through 1292—1st Wash (west of U.S. Highway 666)—1999 (FW)

Location numbers for the March 2000 sampling round were as follows:

Locations 1200 through 1206 and location 1210—San Juan River
Locations 1207 and 1212—Millsite floodplain wetland below seep 425
Locations 1208, 1209, and 1213—Bob Lee Wash/Millsite floodplain wetland (new seep)
Location 1211—Distributary channel pool below 1st Wash (west of U.S. Highway 666)

Field sampling locations for all 1998 and 2000 samples in Figure 4–77 and Plate 5 were established by a Garmin GPS III global positioning system and were converted into state plane coordinates. Sample locations for 1999 were estimated from existing maps, previous sampling locations, or monitor wells.

4.6.2.1 Surface Water and Sediment Sampling Methods

Surface water and sediment samples were collected from September 2 through September 4, 1998, at the millsite floodplain wetland and the upgradient reference wetland. Nine co-located samples of sediment and surface water were collected at the millsite floodplain wetland, but only three samples were collected at the reference wetland, in part because of the small area. In June 1999, three additional sediment samples were collected at the reference wetland. The purpose of the additional samples was to pool both the 1998 and 1999 data sets for improved statistical power based on a larger sample size. Sampling done during September 1999 focused mainly on the area west of U.S. Highway 666. On March 14, 2000, 14 additional surface water

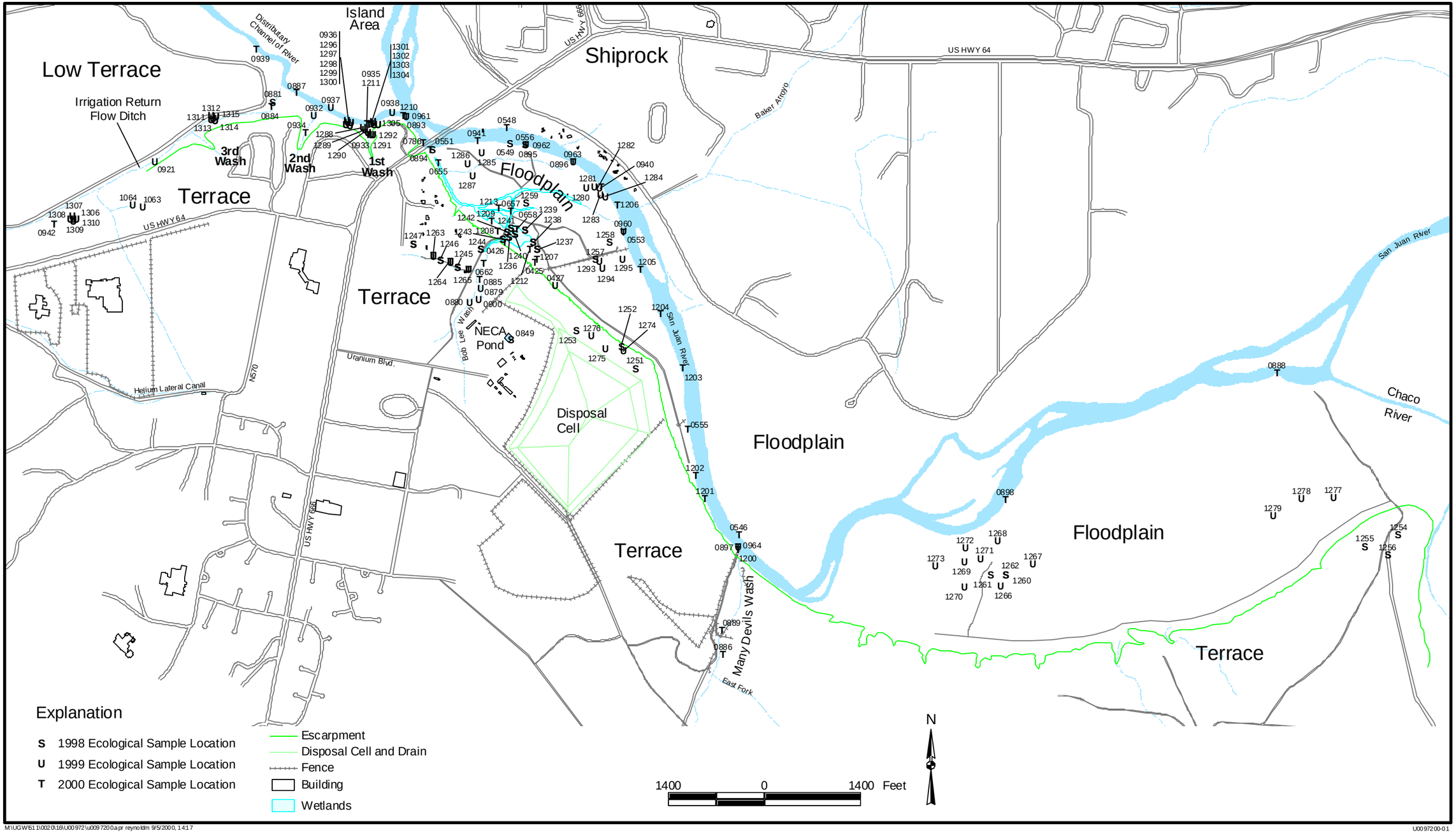


Figure 4-77. Surface Water, Sediment, and Vegetation Sample Locations for the Ecological Risk Assessment

samples were collected at the millsite floodplain, the distributary channel, and the San Juan River to supplement the existing data record. Locations and analytes for these samples were based on comments received on February 29, 2000, from the USFWS. These sample results will provide an improved evaluation of potential risks to protected fish and wildlife species that may occur in the Shiprock area and in the San Juan River.

Consistent with EPA guidance (EPA 1989b), the number of samples satisfied a coefficient of variation (CV) of 15, a minimum detectable relative difference (MDRD) between 10 and 20 percent, a confidence of 80 (Type I error, false positive), and a power of 90 (Type II error, false negative). These values are based on a 1-sided, single sample distribution. Other factors considered in the selection of sample size were the small areal extent of the affected sites and the amount of sample material available for collection. The surface water samples were grab samples; sediment samples were from a nominal depth of 0 to 6 in. below the sediment surface. Surface water sample collection preceded sediment and biota tissue collection. All surface water and sediment sampling containers were certified as precleaned from an industrial supplier.

Surface Water Methods

Both filtered and unfiltered surface water samples were collected at the same locations as the sediment samples for the 1998 sampling season. Surface water samples associated with the 1998 ecological sampling locations were not collected in 1999. The 2000 surface water sampling did not include co-located sediment samples, and the locations are independent of the 1998 and 1999 sampling locations. The filtered sample represents the soluble component for aquatic receptors, and the unfiltered sample represents surface water ingested by terrestrial receptors. Filtered surface water samples were identified with an “F” suffix on the sample identification number, and unfiltered samples received a “U” (unfiltered) suffix. Each sample bottle was first rinsed with the surface water; the rinse water was then discarded prior to sample collection. A sample was collected by immersing the bottle just below the water surface and filling to just below the mouth of the bottle. Samples were then filtered using a 0.45-µm filter and acidified accordingly. Table 4–36 provides a summary of analytes, preservatives, containers, and other information pertaining to surface water sample collection for 1998. The 2000 surface water sample analyses also included ammonium, molybdenum, vanadium, zinc, and gross alpha and gross beta; however, they did not include antimony, arsenic, magnesium, sodium, strontium, Ra-226, and Th-230 from the 1998 analyte list.

Sample labels showing the date, time, location, laboratory bar code, sampler, analyses requested, preservatives, and comments were applied to each container and secured with clear plastic tape. All sample containers were placed in coolers containing ice for transport to the GJO Analytical Chemistry Laboratory. A CoC form was completed for all samples, and a CoC label was placed over each cooler. All samples were maintained under strict CoC.

Table 4–36. Summary of Surface Water Sampling Parameters—1998[a]

Analyte	Preservative	Container	Holding Time	Method
Antimony	HNO_3; pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPMS
Arsenic				ICPAES
Magnesium				ICPAES
Manganese				ICPAES
Ra-226				AS
Selenium				ICPAES
Sodium				ICPAES
Strontium				ICPAES
Th-230				ICPMSFIA
Uranium				ICPMS
Nitrate	H_2SO_4; pH <2; cool 4 °C	125 mL HDPE	28 days	IC
Sulfate	cool 4 °C only	125 mL HDPE	28 days	IC

[a]HDPE– high-density polyethylene.
H_2SO_4–sulfuric acid.
HNO_3–nitric acid.
mL–milliliter.
C–centigrade or Celsius.
AS – Alpha spectrometry.
ICPMS–Inductively coupled plasma-mass spectrometry.
ICPMSFIA–Inductively coupled plasma-mass spectrometry-flow injection analysis.
ICPAES–Inductively coupled plasma-atomic emission spectroscopy.
IC–Ion chromatography.

Sediment Methods

Each sediment sample represented a composite of three or four locations where vegetation material was present. The area for collection was typically a circle with a radius of less than 5 ft. Excess organic matter and larger rocks and pebbles were removed from the sample prior to compositing. The contents of one stainless-steel auger (i.e., one subsample) was collected at each composite location and placed in a large stainless steel mixing pan. All subsamples were mixed thoroughly with a stainless steel spoon prior to removing approximately 4 ounces (114 g) of material for metals analysis. In addition, a 125-mL high-density polyethylene (HDPE) bottle was collected for nitrate and another for sulfate in the 1998 sampling activities. A separate bottle for sulfate and another bottle for both nitrate and ammonia were used in the 1999 field collection. Table 4–37 provides a summary of analytes, preservatives, containers, and other information pertaining to sediment sample collection.

Sample labels were applied to each container and secured with clear plastic tape. All sample containers were placed in coolers containing ice for transport to the GJO Analytical Chemistry Laboratory. A CoC form was completed for all samples and a CoC label placed over each cooler. All samples were maintained under strict CoC. The analytical method for the sediment samples included a complete acid digestion rather than an acid leach as was used in some previous sediment sampling efforts.

Table 4–37. Summary of Sediment Sampling Parameters-1998 and 1999

Analyte	Preservative	Container	Holding Time	Method
Antimony	cool 4 °C	4 ounce amber glass	6 months	ICPMS
Arsenic				ICPAES
Magnesium				ICPAES
Manganese				ICPAES
Ra-226				AS
Selenium				ICPAES
Sodium (1998 only)				ICPAES
Strontium				ICPAES
Th-230				ICPMSFIA
Uranium				ICPMS
Nitrate		125 mL HDPE	28 days	IC
Sulfate		125 mL HDPE	28 days	IC
Ammonia as NH_4 (1999 only)		125 ml HDPE	28 days	SPEC

HDPE–high-density polyethylene.
mL–milliliter.
C–centigrade or Celsius.
ICPMS–Inductively coupled plasma-mass spectrometry.
ICPMSFIA– Inductively coupled plasma-mass spectrometry-flow injection analysis.
ICPAES–Inductively coupled plasma-atomic emission spectroscopy.
IC–Ion chromatography.
SPEC–Spectroscopy.

Quality Control Samples

Field blanks and equipment rinses were collected at only the millsite floodplain wetland (1998). These samples consisted of distilled, deionized water appropriately preserved and cooled in the field. The field blank was prepared by pouring distilled, deionized water directly from the carboy into the appropriate sampling bottle and preserving as necessary. The equipment rinse consisted of pouring distilled deionized water from the carboy over the cleaned sampling equipment (auger, sampling pan, shears, and spoons) and collecting the rinsate in the appropriate sampling containers and preserving and cooling as necessary. Because of the closeness of the millsite and reference wetlands and the small number of samples, no additional equipment rinse and field blank samples were collected.

A field-duplicate surface water sample was collected at the Shiprock wetland location 1243. The field duplicate was identified with “D” suffix appended to the sample identification number. No duplicate samples were collected in June 1999.

4.6.2.2 Plant Tissue Sampling Methods

Vegetation samples collected in 1998 consisted of cattails, bulrush, cottonwood, and greasewood. The 1999 samples consisted of cattail, greasewood, cottonwood, and Russian olive plant tissues. Each sample consisted of material composited from an area around the designated sample location. Cattail and bulrush samples were collected at the same locations as sediment and surface water samples. No sediment or surface water samples were collected at greasewood, cottonwood, or Russian olive locations because no water was present.

Co-located Vegetation Samples

Vegetation samples consisting of cattails and bulrush were collected at both the millsite and reference wetland locations in 1998. During 1999, cattails were collected again at both locations. These samples were co-located with the surface water and sediment samples (1998) and sediment-only samples (1999).

Samples were collected by digging up an entire plant or cluster of plants with a stainless steel shovel. Excess sediment was rinsed off the plants prior to separating the roots and stems. Stems and roots were processed with pruning shears with stainless steel and polyethylene cutting edges. The roots and stems were rinsed thoroughly with sample water, followed by tap and distilled deionized water rinses, until rinsates contained no visible soil or sand particles. All plant materials received a final distilled, deionized water rinse prior to bagging. Stems and roots were composited separately. Stems and roots were double-bagged in clean zip-lock type storage bags. Sample labels were applied to each outermost zip-lock type bag and secured with clear plastic tape. All samples were kept in coolers containing ice for transport to the GJO Analytical Chemistry Laboratory. A CoC form was completed for all samples, and a CoC label was placed over each cooler. All samples were maintained under strict CoC. Samples that could not be processed directly at the laboratory by freeze drying were placed in freezers at 4 °C. Table 4–38 provides a summary of analytes, preservatives, containers, and other information pertaining to biota tissue collection.

Table 4–38. Summary of Biota Sampling Parameters (1998 and 1999)

Analyte	Matrix	Preservative	Container	Holding Time	Method[a]
Antimony	cattail, bulrush, cottonwood, greasewood, Russian olive	cool 4 °C	double 1-gal zip-lock type bags	6 months	ICPMS
Arsenic					ICPAES
Magnesium					ICPAES
Manganese					ICPAES
Ra-226					AS
Selenium					ICPAES
Sodium					ICPAES
Strontium					ICPAES
Th-230					ICPMSFIA
Uranium					ICPMS

[a]AS – Alpha spectrometry.
ICPMS–Inductively coupled plasma- mass spectrometry.
ICPMSFIA–Inductively coupled plasma-mass spectrometry-flow injection analysis.
ICPAES–Inductively coupled plasma-atomic emission spectroscopy.

Cottonwood, Greasewood, and Russian Olive Samples

Six composite greasewood samples consisting primarily of leaves and small stems were collected at the disposal cell terrace in 1998 and 1999. Three greasewood samples were also collected at the terrace reference area in 1998. In 1999, three additional greasewood samples were collected at the other floodplain reference location near wells 850 through 852. Samples were collected by randomly clipping both leaves and stems from three or four plants close together and placing them in zip-lock type bags. Samples were cleaned the same as co-located vegetation samples except that stems and leaves were not segregated but processed together.

Three cottonwood samples were collected at each of three general locations on the millsite floodplain and at the floodplain reference area in 1998 and in 1999. Sample collection and preparation followed the method used for greasewood samples. The areal extent of cottonwood sample collection was larger than for greasewood.

Samples of Russian olive consisting of stems and leaves were collected at the floodplain reference area near wells 850 through 852 and in an area in the north part of the floodplain near well 856 that overlies contaminated ground water. Sample collection and preparation followed the method used for greasewood samples.

Vegetation Sampling Locations and Species

Location numbers and location abbreviations (in parentheses) where plant tissue samples were collected were identified in field books and CoC forms as follows:

Locations 1236 through 1237—Millsite floodplain wetland (FPW)—bulrush
Locations 1238 through 1243—Millsite floodplain wetland (FPW)—cattail
Location 1244—Bob Lee Wash/Millsite floodplain wetland (FPW)—cattail
Locations 1245 through 1247—Reference wetland (REFW)—both bulrush and cattail
Locations 1251 through 1253 and 1253 composite—Terrace terrestrial (TT)—greasewood
Locations 1254 through 1256—Reference terrace (RT)—greasewood
Locations 1257 through 1259—Floodplain terrestrial (FPT)—cottonwood
Locations 1260 through 1262—Floodplain reference (FPR)—cottonwood
Locations 1263 through 1265—Reference wetland (REFW)—cattail only (1999)
Locations 1266 through 1268—Floodplain terrestrial (FPT)—cottonwood (1999)
Locations 1269 through 1273—Floodplain terrestrial (FPT)—Russian olive (1999)
Locations 1274 through 1276—Terrace terrestrial (TT)—greasewood (1999)
Locations 1277 through 1279—Reference terrace (RT)—greasewood on floodplain (1999)
Locations 1280 through 1284—East Contaminated Area on floodplain (ECA)—greasewood (1999)
Locations 1285 through 1287—West Contaminated Area on floodplain (WCA)—Russian olive (1999)
Locations 1288 through 1292—1st wash west of U.S. Highway 666

Cattail samples were uniquely identified by adding an “R” (root) or “S” (stem) suffix to each sample identification. All roots for the same sample identification and field location number were processed as one sample. All stem material for each sample identification and field location number was also processed as one sample. Similarly, all sample bags of greasewood and cottonwood with the same laboratory identification number were processed as a single sample.

Quality Control Samples

A field duplicate cattail sample was collected at Shiprock floodplain wetland location 1243. The field duplicate was identified with a “D” suffix appended to the sample identification number. The equipment rinsate and field duplicate applied to the biota collection as well. No additional quality control samples were collected in 1999.

End of current text

5.0 Site Conceptual Model

This section describes the main physical and chemical characteristics and features of the Shiprock site from a multidisciplinary perspective. Two block diagrams illustrate the hydrological, geochemical, and ecological components of the site conceptual model. The purpose of the block diagrams is to graphically illustrate the site so that it can be described and analyzed more readily. The important physical aspects controlling the movement of water and dissolved contamination are identified, together with ecological systems that are affected by the water.

Figure 5–1 is a block diagram of the entire area affected by the Shiprock site. It illustrates the important physiographic features that define the landscape. Two general areas are illustrated: the terrace and the floodplain. Upland areas south of the disposal site and an escarpment north of the disposal site bound the terrace. The floodplain area is bounded by the escarpment and the San Juan River.

Terrace ground water is hypothesized to be anthropogenic because natural rates of recharge in the terrace are probably insufficient to sustain a water table in the terrace system, which is composed of alluvial sand and gravel and weathered Mancos Shale. Flow modeling sensitivity analyses suggest that a natural recharge rate of at least 10 percent of precipitation is required to create a water table in the Mancos Shale. This is not considered an abnormally high value in most environments; however, in Shiprock the silt loess that covers the gravel is more prone to creating runoff rather than recharge in its natural condition. Wells drilled in a geologic environment thought to be analogous to the one that existed near the disposal cell 60 years ago, located on a broad terrace 2 mi east of the existing disposal cell, indicate that the terrace gravel system is dry and that the Mancos Shale is also dry.

The creation of a terrace ground water system is probably tied to multiple events, including (1) pumping of San Juan River water to the terrace for processing uranium ore at the Shiprock mill, (2) pumping of San Juan River water to the terrace for production and processing of helium at the former Navajo (helium) Plant, (3) slurrying of tailings to the disposal cell during stabilization of the disposal cell, (4) siphoning of San Juan River water to the terrace for irrigation through the Helium Lateral Canal, (5) pumping of San Juan River water to the terrace for municipal water supply, and (6) potential leaks from water supply and sewerage lines. Because of the emergence of so many potential sources of ground water during the past 60 years, it is unrealistic to expect that the entire terrace ground water system could revert back to its pre-1940 conditions.

The terrace ground water is partly contaminated with residual radioactive material (RRM). The highest concentrations of uranium, nitrate, and sulfate occur close to the UMTRA Project disposal cell. These concentrations decrease west and northwest along the direction of plume transport. Most of the RRM-contaminated ground water discharges to the escarpment area east of 3rd Wash (Plate 1). The escarpment thins gradually west of 3rd Wash, and the ground water derived from the various other sources of recharge returns to the San Juan River and its distributary channel through the gap formed between 3rd Wash and the western edge of the terrace system. Ground water from some of the wells west of the edge of the area affected by irrigation from the Helium Lateral Canal contain levels of uranium and selenium that slightly exceed MCLs. These portions of the terrace ground water system are not believed to contain RRM. Rather, the ground water is believed to contain low levels of these constituents resulting from chemical dissociations in the Mancos Shale.

Figure 5–2 presents a site conceptual model that shows the interactions between the terrace system and floodplain alluvial aquifer in the Shiprock area. The former tailings pile, raffinate ponds and current disposal cell are believed to be sources of the ground water contamination. The Shiprock site directly affects both ground water systems because of leaching of RRM from the milling process. Contamination from the milling is transported along ground water flow paths to surface exposure points in Bob Lee Wash and Many Devils Wash, to the wetlands in the floodplain alluvium, and to the San Juan River. Contamination extends from Many Devils Wash on the east to approximately 3rd Wash on the west.

Summaries of water-balance components of both the terrace ground water system and the floodplain alluvial aquifer are presented in Tables 4–5 and 4–8, respectively. In those tables, the alluvial flows were expressed in units of cubic feet per day, and the terrace ground water system flows were expressed in units of cubic feet per year. However, it is also possible to express these components with smaller numeric values. Consequently, the various flow components are now transformed into units of acre-feet per year in Figure 5–2. Table 5–1 is a key that lists the original values from Tables 4–5 and 4–8 and converts them to the desired units.

The terrace ground water system receives no natural recharge from the Mancos Shale; consequently, the southern boundary of the system is considered no-flow, and the remaining boundaries of the flow system are head-dependent flux boundaries. The terrace system receives recharge from internal sources such as areal recharge, the NECA gravel pit, the disposal cell, and irrigation water in the quantities shown on Table 5–1 and Figure 5–2. Contaminated ground water on the terrace is partly contained in a buried ancestral river channel where an estimated 38 million gallons of ground water is stored.

The terrace ground water system contains high uranium, nitrate, and sulfate concentrations near the former ore storage area and the processing site. Discharge from this section of the terrace ground water system flows toward Bob Lee Wash, where seepage and springs in the upper part of the wash deliver high uranium, sulfate, and nitrate concentrations to the surface. Because the contaminated ground water discharge in upper Bob Lee Wash is a potential risk to livestock, interim actions designed to eliminate possible exposure of these receptors to the water were implemented in summer 2000.

South of the disposal cell, the terrace ground water system contains mainly nitrate and sulfate contamination. Discharge from the eastern portion of the buried channel flows east toward Many Devils Wash along the top of a thin, eastward dipping, resistant siltstone bed in the Mancos Shale and in the weathered Mancos Shale immediately below, becoming oxidized as it passes near the NECA gravel pit. Where the ground water discharges into Many Devils Wash at a rate of approximately 0.48 acre-feet per year, a probable livestock-exposure point exists. Interim actions designed to cover the exposed water in this wash were also implemented in summer 2000.

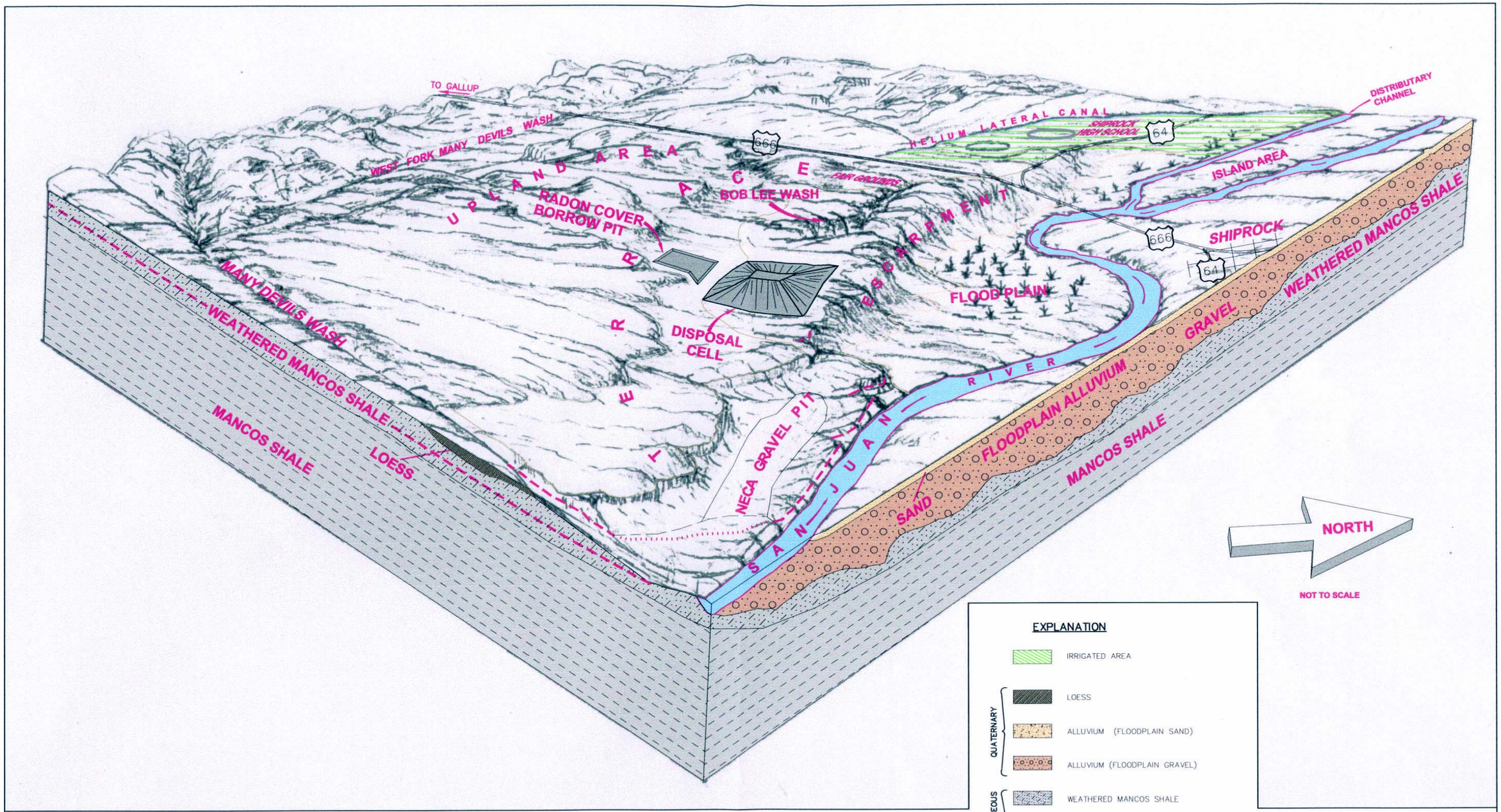


Figure 5–1. Physiographic Block Diagram of the Shiprock Site

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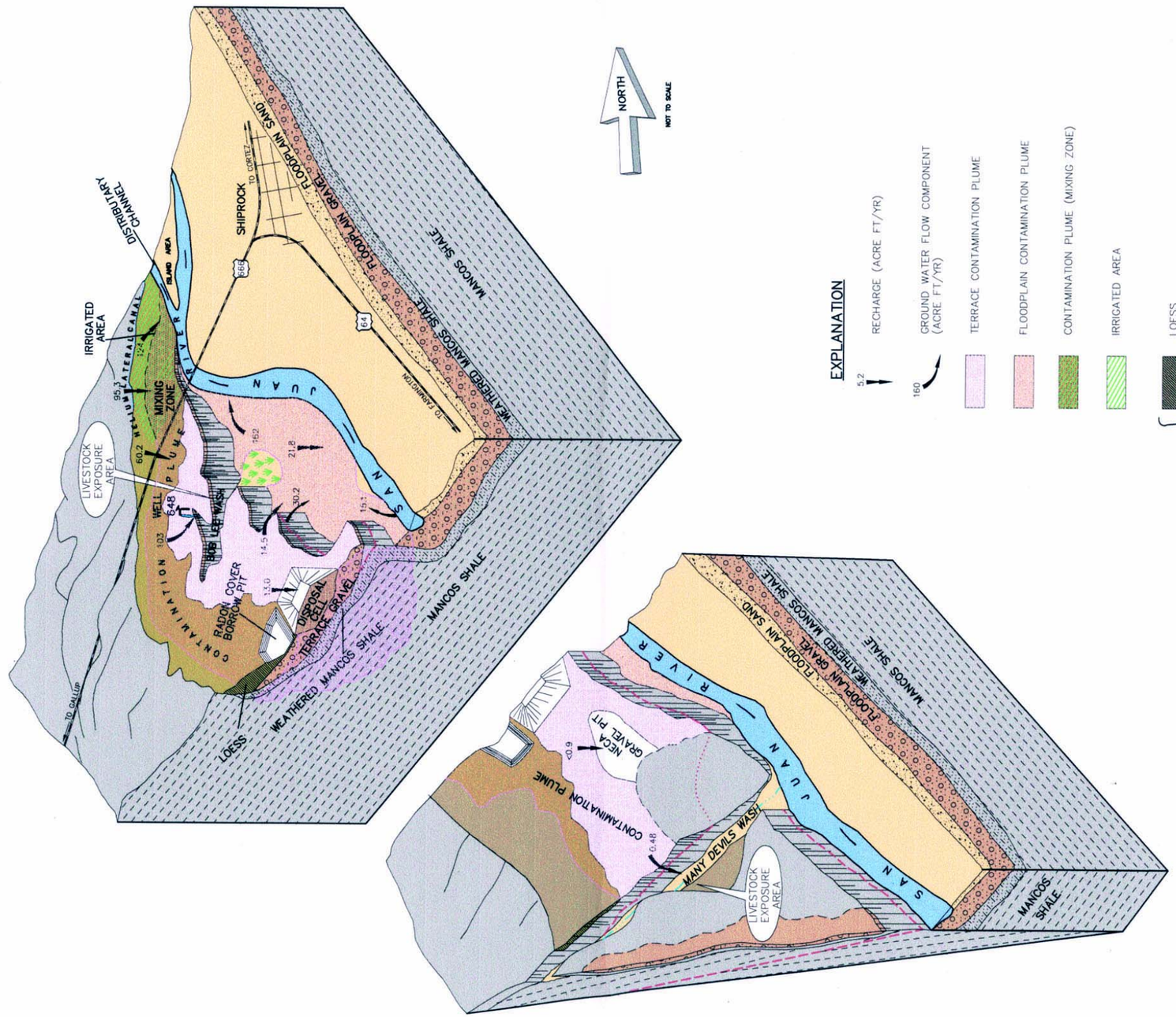

Figure 5–2. Site Conceptual Model of the Shiprock Site

Table 5–1. Estimated Magnitudes of Major Flow Components for Floodplain Alluvial Aquifer and Terrace Ground Water System at the Shiprock Site

Flow Component	Inflow	Inflow (acre-feet per year)	Outflow	Outflow (acre-feet per year)
Floodplain Alluvial Aquifer (See Table 4–5)	(ft^3/day)		(ft^3/day)	
Inflow from San Juan River	1,800	15.1		
Inflow of Recharge	2,600	21.8		
Inflow from Well 648	12,300	103		
Inflow from terrace Ground Water System via Mancos Shale	3,600	30.2		
Outflow to San Juan River	0	0	19,400	162
Subtotal: Floodplain Alluvial Flow Component	20,300	170	19,400	162
Terrace Ground Water System (See Table 4–8)	(ft^3/year)		(ft^3/year)	
Areal Infiltration of Precipitation and Runoff	2,620,000	60.2		
Infiltration of Water from NECA Gravel Pit	<<39,000	0.9		
Drainage of Residual Moisture from the Disposal Cell	568,000	13.0		
Infiltration of Irrigation Water	4,150,000	95.3		
Discharge to Escarpment through Springs			632,000	14.5
Discharge to Many Devils Wash			21,000	0.48
Discharge to Floodplain though Mancos Shale			1,324,000	30.4
Discharge to San Juan River			5,400,000	124
Subtotal: Terrace Ground Water System	7,380,000	169	7,380,000	169
Grand Total (rounded)		339	7,400,000	331

Note: The conversion from cubic feet per day to acre-feet per year is accomplished by multiplying by 0.0084. The conversion from cubic feet per year to acre-feet per year is obtained by dividing by 43,560.

Ground water in the western part of the buried channel system flows west to northwest toward U.S. Highway 666. This area is characterized by declining contaminant concentrations farther northwest where the ground water mixes with the irrigation water from the Helium Lateral Canal. Irrigation water in the mixing zone infiltrates and discharges to the San Juan River either through the irrigation return flow ditch or through ground water discharge directly to the river. Irrigation water is in contact with weathered Mancos Shale and probably leaches sulfate, uranium, and selenium from this bedrock (see Section 4.4.7). Contaminant concentrations exceed MCLs in ground water west of U.S. Highway 666 and, on the basis of sampling results, appear to discharge into the distributary channel. The zone of discharge of the contaminated ground water extends west to approximately 3rd Wash. The ecological risk associated with the ground water discharge to the distributary channel area is evaluated in Section 6.2.

The floodplain alluvial aquifer is bounded by the escarpment along its southern margin and by the San Juan River along its northern margin. The main sources of recharge to the floodplain alluvial aquifer are (1) 103 acre-feet per year from flowing well 648, (2) 15.1 acre-feet per year from the San Juan River, (3) 21.8 acre-feet per year from infiltration of precipitation and runoff, and (4) 30.2 acre-feet per year from ground water discharge off the terrace. Seepage from the

terrace system is a potential source of degradation for the alluvial aquifer. Because the concentrations of COCs next to the escarpment have not changed appreciably over time it is believed that the terrace continues to drain RRM-laden fluids into the floodplain alluvial aquifer. Transport modeling suggests that the floodplain alluvial aquifer would flush readily if the source were removed (DOE 1999g). The water that transports contamination to the terrace and alluvial flow systems is speculated to be drainage of residual moisture from the disposal cell.

The northern margin of the floodplain alluvial aquifer is where the ground water discharges into the San Juan River. The cumulative discharge from the alluvial aquifer is approximately 162 acre-feet per year. Over much of the discharge reach, the ground water is dominantly composed of the 103 acre-foot component from the flowing artesian well 648. Ground water with mill-related contaminants enters the San Juan River upstream of the well-648 discharge reach and, to a much lesser degree, at the extreme downgradient corner of the floodplain alluvial aquifer.

Potential for human exposure to contaminated ground water is greatest where ground water surfaces in seeps and washes. Currently the ground water in the terrace or floodplain is not being extracted for use. However, exposure could occur if water use changes in the future. A characterization of human health risks is evaluated in Section 6.1.

Potential ecological risk owing to ground water contamination occurs in several locations. Ground water discharges to the surface in the seeps, washes, and on the floodplain. Receptors could be exposed to the surface water directly. Also, water flowing across sediments and soils could contribute contamination to these media via adsorption and be taken in by birds and animals that ingest sediment or by burrowing organisms. Plants could take up contaminants from sediment. Deep rooted plants could take up contaminants directly from ground water. Foraging on contaminated plants or preying on contaminated organisms can further propagate contamination through the food chain. Potential ecological risks and a more detailed ecological conceptual model are presented in Section 6.2.

6.0 Baseline Risk Assessment

6.1 Human Health Risks

A BLRA was previously prepared for the Shiprock Site (DOE 1994). Most of the methodology used in that risk assessment followed standard EPA risk assessment protocol (EPA 1989b), though the BLRA did not calculate potential risks for noncarcinogenic constituents. Instead, calculated exposure intakes were compared with a range of contaminant doses associated with various adverse effects. Data used in that report were collected from 1988 to 1993. Since that time, additional data have been collected to more completely characterize the site and to represent more recent site conditions. Updated and revised toxicological data are also available for some site-related constituents. These new data were used to reevaluate COPC identification and assessment of associated risks.

6.1.1 Summary of 1994 BLRA Methodology and Results

As described in previous sections, two different surficial hydrogeologic units are recognized in the vicinity of the Shiprock site—floodplain alluvium and the terrace system consisting of alluvial material and weathered Mancos Shale. While there is likely some contribution of ground water from the terrace system to the floodplain aquifer, these two systems were considered different enough to be evaluated separately in the original BLRA. One of the major distinctions between these two systems is the source of recharge. The floodplain alluvium is a natural aquifer and is recharged primarily from the San Juan River. Ground water from alluvial deposits located on a floodplain upstream from the Shiprock site was sampled to represent the quality of floodplain ground water that existed before milling activities began.

Conversely, it is probable that the terrace ground water system did not exist before the start of milling activities at the Shiprock site and was formed primarily because of discharge of milling-related fluids. Continued recharge to the terrace alluvium is largely from man-made sources (e.g., irrigation and septic systems). Because no pre-millsite terrace ground water likely existed, no background water quality data are available to serve as a baseline in the evaluation of site-related adverse affects and the development of an appropriate compliance strategy for that system.

In addition to ground water from the two systems, the 1994 BLRA also evaluated potential risks associated with direct and indirect exposure to surface water on the floodplain that is contaminated through discharge of ground water to the surface. The following sections provide summaries of the potential risks associated with exposure to ground water in these three different situations, as determined in the 1994 BLRA.

6.1.1.1 Floodplain Ground Water

The 1994 BLRA identified 19 constituents associated with the floodplain aquifer at the Shiprock site as being present at levels statistically above background concentrations for the area. This initial list was screened first to eliminate constituents with concentrations within nutritional ranges and then to eliminate contaminants of low toxicity and high dietary ranges. These two steps eliminated five and three constituents, respectively, resulting in the following COPC list: antimony, arsenic, cadmium, magnesium, manganese, nitrate, selenium, sodium, strontium, sulfate, and uranium. These 11 contaminants were retained for further risk analysis.

A number of potential routes of exposure were evaluated: ingestion of ground water as drinking water in a residential setting, dermal contact with ground water while bathing, and ingestion of garden produce irrigated with ground water. Ingestion of meat and milk from ground water–fed livestock was also considered; however, nitrate and sulfate concentrations in floodplain ground water were so high that livestock could not survive chronic ground water exposure. Therefore, this exposure route was considered not viable and was eliminated from further consideration from a human health perspective. The nitrate and sulfate concentrations ***do*** constitute a real and current risk to livestock in the area even though it is not a significant pathway for human health. Results of the exposure assessment indicated that intakes for all constituents were negligible from exposure routes other than drinking water. Therefore, only exposure through ingestion of ground water as drinking water was retained for more detailed evaluation. Both infants and adults were considered as likely receptors.

Calculated exposure intakes were presented along with contaminant intakes associated with a range of adverse health effects. Potential risks associated with exposure to noncarcinogenic constituents were discussed in a qualitative fashion; carcinogenic risks were quantified and compared to EPA's acceptable risk range of 1×10^{-4} to 1×10^{-6}.

For the noncarcinogenic contaminants nitrate and sulfate, the most sensitive receptor population is infants. Results of the BLRA showed that the most significant health risk is associated with nitrate. If ground water was used for drinking water, the possible exposure greatly exceeds the potential lethal level for infants. Sulfate concentrations were also well above the range expected to result in severe diarrhea or death because of dehydration in infants.

Adult exposure intakes were evaluated for the other noncarcinogenic contaminants. Cadmium and strontium intakes were below oral reference doses (RfDs) established by EPA. Estimated intakes of uranium greatly exceeded its RfD; arsenic, antimony, selenium, and manganese intakes also exceeded their respective RfDs, but to a lesser degree. It was noted, however, that most of the RfDs were established at levels well below those shown to demonstrate actual adverse effects. Exposure estimates for sodium are 3 times greater than the National Research Council's recommended intake and could result in hypertension; magnesium intakes have been shown to be associated with diarrhea in adults, though toxicity data related to more severe effects are unavailable.

Carcinogenic risks calculated for adult exposure to uranium and arsenic both exceeded the upper bound of EPA's acceptable risk range of 1×10^{-4} by approximately 1 order of magnitude.

6.1.1.2 Terrace Ground Water

Because of the lack of background ground water quality data for the terrace alluvium, no statistical comparison could be performed to determine COPCs for terrace alluvial ground water. Instead, COPCs were selected based on their clear association with uranium milling activities and their elevated concentrations with respect to regional waters. Three constituents were evaluated as COPCs—nitrate, sulfate, and uranium. Only the ground water ingestion pathway was considered; infants and adults were evaluated as potential receptors.

Exposure intakes of both nitrate and sulfate exceeded potentially lethal levels for infants. Adult intakes of uranium exceeded the EPA RfD. Carcinogenic risks associated with exposure to uranium are within EPA's acceptable risk range.

6.1.1.3 Floodplain Surface Water

Several pathways were considered likely for exposure to surface water on the floodplain. Sediment ingestion, incidental water ingestion, and dermal absorption of contaminants from surface water were evaluated for children. It was assumed that they would contact the contaminated media during play on the floodplain. Exposure to adults was considered via ingestion of meat and milk obtained from livestock that were watered with contaminated surface water and grazed on contaminated pasture grasses. The contaminants evaluated for the floodplain surface water included selenium, strontium, and uranium.

Exposure intakes calculated for all pathways for noncarcinogenic contaminants were below levels at which adverse health effects would be expected. Carcinogenic risks associated with exposure to uranium were below even the lower bound of EPA's acceptable risk range.

6.1.2 BLRA Update

The 1994 BLRA considered several potential routes of exposure to contaminants and eliminated all but one, ingestion of ground water in a residential setting, as insignificant. However, for the purposes of this update, several other pathways are considered. These include exposure to ammonia through inhalation (see discussion in Section 6.1.2.1) and ingestion of meat and milk from cows grazing in the contaminated areas. Risks from incidental exposure to surface water are presented as well. Several additional COPCs are evaluated for the terrace system compared to the three in the earlier BLRA. All ground water risks discussed in this document are hypothetical; neither floodplain nor the terrace ground water is currently being used for any purpose, and no grazing currently takes place in the contaminated areas. The only potentially complete pathways are for exposure to surface water in the washes and at seeps. Interim actions were completed in summer 2000 to eliminate these exposures (see Section 7.7). Therefore, this assessment concerns mostly potential risks that could exist in the future if land and water use changes.

Risk calculations presented here follow EPA's "Risk Assessment Guidance for Superfund Methodology" (EPA 1989b), which involves determining a point estimate for excess cancer risk from current or potential carcinogenic exposures (risk is equal to lifetime intake times cancer slope factor) and a hazard quotient (HQ) for noncarcinogenic exposures (HQ is equal to exposure intake divided by reference dose). EPA's acceptable carcinogenic risk range is 1×10^{-6} to 1×10^{-4}, which is an excess cancer risk of 1 in 1,000,000 to 1 in 10,000 compared to the general population. Risks exceeding this range are potentially unacceptable. For noncarcinogenic exposures, an HQ exceeding 1 is potentially unacceptable. HQs from multiple contaminants and/or pathways are often summed to estimate cumulative noncarcinogenic risks; these summed HQs are referred to as a hazard index (HI). HIs greater than 1 also represent generally unacceptable exposures. Therefore, it is possible for a number of individual contaminants to each have "acceptable" HQs of less than 1 that, when summed, represent a potentially unacceptable cumulative risk. Figure 6–1 provides exposure intake equations and default assumptions used in ground water and surface water intake calculations for this BLRA update.

Residential Exposure Scenario – Ground Water Ingestion

Chemicals: Intake (chronic daily in mg/kg-d) = (Cw * IRw *EF * ED)/(BW * AT)

Radionuclides: Intake (lifetime in picocuries) = Cw * IRw * EF * ED

Where

Cw	=	contaminant concentration in water
Irw	=	ingestion rate for water (2 liters per day default for adults; 0.64 liter per day for infants)
EF	=	exposure frequency (350 days per year)
ED	=	exposure duration (7 years for adults and 1 year for infants for noncarcinogens; 50 years for carcinogens)
BW	=	body weight (70 kilograms for adults; 4 kilograms for infants)
AT	=	averaging time (365 days * ED for noncarcinogens; 365 days * 70 years for carcinogens)

Incidental Exposure Scenario – Surface Water

Ingestion of Chemicals: Intake (chronic daily in milligrams per kilogram per day) = (Cw * IRw *EF * ED)/(BW * AT)
Absorption of Chemical: Intake (mg/kg-d) = (Cw*SA*PC*ET*EF*ED*CF)/(BW*AT)

Where

Cw	=	contaminant concentration in water
Irw	=	ingestion rate for water (0.05 liter per day for children aged 6–12 years)
EF	=	exposure frequency (3 months per year at 7 days per week = 90 days plus 3 months per year on weekends = 24 days; total = 114 days per year)
ED	=	exposure duration (7 years for children aged 6–12 years playing on floodplain)
ET	=	exposure time (1 hour per day)
BW	=	body weight (38.3 kilograms for children aged 6–12 years)
AT	=	averaging time (365 days * ED for noncarcinogens; 365 days * 70 years for carcinogens)
SA	=	skin surface area available for contact (497 cm^2 body surface area for children 6–12 years old)
PC	=	dermal permeability constant (0.001 cm/h; same rate as water)
CF	=	volumetric conversion factor for water (1 L/1000 cm^3)

Figure 6–1. Ground Water and Surface Water Exposure Intake Equations and Default Assumptions

Toxicological values used to estimate risks (RFDs and slope factors) are conservative values with uncertainty factors built in to be protective of sensitive populations. Therefore, risks presented here are worst-case estimates and are quite likely much higher than those that actually could exist.

In this update, which uses point-exposure doses, single values are used for each parameter required in the risk calculations. Calculations to determine contaminant intakes use standard exposure factors for the adult population (EPA 1989a). The ground water and surface water data used to assess risks in this document are from the last four rounds of sampling at the site—from December 1998 through February 2000. These data were used to give an up-to-date look at the site. Risk calculations performed for floodplain ground water use the 95 percent upper confidence limit (UCL_{95}) on the mean concentrations to provide reasonable worst-case risk estimates for probable future ground water uses. Though future use of the terrace system ground water is unlikely because of its generally poor quality and limited extent, risk estimates using maximum contaminant concentrations are provided for discussion and comparison. If the maximum concentration of a constituent was much higher than the rest of the measured values, a more representative calculation is also provided. Exposure to floodplain and terrace surface water represents the only potentially complete pathway that currently exists. Maximum surface

water concentrations are used in calculations to provide worst-case risk estimates for these possible exposures.

The same methodology was used to calculate carcinogenic risks for this BLRA update as was used in the 1994 BLRA (i.e., receptors are adults and exposure is averaged over 70 years). For all risk calculations, benchmarks for acceptable contaminant intakes (e.g., RFDs and slope factors) are best available data from standard EPA sources (e.g., Integrated Risk Information System, Region III Risk-Based Concentration Table [EPA 2000]).

Analytical results for nitrate presented in this document are concentrations of nitrate reported as NO_3. Other references may report nitrate values as N (nitrogen), also referred to as nitrate-nitrogen. The conversion factor for these different reported quantities is 1 milligram (mg) N (or nitrate-nitrogen) is equal to 4.4 mg nitrate (as NO_3). Thus, the UMTRA ground water standard for nitrate is 10 mg/L as N or 44 mg/L as NO_3. For consistency in this BLRA update and for ease in use of reported analytical data, all concentrations of nitrate are expressed as NO_3.

6.1.2.1 Floodplain Alluvium

This BLRA update uses the COPC list from the original BLRA as a starting point to evaluate current data for ground water in the floodplain alluvium with two exceptions. First, vanadium was eliminated as a COPC in the BLRA because it was detected at similar concentrations to background. On the basis of historical information and because vanadium concentrations appear to be higher south of the San Juan River compared with north of the river, the potential for inclusion of vanadium as a COPC was reevaluated. Ground water samples from the June 1999 sampling event were analyzed for vanadium and were below or just above the detection limit (see Section 4.4). Therefore, vanadium can be eliminated from further consideration as a COPC. Second, ammonium was reevaluated as a COPC because of its conversion to ammonia in ground water. Risks from ammonia can occur from volatilization in a residential setting.

The following 11 COPCs were identified in floodplain alluvial ground water: antimony, arsenic, cadmium, magnesium, manganese, nitrate, selenium, sodium, strontium, sulfate, and uranium. Data from recent sampling of floodplain alluvial wells indicates that concentrations of antimony, arsenic, and cadmium were at or below their respective detection limits in all wells, with few exceptions, and have been at these low levels since 1995. Therefore, these three constituents are eliminated from further evaluation as COPCs. The other nine COPCs are present at levels sufficiently above background to retain them for further evaluation in this BLRA update.

Table 6–1 presents the minimum, maximum, mean, and UCL_{95} values for each COPC in the floodplain alluvial ground water based on the last four rounds of sampling available—December 1998 through February 2000. Though older data are available, only recent data were used to provide the most current estimate of contaminant concentrations in the plume. Also included for comparison are the applicable UMTRA Ground Water Project standards (if available) or other potentially relevant water quality standards or benchmarks, including risk-based concentrations (RBCs) (EPA 2000). The RBC for a given contaminant represents a concentration in drinking water that would be protective of human health provided that

- Residential exposure scenario is appropriate.
- Ingestion of contaminated drinking water is the only exposure pathway.
- The contaminant contributes nearly all the health risk.

- EPA's risk level of 1×10^{-6} for carcinogens and an HQ of 1 for noncarcinogens is appropriate.

If any of these assumptions is *not* true, contaminant levels at or below RBCs cannot automatically be assumed to be protective. For example, if multiple contaminants are present in drinking water, a single contaminant may be below its RBC but still be a significant contributor to the total risk posed by drinking the water. However, if an RBC is exceeded, it is an indication that further evaluation of the contaminant is warranted. RBCs are intended for use in screening-level evaluations.

Table 6–1. Floodplain Alluvial Aquifer Data Summary

Contaminant	No.[a]	Minimum mg/L	Maximum mg/L	Mean mg/L	UCL_{95} mg/L	MCL mg/L	RBC mg/L
Ammonium (as NH_4)							0.28N[b] as NH_3
Background[c]	10/10	0.0037	0.131	0.0648	0.0918		82N as NH_4
Current Plume[d]	74/76	0.0005	113	17.39	23.46		pH 7.2
Historical Plume[e]	8/8	380	542	516			Temp. 12 °C
Magnesium							
Background	10/10	25.6	45.2	36.23	40.11		
Current Plume	76/76	33.7	3570	794.9	997.7		
Historical Plume	7/7	1252	2750	2030			
Manganese							1.7N
Background	10/10	0.72	2.74	1.48	1.87		
Current Plume	76/76	0.39	12.8	3.31	3.91		
Historical Plume	7/7	5.67	9.75	8.08			
Nitrate (as NO_3)						44 (as NO_3)	255N as NO_3
Background	9/10	0.005	0.476	0.221	0.36		58N as N
Current Plume	75/76	0.005	3480	669.6	913		
Historical Plume	7/7	400	5300	3300			
Selenium						0.01 (UMTRA)	0.18N
Background	2/10	0.0001	0.0012	n/a	n/a	0.05 (SDWA)	
Current Plume	67/76	0.00005	1.1	0.11	0.155		
Historical Plume	7/7	0.07	0.599	0.122			
Sodium							
Background	10/10	458	812	582	642		
Current Plume	76/76	143	6400	1737	2045		
Historical Plume	7/7	1650	3810	3500			
Strontium							22N
Background	10/10	1.76	3.18	2.54	2.82		
Current Plume	76/76	0.85	20.1	8.52	9.37		
Historical Plume	7/7	8.94	14	10.1			
Sulfate							
Background	10/10	1140	1590	1464	1612		
Current Plume	76/76	423	25300	6631	7845		
Historical Plume	7/7	6230	15600	13000			
Uranium						0.044	0.11N
Background	10/10	0.0088	0.0274	0.0158	0.0189		
Current Plume	76/76	0.0181	3.95	0.77	0.984		
Historical Plume	7/7	1.64	4.07	2.8			

Background wells: 850, 851, 852
Plume wells: 610, 612, 614–617, 619, 620, 624, 626, 628, 630, 734–736, 853–857
[a]Frequency of detection
[b]N=noncarcinogenic risks
[c]Current background data collected 12/98 through 2/2000
[d]Current plume data collected 12/98 through 2/2000
[e]Historical data collected 1987–1993; sampling was very limited (see DOE 1994)

No standards or benchmarks have been established for magnesium and sodium based on human-health concerns. The secondary standard for sulfate is based on considerations of taste and odor and not on effects to human health. Because of the lack of toxicity data, potential risks from exposure to these three contaminants cannot be quantified. Exposure intakes are calculated for these constituents, but potential adverse effects are considered in only a qualitative fashion.

For pathways evaluated quantitatively in this BLRA update, adults were evaluated as the primary receptor group; infants were also evaluated for exposure to sulfate and nitrate in residential scenarios because they represent the most sensitive receptor population. Children were evaluated for incidental exposure to surface water, as they are most likely to spend time playing in the area. The residential ingestion scenario was evaluated for all contaminants except ammonia. The major risks resulting from ammonia exposure are from inhalation of ammonia in the gaseous form through volatilization from ground water. Risks were calculated using default inhalation exposure parameters for a residential setting (EPA 1991). Analytical results for ammonia were reported as NH_4. The actual amount of ammonia gas, NH_3, available for volatilization was calculated for site-specific temperature and pH using data compiled by Emerson and others (1975). Risks associated with ammonia for a residential setting require that exposure occurs within a closed structure in which volatilized ammonia is trapped through use of ground water for domestic purposes (e.g., drinking, bathing, laundry). Because the higher concentrations of ammonia (via ammonium) occur at locations where it is generally impractical to construct a residence, potential exposures to ammonia should be considered as a worst-case scenario. For exposure scenarios where exposure does not occur in a closed structure (e.g., recreational use), volatilized ammonia would quickly dissipate to the atmosphere and risks would be negligible.

The meat/milk ingestion pathway was considered for livestock exposed to alluvial ground water through grazing and watering. In the original BLRA, this pathway was eliminated because of acutely toxic levels of sulfate and nitrate; livestock could not survive the chronic ingestion of water required to bioaccumulate contaminants for transfer to humans. For this update, the pathway is retained to analyze the effects of other constituents in the event that sulfate and nitrate can be removed. The original BLRA also examined potential human exposure to contaminants through ingestion of ground water–irrigated produce. These exposure intakes present no significant risks. Concentrations of alluvial ground water used in those calculations were all higher than UCL_{95} concentrations presented in this BLRA update. Therefore, risks associated with ingestion of ground water–irrigated produce remain at insignificant levels and were not further quantified.

6.1.2.2 Terrace Ground Water

Manganese, nitrate, sulfate, selenium, and uranium were evaluated for drinking water ingestion in a residential scenario (adults for all contaminants; infants for nitrate and sulfate). Inhalation of ammonia was also considered. Though not considered a viable pathway, meat/milk ingestion of livestock that watered and grazed on the terrace was evaluated for metals. Nearly all contaminant concentrations in samples from terrace wells are below those used in the original BLRA to calculate exposures to ground water–irrigated produce. Therefore, use of terrace ground water for this purpose would be expected to present no significant risks, and this exposure scenario is not considered further. As mentioned in Section 4.4, molybdenum and vanadium were analyzed in samples from the last two sampling events. Molybdenum did not exceed its standard in any sample and all but a few isolated wells for vanadium were below detection. Therefore these constituents were not added as COPCs based on these results.

6.1.2.3 Surface Water

The original BLRA evaluated only exposure to surface water present in floodplain locations. The only contaminants considered were selenium, strontium, and uranium, though concentrations of other constituents are elevated at floodplain surface water locations. Since that time, surface water has been sampled from Bob Lee Wash and Many Devils Wash, located west and east of the former millsite, respectively. Surface water at those locations is fed by ground water from the terrace alluvium. In this BLRA update, surface water from both terrace and floodplain locations are evaluated.

Exposure to surface water in terrace locations was evaluated for the terrace ground water COPCs except ammonia—manganese, nitrate, sulfate, selenium, and uranium. The exposure scenario evaluated was for children playing on the terrace who may experience incidental ingestion of and dermal contact with surface water. Carcinogenic and noncarcinogenic risks were considered. The small quantity of surface water present in these locations is not likely to support significant irrigation; therefore, ingestion of surface water-irrigated produce is not considered a viable pathway.

For floodplain surface water, exposure to children playing on the floodplain was also evaluated. Ingestion and dermal contact were both considered. To be consistent, the evaluation included all COPCs identified in floodplain ground water. Similar calculations for incidental exposure were performed using concentrations of contaminants in both floodplain and terrace alluvial ground water, in the event that this water is routed to the surface for some permissible use (e.g., agricultural, fountains) and is available for incidental ingestion and/or dermal contact. Children were also evaluated for this scenario as representing the most sensitive receptor population.

In the original BLRA, intakes of contaminated sediments associated with surface water were calculated and were identified as constituting an insignificant risk. Sediment intakes were not quantified in this BLRA update. However, it is unlikely that sediment concentrations are significantly different from those used in the previous intake calculations, and it can be assumed that risks associated with incidental sediment ingestion are still low and insignificant.

The meat/milk ingestion pathway was quantitatively evaluated in this BLRA update to confirm results in the original BLRA that no significant risks would be posed from watering livestock on floodplain surface water. Because floodplain surface water contaminant concentrations are lower than those in floodplain ground water, ingestion of garden produce irrigated with surface water should pose no unacceptable risk and is not evaluated further.

6.1.3 Results

6.1.3.1 Floodplain Ground Water

Table 6–2 presents the results of risk calculations for use of floodplain alluvial ground water as drinking water in a residential exposure scenario. Table 6–3 contains calculations for ammonia inhalation for both the floodplain and terrace systems. (Also provided are calculations for a site-specific RBC for ammonium.) The greatest risks posed for a residential scenario are to infants

Table 6–2. Floodplain Aquifer Residential Ground Water Ingestion Risk Calculations

Noncarcinogens? Ground Water Ingestion Only (adults except where noted)										
Contaminant	**Cw95* mg/L**	**IRw L/d**	**EF d/y**	**ED y**	**BW kg**	**AT d**	**Intake mg/kg-d**	**RfD** mg/kg-d**	**HQ**	
Magnesium	998	2	350	7	70	2,555	27.342	n/a	n/a	
Manganese	3.91	2	350	7	70	2,555	0.107	0.047	2.28	
Nitrate	913	2	350	7	70	2,555	25.014	7	3.57	
infants	913	0.64	350	1	4	365	140.077	7	20.01	
Selenium	0.155	2	350	7	70	2,555	0.004	0.005	0.85	
Sodium	2045	2	350	7	70	2,555	56.027	n/a	n/a	
Strontium	9.37	2	350	7	70	2,555	0.257	0.6	0.43	
Sulfate	7845	2	350	7	70	2,555	214.932	n/a	n/a	
infants	7845	0.64	350	1	4	365	1,203.616	n/a	n/a	
Uranium	0.984	2	350	7	70	2,555	0.027	0.003	8.99	
								HI (adult) =	16.12	
Carcinogens? Ground Water Ingestion Only (adults)										
Contaminant		**CW***	**IR**	**EF**	**ED**	**BW**	**AT**	**Intake**	**SF****	**Risk**
U234+238	UCL95	675.024	2	350	50	na	na	2.36E+07	5.32E-11	1.26E-03
(pCi/L)	mean	528.22	2	350	50	na	na	1.85E+07	5.32E-11	9.84E-04

*Data from 12/98 through 2/00 sampling events
**Reference Doses (RfD) and Slope Factors (SF) from best available EPA sources

Table 6–3. Noncarcinogens—Inhalation through Water Use in Residential Setting

Noncarcinogens? Inhalation through water use in residential setting*? Floodplain										
Contaminant		**CA**	**IR**	**EF**	**ED**	**BW**	**AT**	**Intake-max**	**RfD**	**HQ**
Ammonia (UCL95)		0.039882	15	350	30	70	10,950	0.0082	0.0286	0.287
*IR = 15 m^3/d of air default; concentration in air = water concentration x water-to-air volatilization factor x conversion factor Default volatilization factor = .0005; conversion factor is 1,000L/m^3 UCL95 for NH_3 in floodplain alluvial ground water is 0.08 mg/L [1]See										
Noncarcinogens? Inhalation through water use in residential setting*? Terrace										
Contaminant		**CA**	**IR**	**EF**	**ED**	**BW**	**AT**	**Intake-max**	**RfD**	**HQ**
Ammonia (max)		2.958	15	350	30	70	10950	0.6078	0.0286	21.252
*IR = 15 m^3/d of air default; concentration in air = water concentration x water-to-air volatilization factor x conversion factor Default volatilization factor = .0005; conversion factor is 1,000L/m^3 Maximum NH_3 in Shiprock terrace ground water is 5.92 mg/L										
Ammonia		0.1394	15	350	30	70	10,950	0.0286	0.0286	1.002
		RBC = 82 as NH_4, .28 as NH_3 using T = 12°C and pH = 7.2								

from ingestion of nitrate and sulfate. Nitrate concentrations exceed acceptable levels by more than 20 times; predicted intakes are in the range of potentially lethal levels. Though no RfDs have been developed for sulfate, calculated intakes are within the range reported to cause diarrhea and dehydration in sensitive populations (i.e., infants and the elderly), which could be potentially lethal.

The greatest noncarcinogenic risks to adults are posed by ingestion of uranium and, to a lesser degree, nitrate and manganese. Intakes of magnesium and sulfate are within ranges associated with laxative effects in adults, and sodium intakes are within the range associated with hypertension effects; however, RfDs are not available for these constituents and they are evaluated only qualitatively. Calculated HQs for ammonia, selenium, and strontium are all below 1, though selenium approaches this value and is well above its UMTRA standard of 0.01 mg/L. Strontium and ammonium together make up less than 5 percent of the total risk (using infant risks for nitrate) and can probably be eliminated from further consideration as COPCs in floodplain alluvial ground water.

Carcinogenic risks associated with uranium exceed the upper end of EPA's acceptable range for both UCL_{95} and mean ground water concentrations.

Table 6–4 presents results of risk calculations assuming incidental exposure to floodplain ground water. Such exposure could occur if ground water was used for agricultural purposes or in fountains or ponds in a park setting. It is assumed that children are the likely receptors. As shown in the table, incidental exposure (ingestion and dermal contact) would present no unacceptable risk. Table 6–5 contains risk calculations for the meat/milk exposure pathway, assuming cattle are grazed and watered on the floodplain and water is drawn from the ground water system. Maximum contaminant concentrations in floodplain ground water were used as water concentrations; the average contaminant concentration measured in bulrushes was used as the concentration for forage. Sampling of more suitable vegetation for livestock grazing took place recently and these calculations will be updated when those data become available. Current calculations show that noncarcinogenic risks are just above the acceptable HI of 1. Carcinogenic risks are within EPA's acceptable risk range. Because the calculations used maximum ground water concentrations and assume cattle get all of their food and water from the floodplain area, they probably overestimate actual risks. Also, the calculations assume that 75 percent of an individual's diet consists of meat and milk. Therefore, it is likely that risks from meat and milk consumption would be acceptable.

6.1.3.2 Terrace Ground Water

Results of risk calculations for use of terrace alluvial ground water as drinking water in a residential setting are provided in Table 6–6. Ammonia inhalation calculations are in Table 6–3. This exposure scenario is improbable because of the generally poor water quality in this system (even in areas presumably outside the influence of the site) and its questionable sustainability as a regular water source. However, these calculations are useful for comparison.

As with the floodplain ground water, the most severe adverse health effects in this exposure scenario would be associated with intakes of nitrate and sulfate by infants. Calculated exposure intakes are higher than those determined for the floodplain aquifer and likewise are within the range of potentially lethal levels. Adult intake levels for sulfate would also be expected to produce laxative effects. Selenium risks are slightly less than nitrate risks for adults.

Table 6–4. Floodplain Ground Water Incidental Ingestion/Dermal Exposure Pathways

COPC	Cw-95 mg/L	Sa cm^2	Pc cm/h	Cf L/cm^3	ET h/d	EF d/y	ED y	IRw L/d	BW kg	AT d	Intake Ingested mg/kg-d	Intake Absorbed mg/kg-d	Total dose mg/kg-d	RfD mg/kg-d	HQ mg/kg-d
Magnesium	998	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.407	0.004	0.4110	na	na
Manganese	3.91	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.002	0.000	0.0016	0.047	0.034
Nitrate	913	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.372	0.004	0.3760	7	0.054
Selenium	0.155	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.000	0.000	0.0001	0.005	0.013
Sodium	2045	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.834	0.008	0.8421	na	na
Strontium	9.37	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.004	0.000	0.0039	0.6	0.006
Sulfate	7845	497	0.001	0.001	1	114	7	0.05	38.3	2555	3.199	0.032	3.2305	na	na
Uranium	0.984	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.000	0.000	0.0004	0.003	0.135
														HI=	0.242
Carcinogens - Ground Water Ingestion Only (adults)															
COPC		**CW-max**	**IR**	**EF**	**ED**	**BW**	**AT**	**Intake**	**SF****	**Risk**					
U234+238	max	675.024	0.05	114	7	na	na	2.69E+04	5.32E-11	1.43E-06					
(pCi/L)															

*Data based on results of sampling events from 12/98 through 2/00
**Reference Doses (RfD) and Slope Factors (SF) from best available EPA sources

Table 6–5. Meat/Milk Exposure Calculations—Floodplain Ground Water

	Cm	Cb	Irb	Fl	EF	ED	BW	Irm	AT	Intake-Beef	Intake-Milk	Total	RfD/SF	HQ/Risk
Non-carc.														
Manganese	2.436602	2.784688	0.075	0.75	350	7	70	0.3	2555	0.0021457	0.0075101	0.00965581	0.047	0.205443
Selenium	0.471968	1.76988	0.075	0.75	350	7	70	0.3	2555	0.0013638	0.0014547	0.00281847	0.005	0.563695
Strontium	4.61364	0.922728	0.075	0.75	350	7	70	0.3	2555	0.000711	0.0142201	0.01493113	0.6	0.024885
Uranium	0.36186	0.12062	0.075	0.75	350	7	70	0.3	2555	9.294E-05	0.0011153	0.00120827	0.003	0.402755
													HI=	1.196778
Carc.														
Uranium	248.236	82.74532	0.075	0.75	350	70	70	0.3	25550	114033.39	1368400.7	1482434.12	5.32E-11	7.89E-05
						Mn (mg/kg)	2.78469	Cg = 12.8 mg/L		Cs = 384 mg/kg		Fb = .0004	Cp = 321	mg/kg
		Cb = Fb[(Cp x Qp) + (Qs x Cs) + (Qw x Cg)				Se (mg/kg)	1.76988	Cg = 1.1 mg/L		Cs = 1.9 mg/Kg		Fb = .015	Cp = 2.93	mg/kg
						Sr (mg/kg)	0.92273	Cg = 20.1 mg/L		Cs = 407 mg/Kg		Fb = .0003	Cp = 94.5	mg/kg
						U (mg/kg)	0.12062	Cg = 3.95 mg/L		Cs =40 mg/Kg		Fb = .0002	Cp = 19.3	mg/kg
						U (pCi/kg)	82.7453	(U-234 & U-238 combined;assumes 1 mg U is equal to 686 pCi)						
						Mn (mg/kg)	2.4366		Fm = .00035					
		Cm = Fm[(Cp x Qp) + (Qs x Cs) + (Qw x Cg)				Se (mg/kg)	0.47197		Fm = .004					
						Sr (mg/kg)	4.61364		Fm = .0015					
						U (mg/kg)	0.36186		Fm = .0006					
						U (pCi/kg)	248.236							

Cb Contaminant concentration in beef (calculated value; chemical-specific; mg/kg)
Irb Ingestion rate for beef (0.075 kg/day; EPA 1989b)
Fl Fraction of diet (meat/milk) ingested (0.75, unitless; subsistence farm family)
EF Exposure frequency (350 days/year)
ED Exposure duration (7 years for an adult for noncarcinogens; 70 years for carcinogens)
BW Body weight (70 kg for an adult)
AT Averaging time (365 days x ED)
Fb Forage-to-beef transfer coefficient (chemical-specific; unitless)
Cp Contaminant concentration in pasture grasses (measured; chemical-specific; mg/kg)
Qp Quantity of pasture ingested daily by cattle (19 kg dry weight/day)
Qs Quantity of soil ingested daily by cattle (0.38 kg based on 2% of dry matter from feed ingestion rate)
Cg Contaminant concentration in ground water (the highest determined concentration in floodplain; mg/L)
Qw Quantity of water ingested daily by cattle (56 L/day)
Cm Contaminant concentration in milk (calculated value; chemical-specific; mg/kg)
Irm Ingestion rate for milk (0.30 kg/day; EPA 1989b)
Fm Feed-to-milk transfer coefficient (chemical specific; unitless)
Cs Contaminant concentration in soil? highest concentration in terrace or floodplain

All values used are standard EPA defaults or came from the original BLRA

Table 6–6. Terrace Ground Water Residential Ground Water Ingestion Risk Calculations

Noncarcinogens? Ground Water Ingestion Only (adults, except where noted)										
Contaminant	**Cw-max* mg/L**	**IRw L/d**	**EF d/y**	**ED y**	**BW kg**	**AT d**	**Intake mg/kg-d**	**RfD** mg/kg-d**	**HQ**	
Manganese	34.5	2	350	7	70	2555	0.945	0.047	20.11	
Nitrate	10065	2	350	7	70	2555	275.753	7	39.39	
infants	10065	0.64	350	1	4	365	1544.219	7	220.60	
Sulfate	15600	2	350	7	70	2555	427.397	na	na	
infants	15600	0.64	350	1	4	365	2393.425			
Selenium	6.69	2	350	7	70	2555	0.183	0.005	36.66	
Uranium	3.4	2	350	7	70	2555	0.093	0.003	31.05	
mean	0.2567	2	350	7	70	2555	0.007	0.003	2.34	
								HI adult =	127.21	
								(max)		
Carcinogens? Ground Water Ingestion Only (adults)										
Contaminant		**CW***	**IR**	**EF**	**ED**	**BW**	**AT**	**Intake**	**SF****	**Risk**
U234+238	max	2332.4	2	350	50	na	na	8.16E+07	5.32E-11	4.34E-03
(pCi/L)	mean	176.0962	2	350	50	na	na	6.16E+06	5.32E-11	3.28E-04

*Data based on results of sampling events from 12/98 through 2/00

**Reference Doses (RfD) and Slope Factors (SF) from best available EPA sources

The maximum detected concentration of uranium in terrace ground water is associated with significantly elevated risks, both noncarcinogenic and carcinogenic. However, these high concentrations are limited in extent, and risks calculated based on mean aquifer concentrations are much lower. Noncarcinogenic risks based on the mean are approximately double an acceptable HQ of 1. Carcinogenic risks associated with mean concentrations are approximately 3 times the high end of EPA's acceptable risk range.

Excessive risks may occur with the inhalation of ammonia from ammonium in ground water. This could be an important issue if ground water was used in a residence in the vicinity of well 603, which has the highest ammonium concentration in ground water.

Table 6–7 presents results of risk calculations for incidental exposure to terrace ground water. As with the floodplain calculations, it is assumed that children are the receptors. Maximum ground water concentrations were used. Total noncarcinogenic risks are nearly double an acceptable HI of 1 using the maximum uranium concentration; carcinogenic risks are within EPA's acceptable risk range.

Table 6–8 contains calculations for the meat/milk exposure pathway assuming cattle are watered with terrace ground water and grazed on contaminated vegetation. Risks are likely overestimated because of the conservative assumptions used in the calculations (see Section 6.1.3.1) and because vegetation on the terrace probably does not contact ground water and is unlikely to be contaminated. The calculations indicate that noncarcinogenic risks are almost 3 times greater than acceptable, and carcinogenic risks are within EPA's acceptable risk range.

6.1.3.3 Surface Water

Table 6–9 presents the results of risk calculations for incidental ingestion of and dermal exposure to surface water by children playing on the floodplain. All COPCs evaluated for floodplain alluvial ground water were included in the analysis. Carcinogenic and noncarcinogenic risks associated with this exposure pathway are well below potentially unacceptable levels, even based on exposure to maximum detected contaminant concentrations in samples of floodplain surface water. Highest risks are associated with maximum concentrations of manganese, though only a single sample location had a significantly elevated concentration. Calculated sulfate intakes are below levels shown to produce any adverse effects.

Calculations for the meat/milk exposure pathway assuming livestock watering on floodplain surface water are shown in Table 6–10. Even with the same conservative assumptions used previously, risks are below unacceptable levels. These calculations will be updated after results of planned vegetation sampling become available.

Table 6–11 presents risks calculated for incidental ingestion of terrace surface water. COPCs evaluated for terrace alluvial ground water were included. Bob Lee Wash and Many Devils Wash samples are considered separately because of the significantly different concentrations detected in samples from those two locations. Risks calculated for nitrate and uranium at both locations are below potentially unacceptable levels. Intakes calculated for sulfate in Many Devils Wash are at the low end of the range that could result in laxative effects, though these effects would probably be temporary. Calculated risks for both these locations are considered to be worst case, not only because maximum contaminant concentrations were used but also because

Table 6–7. Incidental Terrace Ground Water Ingestion/Dermal Exposure Pathways

Contaminant	Cw-max* mg/L	Sa cm^2	Pc cm/h	Cf L/cm^3	ET hr/d	EF d/y	ED y	IRw L/d	BW kg	AT d	Intake ingested mg/kg-d	Intake absorbed mg/kg-d	Total dose mg/kg-d	RfD mg/kg-d	HQ mg/kg-d
Manganese	34.5	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.014	0.000	0.0142	0.047	0.302
Nitrate	10065	497	0.001	0.001	1	114	7	0.05	38.3	2555	4.104	0.041	4.1447	7	0.592
Selenium	6.69	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.003	0.000	0.0028	0.005	0.551
Sulfate	15600	497	0.001	0.001	1	114	7	0.05	38.3	2555	6.361	0.063	6.4240		
Uranium	3.4	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.001	0.000	0.0014	0.003	0.467
mean	0.2503	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.000	0.000	0.0001	0.003	0.034
														HI (max)=	1.912
														HI (mean)=	1.480
Carcinogens? Ground Water Ingestion Only (adults)															
Contaminant		**CW-max**	**IR**	**EF**	**ED**	**BW**	**AT**	**Intake**	**SF****	**Risk**					
U234+238	max	2332.4	0.05	114	7	na	na	9.31E+04	5.32E-11	4.95E-06					
(pCi/L)															

*Data based on results of sampling events from 12/98 through 2/00
**Reference Doses (RfD) and Slope Factors (SF) from best available EPA sources

Table 6–8. Meat/Milk Exposure Calculations—Terrace Ground Water

	Cm	Cb	Irb	FI	EF	ED	BW	Irm	AT	Intake-Beef	Intake-Milk	Total	RfD/SF	HQ/Risk
Noncarc.														
Manganese	3.264646	2.332624	0.075	0.75	350	7	70	0.3	2555	0.0017974	0.0100623	0.01186	0.047	0.252333
Selenium	1.724128	6.91293	0.075	0.75	350	7	70	0.3	2555	0.0053267	0.0053141	0.010641	0.005	2.128167
Strontium	4.40364	1.912428	0.075	0.75	350	7	70	0.3	2555	0.0014736	0.0135729	0.015046	0.6	0.025077
Uranium	0.34338	0.047048	0.075	0.75	350	7	70	0.3	2555	3.625E-05	0.0010584	0.001095	0.003	0.364872
													HI=	2.77045
Carc.														
Uranium	235.5587	32.27493	0.075	0.75	350	70	70	0.3	25550	44478.885	1298517.2	1342996	5.32E-11	7.14E-05

Cb = Fb[(Cp x Qp) + (Qs x Cs) + (Qw x Cg)

Mn (mg/kg)	2.33262	Cg = 34.5 mg/L	Cs = 262 mg/Kg	Fb = .0004
Se (mg/kg)	6.91293	Cg = 6.69 mg/L	Cs = 1.9 mg/Kg	Fb = .015
Sr (mg/kg)	1.91243	Cg = 17.6 mg/L	Cs = 407 mg/Kg	Fb = .0003
U (mg/kg)	0.04705	Cg = 3.4 mg/L	Cs =40 mg/Kg	Fb =.0002
U (pCi/kg)	32.2749	(U-234 & U-238 combined; assumes 1 mg U is equal to 686 pCi)		

Cm = Fm[(Cp x Qp) + (Qs x Cs) + (Qw x Cg)

Mn (mg/kg)	3.26465	Cp = 384	Fm =.00035
Se (mg/kg)	1.72413	Cp = 1.9	Fm = .004
Sr (mg/kg)	4.40364	Cp = 94.5	Fm = .0015
U (mg/kg)	0.34338	Cp = 19.3	Fm = .0006
U (pCi/kg)	235.559		

Cb Contaminant concentration in beef (calculated value; chemical-specific; mg/kg)
Irb Ingestion rate for beef (0.075 kg/day; EPA 1989a)
FI Fraction of diet (meat/milk) ingested (0.75, unitless; subsistence farm family)
EF Exposure frequency (350 days/year)
ED Exposure duration (7 years for an adult for noncarcinogens; 70 years for carcinogens)
BW Body weight (70 kg for an adult)
AT Averaging time (365 days x ED)
Fb Forage-to-beef transfer coefficient (chemical-specific; unitless)
Cp Contaminant concentration in pasture grasses (measured value; chemical-specific; mg/kg)
Qp Quantity of pasture ingested daily by cattle (19 kg dry weight/day)
Qs Quantity of soil ingested daily by cattle (0.38 kg based on 2% of dry matter from feed ingestion rate)
Cg Contaminant concentration in ground water (the highest determined concentration in terrace; mg/L)
Qw Quantity of water ingested daily by cattle (56 L/day)
Cm Contaminant concentration in milk (calculated value; chemical-specific; mg/kg)
Irm Ingestion rate for milk (0.30 kg/day; EPA 1989a)
Fm Feed-to-milk transfer coefficient (chemical specific; unitless)
Cs Contaminant concentration in soil? highest concentration in terrace or floodplain

All values used are standard EPA defaults or came from the original BLRA

Table 6–9. Floodplain Surface Water Incidental Ingestion/Dermal Exposure Pathways

Contaminant	Cw-max* mg/L	Sa cm^2	Pc cm/h	Cf L/cm^3	ET h/d	EF d/y	ED y	IRw L/d	BW kg	AT d	Intake Ingested mg/kg-d	Intake Absorbed mg/kg-d	Total dose mg/kg-d	RfD mg/kg-d	HQ mg/kg-d
Magnesium	757	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.309	0.003	0.3117	na	na
Manganese	16.4	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.007	0.000	0.0068	0.047	0.144
2nd highest	5.08	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.002	0.000	0.0021	0.047	0.045
Nitrate	552	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.225	0.002	0.2273	7	0.032
Selenium	0.158	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.000	0.000	0.0001	0.005	0.013
Sodium	7320	497	0.001	0.001	1	114	7	0.05	38.3	2555	2.985	0.030	3.0143	na	na
Strontium	19.8	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.008	0.000	0.0082	0.6	0.014
Sulfate	17100	497	0.001	0.001	1	114	7	0.05	38.3	2555	6.972	0.069	7.0417	na	na
2nd highest	5650	497	0.001	0.001	1	114	7	0.05	38.3	2555	2.304	0.023	2.3266	na	na
Uranium	0.682	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.000	0.000	0.0003	0.003	0.094
														HI max =	0.296
Carcinogens? Surface Water Ingestion Only (adults)															
Contaminant		**CW-max**	**IR**	**EF**	**ED**	**BW**	**AT**	**Intake**	**SF****	**Risk**					
U234+238	max	467.852	0.05	114	7	na	na	1.87E+04	5.32E-11	9.93E-07					
(pCi/L)															

*Data based on results of sampling events from 12/98 through 2/00

**Reference Doses (RfD) and Slope Factors (SF) from best available EPA sources

Table 6–10. Meat/Milk Exposure Calculations—Floodplain Surface Water

	Cm	Cb	Irb	FI	EF	ED	BW	Irm	AT	Intake-Beef	Intake-Milk	Total	RfD/SF	HQ/Risk
Noncarc.														
Manganese	2.199383	2.513581	0.075	0.75	350	7	70	0.3	2555	0.00193683	0.00677892	0.00871576	0.047	0.185442
Selenium	0.261408	0.98028	0.075	0.75	350	7	70	0.3	2555	0.00075535	0.00080571	0.00156106	0.005	0.312212
Strontium	3.89964	0.779928	0.075	0.75	350	7	70	0.3	2555	0.00060097	0.01201944	0.01262041	0.6	0.021034
Uranium	0.232903	0.077634	0.075	0.75	350	7	70	0.3	2555	5.9821E-05	0.00071785	0.00077767	0.003	0.259224
													HI=	0.777913
Carc.														
Uranium	159.7716	53.2572	0.075	0.75	350	70	70	0.3	25550	73395.0765	880740.919	954135.995	5.32E-11	5.08E-05

Cb = Fb[(Cp x Qp) + (Qs x Cs) + (Qw x Cg)

Mn (mg/kg)	2.513581	Cg = 0.697mg/L	Cs = 384mg/kg	Fb = .0004	Cp = 321 mg/kg
Se (mg/kg)	0.98028	Cg = 0.160 mg/L	Cs = 1.9 mg/kg	Fb = .015	Cp = 2.93 mg/kg
Sr (mg/kg)	0.779928	Cg = 11.6 mg/L	Cs = 407 mg/kg	Fb = .0003	Cp = 94.5 mg/kg
U (mg/kg)	0.077634	Cg = 0.112 mg/L	Cs = 40 mg/kg	Fb =.0002	Cp = 19.3 mg/kg
U (pCi/kg)	53.2572	(U-234 & U-238 combined; assumes 1 mg U is equal to 686 pCi)			

Cm = Fm[(Cp x Qp) + (Qs x Cs) + (Qw x Cg)

Mn (mg/kg)	2.199383	Fm =.00035
Se (mg/kg)	0.261408	Fm =.004
Sr (mg/kg)	3.89964	Fm =.0015
U (mg/kg)	0.232903	Fm =.0006
U (pCi/kg)	159.7716	

Cb Contaminant concentration in beef (calculated value; chemical-specific; mg/kg)
Irb Ingestion rate for beef (0.075 kg/day; EPA 1989a)
FI Fraction of diet (meat/milk) ingested (0.75, unitless; subsistence farm family)
EF Exposure frequency (350 days/year)
ED Exposure duration (7 years for an adult for noncarcinogens; 70 years for carcinogens)
BW Body weight (70 kg for an adult)
AT Averaging time (365 days x ED)
Fb Forage-to-beef transfer coefficient (chemical-specific; unitless)
Cp Contaminant concentration in pasture grasses (measured; chemical-specific; mg/kg)
Qp Quantity of pasture ingested daily by cattle (19 kg dry weight/day)
Qs Quantity of soil ingested daily by cattle (0.38 kg based on 2% of dry matter from feed ingestion rate)
Cg Contaminant concentration in surface water (representative high concentration in floodplain; mg/L)
Qw Quantity of water ingested daily by cattle (56 L/day)
Cm Contaminant concentration in milk (calculated value; chemical-specific; mg/kg)
Irm Ingestion rate for milk (0.30 kg/day; EPA 1989a)
Fm Feed-to-milk transfer coefficient (chemical specific; unitless)
Cs Contaminant concentration in soil? highest concentration in terrace or floodplain

All values used are standard EPA defaults or came from the original BLRA.

Table 6–11. Incidental Surface Water Ingestion/Dermal Exposure Pathways—Bob Lee and Many Devils Washes

Contaminant	**Cw-max*** mg/L	**Sa** cm^2	**Pc** cm/h	**Cf** L/cm^3	**ET** h/d	**EF** d/y	**ED** y	**IRw** L/d	**BW** kg	**AT** d	**Intake ingested** mg/kg-d	**Intake absorbed** mg/kg-d	**Total dose** mg/kg-d	**RfD** mg/kg-d	**HQ** mg/kg-d
Manganese (BL)	0.01	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.000	0.000	0.0000	0.047	0.000
(MD)	0.129	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.000	0.000	0.0001	0.047	0.001
Nitrate (BL)	2112	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.861	0.009	0.8697	7	0.124
(MD)	8060	497	0.001	0.001	1	114	7	0.05	38.3	2555	3.286	0.033	3.3191	7	0.474
Selenium (BL)	0.013	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.000	0.000	0.0000	0.005	0.001
(MD)	7.01	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.003	0.000	0.0029	0.005	0.577
Sulfate (BL)	12925	497	0.001	0.001	1	114	7	0.05	38.3	2555	5.270	0.052	5.3224		
(MD)	27400	497	0.001	0.001	1	114	7	0.05	38.3	2555	11.172	0.111	11.2831		
Uranium (BL)	2.415	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.001	0.000	0.0010	0.003	0.331
(MD)	0.63	497	0.001	0.001	1	114	7	0.05	38.3	2555	0.000	0.000	0.0003	0.003	0.086
														HI (MD)=	1.139
														HI (BL)=	0.457
Carcinogens? Surface Water Ingestion Only (adults)															
Contaminant		**CW-max**	**IR**	**EF**	**ED**	**BW**	**AT**	**Intake**	**SF****	**Risk**					
U234+238 (pCi/L)	BL	1656.69	0.05	114	7	na	na	6.61E+04	5.32E-11	3.52E-06					
	MD	432.18	0.05	114	7	na	na	1.72E+04	5.32E-11	9.17E-07					

*Data based on results of sampling events from 12/98 through 2/00

**Reference Doses (RfD) and Slope Factors (SF) from best available EPA sources

BL = Bob Lee Wash samples

MD = Many Devil's Wash samples

the poor taste produced by these constituents would discourage ingestion of even small amounts of water. Noncarcinogenic risks for Many Devils Wash are slightly above the acceptable HI of one. Carcinogenic risks associated with both washes are acceptable. This exposure pathway has been eliminated with the recent interim action conducted at the seeps and washes.

Risks from meat/milk ingestion for livestock watered on seeps or in washes (whichever is higher) are presented in Table 6–13. This scenario is unlikely because of the interim action taken to prevent exposure of animals to seeps and washes. However, it is provided for comparison. Calculated risks are virtually the same as those for use of terrace ground water. This is expected, because the surface water is largely present as the surface expression of ground water.

6.1.3.4 Summary and Recommendations

A summary of potential human health risks associated with site-related ground water and surface water is presented in Table 6–12 for the various pathways evaluated either quantitatively or qualitatively in this BLRA update.

Table 6–12. Summary of Potential Human Health Risks

	Drinking Water Residential	Incidental Exposure Dermal/Ingestion	Ingestion of Livestock[a]	Water-Irrigated Produce Ingestion
Floodplain Ground Water	Y[b]	N[c]	Y	N
Terrace Ground Water	Y	Y	Y	N
Floodplain Surface Water	na[d]	N	N	N
Terrace Surface Water	na	Y(MD only)[e]	Y	N

[a]Meat and milk from livestock watered and grazed on contaminated water.
[b]Y = risks are unacceptable.
[c]N = no unacceptable risks.
[d]na = pathway not applicable.
[e]MD = Many Devils Wash

The following observations can be made based on the analysis presented in this document:

- The main unacceptable human health risks associated with the Shiprock site are for use of floodplain and terrace ground water systems for drinking water in a residential setting. In addition, inhalation of ammonia from terrace ground water could present excessive risk if it is used as the primary water source in a residence.

- Conservative risk calculations indicate elevated risks associated with ingestion of milk/meat from cattle watered with both floodplain and terrace ground water and grazed on contaminated vegetation. Risks would also be unacceptable for ingestion of livestock watered exclusively on water from seeps and washes. The interim action undertaken to prevent animal exposure to seeps and washes has eliminated this latter exposure pathway.

Table 6–13. Meat/Milk Exposure Calculations—Terrace Surface Water (seeps and washes)

	Cm	Cb	Irb	FI	EF	ED	BW	Irm	AT	Intake-Beef	Intake-Milk	Total	RfD/SF	HQ/Risk
Noncarc.														
Manganese	2.206736	1.602384	0.075	0.75	350	7	70	0.3	2555	0.00123471	0.00680158	0.008036	0.047	0.170985
Selenium	1.717528	7.18173	0.075	0.75	350	7	70	0.3	2555	0.00553387	0.00529375	0.010828	0.005	2.165524
Strontium	4.27764	1.887228	0.075	0.75	350	7	70	0.3	2555	0.0014542	0.01318451	0.014639	0.6	0.024398
Uranium	0.310452	0.036072	0.075	0.75	350	7	70	0.3	2555	2.7795E-05	0.00095687	0.000985	0.003	0.328223
													HI=	2.689129
Carc.														
Uranium	212.9701	24.74539	0.075	0.75	350	70	70	0.3	25550	34102.2434	1173997.52	1208100	5.32E-11	6.43E-05

Cb = Fb[(Cp x Qp) + (Qs x Cs) + (Qw x Cg)

Mn (mg/kg) 1.602384 Cg = 1.9mg/L Cs = 262 mg/kg Fb =.0004
Se (mg/kg) 7.18173 Cg = 7.01 mg/L Cs = 1.9 mg/kg Fb = .015
Sr (mg/kg) 1.887228 Cg = 16.1 mg/L Cs = 407 mg/kg Fb = .0003
U (mg/kg) 0.036072 Cg = 2.42 mg/L Cs = 40 mg/kg Fb =.0002
U (pCi/kg) 24.74539 (U-234 & U-238 combined; assumes 1 mg U is equal to 686 pCi)

Cm = Fm[(Cp x Qp) + (Qs x Cs) + (Qw x Cg)

Mn (mg/kg) 2.206736 Cp = 321 Fm = 0.00035
Se (mg/kg) 1.717528 Cp = 2.93 Fm = 0.004
Sr (mg/kg) 4.27764 Cp = 94.5 Fm = 0.0015
U (mg/kg) 0.310452 Cp = 19.3 Fm = 0.0006
U (pCi/kg) 212.9701

Cb Contaminant concentration in beef (calculated value; chemical-specific; mg/kg)
Irb Ingestion rate for beef (0.075 kg/day; EPA 1989a)
FI Fraction of diet (meat/milk) ingested (0.75, unitless; subsistence farm family)
EF Exposure frequency (350 days/year)
ED Exposure duration (7 years for an adult for noncarcinogens; 70 years for carcinogens)
BW Body weight (70 kg for an adult)
AT Averaging time (365 days x ED)
Fb Forage-to-beef transfer coefficient (chemical-specific; unitless)
Cp Contaminant concentration in pasture grasses (measured value; chemical-specific; mg/kg)
Qp Quantity of pasture ingested daily by cattle (19 kg dry weight/day)
Qs Quantity of soil ingested daily by cattle (0.38 kg based on 2% of dry matter from feed ingestion rate)
Cg Contaminant concentration in surface water (the highest determined concentration in terrace; mg/L)
Qw Quantity of water ingested daily by cattle (56 L/day)
Cm Contaminant concentration in milk (calculated value; chemical-specific; mg/kg)
Irm Ingestion rate for milk (0.30 kg/day; EPA 1989a)
Fm Feed-to-milk transfer coefficient (chemical specific; unitless)
Cs Contaminant concentration in soil? highest concentration in terrace or floodplain

All values used are standard EPA defaults or came from the original BLRA.

- The contaminants of greatest concern in ground water are nitrate and sulfate, which are present in both floodplain and terrace ground water at levels that are potentially lethal to infants.
- Uranium is the contaminant that poses the greatest noncarcinogenic risks for adult consumption of floodplain ground water; nitrate, manganese, and selenium are of lesser importance. Risks posed by strontium are low enough to be considered insignificant. Noncarcinogenic risks posed by average uranium concentrations in terrace ground water are approximately 2 times the acceptable risk level (HI = 1).
- Concentrations of uranium in both floodplain and terrace ground water result in risk estimates that exceed the high end of EPA's acceptable carcinogenic risk range.
- Exposure to contaminated surface water by children playing in the floodplain area is unlikely to present any unacceptable risk. Calculations indicate that incidental exposure to seeps and washes results in slightly higher than acceptable noncarcinogenic risks. However, the interim action at the seeps and washes has eliminated this exposure pathway.
- Produce irrigated with any site-related water would not accumulate sufficient levels of contamination to produce unacceptable risks through human consumption.

For a compliance strategy for the Shiprock site to be protective of human health, only a few restrictions on water use or access must be imposed. Unacceptable risks to humans would only be posed by use of terrace or floodplain ground water as a primary source of drinking water. Use of any ground water or surface water for agricultural purposes would not present unacceptable risks to humans, though floodplain and terrace ground water and terrace surface water would be unsuitable for watering livestock because of risks to the animals themselves. Incidental exposure to floodplain or terrace surface water by children playing in those areas would not result in unacceptable risks. Therefore, access need not be restricted to prevent this type of exposure.

6.2 Ecological Risk Assessment

Ecological risk assessment (ERA) is a process that evaluates the likelihood that adverse ecological effects are occurring or may occur in the future as a result of exposure to one or more environmental stressors. A stressor is defined as any physical, chemical, or biological entity that can induce an adverse ecological response. The risk assessment process is outlined in EPA guidance documents, particularly the "Guidelines for Ecological Risk Assessment" (EPA 1998) and the "Framework for Ecological Risk Assessment" (EPA 1992). The ERA for the Shiprock site generally follows this EPA framework and guidance.

As shown in Figure 6–2, the framework of the ERA contains three main components: (1) problem formulation, (2) analysis, and (3) risk characterization. The overall goal of the problem formulation is to "set the stage" for the analysis and risk characterization phases of the process. In the problem formulation, the need for a risk assessment is identified and the scope of the problem is defined. Available data are evaluated to identify potential stressors (in this case, the potential stressors are COPCs associated with the ground water at the Shiprock millsite), key ecological receptors, and potential exposure pathways linking the receptors to the stressors. This information is used to develop a site conceptual model and risk hypotheses. Finally, assessment and

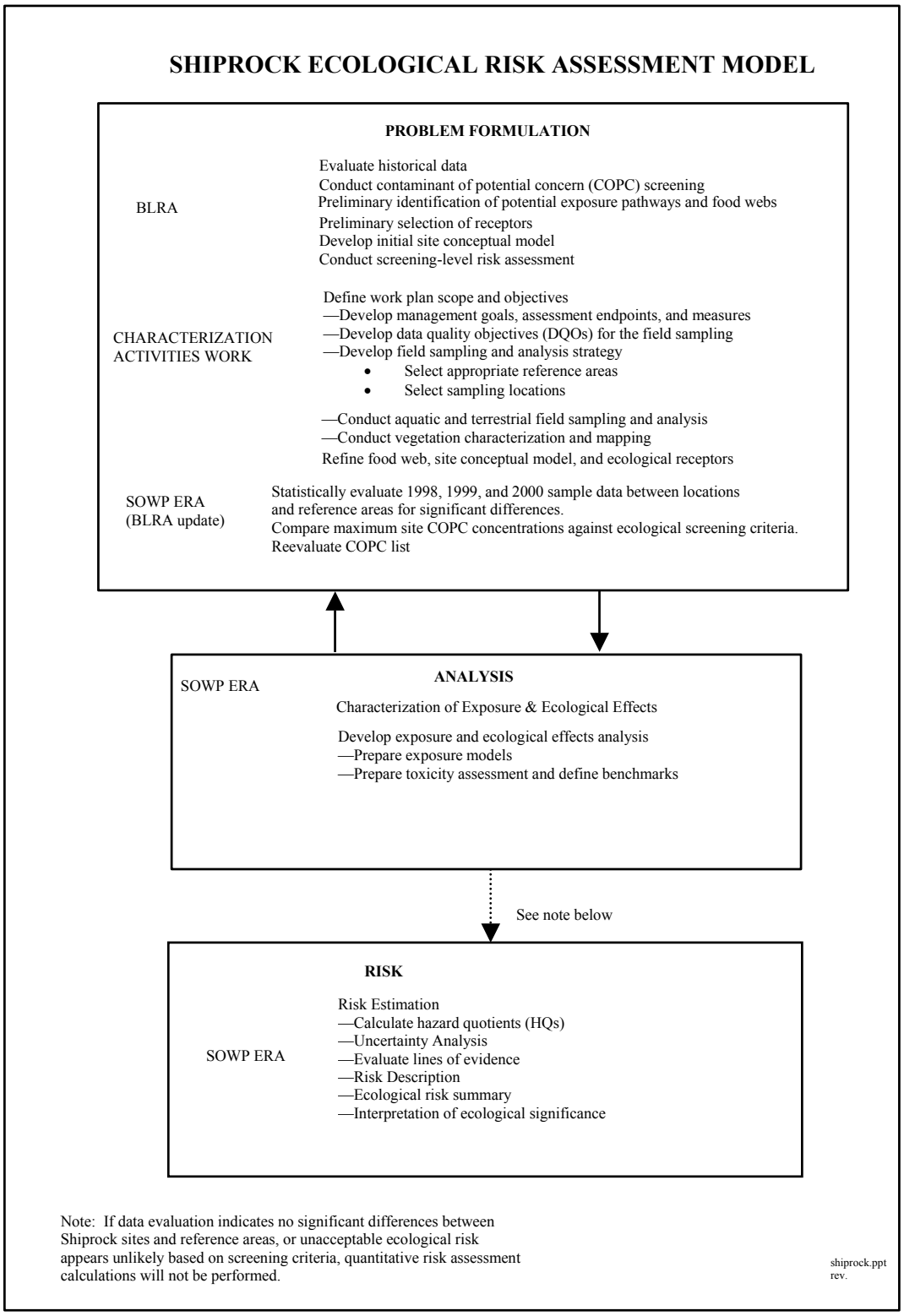

Figure 6–2. Ecological Risk Model for the Shiprock Site

measurement endpoints are defined for the specific determination of risk to these receptors and the environmental resources they represent. These endpoints are directly tied to overall management goals for the site.

The analysis phase of the ERA includes two concurrent steps—the exposure assessment and the effects characterization. In the exposure assessment, the potential for each receptor to be exposed to each stressor is evaluated and, where possible, quantified. The effects characterization describes the potential for the stressor to adversely affect the receptors that are exposed to it. Because the stressors at the Shiprock site are chemical in nature, the principal effects to ecological receptors will be toxicological; however, they may also include physical effects, such as those related to high osmotic potentials and those related to radiation.

The risk characterization phase evaluates (either qualitatively or quantitatively) the combined results of the exposure assessment and effects characterization to determine the potential for risk to the receptors due to their exposure to the stressors. A critical aspect of the risk characterization is the analysis of uncertainties associated with predictions of potential risk. Typically, uncertainties result from data gaps which necessitate the incorporation of assumptions into the analysis and risk characterization phases. In general, these assumptions are conservatively biased toward results that will lead to overestimations rather than underestimations of risk. The uncertainty analysis provides an analysis of these assumptions in terms of their potential for introducing significant bias in the risk estimation.

As described in the EPA guidance (EPA 1998), ERA is an iterative process in which the evaluation of potential risks to ecological receptors is refined as additional data are collected to fill data gaps and reduce uncertainties. At the conclusion of each iteration (or "tier") in the process, decisions are made whether sufficient data have been collected and analyzed to proceed with risk management actions (if required), or whether additional data should be collected. Such a tiered approach to the ERA process was initiated at the Shiprock site in 1994 by the performance of the screening-level BLRA (DOE 1994) and the BLRA supplement (DOE 1996d). Subsequently, additional data have been collected from key environmental media specifically for the purpose of characterizing potential ecological risk. The ERA presented here provides an analysis of these new data as a refinement of the 1994 screening-level assessment and the 1996 BLRA supplement. The recent data collection activities were of two types—visual surveys of biotic communities present at the site and sampling of key environmental media for chemical analysis. The visual surveys (conducted in 1998 and 1999) focused on the identification of special status species, such as threatened and endangered (T&E) species, and the presence of potential habitat for these species at and in the vicinity of the site. Sampling of surface water, sediment, soil, and vegetation for chemical analysis was conducted in 1998, 1999, and 2000, as discussed in Section 4.6, "Ecological Field Investigations."

The overall goal of this risk assessment is to identify ecological COPCs that can be related to the dispersal of contaminants in the ground water underlying the millsite and to characterize the potential for adverse effects of these COPCs on the ecosystem at the Shiprock site based on the results of the recent sampling activities. In particular, potential effects on special status species and sensitive environments are considered. This assessment is an update and expansion of the BLRA screening-level assessment conducted in 1994 (DOE 1994) and the BLRA supplement (DOE 1996d); however, it is still primarily a screening assessment to identify those COPCs and areas for which interim remedial actions may be appropriate and for which future monitoring may be necessary.

6.2.1 Problem Formulation

The problem formulation phase in this risk assessment is represented in part by the information presented in the BLRA (DOE 1994). The BLRA was based on analytical data collected at the Shiprock site prior to 1994. These data were reviewed to determine if concentrations of analytes in ground water, surface water, and sediment may pose a potential ecological risk. Information on the geologic setting, ground water hydrology, geochemistry, and habitats of the Shiprock site were incorporated in the BLRA evaluation. Principal results of the BLRA included an initial screening of chemical analytes as COPCs and an assessment of potential risk to biota, including livestock and irrigated crops. The assessment of potential risk, however, was primarily qualitative. The BLRA provided a basis for the preparation of a characterization work plan (DOE 1998c).

Since the completion of the BLRA, additional abiotic and biotic samples have been collected at Shiprock and at upgradient reference areas. These new analytical data are limited to data obtained from the GJO Analytical Chemistry Laboratory. All available data gathered specifically for the ERA, which include the September 1998, June 1999, September 1999, and March 2000 sampling efforts, have been included in this update. Any other surface data collected after March 2000 will be addressed in the EA for the Shiprock site. All summary statistics and calculations for this ERA are included in Appendix H.

6.2.1.1 Potentially Affected Habitats and Populations

The habitats of the Shiprock site are characterized by the arid climate of the region, the sedimentary geology, and the fluvial processes associated with the San Juan River and its tributaries. The erosional escarpment that runs to the east and north of the disposal cell and the former millsite is a sharp ecological boundary between the arid upland terrace on which the millsite and disposal cell are located and the San Juan River floodplain and channel. The natural vegetation of the upland is Great Basin Desertscrub, although much of the area is urbanized/ industrialized and much of the natural habitat is altered or highly disturbed. To the north of the former millsite, a wide, northerly bend in the San Juan River has left an area (approximately 100 acres) of floodplain between the escarpment and the river channel. The history of this floodplain area with respect to millsite activities is described in Section 3. The vegetation of the floodplain is discussed in Section 4.6.1 and a map of the vegetation of the floodplain is presented in Figure 4–76. The predominant vegetation type of the floodplain is riparian, dominated by saltcedar; however, large areas of wetland and upland vegetation types also occur on the floodplain. Much of the southeastern portion of the floodplain area was disturbed by soil removal during remediation.

Bob Lee Wash and Many Devils Wash are the principal surface drainages west and east (respectively) of the millsite and disposal cell. Bob Lee Wash descends the upland terrace through an eroded ravine and discharges directly onto the floodplain. In the past, discharge from artesian well 648 west of Bob Lee Wash flowed directly into Bob Lee Wash, creating a permanent flow in the lower part of the ravine and creating a 5-acre wetland vegetated by cattails and bulrushes on the floodplain at the mouth of the wash. Discharge from this well has recently been impounded in a small pond on the terrace west of the wash. However, because this small impoundment has a constant inflow of water, but has not filled to capacity, it is likely that infiltration of this water into the ground equals the outflow from the well. Seeps along Bob Lee Wash that continue to feed the flow into the wetland and on the floodplain west of the wetland

may be fed from this infiltration and subsequent percolation of the well water through the terrace gravels and weathered Mancos Shale. North of the wetland, a drainage channel that crosses the floodplain collects outflow from the wetland area and conducts it westward to the northwest corner of the floodplain where it discharges into the San Juan River.

Seeps occur along the base of the escarpment that are fed by terrace ground water. These seeps and other near-surface occurrences of ground water commonly support areas of inland saltgrass. Seep 426, which is just east of the mouth of Bob Lee Wash, flows directly into the wetland.

Many Devils Wash is an ephemeral drainage that descends the upland terrace through a long ravine and discharges directly into the San Juan River just upstream of the southeastern corner of the floodplain area. Most of the channel exhibits surface salt crusts and is sparsely vegetated. The lower portion of the channel (below a distinct knickpoint in the channel) contains standing pools of water, indicating probable contact with ground water. Heavy salt crusts occur along this lower portion of the wash channel, which is devoid of vegetation except at the mouth of the wash near the river where some salt cedars occur.

To the west of U.S. Highway 666, the escarpment gradually disappears and the upland terrace descends more gradually down to the river level. Irrigated fields occur in this lower part of the terrace. West of the U.S. Highway 666 bridge, the San Juan River turns north and again makes a large northerly bend, creating another large area of exposed floodplain to its south. The southern edge of this floodplain (along the base of the escarpment) is marked by a distributary channel that periodically carries river water when the river stage is high. This channel is lined with riparian vegetation (principally saltcedar). Two small washes (1st Wash and 2nd Wash) feed northward into the distributary channel from the terrace. Farther west, an irrigation return flow ditch runs along the base of the remaining escarpment and discharges into the distributary channel. Above this confluence, another small wash (3rd Wash) feeds into the irrigation ditch from the terrace. Seeps and permanent standing water along these various drainages indicate possible contact with ground water from the terrace.

Because surface and near-surface expressions of the contaminated ground water are limited to areas below the escarpment (or ravines deeply incised into the escarpment), the sites that have potential ecological pathways are generally limited to these areas. The areas and media currently considered to have potential ecological pathways, shown in Plate 5, are:

- Area A—The distributary channel and tributaries: surface water and sediment
- Area B—San Juan River (between Many Devils Wash and the distributary channel): surface water and sediment
- Area C—The Shiprock floodplain and associated wetlands: surface water, sediment, soil, plants, and ground water
- Area D—Bob Lee Wash: surface water, sediment, soil, plants, and ground water
- Area E—Many Devils Wash: surface water and sediment
- Area F—The upland (millsite) terrace: deep-rooted plants (contacting ground water)

Seasonally wet sediments and adjacent soils often form surficial salt crusts. These salt crusts are considered to be part of the sediment or soil at each of the sites and are not treated as a separate medium.

General descriptions of the ecological communities existing in these areas are in the BLRA (DOE 1994) and in Section 4.6 of this report. The terrestrial (upland) habitats of the Shiprock site are on the terrace and on portions of the floodplain, especially in the southeastern section and near the base of the escarpment. These habitats are greatly influenced by the low annual precipitation and grazing pressures. Desert shrubs and annual weeds typically dominate the vegetation in these habitats, with little or no grassy understory. On the disposal cell terrace, ground cover is sparse and only a few scattered greasewood shrubs, which are phreatophytes with root systems capable of tapping into the terrace ground water system, are present. Annual plant cover consists primarily of halogeton (*Halogeton spp.*), Russian thistle, kochia, and other weedy species that are not likely to have root systems capable of reaching the ground water. The area around the disposal cell is currently fenced to deter entry by livestock and large terrestrial wildlife species.

Riparian habitats are associated with the San Juan River, the distributary channel, the lower part of Bob Lee Wash, and the various drainage channels that cross the floodplain. Tree cover in these habitats is dominated by saltcedar, but also includes cottonwood and Russian olive. Terrestrial wildlife such as foxes, coyotes, skunks, raccoons, deer, and rodents likely use the riparian habitats for foraging, resting, denning, and other activities. Evidence of beaver use is prevalent in this habitat near the San Juan River. The canopy of the riparian vegetation can also be attractive to a wide variety of birds for feeding, nesting, perching, roosting, and other activities. Birds of the riparian habitats include resident and migratory species. Drinking water sources are commonly available in or near these habitats, adding to their attractiveness to wildlife.

The wetland habitat at the mouth of Bob Lee Wash is principally fed by the outflow from artesian well 648. Much of this area was delineated as jurisdictional wetland by the wetland assessment of the Shiprock site (DOE 1998a). As described in the BLRA (DOE 1994), this wetland and the flows above and below it support a diverse aquatic community, including topminnows (e.g., *Gambusia*), bullfrogs, waterfleas, snails, and a variety of insects (caddis flies, dragonflies, water boatmen, backswimmers, mosquito larvae, damselflies, lacewings, and midge larvae). It is not known whether muskrat inhabit this wetland; however, it is probably used by migrating, and possibly breeding, waterfowl. The aquatic habitat of the San Juan River is also used by waterfowl such as ducks, geese, herons, and egrets.

In 1998 and 1999, surveys for T&E and other sensitive species were conducted at the Shiprock site. The first T&E survey (Ecosphere Environmental Services 1998) was conducted in summer 1998 to determine the potential effects that could occur because of proposed monitor well installation activities. Because most of the proposed wells were at terrace locations, those areas were the primary focus. A second survey (Ecosphere Environmental Services 1999) was conducted in winter 1999 to support proposed well installations and a water distribution system in the floodplain, which was the area of focus.

The surveys found that the Mesa Verde cactus is present in the terrace region but does not have suitable habitat in the floodplain. Western burrowing owls were also observed in the terrace areas. Though not observed, suitable habitat exists in the floodplain area for the southwestern

willow flycatcher. An aquatic survey was not conducted, but the San Juan River is known to be within federally designated Critical Habitat for the Colorado pikeminnow and razorback sucker. Roundtailed chub are also known to be present in the river. To date, there is no recorded evidence that site-related contamination has had adverse effects on any T&E species. Table 6–14 presents a summary of the results of the two T&E surveys.

Table 6–14. Summary of T&E Survey Results

Flora/Fauna	Potential Habitat?		Species Observed?	
	Terrace	Floodplain	Terrace	Floodplain
Bald eagle	Y	N	N	N
Southwestern willow flycatcher	N	Y	N	N
Peregrine falcon[a]	N	N	N	N
Mountain plover	Y	N	N	N
Ferruginous hawk	N	N	N	N
Golden eagle	N	N	N	N
Western burrowing owl	Y	N	Y	N
Rough-legged hawk	Y	N	N	N
Pronghorn	Y	N	N	N
Black-footed ferret	N	N	N	N
Colorado pikeminnow	San Juan River		NA[b]	NA
Razorback sucker	San Juan River		NA	NA
Roundtailed chub	San Juan River		NA	NA
Northern leopard frog	N	Y	N	N
Mesa Verde cactus	Y	N	Y	NA

[a]delisted August 1999.
[b]NA = not available.

6.2.1.2 Summary of the 1994 Ecological Risk Assessment Results

In the 1994 BLRA (DOE 1994), the list of ground water constituents that were present in elevated levels in ground water (based on statistical comparisons between on-site and upgradient well data) was used a starting point for identifying COPCs in those media for which ecological pathways may exist. At that time, the extent of contamination was thought to be limited to the millsite terrace, the floodplain, and the adjacent reach of the San Juan River. The media of concern included ground water, surface water, and sediment from these areas. The water quality of samples from upgradient wells was considered to be representative of background conditions for the floodplain aquifer. These ground-water-based COPCs are listed in Table 6–15. Of this list, ecological COPCs were defined in the BLRA as those constituents that were detected in the surface water or sediment of the San Juan River or the floodplain at concentrations exceeding those in upgradient samples. These site and media-specific ecological COPCs are indicated on Table 6–15. For the floodplain, calcium, chloride, phosphate, potassium, and zinc were excluded as ecological COPCs because they were within nutritional ranges. Similarly, ammonium, boron, and nickel were excluded due to their low toxicities, allowing high dietary intakes by ecological receptors without a significant toxic response. For the BLRA, however, data were not available for Many Devils Wash, the distributary channel, or the irrigation return flow ditch, and these sites were not included in the evaluation of potential ecological risk.

Table 6–15. Summary of Ecological Contaminants of Potential Concern in Ground Water, Surface Water, and Sediments (1994 BLRA)

Constituents Above Background in Ground Water[a]	Constituents Detected in San Juan River Water[b]	Constituents Detected in San Juan River Sediments[b]	Constituents Detected in Floodplain Surface Water and Sediments[c]
Ammonium[d]			
Antimony	X		
Arsenic	X	X	
Boron[d]			
Cadmium			
Calcium[e]			
Chloride[e]			
Magnesium	X		
Manganese		X	X
Nickel[d]			
Nitrate			X
Phosphate[e]			
Polonium-210[f]			
Potassium[e]			
Radium-226[f]		X	
Selenium			X
Sodium	X		
Strontium	X	X	X
Sulfate	X		
Thorium-230[f]	X		
Uranium		X	X
Zinc[e]			

[a]Ground water constituents that exceeded background at the 0.1 significance level.
[b]Ground water constituents were excluded that were either not detected in surface water or sediment or the median concentration adjacent to and downgradient from the site was less than concentrations upgradient of the site.
[c]Selection of constituents analyzed from floodplain pond water and sediment was not based on a comparison to reference areas.
[d]Analyte was considered to be of low toxicity, and therefore, of low potential ecological risk.
[e]Analyte was considered to be within nutritional range on the floodplain.
[f]Analyte is a radioactive decay product of uranium-238

Sampling results for the identified ecological COPCs were compared to applicable water and sediment quality criteria, as available. For ecological risks to occur at the Shiprock site, pathways must exist for exposure of biological receptors to biotic and abiotic media contaminated by ground water. This screening-level assessment of ecological risks evaluated COPCs, potential pathways, receptors, and potential adverse effects (DOE 1994).

Results of the 1994 BLRA indicated that the only risks associated with site-related contamination would be to animals that used surface water from seep 425. The primary risk is associated with exposures to sulfate and nitrate. Concentrations were present that were at acutely toxic levels. Samples from the floodplain area and the San Juan River locations did not exceed any water or sediment quality criteria, though few criteria were found for the identified COPCs. Because soil-to-plant and water-to-plant concentration data were unavailable for many COPCs, the BLRA concluded that it was not possible with existing data to evaluate whether plant tissue

concentrations are phytotoxic or could result in adverse effects to animals foraging on contaminated vegetation. Ground water was unsuitable for crop irrigation because of elevated levels of boron, manganese, and selenium, and because of high salinity. With the exception of the water at seep 425, livestock and wildlife watering on the floodplain would experience no adverse effects. However, ground water pumped into ponds from the most contaminated wells would be unsuitable for livestock and fish. The BLRA also concluded that the potential for COPCs to represent a food chain hazard (via bioaccumulation and biomagnification) was also low, though no tissue samples were analyzed.

Insufficient water quality criteria and sediment quality criteria were available to conduct a thorough evaluation of the adverse effects of surface water, sediment, ground water, and plant uptake on ecological receptors. Additional characterization and evaluation were recommended.

6.2.1.3 Update of the Ecological COPCs

For the current risk assessment, additional data collected and information received subsequent to the issuance of the BLRA were used to reevaluate the list of ecological COPCs that will be further evaluated for potential ecological risk. The initial list of constituents in millsite and floodplain ground water that were identified in the BLRA as exceeding reference site concentrations (as listed in Table 6–15) were retained for reevaluation. Because of their possible occurrence in uranium ore or other process sources, chromium, cobalt, copper, iron, lead, molybdenum, and vanadium were also reevaluated as possible COPCs based on their current data records. However, none of these constituents exceeded reference concentrations in ground water based on the statistical analyses performed for the BLRA.

Constituents that are considered to be essential nutrients (as recognized in EPA 1989b) were excluded as ecological COPCs. These included calcium, magnesium, potassium, and sodium. Among the constituents that were excluded from consideration as ecological COPCs in the BLRA because of their low potential toxicities, chloride and phosphate are still excluded as COPCs for the same reason. It is recognized, however, that at high concentrations in water, these anions and the four cations considered to be essential nutrients can contribute to adverse ecological effects due to high osmotic potentials, and some can affect the use of water by wildlife and livestock by imparting strong tastes to the water. These types of effects, however, are not addressed in this risk assessment.

Sulfate is also an anion of relatively low potential toxicity in biota. High sulfate in water is known to cause diarrhea in humans and livestock; however, some evidence indicates that this effect is temporary and the individual will acclimate to the high sulfate ingestion without long-term adverse effect (EPA 1999). Sulfate-based salts are commonly used to test the toxicity of cationic elements, indicating a general lack of toxic potential of the sulfate anion, which would otherwise interfere with the test results. However, because of its high concentrations in the ground water associated with the millsite, sulfate has not been excluded from consideration as an ecological COPC.

The radioactive elements in the decay chain of uranium-238 that have sufficiently long half-lives to accumulate at detectable levels in the environment are not specifically included in this evaluation. These radionuclides, which include thorium-230, radium-226 (including radium-228), polonium-210, and lead-210, are assumed to be COPCs at all sites where uranium is identified as a COPC; however, the principal risk to ecological receptors from these elements is from radiation resulting from their decay rather than their individual chemical toxicities.

Ecological COPCs were identified from the remaining list of constituents based on their detection in recent analytical data from the Shiprock site and comparisons of these detected concentrations to equivalent data from a designated reference site or other background values. These comparisons to background were performed separately for each of the six areas where ecological pathways may exist (as described in Section 6.2.1.2). "Recent" data was considered to be data from samples collected in 1998, 1999, and 2000, or the most recent year for which data are available for the analyte. Because a relatively small number of data points were available for the reference locations, statistical comparisons were not possible. Therefore, a comparison of maximum values was used to identify COPCs. A constituent was retained as an ecological COPC if the maximum concentration detected in the surface water, sediment, soil, or (in the case of the upland terrace) vegetation samples from the site in question was greater than the maximum detected reference site concentration.

In some cases, a lack of detections was the criterion for eliminating a constituent from further consideration as an ecological COPC. As described in Section 6.1.2.1, the 1998–2000 sampling of the floodplain alluvial ground water showed that concentrations of antimony, arsenic, and cadmium were at or below their respective detection limits and were therefore dropped from consideration as COPCs in the human health risk assessment. For the same reason, these constituents are also dropped from consideration as ecological COPCs. Concentrations of chromium, cobalt, copper, lead, nickel, and zinc in filtered surface water samples from the Shiprock site have been at or below their respective detection limits, and these constituents were also not retained as ecological COPCs. Boron, molybdenum, iron, and vanadium have generally few detections in the sampling record for the Shiprock site, making their identification as millsite-related contaminants questionable. Molybdenum, iron, and vanadium are possible constituents of uranium ore and were therefore retained as potential COPCs pending comparison to reference data. In the case of boron, the highest detections are limited to water at seeps 425 and 426 (on the Shiprock floodplain); however, because there are no analyses of boron in reference samples to provide a comparative value for dropping this constituent as a COPC at this time, boron was retained as an ecological COPC for the floodplain (see Area C in Plate 5).

For the water and sediment data from the San Juan River samples, the 1998 and 1999 data from upstream sampling locations 888 and 898 were used as reference data. For the water and sediment data from the sites associated with the escarpment outflow (i.e., Many Devils Wash, Bob Lee Wash, the floodplain seeps, 1st Wash, 2nd Wash, 3rd Wash, the distributary channel, and the irrigation return flow ditch), the data from the water and sediment samples associated with the outflow of well 648 were used as the reference data. For the soil samples from the floodplain, the data from the soil samples collected from the upgradient floodplain (east of Many Devils Wash) were used as the reference data. The water from well 648 and the San Juan River water below the Chaco River may not be representative of average regional background conditions; however, they do represent local variability in analytes within a region that is strongly influenced by sedimentary geology.

For the vegetation samples from the upland (millsite) terrace, conspecific vegetation samples from the terrace east (upgradient) of the millsite were used as the reference. Although the 1998–2000 ecological field sampling included other vegetation samples (e.g., plants from the wetland, riparian zone, and distributary channel), these were not used to identify COPCs because data from other accessible media (surface water, sediment, and soil) are available for these areas and direct contact

between the vegetation and these media is known to occur. These data are incorporated in the evaluation of risk associated with the ecological COPCs identified for the areas.

The following sections describe the area-specific (Plate 5) identification of ecological COPCs for the Shiprock site.

Area A: The Distributary Channel and Tributaries. The ecological COPCs identified for the distributary channel surface water and sediment are ammonium, manganese, molybdenum, nitrate, selenium, strontium, sulfate, uranium, and vanadium (Table 6–16). Recent surface water data indicate that iron concentrations are within the range of the reference data, and zinc concentrations are below its detection limit in surface water in Area A. Although no data have been collected for lead-210 and thorium-230, the low concentrations of polonium-210, radium-226, and radium-228 at this area indicate that the concentrations of these radionuclides are not elevated.

Area B: The San Juan River. The ecological COPCs identified for the San Juan River surface water and sediment are ammonium, manganese, molybdenum, nitrate, selenium, strontium, sulfate, uranium, and vanadium (Table 6–17). Based on recent data from filtered water samples, boron, chromium, cobalt, copper, lead, nickel, and zinc have been near or below detection limits and have not been included as COPCs for this area. Although iron has been detected in some water samples from the San Juan River, only one of these detections exceeded the range of reference site data. Based on the low frequency of detection and generally low concentrations when detected, iron was not included as an ecological COPC for this area. Data for the radiological constituents indicate that lead-210, radium-226, and thorium-230 concentrations may be elevated in this area, although reference data for lead-210 and thorium-230 are not available.

Area C: The Shiprock Floodplain. The ecological COPCs identified for the floodplain area surface water, sediment, and soil are ammonium, boron, manganese, molybdenum, nitrate, selenium, strontium, sulfate, uranium, and vanadium (Table 6–18). With the exception of one detection in 1986, chromium concentrations have been below the detection limit in all surface water samples from the floodplain, including seeps 425 and 426. Maximum manganese, selenium, strontium, and uranium concentrations in sediment samples from the floodplain exceed those from the reference samples. Ammonium, nitrate, sulfate, and uranium concentrations were also elevated in the soil. All radiological constituents except polonium-210 were identified as COPCs. Reference data are not available for lead-210.

Area D: Bob Lee Wash. The ecological COPCs identified for Bob Lee Wash surface water, sediment, and soil are ammonium, manganese, molybdenum, nitrate, selenium, strontium, sulfate, uranium, and vanadium (Table 6–19). Chromium, iron, lead, nickel, and zinc concentrations are below detection limits in the surface water at this site. Manganese concentrations were elevated in sediment samples from Bob Lee Wash. Ammonium, manganese, nitrate, selenium, strontium, sulfate, and uranium concentrations were elevated in the soil. Of the radiological constituents, radium-226 and radium-228 were detected at a concentration exceeding the reference data; however, lead-210 was also identified as a COPC because no reference data are available for this constituent.

Table 6–16. Summary of Ecological Contaminants of Potential Concern in Area A: the Distributary Channel and Tributaries (based on data collected from 1998 through 2000)

Constituent	Maximum Concentration in Surface Water[a]		Maximum Concentration in Sediments[b]		Selected as COPC?	Reason
	Site	Ref.	Site	Ref.		
Nonradiological Constituents						
Ammonium	3.0	0.6	49.1	8.3	Yes	Elevated in water and sediment
Antimony	< 0.0024	<0.003	0.32	0.18	No	Ground water data do not indicate antimony is a millsite contaminant. Not detected in surface water.
Arsenic	< 0.001	0.008	6.4	7.8	No	Ground water data do not indicate arsenic is a millsite contaminant. Not detected in surface water.
Boron	ND	ND	ND	ND	No	Not expected to be a COPC based on low concentrations at Shiprock floodplain.
Cadmium	< 0.001	0.001	0.81	ND	No	Ground water data do not indicate cadmium is a millsite contaminant. Not detected in surface water.
Calcium	586	212	ND	ND	No	Nutrient
Chloride	252	57.7	ND	ND	No	Low toxicity
Chromium	ND	< 0.01	ND	ND	No	Not expected to be a COPC based on data from the Shiprock floodplain.
Cobalt	ND	< 0.05	ND	ND	No	Not expected to be a COPC based on data from the Shiprock floodplain.
Copper	ND	< 0.02	ND	ND	No	Not expected to be a COPC based on data from the Shiprock floodplain.
Iron	0.197	0.98	ND	ND	No	Within range of reference samples.
Lead	ND	< 0.003	ND	ND	No	Not expected to be a COPC based on data from the Shiprock floodplain.
Magnesium	787	12.8	12,200	13,200	No	Nutrient
Manganese	1.90	0.5	368	365	Yes	Elevated in water and sediment.
Molybdenum	0.0197	< 0.01	ND	ND	Yes	Elevated in water.
Nickel	ND	ND	ND	ND	No	Not expected to be a COPC based on data from the Shiprock floodplain.
Nitrate	515	23.9	243	189	Yes	Elevated in water and sediment.
Phosphate	ND	ND	ND	ND	No	Low toxicity
Potassium	23.0	8.63	ND	ND	No	Nutrient
Selenium	0.428	< 0.005	23.1	1.0	Yes	Elevated in water and sediment.
Sodium	1,030	1,400	504	1,880	No	Nutrient
Strontium	9.78	13.5	407	312	Yes	Elevated in sediment.
Sulfate	5,670	2,520	27,800	1,890	Yes	Elevated in water and sediment.
Uranium	0.102	< 0.001	5.4	4.5	Yes	Elevated in water and sediment.
Vanadium	0.0025	< 0.01	ND	ND	Yes	Elevated in water.
Zinc	< 0.0076	0.01	ND	ND	No	Not detected in water.
Radiological Constituents						
Lead-210	ND	ND	ND	ND	No	Not expected to be a COPC based on information for other uranium daughters.
Polonium-210	< 0.2	1.2	ND	ND	No	Not detected in water.
Radium-226	0.83	1.0	1.22	2.68	No	Within background range.
Radium-228	< 0.97	1.1	ND	ND	No	Not detected in water.
Thorium-230	ND	1.0	1.5	3.9	No	Sediment within background range.

[a]In mg/L for nonradiological constituents and pCi/L for radiological constituents.
[b]In mg/kg for nonradiological constituents and pCi/g for radiological constituents.
ND = no data

Table 6–17. Summary of Ecological Contaminants of Potential Concern in Area B: the San Juan River (based on data collected from 1998 through 2000)

Constituent	Maximum Concentration in Surface Water[a]		Maximum Concentration in Sediment[b]		Selected as COPC?	Reason
	Site	Ref.	Site	Ref.		
Nonradiological Constituents						
Ammonium	0.164	0.0444	1.8	1.0	Yes	Elevated in water and sediment.
Antimony	< 0.0022	<0.001	0.17	0.16	No	Ground water data do not indicate antimony is a millsite contaminant.
Arsenic	0.001	< 0.001	1.0	0.78	No	Ground water data do not indicate arsenic is a millsite contaminant.
Boron	< 0.1	ND	ND	ND	No	Not detected in recent samples.
Cadmium	< 0.001	< 0.001	0.14	0.21	No	Ground water data do not indicate cadmium is a millsite contaminant.
Calcium	398	92.1	ND	ND	No	Nutrient
Chloride	125	43.0	ND	ND	No	Low toxicity
Chromium	< 0.01	ND	ND	ND	No	Not detected in most recent water samples
Cobalt	< 0.05	ND	ND	ND	No	Not detected in recent water samples
Copper	0.02	ND	ND	ND	No	At or below detection limit in recent water samples
Iron	0.0865	< 0.0404	ND	ND	No	Only one recent sample detected above range of reference data
Lead	0.02	ND	ND	ND	No	Water data near or below detection limit
Magnesium	201	42.4	683	640	No	Nutrient
Manganese	0.592	0.362	229	176	Yes	Elevated in water and sediment
Molybdenum	0.0099	0.0075	< 27.0	ND	Yes	One sample exceeded the maximum reference concentration in water. Molybdenum may be associated with uranium ore.
Nickel	< 0.04	ND	ND	ND	No	Not detected in water samples
Nitrate	104	3.63	39.0	26.2	Yes	Elevated in water and sediment
Potassium	13.7	6.5	ND	ND	No	Nutrient
Selenium	0.0787	0.0018	< 0.2	< 0.2	Yes	Elevated in water.
Sodium	1,330	129	581	241	No	Nutrient
Strontium	14.6	1.55	45.0	36.2	Yes	Elevated in water and sediment.
Sulfate	4,190	491	2,660	1,950	Yes	Elevated in water and sediment.
Uranium	0.112	0.0038	0.25	0.21	Yes	Elevated in water and sediment.
Vanadium	0.0017	0.0012	19.4	ND	Yes	Elevated in water.
Zinc	< 0.0076	ND	ND	ND	No	Not detected in recent water samples
Radiological Constituents						
Lead-210	2.6	ND	ND	ND	Yes	No reference site data.
Polonium-210	< 0.52	< 0.26	ND	ND	No	Not detected in water samples.
Radium-226	0.68	< 0.14	1.71	ND	Yes	Elevated in water.
Radium-228	< 1.0	< 0.84	ND	ND	No	Not detected in water samples.
Thorium-230	0.5	ND	ND	ND	Yes	No reference site data.

[a]In mg/L for nonradiological constituents and pCi/L for radiological constituents.
[b]In mg/kg for nonradiological constituents and pCi/g for radiological constituents.
ND = no data

Table 6–18. Summary of Ecological Contaminants of Potential Concern in Area C: the Shiprock Floodplain (based on data collected from 1998 through 2000)

Constituent	Maximum Concentration in Surface Water[a]		Maximum Concentration in Sediment or Soil[b]		Selected as COPC?	Reason
	Site	Ref.	Site	Ref.		
Nonradiological Constituents						
Ammonium	1.02	0.6	25.9	8.9	Yes	Elevated in water and soil
Antimony	< 0.0041	<0.003	0.38	0.19	No	Ground water data do not indicate antimony is a millsite contaminant.
Arsenic	0.0072	0.008	8.5	7.8	No	Ground water data do not indicate arsenic is a millsite contaminant.
Boron	0.56	ND	ND	ND	Yes	Boron detected at seeps 425 and 426. No data from reference locations.
Cadmium	0.0016	0.001	1.17	0.12	No	Ground water data do not indicate cadmium is a millsite contaminant.
Calcium	478	212	ND	ND	No	Nutrient
Chloride	547	57.7	ND	ND	No	Low toxicity
Chromium	< 0.01	< 0.01	ND	ND	No	Not detected in recent water samples
Cobalt	< 0.05	< 0.05	ND	ND	No	Not detected in recent water samples
Copper	< 0.021	< 0.02	ND	ND	No	Not detected in recent water samples
Iron	0.402	0.98	ND	ND	No	Detections within range of reference data
Lead	< 0.03	< 0.003	ND	ND	No	Not detected in recent water samples
Magnesium	757	14.7	14,300	17,300	No	Nutrient
Manganese	16.4	0.5	1,190	365	Yes	Elevated in water and sediment
Molybdenum	0.0124	< 0.023	ND	ND	Yes	Background concentrations uncertain. Retained because molybdenum may be associated with uranium ore.
Nickel	< 0.061	ND	ND	ND	No	Not detected in water samples
Nitrate	2,460	23.9	1,010	189	Yes	Elevated in water and soil
Potassium	44.0	8.63	ND	ND	No	Nutrient
Selenium	0.137	< 0.005	4.2	1.0	Yes	Elevated in water and sediment.
Sodium	7,320	1,400	2,430	1,880	No	Nutrient
Strontium	19.8	13.5	1,620	312	Yes	Elevated in water and sediment
Sulfate	17,100	2,520	42,300	1,890	Yes	Elevated in water and soil
Uranium	0.682	< 0.001	43.5	4.5	Yes	Elevated in water, sediment, and soil
Vanadium	0.0056	< 0.01	ND	ND	Yes	Background concentrations uncertain. Retained because vanadium may be associated with uranium ore.
Zinc	< 0.0076	0.01	ND	ND	No	Not detected in recent water samples
Radiological Constituents						
Lead-210	0.7	ND	ND	ND	Yes	No reference site data.
Polonium-210	< 0.53	1.2	ND	ND	No	Within background range.
Radium-226	0.99	1.0	12.87	2.68	Yes	Elevated in sediment
Radium-228	1.13	1.1	ND	ND	Yes	Elevated in water.
Thorium-230	< 1.2	1.0	84.0	3.9	Yes	Elevated in sediment.

[a]In mg/L for nonradiological constituents and pCi/L for radiological constituents.
[b]In mg/kg for nonradiological constituents and pCi/g for radiological constituents.
ND = no data

Table 6–19. Summary of Ecological Contaminants of Potential Concern in Area D Bob Lee Wash (based on data collected from 1998 through 2000)

Constituent	Maximum Concentration in Surface Water[a]		Maximum Concentration in Sediment or Soil[b]		Selected as COPC?	Reason
	Site	Ref.	Site	Ref.		
Nonradiological Constituents						
Ammonium	5.0	0.6	14.7	8.9	Yes	Elevated in water and soil
Antimony	< 0.0011	<0.003	< 0.1	0.19	No	Ground water data do not indicate antimony is a millsite contaminant.
Arsenic	< 0.001	0.008	1.72	7.8	No	Ground water data do not indicate arsenic is a millsite contaminant.
Boron	0.1	ND	ND	ND	No	The detected value is at detection limit.
Cadmium	< 0.001	< 0.001	0.47	0.12	No	Ground water data do not indicate cadmium is a millsite contaminant.
Calcium	619	212	ND	ND	No	Nutrient
Chloride	332	57.7	ND	ND	No	Low toxicity
Chromium	< 0.01	< 0.01	ND	ND	No	Not detected in recent water samples
Cobalt	ND	< 0.05	ND	ND	No	Not expected to be a COPC based on information from the Shiprock floodplain.
Copper	ND	< 0.02	ND	ND	No	Not expected to be a COPC based on information from the Shiprock floodplain.
Iron	< 0.0592	0.98	ND	ND	No	Not detected in recent water samples.
Lead	< 0.03	< 0.003	ND	ND	No	Not detected in recent water samples.
Magnesium	506	14.7	11,000	17,300	No	Nutrient
Manganese	0.0568	0.5	533	365	Yes	Elevated in sediment
Molybdenum	0.0205	< 0.023	ND	ND	Yes	Background concentration uncertain. Retained because molybdenum may be associated with uranium ore.
Nickel	< 0.04	ND	ND	ND	No	Not detected in water samples
Nitrate	2,110	23.9	1,120	189	Yes	Elevated in water and soil
Potassium	26.1	8.63	ND	ND	No	Nutrient
Selenium	0.119	< 0.005	0.57	1.0	Yes	Elevated in water.
Sodium	1,290	1,400	3,710	1,880	No	Nutrient
Strontium	12.5	13.5	407	312	Yes	Elevated in soil.
Sulfate	12,900	2,520	50,200	1,890	Yes	Elevated in water and soil
Uranium	2.42	< 0.001	40.2	4.5	Yes	Elevated in water and soil
Vanadium	0.0052	< 0.01	ND	ND	Yes	Background concentration uncertain. Retained because vanadium may be associated with uranium ore.
Zinc	< 0.05	0.01	ND	ND	No	Not detected in recent water samples
Radiological Constituents						
Lead-210	0.3	ND	ND	ND	Yes	No reference site data.
Polonium-210	< 0.29	1.2	ND	ND	No	Not detected in recent water samples.
Radium-226	1.88	1.0	ND	2.68	Yes	Elevated in water.
Radium-228	1.17	1.1	ND	ND	Yes	Elevated in water.
Thorium-230	ND	1.0	ND	3.9	No	No data.

[a]In mg/L for nonradiological constituents and pCi/L for radiological constituents.
[b]In mg/kg for nonradiological constituents and pCi/g for radiological constituents.
ND = no data

Table 6–20. Summary of Ecological Contaminants of Potential Concern in Area E: Many Devils Wash (based on data collected from 1998 through 2000)

Constituent	Maximum Concentration in Surface Water[a]		Maximum Concentration in Sediment[b]		Selected as COPC?	Reason
	Site	Ref.	Site	Ref.		
Nonradiological Constituents						
Ammonium	2.05	0.6	11.7	8.9	Yes	Elevated in water and sediment.
Antimony	0.0012	<0.003	0.18	0.19	No	Ground water data do not indicate antimony is a millsite contaminant.
Arsenic	< 0.0011	0.008	1.05	7.8	No	Ground water data do not indicate arsenic is a millsite contaminant.
Boron	ND	ND	ND	ND	No	No data.
Cadmium	< 0.001	< 0.001	0.26	0.12	No	Ground water data do not indicate cadmium is a millsite contaminant.
Calcium	474	212	ND	ND	No	Nutrient
Chloride	2,690	57.7	ND	ND	No	Low toxicity
Chromium	ND	< 0.01	ND	ND	No	Not expected to be a COPC based on information from the Shiprock floodplain.
Cobalt	ND	< 0.05	ND	ND	No	Not expected to be a COPC based on information from the Shiprock floodplain.
Copper	ND	< 0.02	ND	ND	No	Not expected to be a COPC based on information from the Shiprock floodplain.
Iron	0.0086	0.98	ND	ND	No	Within range of reference samples.
Lead	ND	< 0.003	ND	ND	No	Not expected to be a COPC based on information from the Shiprock floodplain.
Magnesium	3,610	14.7	11,900	17,300	No	Nutrient
Manganese	0.129	0.5	114	365	No	Within range of reference samples.
Molybdenum	0.135	< 0.023	ND	ND	Yes	Elevated in water.
Nickel	ND	ND	ND	ND	No	Not expected to be a COPC based on information from the Shiprock floodplain.
Nitrate	8,060	23.9	1,300	189	Yes	Elevated in water and sediment.
Potassium	190	8.63	ND	ND	No	Nutrient
Selenium	7.01	< 0.005	0.44	1.0	Yes	Elevated in water.
Sodium	28,300	1,400	3,660	1,880	No	Nutrient
Strontium	16.1	13.5	184	312	Yes	Elevated in water.
Sulfate	72,800	2,520	19,600	1,890	Yes	Elevated in water and sediment.
Uranium	0.630	< 0.001	0.86	4.5	Yes	Elevated in water.
Vanadium	< 0.0006	< 0.01	ND	ND	No	Not detected in recent water samples.
Zinc	ND	0.01	ND	ND	No	No data
Radiological Constituents						
Lead-210	ND	ND	ND	ND	No	No data.
Polonium-210	< 0.41	1.2	ND	ND	No	Within background range.
Radium-226	0.91	1.0	ND	2.68	No	Within background range.
Radium-228	1.86	1.1	ND	ND	Yes	Elevated in water.
Thorium-230	ND	1.0	ND	3.9	No	No data.

[a]In mg/L for nonradiological constituents and pCi/L for radiological constituents.
[b]In mg/kg for nonradiological constituents and pCi/g for radiological constituents.
ND = no data

Area E: Many Devils Wash. The ecological COPCs identified for Many Devils Wash (surface water and sediment) as based on comparisons with the well 648 reference data are ammonium, molybdenum, nitrate, selenium, strontium, sulfate, and uranium (Table 6–20). Boron and zinc, however, have not been included as analytes for Many Devils Wash. Of the radiological analytes, radium-228 concentration was slightly elevated in water. Although no data have been collected for lead-210 and thorium-230, the low concentrations of the other radionuclides at this area indicate that concentrations of uranium-238 decay-chain constituents are generally low.

Area F: The Upland Terrace. Of the eight metals and two radionuclides that were analyzed in greasewood samples from the millsite terrace, the maximum concentrations of arsenic, manganese, strontium, and uranium exceeded the maximum concentrations of the reference greasewood samples (see Appendix H). Arsenic was not included as a COPC because the maximums were very close in magnitude (0.46 mg/kg dry weight at the millsite versus 0.37 mg/kg dry weight at the reference site) and the frequency of detection, mean, and UCL_{95} of the mean were all less for the millsite than for the reference site. Neither of the radionuclides (radium-226 and thorium-230) was detected at either site. Nitrate and sulfate were not included as analytes in these samples.

Summary. A summary of the results of the reevaluation of ecological COPCs is presented in Table 6–21. These COPCs are site-specific but are not specific to particular media. Therefore, the list of COPCs for a site applies at all media of concern at that site.

Table 6–21. Summary of Ecological Contaminants of Potential Concern at Areas Associated with the Shiprock Millsite Ground Water Based on Most Recent Analytical Data

Area A: Distributary Channel and Tributaries	Area B: San Juan River	Area C: Shiprock Floodplain	Area D: Bob Lee Wash	Area E: Many Devils Wash	Area F: Upland Terrace
Media: surface water sediment	Media: surface water sediment	Media: surface water sediment soil	Media: surface water sediment soil	Media: surface water sediment	Media: vegetation
Ammonium Manganese Molybdenum Nitrate Selenium Strontium Sulfate Uranium Vanadium	Ammonium Manganese Molybdenum Nitrate Selenium Strontium Sulfate Uranium Vanadium	Ammonium Boron Manganese Molybdenum Nitrate Selenium Strontium Sulfate Uranium Vanadium	Ammonium Manganese Molybdenum Nitrate Selenium Strontium Sulfate Uranium Vanadium	Ammonium Molybdenum Nitrate Selenium Strontium Sulfate Uranium	Manganese Strontium Uranium

6.2.1.4 Ecological Site Conceptual Model

The conceptual model for an ERA is developed from information about stressors, predicted exposure pathways, and the potential effects of exposure on ecological receptors. Conceptual models consist of two principal components (EPA 1998):

- A set of risk hypotheses that provide descriptions of predicted relationships among stressor, exposure, and assessment endpoint response, along with the rationale for their selection.

- A diagram that illustrates the relationships presented in the risk hypotheses.

A complete exposure pathway is the mechanism by which a contaminant in an environmental medium (i.e., the source) can contact an ecological receptor. A complete exposure pathway includes

- Contaminant source

- Release mechanism that allows contaminants to become mobile or accessible

- Transport mechanism that moves contaminants away from the release

- Ecological receptor

- Route of exposure (e.g., dermal or direct contact, inhalation, or ingestion).

Because the stressors at the Shiprock site are chemical contaminants, the risk hypotheses are considered to be stressor-initiated.

As part of the initial problem formulation in the BLRA, a generalized site conceptual model was developed for the Shiprock site. That model has since been revised to address current and potential exposure pathways based on all the available data (Figure 6–3). At the Shiprock site, the movement of contaminated ground water from the millsite in various directions has resulted in surface and near surface expressions of this ground water at several locations, each with differing stressors (COPCs), exposure pathways, habitats, and potential receptors. For this reason, risk hypotheses are developed separately for each of these locations (shown in Plate 5). These evaluation regions are

- The distributary channel and its tributaries (all west of U.S. Highway 666)

- The San Juan River

- The Shiprock floodplain

- Bob Lee Wash

- Many Devils Wash

- The upland (disposal cell/millsite) terrace.

The distributary channel area has only recently been identified as being potentially affected by milling operations based on the results of limited sampling conducted in this region in the spring and fall of 1999 and in 2000.

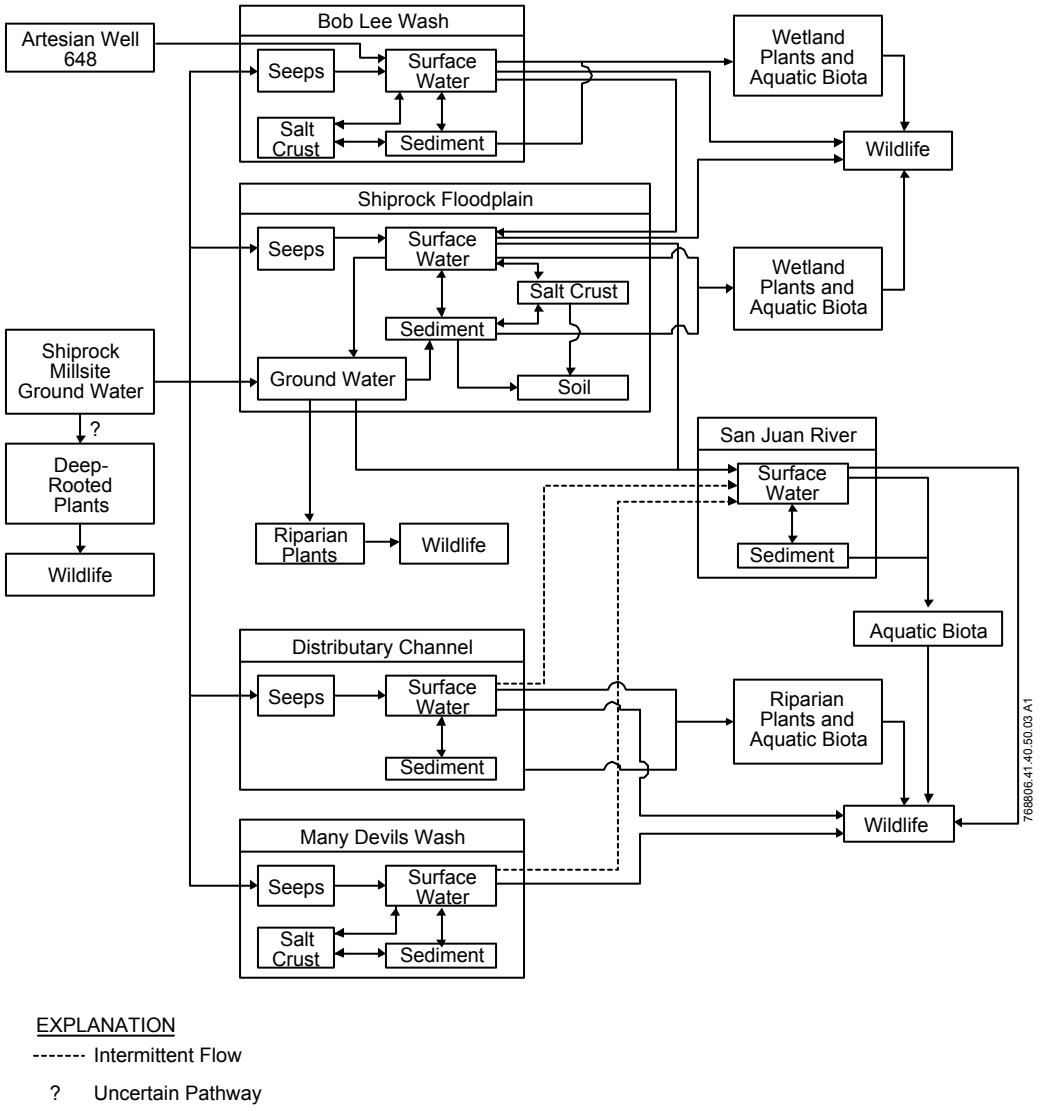

Figure 6–3. Shiprock Ecological Site Conceptual Model

Risk Hypotheses Based on Current Exposure Scenarios

The following are the risk hypotheses proposed for the six areas of the Shiprock site where complete exposure pathways to ecological receptors may exist based on the current site conditions.

The distributary channel and tributaries: The distributary channel crosses the south side of the San Juan River floodplain west (downstream) of the U.S. Highway 666 bridge. Contaminants that have entered the river from the floodplain area adjacent to the former millsite may be present in the river water that enters the distributary channel during periods of high river stage. However, a more significant potential input of millsite contaminants is from ground water seepage along the escarpment that forms the southern bank of the channel, and especially at the incised channels of two small washes (1st Wash and 2nd Wash) that flow into the channel from the escarpment. Pools of permanent standing water at the mouths of these washes indicate probable contact with ground water. The migration of ground water westward from the former millsite may have reached the area of these seeps. If that is the case, these pools provide for complete ecological exposure pathways for organisms (particularly wildlife) that may drink from the pools. Aquatic organisms may become established in the pools and become food-chain links for additional exposures to wildlife. The vegetation along the distributary channel is dense riparian woodland, dominated by saltcedar, providing habitat for birds, such as flycatchers, that may feed on invertebrates that have been exposed to the water in the pools. Flows within the channel will occasionally flush the contaminants downstream (to the San Juan River). Such flows may carry aquatic organisms (e.g., fish) from the river, which may be entrapped in the pools as the water in the channel again recedes.

The irrigation return flow ditch is a tributary of the distributary channel and essentially forms the western boundary of the upland terrace. As with the distributary channel, contaminants may be entering the ditch from ground water seepage, especially at the mouth of a small wash (3rd Wash) that flows into the channel from the escarpment and further west from flows from former gravel pits (sample location 942). The migration of ground water westward from the former millsite may have reached the area of this ditch. If that is the case, surface water in the ditch provides a complete ecological exposure pathways for organisms that may drink or feed along its length. Aquatic organisms in the ditch may be exposed to contaminants in the ditch water and sediments, and become food-chain links for additional exposures to wildlife. The vegetation along the ditch may also serve as links in these exposure pathways.

The San Juan River: Hydrogeologic information regarding plume migration suggests that contamination might be reaching the San Juan River by subsurface flow through the floodplain alluvium. In addition, contaminants may be reaching the river by surface and subsurface flow from Many Devils Wash and from the wetland area drainages that discharge into the river at the northwestern corner of the floodplain. These discharges could result in the direct or indirect exposure of wildlife and riparian plant receptors that use or inhabit the river to millsite contaminants. More significantly, however, is the potential for direct exposure to aquatic receptors that live in the river. These potentially include two endangered fish species (the Colorado pikeminnow and the razorback sucker).

The Shiprock floodplain: Process water from the milling operations may have mounded in the alluvium of the terrace and in the underlying weathered Mancos Shale, creating a ground water

plume that is thought to be migrating radially from the former millsite, especially in a northerly direction toward the San Juan River escarpment. This water is discharging as seeps along the escarpment above the Shiprock floodplain as well as affecting the ground water underlying the floodplain.

The Shiprock floodplain contains a variety of habitat types, including terrestrial, wetland, and riparian. Bob Lee Wash flows onto the floodplain and forms a small perennial wetland. This surface flow is primarily maintained by discharge from artesian well 648 located on the terrace west of the wash. This water may also contain mill-related contaminants from ground water seeps along the wash. Ground water seeps along the escarpment are a significant source of mill-related contaminants to both the surface water and sediments of the wetland and its downstream drainages. Evaporation of contaminated water from the soil and seasonally wet sediment at and around the seeps results in the accumulation of these contaminants within salt crusts that form on their surfaces. In addition, phreatophytes growing on the floodplain may take up contaminants directly from the ground water of the floodplain through their root systems, thereby creating a potential exposure pathway to herbivores and their predators and creating a potential for the surface deposition of the contaminants through litterfall. Therefore, seeps and shallow ground water create potentially complete pathways between the millsite contaminants and ecological receptors of the floodplain. These receptors include species associated with terrestrial, riparian, and wetland habitats.

Bob Lee Wash: Northwest of the former millsite, the escarpment is incised by Bob Lee Wash. Ground water seeps along Bob Lee Wash provide a potential source of mill-related contaminants to the both the surface water and sediments of this drainage. Seasonal pools of standing water in the upper reaches of the wash indicate surface contact with the ground water and a potential source of contaminants in soils of that area. Although the upper part of the wash is arid and sparsely vegetated, the lower part (fed by the discharge of well 648) is densely vegetated with saltcedars, wetland grasses, and cattails. Potential receptors in the upper reaches are probably limited to terrestrial wildlife and upland plants (also some inland saltgrass); the lower reach includes aquatic and wetland receptors.

Many Devils Wash: The migration of ground water east from the former millsite may have reached Many Devils Wash, which has a channel that is deeply eroded into the escarpment. Standing pools of surface water occur in the channel between an erosional knickpoint and the mouth of the wash (at the San Juan River), and for several hundred feet upstream from the knickpoint, indicating probable contact with ground water. These pools are lined with salt crusts. With the exception of a few scattered saltcedars, the channel of this wash is nearly devoid of vegetation. These pools may provide complete ecological exposure pathways for organisms (particularly wildlife) that may drink from them. Because of the lack of plants and aquatic organisms in and around the pools, the potential for exposure through the food chain is minimal. Flows within the wash can flush contaminants into the San Juan River. Such flows, however, appear to be rare.

Upland terrace: Milling operations at the Shiprock site resulted in a variety of contaminants in the ground water underlying the terrace on which the mill was located (and which is currently the site of the disposal cell). The habitats of the upland terrace area are terrestrial, and the depth to the ground water limits the potential contact with ecological receptors. Deep-rooted plants (phreatophytes) growing on the terrace, however, may contact this ground water and take up contaminants through their root systems, thereby creating a potential exposure pathway to

herbivores and organisms higher in the food chain. Risk to the organisms trophically linked to these plants may therefore exist at this site.

Hypothetical Future Exposure Scenario: Without institutional controls, ground water could possibly be pumped and used for irrigation, livestock watering, or industrial uses. This practice would create a source for ground water and surface water ingestion, direct contact with terrestrial vegetation, and deposition of ground water and surface water on the soil. The soil would then represent an additional source medium for ingestion and direct contact. Large-scale irrigation with ground water is not considered a likely future pathway because surface water is the main source of irrigation water in the Shiprock area. As long as there is the possibility of pumping ground water for agricultural purposes, it is assumed that the potential exists for these two exposure pathways.

6.2.1.5 Ecological Receptors

Ecological receptors that could potentially be exposed to COPCs were identified in the BLRA (DOE 1994) and include mammalian and avian species. Section 6.2.1.1. summarizes the habitats and populations that may be affected by exposures to COPCs at the Shiprock site. The food web for the Shiprock site (Figure 6–4) illustrates the significant dietary interactions among and between the terrestrial and aquatic receptors. The food web also depicts the major trophic interactions and shows nutrient flow and transfer of matter and energy through the trophic levels. This food web model was developed from the species lists and consideration of the exposure pathways. The food web diagram was used to portray potential routes of COPCs from the ground water to biota at various trophic levels, with potential receptor species being specific to each of the six areas identified as having potentially complete ecological exposure pathways. These potential receptors are as follows:

The distributary channel and tributaries. The habitat of the distributary channel and it tributaries is primarily riparian. Although standing water does occur along distributary channel when it is not flowing, the development of a wetland community is curtailed by periodic flushing and scouring when the river water flows through the channel. The absence of high, scouring flows along the irrigation return flow ditch allows a greater development of a wetland community than in the distributary channel. The potential receptors of these areas include:

- Plants—Riparian plants that grow along the channel course and in the mouths of the secondary washes include saltcedar, cottonwood, and Russian olive.

- Aquatic receptors—Because of the periodic flushing of the distributary channel by the river, the aquatic receptors are probably limited to fish (trapped in the pools as the river flow recedes) and aquatic invertebrates, including insects (some as larvae only). Water flows in the irrigation ditch may be less dynamic, allowing greater development of an aquatic community.

- Terrestrial wildlife—Terrestrial wildlife may be exposed to COPCs in these channels as a result of drinking surface water (especially from the standing pools) and feeding on the aquatic organisms from these pools. The latter exposure route is most likely to affect insectivorous birds, such as swallows, flycatchers, and shorebirds (e.g., killdeer). Both birds and mammals may be attracted to the pools for drinking.

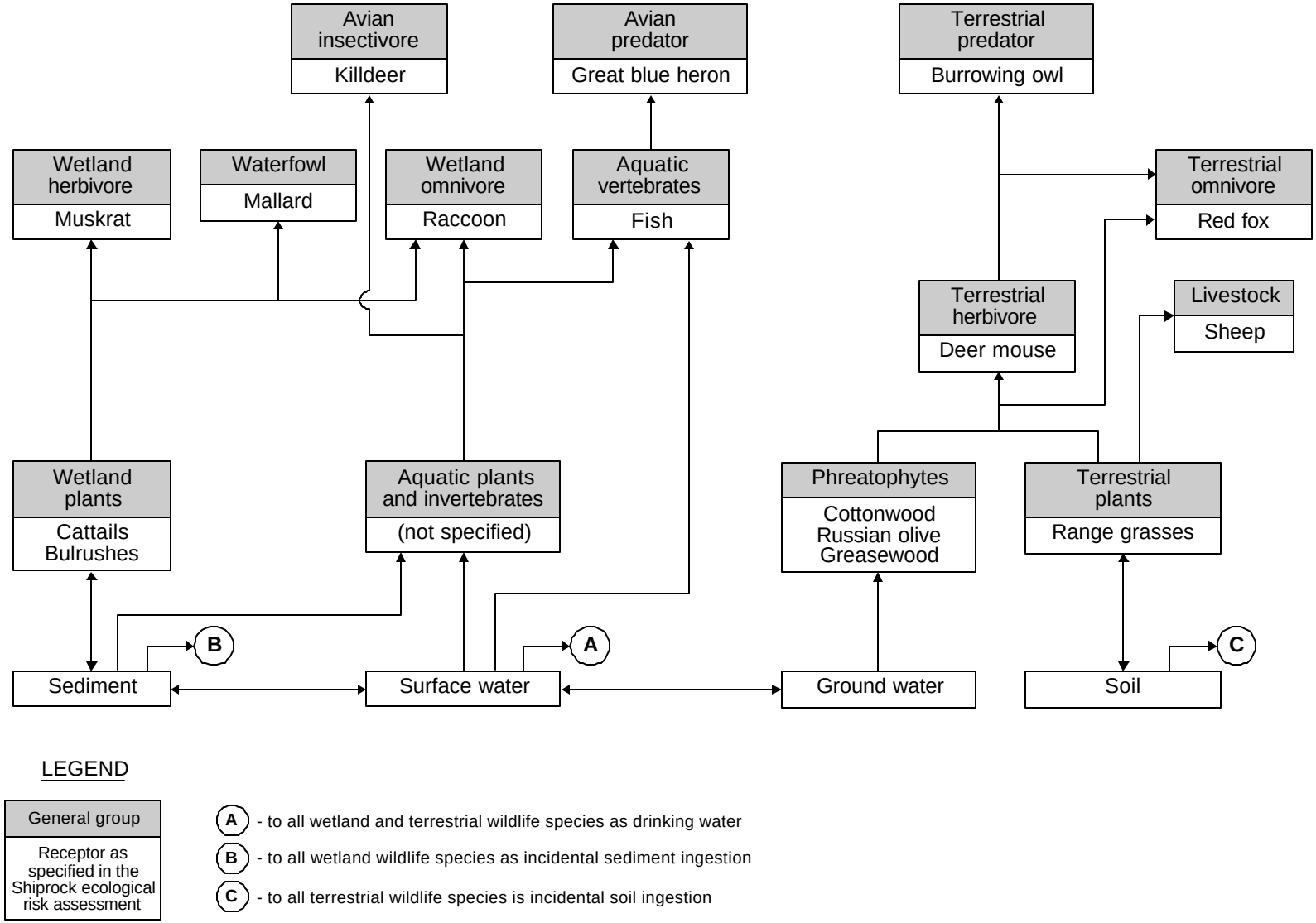

Figure 6-4. Generalized Food Web for Shiprock Ecological Receptors

Based on habitat conditions along the distributary channel, the endangered southwestern willow flycatcher may occur there. This species is considered a potential receptor at this location. Also, individuals of the endangered Colorado pikeminnow or razorback sucker (especially young individuals) may be trapped in the pools as the river water recedes from the channel. Therefore, these species, too, are considered potential receptors.

The San Juan River. The habitat of the San Juan River is considered to be aquatic. The potential receptors of this area include aquatic (free-swimming and benthic) organisms that live in the river and its sediments and predators that feed on these organisms. These predators are primarily avian, and may include herons, egrets, bald eagles, ospreys, shorebirds, and aerial insectivores (e.g., swallows and flycatchers). Ducks, geese, and other waterfowl also use the river during migration. The San Juan River is designated as Critical Habitat for two endangered fish species (see Section 6.2.1.1). These species are therefore considered to be potential receptors, although their presence in the Shiprock area has not been confirmed.

The Shiprock floodplain. The habitats of the Shiprock floodplain includes both terrestrial and aquatic elements. These habitats can be categorized as upland, riparian, and wetland. The potential receptors of the floodplain area include:

- Plants—Phreatophytes and wetland plants are capable of contacting contaminated water, either by direct contact with the alluvial ground water or by contact with water discharged from seeps. Saltcedar, cottonwood, Russian olive, cattail, bulrush, and inland saltgrass are examples of potential plant receptors of this area.

- Terrestrial herbivores—The terrestrial wildlife that may be exposed to COPCs through the consumption of phreatophytes and wetland plants include rodents (e.g., white-footed mice, voles, and ground squirrels), lagomorphs (cottontails and jackrabbits), and mule deer. Evidence of beaver use along the San Juan River indicates that this herbivorous rodent is a potential receptor in the riparian habitat of the floodplain. Additional exposure in these receptors may result from the ingestion of water, sediment, and soil from the site.

- Terrestrial predators—Predators that may be exposed to COPCs through the consumption of terrestrial herbivorous prey include foxes, coyotes, skunks, snakes, and raptors. Many mammalian predators will also consume plant material, making them omnivores rather than strict carnivores. Additional exposure in these receptors may result from the ingestion of water, sediment, and soil from the site.

- Aquatic receptors—The aquatic receptors include those organisms that live in or are dependent upon surface water for survival and/or reproduction. On the Shiprock floodplain, these receptors are limited to the wetland at the mouth of Bob Lee Wash and the surface flows into and out of this wetland. The aquatic receptors include fish (minnows), aquatic insects (some as larvae only) and other invertebrates, bullfrogs (and possibly other frogs and toads), and aquatic plants (e.g., algae).

- Wetland herbivores—The herbivorous wildlife that may be exposed to COPCs through close association with wetland plants include mice, voles, cottonrats, and possibly muskrats. Waterfowl, especially dabbling ducks, may also consume plants from the wetland. Additional exposure in these receptors may result from the ingestion of water, sediment, and soil from the site.

- Wetland predators—The terrestrial predators described above are also potential predators in the wetland habitat. Predators that are more closely associated with wetlands include raccoons, herons, and egrets. A wide range of omnivorous and insectivorous birds may also be associated with the wetland habitat, including swallows, flycatchers, red-winged blackbirds, sparrows, wrens, shrikes, and shorebirds (e.g., killdeer). These may include breeding pairs, wintering individuals, or migrants. Additional exposure in these receptors may result from the ingestion of water, sediment, and soil from the site.

The endangered southwestern willow flycatcher may occur in the riparian woodlands of the floodplain (it was possibly heard during the T&E surveys conducted in 1998 and 1999, but not confirmed). This species is considered a potential receptor based on the presence of potentially favorable habitat and the unconfirmed auditory observation. The wetland may also be a potential habitat for the endangered northern leopard frog; however, because there are no supporting observations indicating its presence, it is not considered a potential receptor.

Bob Lee Wash. As described above, the habitat of the upper reaches of Bob Lee Wash is primarily terrestrial and that of the lower reach (near the mouth of the incised channel) is wetland/riparian. The potential receptors of this area include:

- Plants—Terrestrial plants in the upper reaches of the wash may be exposed to COPCs in soil. Riparian and wetland plants growing in the lower part of the wash may be exposed to COPCs in the water and sediment.

- Aquatic receptors—Small fish and aquatic invertebrates are potentially exposed to COPCs in the water of the lower reach of the wash.

- Terrestrial wildlife—Terrestrial wildlife may be exposed to COPCs in the wash as a result of drinking surface water (especially from the standing pools) and feeding on the aquatic organisms from the water. The latter exposure route is most likely to affect insectivorous birds.

Based on habitat conditions along Bob Lee Wash, the endangered southwestern willow flycatcher may occur there. This species is, therefore, considered a potential receptor at this location.

Many Devils Wash. Because the highly saline standing water in Many Devils Wash inhibits the growth of vegetation and the occurrence of aquatic organisms, the habitat of this area is considered to be essentially terrestrial. The potential receptors of this area include wildlife species, such as coyotes, foxes, and birds that may occasionally try to drink the water.

Upland terrace. The habitat of the upland (millsite) terrace is terrestrial. The potential receptors of this area include:

- Plants—Only deep-rooted plants (phreatophytes) are capable of contacting contaminated ground water from the terrace surface and are therefore essential in completing the potential exposure pathway for this area. Black greasewood is an example of such a plant that grows on the terrace.

- Herbivores—The herbivorous wildlife that may be exposed to COPCs taken up by phreatophytes include rodents (e.g., deer mice, pocket mice, and ground squirrels), lagomorphs (cottontails and jackrabbits), and mule deer. Rodents may opportunistically eat insects, which may, in turn, have ingested contaminants in the plants.

- Predators—Predators that may be exposed to COPCs through the consumption of herbivorous prey include foxes, coyotes, snakes, and raptors (e.g., hawks, owls, and eagles). Many mammalian predators, such as foxes and coyotes, will also consume plant material, making them omnivores rather than strict carnivores.

The endangered Mesa Verde cactus was recorded on the terrace during the T&E surveys conducted in 1998 and 1999 (see Section 6.2.1.1). This species is not considered a potential receptor because it is not likely to root deep enough to contact contaminants in the ground water. The western burrowing owl was also recorded on the terrace during the T&E surveys and may be exposed to COPCs through the food chain if phreatophytes are transporting the ground water contaminants to their accessible (aboveground) tissues. Although not listed as threatened or endangered, the western burrowing owl is a species of management concern.

6.2.1.6 Management Goals and Endpoints

Table 6–22 presents the primary goals for the protection of environmental resources at the Shiprock site with respect to contaminants associated with ground water, and the assessment and measurement endpoints that will be used to evaluate potential risk to these resources in support of achieving these goals.

Table 6–22. Management Goals, Assessment Endpoints, and Measurement Endpoints for the Evaluation of Ecological Risks at the Shiprock Site

Management Goals	Assessment Endpoints	Measurement Endpoints
Protect the quality of critical habitat for endangered fish species in the San Juan River	Surface water quality of the San Juan River	Concentrations of ecological COPCs in the surface water of the San Juan River meet applicable water quality criteria or equivalent benchmarks for the protection of aquatic life.
	Sediment quality of the San Juan River	Concentrations of ecological COPCs in the sediment of the San Juan River meet applicable sediment quality benchmarks for the protection of benthic organisms.
Maintain habitat quality of the floodplain and other areas of potential ground water discharge for the protection of wildlife diversity	Potential for adverse effects on survival and reproduction in wildlife from exposures to COPCs in various environmental media	Hazard quotients comparing estimated exposure to toxicity benchmarks for key indicator receptor species are less than unity.
Maintain quality of the range resources of the floodplain for livestock production	Forage and surface water quality	Hazard quotients comparing estimated exposure to toxicity benchmarks for sheep are less than unity.

6.2.2 Analysis

6.2.2.1 Exposure Assessment

Exposure Modeling and Assumptions

Only complete exposure pathways are quantitatively and qualitatively evaluated in an ERA. In this assessment, the following potential exposure pathways were considered for evaluation:

- Surface water—ingestion and direct contact
- Soil—ingestion and direct contact
- Sediment—ingestion and direct contact
- Dietary—ingestion of forage or prey, as appropriate, by receptor

The contaminants associated with the Shiprock site are inorganics and are principally associated with water (in dissolved form) and sediment/soil (adsorbed to particles in these media). Estimations of potential exposures to key ecological receptors are based on the dominant pathways to these media for the specific receptor. Exposures in plants (both terrestrial plants and emergents) are dominated by direct contact with the soil or sediment in which they are rooted. Exposures to aquatic organisms (those that live within the water column) and benthic organisms (those that live within the sediment) are dominated by direct contact with the external media (water and sediment) in which they live, but in the cases of aquatic and benthic animals also include the ingestion of food associated with these media. In all of these cases (plants and animals), potential exposure to a COPC is based on the concentration of that COPC in the media of principal contact (water, sediment, or soil).

Exposures in terrestrial and wetland wildlife involve multiple potential pathways that may include ingestion of food, water, and sediment/soil; direct contact and dermal absorption; and inhalation. In this assessment, the inhalation and dermal absorption pathways are assumed to be minor pathways with respect to the combined exposures based on ingestion (food, water, and sediment/soil ingestion). Most wildlife of the area have very little and infrequent direct dermal contact with potentially contaminated media due to their protective covers of feathers or fur and their habits and behaviors, such as preening and grooming, and (in the cases of most birds) living principally in trees and shrubs. Because the COPCs at the Shiprock site are not highly volatile, their occurrence in the air is principally related to dust particles. For the assessment of exposures to wildlife, however, dust inhalation was considered a minor exposure pathway relative to soil or sediment ingestion. Although both dermal absorption and inhalation will contribute to the overall exposure in these receptors, these contributions are assumed to be included within the conservatisms incorporated in the estimation of exposures through the ingestion pathways.

In the estimation of ingestion-related exposure for the terrestrial and wetland wildlife receptors, the COPCs are assumed to be 100 percent bioavailable and the receptors are assumed to be exposed only at the selected exposure point concentration, regardless of home range size or seasonal use patterns. The exposure through multiple ingestion pathways is modeled using the methods described in the EPA's *Wildlife Exposure Factors Handbook* (EPA 1993). The basic model for estimating the daily intake of a COPC per kilogram of body weight (i.e., the estimated daily dose of the COPC) through these ingestion pathways is

$$D_x = \frac{\sum_{k=1}^{m}(C_k \cdot F_k \cdot I_k) + C_s \cdot F_s \cdot I_s + C_w \cdot F_w \cdot I_w}{W}$$

where

D_x is the estimated daily dose (mg/kg-day) of COPC *x,*
C_k is the concentration of COPC x in the kth food type (mg/kg dry weight),
F_k is the fraction of the kth food type that comes from the site,
I_k is the ingestion rate of the kth food type (kg dry weight/day),
m is the number of food items in the receptor's diet,
C_s is the concentration of COPC *x* in the sediment or soil (mg/kg dry weight),
F_s is the fraction of ingested sediment or soil that comes from the site,
I_s is the ingestion rate of sediment or soil (kg dry weight/day),
C_w is the concentration of COPC *x* in water (mg/L),
F_w is the fraction of the ingested water that comes from the site,
I_w is the ingestion rate of water (L/day), and
W is the body weight of the receptor (kg wet weight).

F_k, F_s, and F_w are commonly assumed to be the area use factor (the area of the site divided by the home range of the receptor or 1, whichever is smaller) but may also be modified by a seasonal use factor (number of days at the site divided by 365 days per year) if the home range is used for only part of the year. For estimating risk in this assessment, both area use and seasonal use are conservatively assumed to be 100 percent; therefore, F_k, F_s, and F_w are assumed to be 1.

For the purposes of estimating exposure in terrestrial wildlife, the COPC concentrations in plants and small mammals were principally based on the empirically-derived uptake models (nonlinear or linear) as recommended by Oak Ridge National Laboratory (Bechtel Jacobs Company 1998a and Sample and others 1998). The nonlinear form of the uptake model is

$$C_{organism} = B_0 \cdot C_{soil}^{\ B_1}$$

where

$C_{organism}$ is the concentration of the COPC in the plant or small mammal (mg/kg dry weight),
C_{soil} is the soil concentration of the COPC (mg/kg dry weight), and
B_0 and B_1 are empirically derived model parameters for the COPC and organism.

In the linear form of this model, B_1 is assumed to be exactly 1 and B_0 becomes a soil-to-organism transfer factor, where

$$C_{organism} = B_0 \cdot C_{soil}$$

In cases where parameters were not available in the Oak Ridge National Laboratory uptake model documents, soil-to-plant transfer factors from other literature sources (e.g., Baes and others 1984) were used in this linear model. For small mammals, soil-to-mammal transfer factors

based on Sandia National Laboratories data (SNL 1999) were used to supplement the modeling information available in Sample and others (1998). In some cases (e.g., exposures estimated for the upland terrace area), small mammal concentrations were modeled from plant concentrations using food-to-mammal transfer factors from Baes and others (1984), IAEA (1994), or SNL (1999). In this case, the model is of the form:

$$C_{mammal} = B_0 \cdot C_{plant}$$

where:

C_{mammal} is the concentration of the COPC in an herbivorous mammal (mg/kg dry weight),
C_{plant} is the concentration of the COPC in the plant material eaten by the mammal (mg/kg dry weight), and
B_0 is the food-to-mammal transfer factor (converted as necessary to be on a dry-weight to dry-weight basis).

For aquatic prey species (invertebrates and fish), linear uptake models based on bioaccumulation factors (BAFs) were used to estimate concentrations of COPCs in tissues. These models are of the form:

$$C_{organism} = BAF \cdot C_{water}$$

where:

$C_{organism}$ is the concentration of the COPC in the invertebrate or fish prey species(mg/kg dry weight),
C_{water} is the concentration of the COPC in the water (mg/L), and
BAF is the bioaccumulation factor for the COPC.

BAFs account for all exposure pathways (dermal absorption, uptake through respiratory organs, and ingestion). In contrast, bioconcentration factors (BCFs) account for uptake through pathways other than ingestion. However, for most inorganic constituents, uptake through ingestion of water is insignificant, and BAFs are considered to be equal to BCFs. Therefore, BCFs are used as BAFs in this assessment when the latter values are not available. Whenever possible, however, BAFs and BCFs specific to either invertebrates or fish were used to model the concentrations in these respective prey types.

Table 6–23 presents the uptake model parameters (B_0, B_1, BAF, and/or BCF values) used in modeling the concentrations of COPCs through the food chain at the Shiprock site.

Table 6–23. Uptake Model Parameters and Bioaccumulation Factors for Ecological Contaminants of Potential Concern

Contaminant of Potential Concern	Uptake Model Parameters				Bioaccumulation Factors	
	Plants		Small mammals		Invertebrates	Fish
	B_0	B_1	B_0	B_1		
Ammonium	---	---	---	---	---	---
Boron	4.0[a]	1.0[b]	0.0008[a,c]	1.0[b]	---	---
Manganese	3.0[d]	1.0[b]	0.0205[e]	1.0[b]	65[f]	17.8[f]
Molybdenum	0.8[g]	1.0[b]	0.001[g,c]	1.0[b]	10[h]	10[h]
Nitrate	---	---	---	---	---	---
Selenium	0.508[l]	1.10[i]	0.660[e]	0.376[e]	269[j]	129[k]
Strontium	2.5[a]	1.0[b]	0.008[a,c]	1.0[b]	9.5[h]	9.5[f]
Sulfate	---	---	---	---	---	---
Uranium	0.023[g]	1.0[b]	0.033[l]	1.0[b]	27.1[h]	27.1[f]
Vanadium	0.0055[a]	1.0[b]	0.0123[e]	1.0[b]	3,000[m]	3,000[n]

[a]From Baes and others (1984).
[b]The uptake model is linear; therefore, B_1 = 1.0.
[c]Based on uptake from food.
[d]From NCRP (1989).
[e]From Sample and others (1998).
[f]From AQUIRE (2000).
[g]From IAEA (1994).
[h]Invertebrate bioaccumulation factor based on fish bioaccumulation factor.
[l]From Bechtel Jacobs Company (1998 a).
[j]Geometric mean of selenite bioaccumulation factors for water fleas based on 14-day exposure from AQUIRE (2000).
[k]From NMED (2000).
[l]From SNL (1999).
[m]From Neumann (1985).
[n]Fish bioaccumulation factor based on invertebrate bioaccumulation factor.
--- = No data available.

Key Indicator Receptors

The receptors used to evaluate potential risks at each site were selected based on their potential presence in the habitats of the site, their potential for exposure to COPCs in the media at the site, and their potential for conservatively representing potential exposures to a range of other receptors at the site. Potential receptors for the habitats present at each of the six sites identified as having potentially complete ecological pathways are discussed in Section 6.2.1.5. The indicator receptors are representative of key links in the food webs associate with these habitats. These indicator receptors are as follows:

- Terrestrial habitats—plant, deer mouse (herbivorous), red fox, burrowing owl
- Wetland habitats—plant, muskrat, mallard, raccoon, killdeer, great blue heron
- Aquatic habitats—aquatic and benthic organisms

Terrestrial exposure pathways are found on the upland terrace and on the floodplain. For the former, deep-rooted plants (e.g., greasewood) are considered only as the transport mechanism between the COPCs in the ground water and the receptors (herbivores and their predators) on the surface of the terrace. Therefore, risk to these deep-rooted plants is not evaluated. For the

terrestrial systems of the floodplain, soil is considered to be the primary source medium for COPC exposures, and therefore, risks to all terrestrial receptors listed above are evaluated. The deer mouse is used to represent herbivorous species, and the red fox and burrowing owl are used to represent omnivorous and carnivorous species.

For the wetland habitats, emergent plants, such as cattails, are considered to be the primary producers and the muskrat and mallard are considered to be representative of herbivores that may consume such plants (both will also eat some animal prey). The raccoon represents an omnivore in this habitat. The killdeer represents an insectivorous bird, and the great blue heron represents a predatory bird. All animal prey of these wildlife receptors (the muskrat is the only one to be assumed to be purely herbivorous) are assumed to be aquatic (invertebrates or fish).

Receptors in the aquatic habitats are not specified. Risk to these receptors is based on comparisons of the media COPC concentrations (water and sediment) to broad-based benchmark values, such as ambient water quality criteria (AWQC), that are protective of a wide range of aquatic and benthic organisms. For the San Juan River and the distributary channel, where endangered fish may be exposed, fish are assumed to be included as potential aquatic receptors within this broad categorization. All wildlife receptors are modeled as potential receptors of COPCs in surface water through the consumption of that water at all sites where surface water is present as a medium of concern.

The species-specific parameters used to model exposures to these key indicator receptors (wildlife only) are presented in Table 6–24.

6.2.2.2 Effects Characterization

The potential for adverse effects to ecological receptors resulting from exposures to COPCs at the Shiprock site was evaluated through the comparison of the potential exposure in the receptor to a toxicity-based benchmark of exposure representing the threshold of potential adverse effects.

For aquatic and benthic receptors and plants, the exposure to a COPC is characterized by the concentration of that COPC in the medium (water, sediment, or soil) with which the receptor is principally in direct contact. Therefore, the benchmarks by which the potential for adverse effects is evaluated are also based on media concentrations. For surface water, either AWQC (EPA 1999, Buchman 1999) or Navajo Nation Surface Water Quality Standards (NNSWQSs) for warm water habitat (whichever was lesser) were used as the principal benchmarks for evaluating potential risk to aquatic life. When neither was available for a COPC, Tier II secondary values (Suter and Tsao 1996) or other values (e.g., Haines and others 1994) were used. Sediment benchmarks were principally based on the lowest threshold effect levels (TELs) as presented in Buchman (1999), and supplemented from other sources (e.g., EPA 1996, Jones and others 1997, and Haines and others 1994). Table 6–25 presents these water quality benchmark values.

Table 6–24. Exposure Parameters for Livestock and Wildlife Receptors

Receptor	Body weight (kg)[a]	Food ingestion rate (kg [dry wt.]/day)[b]	Soil/sediment ingestion rate (percent of food ingestion)[c]	Water ingestion rate (L/day)[d]	Dietary Composition (percent)[e]
Sheep (*Ovis aries)*	50[f]	2.0[f]	6.8[g]	5.0[f]	Plant (grasses): 100
Deer mouse (*Peromyscus maniculatus*)	0.0239[h]	0.00372	2.0[i]	0.00344	Plant: 100
Red fox (*Vulpes vulpes*)	4.54	0.238	2.8	0.386	Plant: 20 Mouse: 80
Muskrat (*Ondatra zibethicus*)	1.135	0.0772[j]	9.4[k]	0.111	Plant (cattails): 100
Raccoon (*Procyon lotor*)	5.74	0.289	9.4	0.477	Plant: 40 Invertebrate: 50 Fish: 10
Burrowing owl (*Speotyto cunicularia*)	0.155[l]	0.0173	2.0[m]	0.000283	Mouse: 100
Mallard (*Anas platyrhynchos*)	1.134	0.0592	3.3	0.0642	Plant: 90 Invertebrate: 10
Killdeer (*Charadrius vociferus*)	0.0966[l]	0.00932	18[n]	0.0123	Invertebrate: 100
Great blue heron (*Ardea herodias*)	2.229	0.0963[o]	2.0[m]	0.101	Invertebrate: 50 Fish: 50

[a]From EPA (1993), except where noted.
[b]Based on allometric equations from Nagy (1987), as presented in EPA (1993), except where noted.
[c]From Beyer and others (1994). Data are species-specific except where noted.
[d]Based on allometric equations from Calder and Braun (1983), as presented in EPA (1993), except where noted.
[e]Diets of sheep, deer mouse, muskrat, burrowing owl, killdeer, and great blue heron are generalized to emphasize specific trophic levels. Dietary compositions of the red fox, raccoon, and mallard are based on species-specific information presented in EPA (1993) and Martin and others (1951) and have been rounded to increments of 10 percent.
[f]From IAEA (1994).
[g]Based on soil ingestion for bison from Beyer and others (1994).
[h]Weight of deer mouse based on data specific to New Mexico from Silva and Downing (1995).
[i]Based on soil ingestion for the white-footed mouse from Beyer and others (1994).
[j]Based on species-specific food intake rate from EPA (1993), with assumed water content of food of 80 percent.
[k]Based on soil/sediment ingestion for raccoon from Beyer and others (1994).
[l]From Dunning (1993).
[m]No data available. Assumed value of 2 percent is based on the detection limit of the method used by Beyer and others (1994).
[n]Based on the mean soil/sediment ingestion rate of four species of sandpipers as reported by Beyer and others (1994).
[o]Based on species-specific food intake rate from EPA (1993), with assumed water content of food of 76 percent.

Table 6–25. Surface Water and Sediment Quality Benchmarks for Ecological Contaminants of Potential Concern for the Protection of Freshwater Aquatic Life

Contaminant of Potential Concern	Water Quality Benchmarks (mg/L)				Sediment Quality Benchmarks (mg/kg)	
	AWQC[a]	NNSWQS[b]	Tier II[c]	Other	TEL[d]	Other
Ammonium	--	--	--	0.18[e]	--	75[f]
Boron	--	--	0.0016	1.0[g]	--	--
Manganese	--	--	0.08	--	630	--
Molybdenum	--	--	0.24	--	--	4.0[h]
Nitrate	--	--	--	177[i]	--	2,440[j]
Selenium	0.005	0.002	--	--	--	5.0[k]
Strontium	--	--	1.5	--	--	--
Sulfate	--	--	--	100[l]	--	--
Uranium	--	--	0.0026	0.30[m]	--	--
Vanadium	--	--	0.019	--	--	--

[a]EPA ambient water quality criteria (EPA 1999, Buchman 1999). Hardness of 100 mg/L $CaCO_3$ was used for all hardness-dependent values.
[b]Navajo Nation Surface Water Quality Standard for warm water aquatic life.
[c]Tier II secondary chronic value from Suter and Tsao (1996).
[d]Threshold effect level (lowest) from Buchman (1999).
[e]State of New Mexico Standard for Interstate and Intrastate Streams (WQCC 2000), based on pH range of 6.6–8.8 and maximum temperature of 32.2° C, and converted from mg N (as NH_3)/L to mg NH_4/L.
[f]EPA Region V guideline for Great Lakes harbor sediment, nonpolluted (from Haines and others 1994).
[g]From Eisler (1994).
[h]Sediment quality guideline for the protection of agricultural uses (from Haines and others 1994).
[i]Guideline from British Columbia (Haines and others 1994) converted from μg N/L to mg NO_3/L.
[j]Lowest effect level (Ontario) for total kjeldahl nitrogen (from Haines and others 1994) and converted from mg N/L to mg NO_3/L.
[k]Sediment quality criterion from British Columbia (Haines and others 1994).
[l]Maximum concentration value (tentative) from British Columbia for the protection of aquatic life (Haines and others 1994).
[m]Maximum concentration value (British Columbia) for total uranium (from Haines and others 1994), presented in contrast to the Tier II secondary chronic value.
-- = No value available.

For plants, toxicity benchmarks are based primarily on the information provided in Efroymson and others (1997). These benchmarks are based on lowest-observed-adverse-effect levels (LOAELs) using 20 percent reduction in growth as the endpoint. Only the soil-based (rather than solution-based) benchmarks were used. Although based on LOAELs, these benchmarks are considered conservative. The endpoint is sublethal and reductions in plant growth may have no significant effect on the reproductive potential or the continued existence of a plant population. Further, these benchmarks are primarily based on studies in which the chemical of interest is added freshly to a soil (often as a soluble salt) and is typically more bioavailable than the COPCs in field situations where they have had time to bind more strongly with soil particles. The plant toxicity benchmarks are presented in Table 6–26.

Table 6–26. Plant Toxicity Benchmarks for Ecological Contaminants of Potential Concern

Contaminant of Potential Concern	Plant Toxicity Benchmark (mg/kg soil)[a]
Ammonium	---
Boron	0.5
Manganese	500
Molybdenum	2.0
Nitrate	---
Selenium	1.0
Strontium	---
Sulfate	---
Uranium	5.0
Vanadium	2.0

[a]From Efroymson and others (1997).
--- = No benchmark available.

For the wildlife receptors, no-observed-adverse-effect levels (NOAELs) for chronic oral exposure are used as benchmarks for toxic effects. The endpoints of particular interest in this assessment are those associated with reproductive health, development, and mortality. Therefore, NOAELs are defined as the maximum dosage tested that produced no effect that would be considered adverse to the receptor's survival, growth, or reproductive capacity. Because the NOAELs for the wildlife receptor species are based on NOAELs from test species, the latter are scaled to NOAELs specific to the wildlife receptor species using a power function of the ratio of body weights, as described by Sample and others (1996) and Sample and Arenal (1999). This scaling is based on the equation:

$$NOAEL_W = NOAEL_T \left(\frac{BW_T}{BW_W}\right)^s$$

where

$NOAEL_W$ is the no-observed-adverse-effect level for the wildlife receptor species (mg/kg-day),
$NOAEL_T$ is the no-observed-adverse-effect level for the test species (mg/kg-day),
BW_T is the body weight of the test species (kg),
BW_W is the body weight of the wildlife receptor species (kg), and
s is the body weight scaling factor; (s = 0.06 for mammals and s = -0.2 for birds (Sample and Arenal 1999).

Toxicity studies were considered to be chronic if they are conducted over a period of 26 weeks (one-half year) or more. This period represents the period of seasonal use by migratory and hibernating species and is sufficient time for small animals to complete their reproductive cycles. Studies of lesser duration (i.e., 1 to 25 weeks) are considered subchronic, unless they specifically included reproductive effects as endpoints (Sample and others 1996). When only subchronic oral $NOAEL_T$ values were available, these are converted to chronic $NOAEL_T$ values by applying an uncertainty factor of 0.1 (Sample and others 1996).

When only a chronic LOAEL value was available for test data, an uncertainty factor of 0.1 was used to convert it to the chronic $NOAEL_T$. If only a subchronic LOAEL was available, then an uncertainty factor of 0.01 was used to estimate the chronic $NOAEL_T$. This uncertainty factor is the product of two uncertainty factors of 0.1, one to convert the subchronic value to a chronic value and the other to convert the LOAEL to an NOAEL.

When possible, NOAELs for the wildlife receptor species are derived from test species that are taxonomically close to the target receptor. NOAELs were not determined if toxicity data could not be found for test species within the same class. Therefore, NOAELs for mammalian receptors are derived only from mammalian test species data and NOAELs for avian receptors are derived only from avian test species data. These data are presented in Table 6–27 and Table 6–28.

Table 6–27. Mammal Toxicity Benchmarks for Ecological Contaminants of Potential Concern

Contaminant of Potential Concern	Mammalian Test Data[a]			Mammalian Receptor NOAELs (mg/kg-day)				
	Test Species	Body weight (kg)	NOAEL (mg/kg-day)	Sheep	Deer mouse	Red fox	Muskrat	Raccoon
Ammonium	---	---	---	---	---	---	---	---
Boron	Rat	0.35	28.0	20.8	32.9	24.0	26.1	23.7
Manganese	Rat	0.35	88.0	65.3	103	75.5	82.0	74.4
Molybdenum	Mouse	0.03	0.26	0.167	0.264	0.192	0.209	0.190
Nitrate	Guinea pig	0.86	507	397	629	459	499	452
Selenium	Rat	0.35	0.20	0.149	0.235	0.171	0.186	0.169
Strontium	Rat	0.35	263	195	309	226	245	222
Sulfate	---	---	---	---	---	---	---	---
Uranium	Mouse	0.028	3.07	1.96	3.10	2.26	2.46	2.23
Vanadium	Rat	0.26	0.21	0.153	0.242	0.177	0.192	0.174

[a]From Sample and others (1996).

Table 6–28. Avian Toxicity Benchmarks for Ecological Contaminants of Potential Concern

Contaminant of Potential Concern	Avian Test Data[a]			Avian Receptor NOAELs (mg/kg-day)			
	Test Species	Body weight (kg)	NOAEL (mg/kg-day)	Burrowing owl	Mallard	Killdeer	Great blue heron
Ammonium	---	---	---	---	---	---	---
Boron	Mallard	1.0	28.8	19.8	29.5	18.0	33.8
Manganese	Japanese quail	0.072	977	1,140	1,700	1,040	1,940
Molybdenum	Chicken	1.5	3.53	2.24	3.34	2.04	3.82
Nitrate	---	---	---	---	---	---	---
Selenium	Mallard	1.0	0.40	0.276	0.410	0.251	0.470
Strontium	---	---	---	---	---	---	---
Sulfate	---	---	---	---	---	---	---
Uranium	Black duck	1.25	16.0	10.5	15.7	9.59	18.0
Vanadium	Mallard	1.17	11.4	7.61	11.3	6.92	13.0

[a]From Sample and others (1996).

6.2.3 Risk Characterization

The potential for risk to ecological receptors is determined through hazard quotients (HQs). HQs are specific to a particular receptor for exposure to a particular COPC. An HQ is defined by:

$$HQ = \frac{Exposure}{Benchmark}$$

For aquatic and benthic organisms and plants, exposures are equivalent to media concentrations (surface water for aquatic organisms and sediment or soil for benthic organisms and plants). For wildlife and livestock, exposures are modeled from multiple pathways by the methods described in Section 6.2.2.1. The methods for determining toxicity benchmark values for these receptors are discussed in Section 6.2.2.2.

The value of the HQ is greater than 1.0 if the magnitude of the exposure is greater than the corresponding benchmark, and conversely, the HQ is less than or equal to 1.0 if the exposure is less than or equal to the benchmark. An HQ value less than or equal to 1.0 is interpreted as evidence of no potential risk to that receptor for that COPC. If the HQs for a COPC are less than unity for all receptors, that COPC is eliminated from further consideration at that site. However, because exposure for the screening of COPCs is conservatively estimated, an HQ value greater than unity is not interpreted as evidence of risk, but only as evidence that the potential for risk cannot be ruled out.

For the purposes of this evaluation, potential exposures are conservatively based on the maximum measured COPC in each medium of ecological concern (surface water, sediment, and soil), as appropriate to each area. In addition, the UCL_{95} concentrations were used to calculate HQs that better reflect average (yet still conservatively estimated) risks to receptors in these areas. Measured concentrations of COPCs in wetland and upland plants were used in the calculation of exposures to herbivores when such data were available. The following are area-specific summaries of these results.

6.2.3.1 Area A: The Distributary Channel and Tributaries

Table 6–29 summarizes the water and sediment data for Area A as used to evaluate potential ecological risks. The surface water data for the distributary channel and its tributaries show exceedences of the water quality benchmarks for ammonium, manganese, nitrate, selenium, strontium, sulfate, and uranium (Table 6–30). The maximum HQ based on comparisons of maximum measured COPC concentrations in water to corresponding water quality benchmarks is 214 for selenium. Those of nitrate and strontium are less than 10. When based on the UCL_{95} concentrations, the maximum HQ (again for selenium) drops to 92.5, and all except selenium, sulfate, and uranium drop to values less than 10. The HQs for molybdenum and vanadium were all less than unity.

In the sediment data from this area, only selenium exceeded its sediment quality benchmark, with all HQs being less than 10 (Table 6–30). Sediment quality benchmark values are not available for strontium, sulfate, uranium, and vanadium. Based on the maximum sediment concentrations, HQs for wetland plants at the area exceeded unity for selenium, uranium, and vanadium, with a maximum HQ (for selenium) of 23.1. Those for uranium and vanadium were near unity and

Table 6–29. Summary of Surface Water and Sediment Data from the Distributary Channel and Tributaries (Area A)

Contaminant of Potential Concern	Surface Water (mg/L)		Sediment (mg/kg)	
	Maximum	UCL_{95}	Maximum	UCL_{95}[a]
Ammonium	3.00	0.458	49.1	24.3
Manganese	1.90	0.315	368	303
Molybdenum	0.0197	0.00870	0.394[b]	0.174[b]
Nitrate	515	194	243	163
Selenium	0.428	0.185	23.1	11.8
Strontium	9.78	5.81	407	339
Sulfate	5,670	2,730	27,800	21,400
Uranium	0.102	0.0518	5.40	4.64
Vanadium	0.00250	0.00114	2.50[b]	1.14[b]

[a]Upper confidence limit (one-tailed). One-half the detection limit used for nondetects. Confidence limit only presented when less than the maximum measured concentration.
[b]Concentration is estimated from surface water concentration based on the distribution coefficient.

Table 6–30. Hazard Quotients for Aquatic Organisms, Benthic Organisms, and Plants at the Distributary Channel and Tributaries (Area A) Based on Comparisons of Media Concentrations to Water Quality, Sediment Quality, and Plant Toxicity Benchmarks

Contaminant of Potential Concern	Aquatic Organisms		Benthic Organisms	
	HQ based on maximum surface water concentration	HQ based on UCL_{95}[a] of surface water concentrations	HQ based on maximum sediment concentration	HQ based on UCL_{95}[a] of sediment concentrations
Ammonium	**16.7**	**2.54**	0.655	0.324
Manganese	**23.8**	**3.94**	0.584	0.481
Molybdenum	0.0821	0.0363	0.0985[b]	0.0435[b]
Nitrate	**2.91**	**1.10**	0.0996	0.0668
Selenium	**214**	**92.5**	**4.62**	**2.36**
Strontium	**6.52**	**3.87**	NB	NB
Sulfate	**56.7**	**27.3**	NB	NB
Uranium	**39.2**	**19.9**	NB	NB
Vanadium	0.132	0.0600	NB	NB

Contaminant of Potential Concern	Wetland Plants	
	HQ based on maximum sediment concentration	HQ based on UCL_{95}[a] of sediment concentrations
Ammonium	NB	NB
Manganese	0.736	0.606
Molybdenum	0.197[b]	0.0870[b]
Nitrate	NB	NB
Selenium	**23.1**	**11.8**
Strontium	NB	NB
Sulfate	NB	NB
Uranium	**1.08**	0.928
Vanadium	**1.25[b]**	0.570[b]

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
[b]Based on sediment concentration estimated from the surface water concentration by using the distribution coefficient.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

dropped to less than unity when based on the UCL_{95} concentrations (the concentrations of vanadium in sediment were estimated from surface water concentrations based on the distribution coefficient). The HQ for wetland plants exposed to selenium in sediment was still greater than 10 when based on the UCL_{95}.

Selenium was the only COPC to show HQs greater than unity for terrestrial wildlife receptors and livestock (Table 6–31). Based maximum selenium concentration data for surface water and sediment, HQs greater than unity were found for the deer mouse (HQ = 2.12) and sheep (HQ = 1.04). The UCL_{95} concentrations reduced the former to 1.62 and the latter to less than unity. For the wetland wildlife receptors (Table 6–32), all selenium HQs were greater than unity for both the maximum and UCL_{95}-based exposure point concentrations. The three chiefly predatory species (raccoon, killdeer, and great blue heron) showed the highest HQs (46.1 for the killdeer was the maximum). Vanadium was the only other COPC for this area that showed an HQ greater than unity for the wetland wildlife receptors. The HQ for vanadium exposure in the raccoon based on maximum media concentrations (again, the concentration of vanadium in sediment was estimated from surface water concentrations) was 2.02. This dropped to below unity when based on the UCL_{95}. Site-specific data were available for the concentrations of manganese, selenium, strontium, and uranium in wetland plants (cattails and bulrushes) for this area, and these data were used in the modeling of exposure and risk in the wetland wildlife receptors.

Table 6–31. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors at the Distributary Channel and Tributaries (Area A)

Contaminant of Potential Concern	Sheep		Deer Mouse	
	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations
Ammonium	NB	NB	NB	NB
Manganese	0.0302	0.0206	0.0698	0.0500
Molybdenum	0.0118	0.00522	0.0107	0.00475
Nitrate	0.130	0.0488	0.118	0.0444
Selenium	**1.04**	0.739	**2.12**	**1.62**
Strontium	0.0341	0.0266	0.0761	0.0607
Sulfate	NB	NB	NB	NB
Uranium	0.0211	0.0125	0.0439	0.0266
Vanadium	0.00163	7.44×10^{-4}	0.00148	6.77×10^{-4}

Contaminant of Potential Concern	Red Fox		Burrowing Owl	
	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations
Ammonium	NB	NB	NB	NB
Manganese	0.00835	0.00493	3.04×10^{-6}	5.05×10^{-7}
Molybdenum	0.00871	0.00385	1.60×10^{-5}	7.08×10^{-6}
Nitrate	0.0955	0.0360	NB	NB
Selenium	0.384	0.231	0.00284	0.00123
Strontium	0.0103	0.00756	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.00746	0.00419	1.77×10^{-5}	8.97×10^{-6}
Vanadium	0.00120	5.48×10^{-4}	6.00×10^{-7}	2.73×10^{-7}

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

Table 6–32. Hazard Quotients for Riparian/Wetland Wildlife Receptors at the Distributary Channel and Tributaries (Area A)

Contaminant of Potential Concern	Muskrat		Raccoon		Mallard	
	HQ based on maximum concentration	HQ based on UCL_{95} [a] concentration	HQ based on maximum concentration	HQ based on UCL_{95} [a] concentration	HQ based on maximum concentration	HQ based on UCL_{95} [a] concentration
Ammonium	NB	NB	NB	NB	NB	NB
Manganese	0.339	0.159	0.177	0.0722	0.0111	0.00490
Molybdenum	0.124[b]	0.0547[b]	0.0990[b]	0.0437[b]	0.00529[b]	0.00233[b]
Nitrate	0.137	0.0624	0.240	0.0942	NB	NB
Selenium	**17.5**	**5.68**	**29.9**	**12.4**	**6.80**	**2.34**
Strontium	0.106	0.0494	0.0610	0.0331	NB	NB
Sulfate	NB	NB	NB	NB	NB	NB
Uranium	0.165	0.0556	0.119	0.0538	0.0178	0.00566
Vanadium	0.0893[b]	0.0407[b]	**2.02[b]**	0.920[b]	0.00391[b]	0.00178[b]

Contaminant of Potential Concern	Killdeer		Great Blue Heron	
	HQ based on maximum concentration	HQ based on UCL_{95} [a] concentration	HQ based on maximum concentration	HQ based on UCL_{95} [a] concentration
Ammonium	NB	NB	NB	NB
Manganese	0.0179	0.00702	0.00309	6.20×10^{-4}
Molybdenum	0.0139[b]	0.00614[b]	0.00589[b]	0.00260[b]
Nitrate	NB	NB	NB	NB
Selenium	**46.1**	**20.1**	**15.5[b]**	**6.72**
Strontium	NB	NB	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.0389	0.0232	0.0171	0.00879
Vanadium	0.111[b]	0.0505	0.0626[b]	0.0286[b]

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
[b]Based on sediment concentration estimated from the surface water concentration by using the distribution coefficient.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

In summary, selenium is the principal risk driver in Area A. Potential risks from selenium exposure may exist for aquatic and benthic organisms, wetland plants, and wildlife (especially predators) that are associated with the wetland habitats. Risks to terrestrial wildlife and livestock that may use the area (principally being exposed through drinking water) are minimal. Ammonium, manganese, nitrate, strontium, sulfate, and uranium are also of potential risk to aquatic organisms in this area. No potential for risk from exposure to molybdenum was found, and the potential for risk from vanadium exposure (limited to maximum HQs of 2.02 or less in the raccoon and in wetland plants) is negligible.

6.2.3.2 Area B: The San Juan River

Table 6–33 summarizes the water and sediment data for Area B as used to evaluate potential ecological risks. The surface water data for the San Juan River show exceedences of the water quality benchmarks for manganese, selenium, strontium, sulfate, and uranium (Table 6–34). The maximum HQ based on comparisons of maximum measured COPC concentrations in water to corresponding water quality benchmarks is 43.1 for uranium, although those of sulfate and selenium are near this value. Those for manganese and strontium are less than 10. When based on the UCL_{95} concentrations, all of these HQs drop to values less than 10 (the HQ for manganese drops to less than 1); those for selenium, sulfate, and uranium are less than 3.5. The HQs for ammonium, molybdenum, nitrate, and vanadium are all less than unity.

Table 6–33. Summary of Surface Water and Sediment Data from the San Juan River (Area B)

Contaminant of Potential Concern	Surface Water (mg/L)		Sediment (mg/kg)	
	Maximum	UCL_{95}[a]	Maximum	UCL_{95}[a]
Ammonium	0.164	0.0393	1.80	1.52
Manganese	0.592	0.0518	229	224
Molybdenum	0.00990	0.00220	< 27.0[b]	< 27.0[b]
Nitrate	104	6.04	39.0	30.2
Selenium	0.0787	0.00411	< 0.20[b]	< 0.20[b]
Strontium	14.6	1.52	45.0	44.9
Sulfate	4,190	346	2,660	2,360
Uranium	0.112	0.00713	0.250	0.235
Vanadium	0.00170	0.000968	19.4	17.6

[a]Upper confidence limit (one-tailed). One-half the detection limit used for nondetects. Confidence limit only presented when less than the maximum measured concentration.
[b]Analyte not detected at the indicated detection limit.

Table 6–34. Hazard Quotients for Aquatic Organisms, Benthic Organisms, and Plants at the San Juan River (Area B) Based on Comparisons of Media Concentrations to Water Quality, Sediment Quality, and Plant Toxicity Benchmarks

Contaminant of Potential Concern	Aquatic Organisms		Benthic Organisms	
	HQ based on maximum surface water concentration	HQ based on UCL_{95}[a] of surface water concentrations	HQ based on maximum sediment concentration	HQ based on UCL_{95}[a] of sediment concentrations
Ammonium	0.911	0.218	0.0240	0.0203
Manganese	**7.40**	0.648	0.363	0.356
Molybdenum	0.0413	0.00917	**3.38**[b]	**3.38**[b]
Nitrate	0.588	0.0341	0.0160	0.0124
Selenium	**39.4**	**2.06**	0.0200[b]	0.0200[b]
Strontium	**9.73**	**1.01**	NB	NB
Sulfate	**41.9**	**3.46**	NB	NB
Uranium	**43.1**	**2.74**	NB	NB
Vanadium	0.0895	0.0509	NB	NB

Contaminant of Potential Concern	Wetland Plants	
	HQ based on maximum sediment concentration	HQ based on UCL_{95}[a] of sediment concentrations
Ammonium	NB	NB
Manganese	0.458	0.448
Molybdenum	**6.75**[b]	**6.75**[b]
Nitrate	NB	NB
Selenium	0.100[b]	0.100[b]
Strontium	NB	NB
Sulfate	NB	NB
Uranium	0.0500	0.0470
Vanadium	**9.70**	**8.80**

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
[b]Analyte was not detected in sediment. Hazard quotient is based on one-half the detection limit for the sediment concentration.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

In the sediment data from this area, only molybdenum concentrations exceeded the sediment quality benchmark, with an HQ of 3.38 (Table 6–34). Molybdenum was not detected in sediment samples from the San Juan River and this HQ is based on one-half the detection limit of 27.0 mg/kg, or 13 mg/kg. The K_d of molybdenum is 20 L/kg, and the maximum concentration of molybdenum measured in the San Juan River water was 0.0099 mg/L. The estimated maximum sediment concentration is 0.198 mg/kg; it is highly likely that this detection limit greatly exceeded the actual molybdenum concentrations, and the HQ for benthic organisms is probably less than unity. Similarly, the HQs for wetland plants exposed to molybdenum in sediments (6.75) are also exaggerated by the use of one-half the detection limit as the exposure point concentration and are also probably less than unity. Vanadium also showed HQs greater than unity for wetland plants; however, the maximum vanadium concentration in sediment from this area (19.4 mg/kg) is much less than the typical background concentration of 50 mg/kg in sediment reported by Buchman (1999). It is therefore likely that these HQs are the result of a highly conservative plant toxicity benchmark for vanadium.

No HQs greater than unity were found for the terrestrial wildlife receptors and livestock (Table 6–35). For the wetland wildlife receptors (Table 6–36), HQs greater than unity were found for molybdenum, selenium, and vanadium. None of these exceeded 10. As discussed above, the HQs for molybdenum are probably overestimated by the use of one-half the detection limit for the sediment concentration. HQs greater than 1 for vanadium were limited to the raccoon and did not exceed 1.86 based on maximum media concentrations. As with molybdenum, selenium was not detected in sediment, and one-half the detection limit of 0.20 mg/kg was used as the sediment concentration for the risk analysis. HQs greater than 1 for selenium were limited to the three predatory species (raccoon, killdeer, and great blue heron) when based on the maximum media concentrations. All of these dropped to below unity when based on the UCL_{95}.

In summary, potential ecological risks at Area B are principally limited to aquatic receptors. In recent sampling rounds, concentrations of selenium, sulfate, and uranium have exceeded the water quality benchmarks for these analytes by factors of about 40, and those of manganese and strontium have also exceeded their benchmark values, but to a lesser degree. The UCL_{95}s of these COPCs, however, indicate that these maximum HQs do not represent the general water quality conditions of the river. In the case of selenium, for example, only 2 of the 60 filtered water sample data points were greater than the benchmark (0.002 mg/L). Both were from location 894, where the drainage channel from the floodplain (Area C) enters the river. All of the other data points were not only less than the water quality benchmark, but were also less than the maximum selenium concentration from the background river samples (0.0018 mg/L).

In the case of sulfate, 62 of the 70 data points exceeded the water quality benchmark of 100 mg/L; however, all but three were less than 300 mg/L. As with selenium, these two highest concentrations (both of which exceeded 1,000 mg/L) were from samples collected at location 894. The third highest (at location 940) was recorded during the low-flow sampling in February 2000. Because the UCL_{95} of the San Juan River reference samples was 329 mg/L, it can be concluded that all but these three highest data points are within background for the river. Similarly, in the case of manganese, only three samples (out of 60) exceeded the water quality benchmark, and only the two highest of these exceeded the range of the background data for the river. The two highest were from sampling location 894, and the third highest was from sampling location 551, which is adjacent to location 894. For strontium, only two samples (out of 52) exceeded the background range and the water quality benchmark.

Table 6–35. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors at the San Juan River (Area B)

Contaminant of Potential Concern	Sheep		Deer Mouse	
	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations
Ammonium	NB	NB	NB	NB
Manganese	9.06×10^{-4}	7.93×10^{-5}	8.24×10^{-4}	7.21×10^{-5}
Molybdenum	0.00594	0.00132	0.00540	0.00120
Nitrate	0.0262	0.00152	0.0238	0.00138
Selenium	0.0530	0.00277	0.0482	0.00252
Strontium	0.00748	7.78×10^{-4}	0.00680	7.08×10^{-4}
Sulfate	NB	NB	NB	NB
Uranium	0.00572	3.64×10^{-4}	0.0520	3.31×10^{-4}
Vanadium	0.00111	6.32×10^{-4}	0.00101	5.74×10^{-4}

Contaminant of Potential Concern	Red Fox		Burrowing Owl	
	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations
Ammonium	NB	NB	NB	NB
Manganese	6.68×10^{-4}	5.84×10^{-5}	9.49×10^{-7}	8.30×10^{-8}
Molybdenum	0.00438	9.73×10^{-4}	8.06×10^{-6}	1.79×10^{-6}
Nitrate	0.0193	0.00112	NB	NB
Selenium	0.0391	0.00204	5.21×10^{-4}	2.72×10^{-5}
Strontium	0.00551	5.74×10^{-4}	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.00421	2.68×10^{-4}	1.94×10^{-5}	1.23×10^{-6}
Vanadium	8.18×10^{-4}	4.66×10^{-4}	1.34×10^{-6}	2.32×10^{-7}

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

For uranium, 12 of the 60 data points (excluding the February 2000 data points) exceeded the water quality benchmark of 0.0026 mg/L. These were generally from sampling locations along the left (or millsite) bank of the river from location 1206 to location 1210. The two highest uranium concentrations were from sampling location 894. All other locations had concentrations of 0.0048 mg/L or less, which would result in HQs of less than 2. The data from February 2000 (Figure 4–19) show uranium concentrations remaining fairly consistent with those at the upstream locations except at location 940, where an HQ near 20 would be indicated. Therefore, uranium appears to be entering the river at concentrations exceeding its benchmark along a stretch of the left riverbank rather than from a point source as is indicated for the other COPCs in the river water.

No HQs greater than unity were found for ammonium and nitrate. The HQs greater than unity for molybdenum and vanadium are negligible or can be ascribed to conservatisms in the exposure point concentration (e.g., the use of one-half the detection limit for nondetects) or to conservative benchmarks (e.g., the plant toxicity benchmark for vanadium). No risk to terrestrial receptors or livestock was indicated for this area, and the potential risks to wetland predators from exposure to

selenium is expected to be isolated to the area of sampling location 894, because selenium was within background concentrations at all other locations along the river.

Table 6–36. Hazard Quotients for Riparian/Wetland Wildlife Receptors at the San Juan River (Area B)

Contaminant of Potential Concern	Muskrat		Raccoon		Mallard	
	HQ based on maximum concentration	HQ based on UCL_{95}[a] concentration	HQ based on maximum concentration	HQ based on UCL_{95}[a] concentration	HQ based on maximum concentration	HQ based on UCL_{95}[a] concentration
Ammonium	NB	NB	NB	NB	NB	NB
Manganese	0.588	0.575	0.217	0.198	0.0194	0.0189
Molybdenum	**3.93**[b]	**3.93**[b]	**1.51**[b]	**1.49**[b]	0.159[b]	0.159
Nitrate	0.0262	0.00569	0.0479	0.00432	NB	NB
Selenium	0.0593[b]	0.0202[b]	**4.41**[b]	0.237[b]	0.285[b]	0.0196[b]
Strontium	0.0382	0.0329	0.0449	0.0146	NB	NB
Sulfate	NB	NB	NB	NB	NB	NB
Uranium	0.00526	0.00104	0.0664	0.00474	0.00146	1.32×10^{-4}
Vanadium	0.684	0.620	**1.86**	**1.24**	0.00575	0.00442

Contaminant of Potential Concern	Killdeer		Great Blue Heron	
	HQ based on maximum concentration	HQ based on UCL_{95}[a] concentration	HQ based on maximum concentration	HQ based on UCL_{95}[a] concentration
Ammonium	NB	NB	NB	NB
Manganese	0.00749	0.00407	0.00101	1.79×10^{-4}
Molybdenum	0.120[b]	0.116[b]	0.00597[b]	0.00370[b]
Nitrate	NB	NB	NB	NB
Selenium	**8.19**[b]	0.434[b]	**2.85**[b]	0.149[b]
Strontium	NB	NB	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.0325	0.00246	0.0185	0.00119
Vanadium	0.120	0.0846	0.0438	0.0254

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
[b] Analyte was not detected in sediment. Hazard quotient is based on one-half the detection limit for the sediment concentration.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

6.2.3.3 Area C: Shiprock Floodplain

Table 6–37 summarizes the water, sediment, and soil data for Area C as used to evaluate potential ecological risks. The surface water data for the Shiprock floodplain area show exceedences of the water quality benchmarks for ammonium, manganese, nitrate, selenium, strontium, sulfate, and uranium (Table 6–38). The maximum HQ based on comparisons of maximum measured COPC concentrations in water to corresponding water quality benchmarks is 262 for uranium. Those of manganese and sulfate also exceeded 100. When based on the UCL_{95} concentrations, the maximum HQ (again for uranium) decreases to 88.8. The HQ for ammonium decreases to below unity. The HQs for boron, molybdenum, and vanadium were all less than unity.

Table 6–37. Summary of Surface Water, Sediment, and Soil Data from the Shiprock Floodplain (Area C)

Contaminant of Potential Concern	Surface Water (mg/L)		Sediment (mg/kg)		Soil (mg/kg)	
	Maximum	UCL_{95}[a]	Maximum	UCL_{95}[a]	Maximum	UCL_{95}[a]
Ammonium	1.02	0.168	31.6	5.21	25.9	15.6
Boron	0.560	> maximum	1.68[b]	> maximum	ND	ND
Manganese	16.4	1.36	1,190	578	723	398
Molybdenum	0.0124	0.00859	0.248[b]	0.172[b]	ND	ND
Nitrate	2,460	280	83.5	40.8	1,010	444
Selenium	0.137	0.0361	4.20	2.76	2.00	1.13
Strontium	19.8	11.3	1,620	545	349	188
Sulfate	17,100	4,790	12,300	7,670	42,300	31,400
Uranium	0.682	0.231	43.5	14.1	35.6	14.6
Vanadium	0.0056	0.00191	5.60[b]	1.91[b]	ND	ND

[a]Upper confidence limit (one-tailed). One-half the detection limit used for nondetects. Confidence limit only presented when less than the maximum measured concentration.
[b]Concentration is estimated from surface water concentration based on the distribution coefficient.
ND = no data

In the sediment data from this area, only the maximum concentration of manganese resulted in an HQ greater than 1 (Table 6–38). The HQ drops to less than 1 when based on the UCL_{95} concentration of manganese in sediment. Sediment quality benchmark values are not available for strontium, sulfate, uranium, and vanadium. Based on the maximum sediment concentrations, HQs for wetland plants in the floodplain area exceeded unity for boron, manganese, selenium, uranium, and vanadium. The maximum HQ was 8.70 (for uranium). With the exception of boron, which was not further assessed, all of these HQs dropped to less than 3 when based on the UCL_{95} concentrations, and that for vanadium dropped to less than unity. (The concentrations of boron, molybdenum, and vanadium in sediment were estimated from surface water concentrations based on the distribution coefficient.) Risk to upland plants was assessed based on comparisons of soil concentrations to plant toxicity benchmarks. The HQs for upland plants were greater than 1 for manganese, selenium, and uranium when based on the maximum soil concentrations of these COPCs, and were greater than 1 for selenium and uranium when based on the UCL_{95} concentrations (Table 6–38). Uranium had the highest HQs for upland plants, at 7.12 and 2.92 for the maximum and UCL_{95} concentrations, respectively.

Potential exposures to manganese and selenium showed HQs greater than unity for the deer mouse (Table 6–39). In neither case did the HQs exceed a value of 3. Maximum manganese concentrations also resulted in an HQ of 1.16 for sheep, but this dropped to below unity when based on the UCL_{95} concentrations. No risks were predicted for the red fox or burrowing owl. For the wetland wildlife receptors (Table 6–40), all selenium HQs were greater than unity for the maximum concentrations, but these were limited to the predatory species (raccoon, killdeer, and great blue heron) when based on the UCL_{95} concentrations. The highest HQs were for the killdeer, which ranged from 14.5 to 3.95 (based on the maximum and UCL_{95} concentrations, respectively). Vanadium showed HQs greater than unity for the raccoon (ranging from 4.52 to 1.54 for the maximum and UCL_{95} concentrations, respectively). The maximum HQs exceeding unity for nitrate in the raccoon and for uranium in the muskrat were considered negligible. Both are less than unity when based on the UCL_{95} concentrations for this area. Site-specific data were available for the concentrations of manganese, selenium, strontium, and uranium in wetland plants (cattails and bulrushes) and upland plants (greasewood, cottonwood, and Russian olive) for this area, and these

data were used in the modeling of exposure and risk in the wetland and terrestrial wildlife receptors, respectively.

Table 6–38. Hazard Quotients for Aquatic Organisms, Benthic Organisms, and Plants at the Shiprock Floodplain (Area C) Based on Comparisons of Media Concentrations to Water Quality, Sediment Quality, and Plant Toxicity Benchmarks

Contaminant of Potential Concern	Aquatic Organisms		Benthic Organisms	
	HQ based on maximum surface water concentration	HQ based on UCL_{95}[a] of surface water concentrations	HQ based on maximum sediment concentration	HQ based on UCL_{95}[a] of sediment concentrations
Ammonium	**5.67**	0.933	0.422	0.0694
Boron	0.56	NA	NB	NB
Manganese	**205**	**17.0**	**1.89**	0.917
Molybdenum	0.0517	0.0358	0.0620[b]	0.0430[b]
Nitrate	**13.9**	**1.58**	0.0342	0.0167
Selenium	**68.5**	**18.1**	0.840	0.552
Strontium	**13.2**	**7.53**	NB	NB
Sulfate	**171**	**47.9**	NB	NB
Uranium	**262**	**88.8**	NB	NB
Vanadium	0.295	0.101	NB	NB

Contaminant of Potential Concern	Wetland Plants		Upland Plants	
	HQ based on maximum sediment concentration	HQ based on UCL_{95}[a] of sediment concentrations	HQ based on maximum soil concentration	HQ based on UCL_{95}[a] of soil concentrations
Ammonium	NB	NB	NB	NB
Boron	**3.36**[b]	NA	ND	ND
Manganese	**2.38**	**1.16**	**1.45**	0.796
Molybdenum	0.124[b]	0.0859[b]	ND	ND
Nitrate	NB	NB	NB	NB
Selenium	**4.20**	**2.76**	**2.00**	**1.13**
Strontium	NB	NB	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	**8.70**	**2.82**	**7.12**	**2.92**
Vanadium	**2.80**[b]	0.955[b]	ND	ND

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
[b]Sediment concentration was estimated from surface water concentration based on the distribution coefficient.
NA = Not assessed because UCL_{95} exceeded the maximum measured concentration.
NB = No benchmark available
ND = No data for the specified medium available
Bold indicates hazard quotient greater than 1.

In summary, manganese, selenium, sulfate, and uranium are the principal risk drivers in surface water at Area C. Although potential risks to benthic organisms appear to be low in this area, manganese, selenium, and uranium in sediments and soil may pose risks to wetland and upland plant communities. Minor risks to terrestrial herbivores may exist on the floodplain from selenium and manganese exposures, and potential risks to wetland predators may exist from exposures to selenium in the food chain. No potential for risk from exposure to molybdenum was found, and the potential for risks from ammonium, boron, nitrate, strontium, and vanadium exposures at this area are negligible.

Table 6–39. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors at the Shiprock Floodplain (Area C)

Contaminant of Potential Concern	Sheep		Deer Mouse	
	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations
Ammonium	NB	NB	NB	NB
Boron	0.00269	NA	0.00245	NA
Manganese	**1.16**	0.421	**2.75**	1.00
Molybdenum	0.00744	0.00516	0.00677	0.00469
Nitrate	0.729	0.118	0.819	0.176
Selenium	0.991	0.780	**2.23**	**1.85**
Strontium	0.110	0.0707	0.246	0.160
Sulfate	NB	NB	NB	NB
Uranium	0.0963	0.0406	0.0970	0.0464
Vanadium	0.00366	0.00125	0.00332	0.00113

Contaminant of Potential Concern	Red Fox		Burrowing Owl	
	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations
Ammonium	NB	NB	NB	NB
Boron	0.00198	NA	5.15×10^{-5}	NA
Manganese	0.291	0.105	0.00289	0.00158
Molybdenum	0.00548	0.00380	1.01×10^{-5}	6.99×10^{-6}
Nitrate	0.483	0.0635	NB	NB
Selenium	0.491	0.364	0.364	0.289
Strontium	0.0314	0.0196	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.0733	0.0291	0.0201	0.00822
Vanadium	0.00269	9.19×10^{-4}	1.34×10^{-6}	4.58×10^{-7}

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
NA = Not assessed because UCL_{95} exceeded the maximum measured concentration.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

6.2.3.4 Area D: Bob Lee Wash

Table 6–41 summarizes the water, sediment, and soil data for Area D as used to evaluate potential ecological risks. The surface water data for Bob Lee Wash show exceedences of the water quality benchmarks for ammonium, nitrate, selenium, strontium, sulfate, and uranium (Table 6–42). The maximum HQ based on comparisons of maximum measured COPC concentrations in water to corresponding water quality benchmarks is 929 for uranium. Those of selenium and sulfate were also high (greater than 50). When based on the UCL_{95} concentrations, the maximum HQ (again for uranium) decreases to 468. The HQs for manganese, molybdenum, and vanadium were all less than unity.

Table 6–40. Hazard Quotients for Riparian/Wetland Wildlife Receptors at the Shiprock Floodplain (Area C)

Contaminant of Potential Concern	Muskrat		Raccoon		Mallard	
	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.
Ammonium	NB	NB	NB	NB	NB	NB
Boron	0.0200	NA	0.0108	NA	0.0123	NA
Manganese	0.696	0.263	0.724	0.145	0.0246	0.0123
Molybdenum	0.0779	0.0540	0.0623	0.0432	0.00333	0.00230
Nitrate	0.496	0.0610	**1.09**	0.125	NB	NB
Selenium	**2.55**	0.697	**8.54**	**2.29**	**1.24**	0.324
Strontium	0.389	0.224	0.191	0.105	NB	NB
Sulfate	NB	NB	NB	NB	NB	NB
Uranium	**1.66**	0.328	0.989	0.258	0.178	0.0350
Vanadium	0.200	0.0682	**4.52**	**1.54**	0.00875	0.00299

Contaminant of Potential Concern	Killdeer		Great Blue Heron	
	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.
Ammonium	NB	NB	NB	NB
Boron	0.0175	NA	0.00330	NA
Manganese	0.121	0.0181	0.0258	0.00235
Molybdenum	0.00875	0.00606	0.00371	0.00257
Nitrate	NB	NB	NB	NB
Selenium	**14.5**	**3.95**	**4.97**	**1.31**
Strontium	NB	NB	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.274	0.0916	0.115	0.0389
Vanadium	0.248	0.0847	0.140	0.0479

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
NA = Not assessed because UCL_{95} exceeded the maximum measured concentration.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

Table 6–41. Summary of Surface Water, Sediment, and Soil Data from Bob Lee Wash (Area D)

Contaminant of Potential Concern	Surface Water (mg/L)		Sediment (mg/kg)		Soil (mg/kg)	
	Maximum	UCL_{95}[a]	Maximum	UCL_{95}[a]	Maximum	UCL_{95}[a]
Ammonium	5.00	1.13	14.7[b]	> maximum[b]	14.7	> maximum
Manganese	0.0568	0.0256	533	NC[c]	262	> maximum
Molybdenum	0.0205	0.0158	0.410[d]	0.316[d]	ND	ND
Nitrate	2,110	628	1,120[b]	> maximum[b]	1,120	> maximum
Selenium	0.119	0.0392	0.450	NC[c]	0.570	> maximum
Strontium	12.5	11.9	232	NC[c]	407	> maximum
Sulfate	12,900	5,650	50,200[b]	> maximum[b]	50,200	> maximum
Uranium	2.42	1.22	3.84	NC[c]	40.2	> maximum
Vanadium	0.00520	> maximum	5.20[d]	> maximum[d]	ND	ND

[a]Upper confidence limit (one-tailed). One-half the detection limit used for nondetects. Confidence limit only presented when less than the maximum measured concentration.
[b]Not analyzed in sediment. Soil data used to approximate sediment concentration.
[c]Confidence limit not calculated because only one sediment sample has been collected at this area.
[d]Not analyzed in sediment. Sediment concentration estimated from surface water concentration based on the distribution coefficient.
ND = no data
NC = not calculated

Table 6–42. Hazard Quotients for Aquatic Organisms, Benthic Organisms, and Plants at Bob Lee Wash (Area D) Based on Comparisons of Media Concentrations to Water Quality, Sediment Quality, and Plant Toxicity Benchmarks

Contaminant of Potential Concern	Aquatic Organisms		Benthic Organisms	
	HQ based on maximum surface water concentration	HQ based on UCL_{95}[a] of surface water concentrations	HQ based on maximum sediment concentration	HQ based on UCL_{95}[a] of sediment concentrations
Ammonium	**27.8**	**6.28**	0.196[b]	NA[b,c]
Manganese	0.710	0.320	0.846	NA[d]
Molybdenum	0.0854	0.0658	0.103[e]	0.0790[e]
Nitrate	**11.9**	**3.55**	0.459[b]	NA[b,c]
Selenium	**59.5**	**19.6**	0.0900	NA[d]
Strontium	**8.33**	**7.93**	NB	NB
Sulfate	**129**	**56.5**	NB	NB
Uranium	**929**	**468**	NB	NB
Vanadium	0.274	NA[c]	NB	NB

Contaminant of Potential Concern	Wetland Plants		Upland Plants	
	HQ based on maximum sediment concentration	HQ based on UCL_{95}[a] of sediment concentrations	HQ based on maximum soil concentration	HQ based on UCL_{95}[a] of soil concentrations
Ammonium	NB	NB	NB	NB
Manganese	**1.07**	NA[d]	0.524	NA[c]
Molybdenum	0.205[e]	0.158[e]	ND	ND
Nitrate	NB	NB	NB	NB
Selenium	0.450	NA[d]	0.570	NA[c]
Strontium	NB	NB	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.768	NA[d]	**8.04**	NA[c]
Vanadium	**2.60[e]**	NA[c]	ND	ND

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
[b]Soil data used to approximate sediment concentration.
[c]Not assessed because UCL_{95} exceeded the maximum measured concentration.
[d]Not assessed because insufficient samples collected to calculate the UCL_{95}.
[e]Based on sediment concentration estimated from surface water concentration using the distribution coefficient.
NA = Not assessed
NB = No benchmark available
ND = No data for the specified medium available
Bold indicates hazard quotient greater than 1.

None of the measured COPC concentrations in sediment resulted in HQs greater than 1 (Table 6–42). Sediment quality benchmark values are not available for strontium, sulfate, uranium, and vanadium. Based on the sediment concentrations, HQs for wetland plants in the floodplain area exceeded unity for manganese and vanadium, with a maximum HQ (for vanadium) of 2.60. (The concentration of vanadium in sediment was estimated from surface water concentrations based on the distribution coefficient.) Risk to upland plants was assessed based on comparisons of soil concentrations to plant toxicity benchmarks. Based on the maximum soil concentration, the HQ for upland plants exposed to uranium was 8.04. Neither of the other two COPCs for which HQs could be calculated for upland plants (manganese and selenium) showed potential risk.

Of the terrestrial receptors, HQs greater than unity were limited to manganese exposure in the deer mouse (Table 6–43). This exceedence (HQ = 1.19) is negligible. Among the wetland wildlife receptors (Table 6–44), selenium HQs were greater than unity for the predatory species (raccoon, killdeer, and great blue heron). The highest HQs were for the killdeer, which ranged from 12.4 to 4.11 (based on the maximum and UCL_{95} concentrations, respectively). Uranium and vanadium showed HQs greater than unity for the raccoon. In the case of uranium, the maximum HQ was low (1.43) and did not exceed unity when based on the UCL_{95}. For vanadium, the HQ was 4.20 for both the maximum and UCL_{95} concentrations, and is also considered of low magnitude. Site-specific data were available for the concentrations of manganese, selenium, strontium, and uranium in wetland plants (cattails and bulrushes) for this area, and these data were used in the modeling of exposure and risk in the wetland wildlife receptors.

Table 6–43. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors at Bob Lee Wash (Area D)

Contaminant of Potential Concern	Sheep		Deer Mouse	
	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations
Ammonium	NB	NB	NB	NB
Manganese	0.492	0.492	**1.19**	**1.19**
Molybdenum	0.0123	0.00948	0.0112	0.00862
Nitrate	0.652	0.278	0.766	0.427
Selenium	0.164	0.110	0.261	0.212
Strontium	0.220	0.220	0.522	0.522
Sulfate	NB	NB	NB	NB
Uranium	0.198	0.137	0.199	0.143
Vanadium	0.00339	0.00339	0.00309	0.00309

Contaminant of Potential Concern	Red Fox		Burrowing Owl	
	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations
Ammonium	NB	NB	NB	NB
Manganese	0.118	0.117	0.00104	0.00104
Molybdenum	0.00907	0.00699	1.67×10^{-5}	1.29×10^{-5}
Nitrate	0.421	0.146	NB	NB
Selenium	0.211	0.172	0.221	0.221
Strontium	0.0547	0.0545	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.146	0.101	0.0230	0.0227
Vanadium	0.00250	0.00250	1.25×10^{-6}	1.25×10^{-6}

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

In summary, selenium, sulfate, and uranium are the principal risk drivers in surface water at Area D. No risks to benthic organisms were found; however, uranium in soil may pose a risk to plant communities. No or negligible risks to terrestrial wildlife and livestock receptors were found for this area. Potential risks to wetland predators may exist from exposures to selenium in the food chain. No potential for risk from exposure to molybdenum was found, and the potential for risks from ammonium, nitrate, strontium, and vanadium exposures at this area are negligible.

Table 6–44. Hazard Quotients for Riparian/Wetland Wildlife Receptors at Bob Lee Wash (Area D)

Contaminant of Potential Concern	Muskrat		Raccoon		Mallard	
	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.
Ammonium	NB	NB	NB	NB	NB	NB
Manganese	0.227	0.182	0.0958	0.0803	0.00674	0.00523
Molybdenum	0.129	0.0993	0.103	0.0794	0.00550	0.00424
Nitrate	0.581	0.290	0.990	0.338	NB	NB
Selenium	0.154	0.0925	**6.69**	**2.22**	0.450	0.159
Strontium	0.549	0.350	0.209	0.143	NB	NB
Sulfate	NB	NB	NB	NB	NB	NB
Uranium	0.127	0.0729	**1.43**	0.729	0.0332	0.0174
Vanadium	0.186	0.186	**4.20**	**4.20**	0.00813	0.00813

Contaminant of Potential Concern	Killdeer		Great Blue Heron	
	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.
Ammonium	NB	NB	NB	NB
Manganese	0.00928	0.00909	3.25×10^{-4}	2.77×10^{-4}
Molybdenum	0.0145	0.0111	0.00613	0.00472
Nitrate	NB	NB	NB	NB
Selenium	**12.4**	**4.11**	**4.31**	**1.42**
Strontium	NB	NB	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.697	0.355	0.400	0.202
Vanadium	0.230	0.230	0.130	0.130

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
NA = Not assessed because UCL_{95} exceeded the maximum measured concentration.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

6.2.3.5 Area E: Many Devils Wash

Table 6–45 summarizes the water and sediment data for Area E as used to evaluate potential ecological risks. Probably because of the high salinity of the surface water that pools in Many Devils Wash (and possibly also because of the potentially toxic concentrations of some COPCs), aquatic organisms do not inhabit in these pools and wetland vegetation does not grow in or around them. Therefore, exposures to COPCs through food-chain transfers are not expected to occur at this area, and none of the wetland wildlife receptors (muskrat, raccoon, mallard, killdeer, and great blue heron) or plants are expected to be exposed at this site. However, because these waters and sediments could be flushed into the San Juan River by a future flow event in the wash, exposure and risk modeling were performed on the concentration data from this area for all aquatic and wetland receptors to identify the maximum potential risk to these receptors from such an event. Potential exposures and risks to the terrestrial wildlife receptors and livestock (sheep) were also calculated based on the unlikely scenario that the pools are used as a drinking water source by these animals.

The surface water data for Many Devils Wash show exceedences of the water quality benchmarks for ammonium, nitrate, selenium, strontium, sulfate, and uranium (Table 6–46). The maximum HQ based on comparisons of maximum measured COPC concentrations in water to corresponding water quality benchmarks is 3,510 for selenium. Those of uranium and sulfate were also high (greater than 200). When based on the UCL_{95} concentrations, the maximum HQ (again for selenium) decreases to 2,010, and those for uranium and sulfate are still greater than 100. The HQs for molybdenum did not exceed unity.

Table 6–45. Summary of Surface Water and Sediment Data from Many Devils Wash (Area E)

Contaminant of Potential Concern	Surface Water (mg/L)		Sediment (mg/kg)	
	Maximum	UCL_{95}[a]	Maximum	UCL_{95}[a]
Ammonium	2.05	0.839	11.7	NC[b]
Molybdenum	0.135	0.120	2.70[c]	2.40[c]
Nitrate	8,060	4,900	1,300	NC[b]
Selenium	7.01	4.02	0.440	NC[b]
Strontium	16.1	12.4	184	NC[b]
Sulfate	72,800	35,600	19,600	NC[b]
Uranium	0.630	0.302	0.860	NC[b]

[a]Upper confidence limit (one-tailed). One-half the detection limit used for nondetects.
[b]UCL not calculated because only one sediment sample has been collected from this area.
[c]Concentration is estimated from surface water concentration based on the distribution coefficient.
NC = Not calculated

Table 6–46. Hazard Quotients for Aquatic Organisms, Benthic Organisms, and Plants at Many Devils Wash (Area E) Based on Comparisons of Media Concentrations to Water Quality, Sediment Quality, and Plant Toxicity Benchmarks

Contaminant of Potential Concern	Aquatic Organisms		Benthic Organisms	
	HQ based on maximum surface water concentration	HQ based on UCL_{95}[a] of surface water concentrations	HQ based on maximum sediment concentration	HQ based on UCL_{95}[a] of sediment concentrations
Ammonium	**11.4**	**4.66**	0.156	NA[b]
Molybdenum	0.563	0.499	0.675[c]	0.599[c]
Nitrate	**45.1**	**27.7**	0.533	NA[b]
Selenium	**3,510**	**2,010**	0.0880	NA[b]
Strontium	**10.7**	**8.27**	NB	NB
Sulfate	**728**	**356**	NB	NB
Uranium	**242**	**116**	NB	NB

Contaminant of Potential Concern	Wetland Plants	
	HQ based on maximum sediment concentration	HQ based on UCL_{95}[a] of sediment concentrations
Ammonium	NB	NB
Molybdenum	**1.35**[c]	**1.20**[c]
Nitrate	NB	NB
Selenium	0.440	NA[b]
Strontium	NB	NB
Sulfate	NB	NB
Uranium	0.172	NA[b]

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
[b]Not assessed because insufficient samples collected to calculate the UCL_{95}.
[c]Based on sediment concentration estimated from surface water concentration using the distribution coefficient.
NA = Not assessed
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

None of the measured COPC concentrations in sediment resulted in HQs greater than 1 (Table 6–46), although sediment quality benchmark values were not available for strontium, sulfate, uranium, and vanadium. The HQs for wetland plants exceeded unity only for molybdenum, with a maximum HQ of 1.35. The concentration of molybdenum in sediment, however, was estimated from the surface water concentrations based on the distribution coefficient.

Of the terrestrial receptors, HQs greater than unity were limited to nitrate and selenium (Table 6–47). Potential exposures of sheep to selenium in drinking water from this area resulted in the highest HQs among these receptors, with a maximum HQ of 4.72. HQs greater than unity were also found for the deer mouse and red fox. For the wetland wildlife receptors (Table 6–48), HQs greater than unity were again limited to selenium and nitrate. The predatory species (raccoon, killdeer, and great blue heron) showed the highest HQs, which were associated with exposures to selenium. The highest HQs were for the killdeer, which ranged from 729 to 418 (based on the maximum and UCL_{95} concentrations, respectively). The HQs for nitrate exposures in the muskrat and raccoon were all less than 4. (Nitrate HQs for the avian receptors could not be determined because no toxicity benchmark could be found.)

Table 6–47. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors at Many Devils Wash (Area E)

Contaminant of Potential Concern	Sheep		Deer Mouse	
	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations
Ammonium	NB	NB	NB	NB
Molybdenum	0.0810	0.0719	0.0737	0.00654
Nitrate	**2.03**	**1.23**	**1.84**	**1.12**
Selenium	**4.72**	**2.71**	**4.29**	**2.46**
Strontium	0.00824	0.00635	0.00749	0.00577
Sulfate	NB	NB	NB	NB
Uranium	0.0322	0.00154	0.0292	0.00140

Contaminant of Potential Concern	Red Fox		Burrowing Owl	
	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations	HQ based on maximum concentrations	HQ based on UCL_{95}[a] concentrations
Ammonium	NB	NB	NB	NB
Molybdenum	0.0597	0.0530	1.10×10^{-4}	9.75×10^{-5}
Nitrate	**1.49**	0.910	NB	NB
Selenium	**3.48**	**1.99**	0.0464	0.0266
Strontium	0.00608	0.00468	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.0237	0.0114	1.09×10^{-4}	5.23×10^{-5}

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

Table 6–48. Hazard Quotients for Riparian/Wetland Wildlife Receptors at Many Devils Wash (Area E)

Contaminant of Potential Concern	Muskrat		Raccoon		Mallard	
	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.
Ammonium	NB	NB	NB	NB	NB	NB
Molybdenum	0.848[b]	0.753[b]	0.678[b]	0.602[b]	0.0362[b]	0.0321[b]
Nitrate	**1.77**	**1.16**	**3.61**	**2.23**	NB	NB
Selenium	**3.77**	**2.20**	**392**	**225**	**25.0**	**14.4**
Strontium	0.139	0.137	0.0827	0.0742	NB	NB
Sulfate	NB	NB	NB	NB	NB	NB
Uranium	0.0278	0.0148	0.372	0.179	0.00811	0.00397

Contaminant of Potential Concern	Killdeer		Great Blue Heron	
	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.	HQ based on maximum conc.	HQ based on UCL_{95}[a] conc.
Ammonium	NB	NB	NB	NB
Molybdenum	0.0953[b]	0.0845[b]	0.0404[b]	0.0358[b]
Nitrate	NB	NB	NB	NB
Selenium	**729**	**418**	**254**	**146**
Strontium	NB	NB	NB	NB
Sulfate	NB	NB	NB	NB
Uranium	0.182	0.0879	0.104	0.0500

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
[b]Based on sediment concentration estimated from the surface water concentration by using the distribution coefficient.
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

In summary, selenium, sulfate, and uranium are the principal risk drivers in surface water at Area E. No risks to benthic organisms were found, and the potential for risk to plants from COPCs in sediments is probably negligible. Potential risks to terrestrial wildlife and livestock receptors were found for selenium and nitrate at this area from the ingestion of surface water. Potential risks to wetland wildlife receptors may exist from exposures to these same two COPCs. In the case of selenium, these risks are increased in predatory species through accumulation in the food chain. The potential for risk from exposures to molybdenum at this area is negligible.

6.2.3.6 Area F: Upland Terrace

Summary statistics were calculated for each analyte evaluated in the samples of greasewood leaves and stems collected from the (millsite) upland terrace area and the reference terrace area. These statistics are provided in Tables 1 through 28 in Appendix H, "Data Evaluation of Ecological Risk Assessment." Based on the comparison of the maximum measured concentrations in the 1998 and 1999 samples of greasewood from these areas, manganese, strontium, and uranium were identified as being elevated in the millsite area and were therefore identified as COPCs. Table 6–49 presents maximum measured concentrations of these COPCs in the millsite upland terrace samples and the results of the risk analysis based on these maximum concentrations. None of the three COPCs resulted in HQs greater than unity in any of the three terrestrial wildlife receptors (deer mouse, red fox, and burrowing owl). These results indicate that although uptake of some millsite-related constituents by plants may be occurring, the resulting concentrations of these constituents in the aboveground tissues of these plants are not sufficient to pose a risk to wildlife on the terrace.

Table 6–49. Hazard Quotients for Terrestrial Wildlife and Livestock Receptors on the Upland Terrace (Area F)

Contaminant of Potential Concern	Maximum Concentration in Plants (mg/kg dw)[a]	Hazard Quotient for Sheep	Hazard Quotient for the Deer Mouse	Hazard Quotient for the Red Fox	Hazard Quotient for the Burrowing Owl
Manganese	200	0.122	0.301	0.0279	4.81×10^{-6}
Strontium	285	0.0584	0.144	0.0133	NB
Uranium	1.70	0.0347	0.0854	0.00790	8.69×10^{-6}

[a]Milligrams per kilogram dry weight of greasewood leaves and stems.
NB = No benchmark available

6.2.3.7 Potential Risks from Salt Crust Ingestion

Samples of surficial salt crusts were collected from Areas A, C, D, and E and were analyzed for a number of different constituents (see Section 4.4.5). Only nitrate, sulfate, ammonium, and uranium were evaluated for ecological risk because they were present most consistently and at highest concentrations. Because these salt crusts typically form as a result of evaporation of water at the surface of drying sediments and soils, aquatic and benthic organisms and rooted plants are not potentially exposed to these salts. (At Many Devils Wash, the surface waters may be at saturation for some salts, resulting in salts precipitating on the bottom of the surface water; however, the high osmotic potential of these waters precludes the existence of aquatic organisms and wetland plants as potential receptors at this area.) Wildlife, however, may be exposed to these salts through their incidental ingestion with soil and sediment. To evaluate the potential for additional risk to these receptors by the ingestion of salt crust material at these four areas, the risk evaluations for the wetland and terrestrial wildlife and livestock receptors described in Sections 6.2.3.1, 6.2.3.3, 6.2.3.4, and 6.2.3.5 were recalculated using the measured concentrations of nitrate and uranium in salt crusts to represent the concentrations of these analytes in the ingested soil or sediment of these receptors. These calculations conservatively assume that all ingested soil or sediment is represented by salt crust. Sulfate and ammonium were not included in these analyses because of the lack of a toxicity benchmark for these receptors. Maximum salt concentrations were used in these analyses for all four areas, and sufficient data were available for the calculation of UCL_{95}s for Areas C and D.

Table 6–50 summarizes the results of the salt crust analyses. Table 6–51 presents the HQs based on the assumption of 100 percent incidental salt crust ingestion for the sediment portion of their diet by wildlife and livestock receptors at Areas A, C, D, and E. Based on the maximum concentrations of nitrate and uranium in salt crusts, no HQs greater than unity were found for Area A; however, HQs greater than unity were found for nitrate at Areas C, D, and E, and for uranium at Areas C and D. At Area C, the maximum HQ for nitrate was 2.65, and that for uranium was 1.68. At Area D, these maximums were 1.83 and 1.59, respectively. At Area E, the maximum HQ was 4.18 for nitrate (the maximum HQ for wildlife at this area based on sediment concentrations was 3.61). Based on the UCL_{95}s, all HQs at Areas C and D dropped below unity. Therefore, the potential for increased risk by the incidental ingestion of salt crusts in these areas by wildlife and livestock is considered to be low.

Table 6–50. Summary of Salt Crust Analysis Data from Areas A, C, D, and E

Contaminant of Potential Concern	Area A: Distributary Channel and Tributaries		Area C: Shiprock Floodplain		Area D: Bob Lee Wash		Area E: Many Devils Wash	
	Maximum concentration (mg/kg)	UCL_{95}[a] concentration (mg/kg)	Maximum concentration (mg/kg)	UCL_{95}[a] concentration (mg/kg)	Maximum concentration (mg/kg)	UCL_{95}[a] concentration (mg/kg)	Maximum concentration (mg/kg)	UCL_{95}[a] concentration (mg/kg)
Nitrate	8,527	NA[b]	149,096	41,387	81,717	38,214	55,748	NA[b]
Uranium	4.90	NA[b]	53.38	16.3	76.0	59.3	1.79	NA[b]

[a]Upper confidence limit (one-tailed) using one-half the detection limit for nondetects.
NA = Not assessed because insufficient samples collected to calculate the UCL_{95}.

Table 6–51. Hazard Quotients for Terrestrial and Wetland Wildlife and Livestock Receptors Based on the Assumption of 100 Percent Salt Crust Ingestion at Areas A, C, D, and E

Contaminant of Potential Concern	Hazard Quotients for Terrestrial Receptors				Wetland Receptors				
	Sheep	Deer Mouse	Red Fox	Burrowing Owl	Muskrat	Raccoon	Mallard	Killdeer	Great Blue Heron
Area A: Distributary Channel and Tributaries									
Nitrate (maximum)	0.188	0.160	0.123	NB	0.243	0.326	NB	NB	NB
Uranium (maximum)	0.0279	0.0488	0.0106	0.00105	0.163	0.118[b]	0.0177	0.0380	0.0171
Area C: Shiprock Floodplain									
Nitrate (maximum)	**1.74**	**1.55**	0.958	NB	**2.41**	**2.65**	NB	NB	NB
Nitrate (UCL_{95}[a])	0.399	0.379	0.195	NB	0.591	0.558	NB	NB	NB
Uranium (maximum)	0.121	0.115	0.0849	0.0238	**1.68**	**1.01**	0.179	0.292	0.115
Uranium (UCL_{95})	0.0430	0.0481	0.0302	0.00858	0.334	0.262	0.0353	0.0956	0.0390
Area D: Bob Lee Wash									
Nitrate (maximum)	**1.20**	**1.17**	0.679	NB	**1.61**	**1.83**	NB	NB	NB
Nitrate (UCL_{95})	0.532	0.610	0.265	NB	0.766	0.726	NB	NB	NB
Uranium (maximum)	0.248	0.235	0.169	0.0305	0.314	**1.59**	0.0411	0.828	0.403
Uranium (UCL_{95})	0.163	0.163	0.113	0.0268	0.217	0.846	0.0235	0.456	0.204
Area E: Many Devils Wash									
Nitrate (maximum)	**2.41**	**2.12**	**1.67**	NB	**2.47**	**4.18**	NB	NB	NB
Uranium (maximum)	0.0346	0.0310	0.0249	4.88×10^{-4}	0.0303	0.374	0.00821	0.183	0.104

[a]Upper confidence limit (one-tailed).
NB = No benchmark available
Bold indicates hazard quotient greater than 1.

6.2.3.8 Potential Risks from Radionuclides

Potential risks from radiological COPCs were evaluated using the screening-level benchmarks for aquatic biota (specifically large and small fish) derived for Oak Ridge National Laboratory (Bechtel Jacobs 1998b), as based on the methodology for estimating dose rates for aquatic biota developed by Blaylock and others (1993). Radiological analyses in surface water and sediment samples from the Shiprock site have included four uranium-238 daughters (radium-226, thorium-230, lead-210, and polonium-210), as well as radium-228, gross alpha, and gross beta activities. As shown in Table 6–16 through Table 6–20, radium-226, radium-228, thorium-230, and lead-210 have been identified as COPCs at one or more of the areas where surface water is an exposure medium (Areas A, B, C, D, and E) based on comparisons to reference site data. Table 6–52 presents the comparison (as HQs) of the sitewide maximum concentrations of these radionuclides to their screening benchmark values. Although no benchmark was available for radium-228, it is clear from the HQs for the other radiological COPCs that doses to aquatic biota (particularly to fish) from uranium-238 daughters at the Shiprock site are negligible.

Table 6–52. Hazard Quotients for Radiological COPCs

Contaminant of Potential Concern	Surface Water			Sediment		
	Benchmark Value[a] (pCi/L)[b]	Maximum Measured Activity (pCi/L)	Hazard Quotient	Benchmark Value[a] (pCi/kg)[c]	Maximum Measured Activity (pCi/kg)	Hazard Quotient
Lead-210	30,600	2.6	8.50×10^{-5}	9.77×10^{6}	ND	NA
Radium-226	160	1.88	0.0118	28,200	12.9	4.57×10^{-4}
Radium-228	NB	1.86	NB	NB	ND	NA
Thorium-230	413	< 1.2	< 0.00291	1.12×10^{8}	84.0	7.50×10^{-7}

[a]Benchmark is the minimum for large and small fish (from Bechtel Jacobs 1998b)
[b]Picocuries per liter
[c]Picocuries per kilogram
NB = No benchmark available

The concentrations of uranium isotopes have not been directly measured in surface water at the site. Because potential dose is radionuclide-specific and not particle-type-specific, dose to biotic receptors cannot be estimated from gross alpha and gross beta measurements. However, if it is conservatively assumed that all gross alpha activity is from uranium-238, the potential for risk from this isotope can be rejected if the maximum gross alpha activity is less than the benchmark value for the isotope. The maximum gross alpha activity that has been measured in surface water at the Shiprock site is 1,147 pCi/L (at Bob Lee Wash). This is about a factor of four less than the benchmark of 4,550 pCi/L (Bechtel Jacobs 1998b). In the San Juan River (Area B), the maximum measured gross alpha activity in surface water is 26.9 pCi/L, and in the distributary channel (Area A), it is 92.9 pCi/L. Therefore, radiological exposures are not expected to adversely affect these areas as critical habitat for endangered fish.

6.2.3.9 Comparison of Surface Water Data to Livestock Drinking Water Standards

Although risks to livestock were evaluated by estimating exposures in sheep through multiple ingestion pathways (food, water, and incidental soil ingestion), water quality standards exist for some of the COPCs at the Shiprock site that are specific for use as drinking water for livestock and wildlife. Navajo Nation surface water quality standards for livestock and wildlife include a

standard of 0.05 mg/L for selenium and 0.10 mg/L for vanadium. Surface water data from Areas A through E (see Sections 6.2.3.1 through 6.2.3.5) show that recent vanadium concentrations do not exceed this benchmark at any of these areas. Although the maximum selenium concentrations exceed the standard at all five areas, the UCL_{95} concentrations do not exceed the standard at Areas B, C, and D. Therefore, risk to livestock from selenium in drinking water may be limited to the area west of U.S. Highway 666 and Many Devils Wash. The BLRA (DOE 1994) evaluated potential risks to livestock from nitrate and sulfate exposures in drinking water based on the EPA standards of 100 and 1,000 mg/L, respectively. The maximum concentrations of both of these COPCs exceeded the standards at all five areas (A through E), and only at Area B (the San Juan River) were the UCL_{95} concentrations less than the standards. These results indicate that surface water quality may not be acceptable for use as livestock drinking water at Areas A and E because of high nitrate, selenium, and sulfate concentrations, and at Areas C and D because of high nitrate and sulfate concentrations.

6.2.3.10 Potential Risks to Sensitive Species

As described in Section 6.2.1.1 and Table 6–14, several sensitive species have the potential for occurring at or near the Shiprock site. Of particular concern are the endangered Colorado pikeminnow and razorback sucker, for which the San Juan River is currently designated as Critical Habitat, and the southwestern willow flycatcher, which may use the riparian woodlands of Areas A, B, C, and D. In addition, the wetlands of Areas C and D are potential habitats for the northern leopard frog. The burrowing owl, which is considered to be a sensitive species but is not listed as threatened or endangered, was included as a receptor in this risk assessment. Potential risks to other sensitive species are considered to be included within the risk evaluations described in Sections 6.2.3.1 through 6.2.3.6. The potential risks to the Colorado pikeminnow and razorback sucker are considered through the HQs for aquatic organisms in Areas A and B. Potential risk to the northern leopard frog would be included through the HQs for aquatic organisms at Areas C and D. Because exposures in the killdeer are high due to its diet of aquatic invertebrates and high sediment ingestion rate, risk to this receptor is expected to be inclusive of risks to other migratory birds in the wetland habitats, including the southwestern willow flycatcher. As noted in Section 6.2.1.1, the Mesa Verde cactus is the only threatened or endangered species that is known to occur at the Shiprock site; however, no exposure pathway is expected to exist for this species due to its occurrence on the upland terrace, and its lack of roots capable of reaching contaminants in the ground water.

6.2.3.11 Ecological Risk Summary

For the purpose of summarization, the receptors are categorized into six groups: aquatic organisms, benthic organisms, upland plants, wetland plants, terrestrial wildlife and livestock, and wetland wildlife. Further, the potential risk to each group as based on the HQs presented earlier in this section was categorized as follows:

- None: HQs less than or equal to 1 for both the maximum and UCL_{95} concentrations
- Very low: Maximum HQs less than 10 but greater than 1; UCL_{95}-based HQs less than 1
- Low: Both maximum and UCL_{95}-based HQs less than 10, but greater than 1

- Medium-Low: Maximum HQ greater than or equal to 10 but less than 100; UCL_{95}-based HQs less than 10
- Medium: Both maximum and UCL_{95}-based HQs greater than or equal to 10 but less than 100
- High: Maximum HQ greater than or equal to 100 but less than 1,000; UCL_{95}-based HQs greater than 10
- Very high: Maximum HQs greater than or equal to 1,000.

The results of this categorization of potential risk are presented in Table 6–53. In the cases where multiple receptors are included in the receptor group (i.e., the terrestrial and wetland wildlife groups), the risk is based on the highest worst-case risk result among the receptors. Because many conservatisms were incorporated in the calculation of these HQs, including the use of maximum and UCL_{95} values as exposure point concentrations, the use of conservative toxicity benchmarks, such as water quality criteria and NOAELs, and the assumption of 100 percent area and seasonal use, the HQs are expected to overestimate actual risk to most individual receptors, and therefore, risks categorized as medium-low to none are not expected to represent significant potential risks to populations of nonsensitive species. However, for those receptor groups that may include sensitive species, risk categorizations of medium-low to very low are still considered to be of concern. No potential risks were identified for Area F, and therefore, this area is not included in Table 6–53.

Because the surface water in Area A may be used by endangered fish from the San Juan River, all of the original COPCs except molybdenum and vanadium are considered to be of concern in this area. Selenium is also of concern because of potential effects on wetland plants and for its potential to accumulate in the food chain, leading to the potential for adverse effects on wetland predators, which may include the southwestern willow flycatcher.

For Area B, which is designated as Critical Habitat for the Colorado pikeminnow and razorback sucker, manganese, selenium, strontium, sulfate, and uranium are still considered as COPCs for aquatic organisms. As discussed in Section 6.2.3.2, however, all of these except uranium are only elevated above background near the point where surface water in a drainage channel from the Shiprock floodplain discharges into the river (i.e., at sampling location 894). Uranium is of concern in the river along the lower (downstream) half of the floodplain shoreline. Selenium may be of concern to wetland predators (potentially including the southwestern willow flycatcher) at the area of location 894; however, no risk is indicated for larger, piscivorous birds (e.g., the bald eagle) from feeding over large reaches of the river channel. Although low risk from vanadium is indicated by the HQs, these are principally driven by sediment concentrations, which are well below typical background concentrations. Therefore, risk from vanadium is not considered potentially significant.

In the surface water of Area C, manganese, selenium, sulfate, and uranium are of primary concern for the protection of aquatic communities. If sensitive species are present (e.g., the northern leopard frog), ammonium, nitrate, and strontium may also be of concern in surface water. Because of accumulation in the food chain, selenium may be of concern to wetland predators in this area. These may include the southwestern willow flycatcher. As discussed for Area B, the low risk associated with vanadium is not considered to be potentially significant because the sediment concentration (estimated from the surface water concentration) is well below typical background values.

Table 6–53. Summary of Potential Ecological Risks at the Shiprock Site (see text for definition of risk categories)

Contaminant of Potential Concern	Aquatic Organisms	Benthic Organisms	Upland Plants	Wetland Plants	Terrestrial Wildlife and Livestock	Wetland Wildlife
(principal exposure media)	surface water	sediment	soil	sediment	surface water soil food	surface water sediment food
Area A: Distributary Channel and Tributaries						
Ammonium	Medium-Low	None	NA	--	--	--
Manganese	Medium-Low	None	NA	None	None	None
Molybdenum	None	None	NA	None	None	None
Nitrate	Low	None	NA	--	None[a]	None[a]
Selenium	High	Low	NA	Medium	Low	Medium
Strontium	Low	--	NA	--	None	None[a]
Sulfate	Medium	--	NA	--	--	--
Uranium	Medium	--	NA	Very low	None	None
Vanadium	None	--	NA	Very low	None	None
Area B: San Juan River						
Ammonium	None	None	NA	--	--	--
Manganese	Low	None	NA	None	None	None
Molybdenum	None	Low-None[b]	NA	Low-None[b]	None	Low-None[b]
Nitrate	None	None	NA	--	None[a]	None[a]
Selenium	Medium-Low	None	NA	None	None	Very low
Strontium	Low	--	NA	--	None	None[a]
Sulfate	Medium-Low	--	NA	--	--	--
Uranium	Medium-Low	--	NA	None	None	None
Vanadium	None	--	NA	Low	None	Low
Area C: Shiprock Floodplain						
Ammonium	Medium-Low	None	--	--	--	--
Boron	None	--	--	Low	None	
Manganese	High	Very low	Very low	Low	Very low	None
Molybdenum	None	None	--	None	None	None
Nitrate	Medium-Low	None	--	--	None[a]	Very low[a]
Selenium	Medium	None	Low	Low	Low	Medium-Low
Strontium	Medium-Low	--	--	--	None	None[a]
Sulfate	High	--	--	--	--	--
Uranium	High	--	Low	Low	None	None
Vanadium	None	--	--	Very low	None	Low
Area D: Bob Lee Wash						
Ammonium	Medium-Low	None	--	--	--	--
Manganese	None	None	None	Very low	None	None
Molybdenum	None	None	--	None	Low	None
Nitrate	Medium-Low	None	--	--	None[a]	None[a]
Selenium	Medium	None	None	None	None	Medium-Low
Strontium	Low	--	--	--	None	None[a]
Sulfate	High	--	--	--	--	--
Uranium	High	--	Low	None	None	Very low
Vanadium	None	--	--	Low	None	Low
Area E: Many Devils Wash						
Ammonium	Medium-Low	None	NA	--	--	--
Molybdenum	None	None	NA	Low	None	None
Nitrate	Medium	None	NA	--	Low[a]	Low[a]
Selenium	Very high	None	NA	None	Low	High
Strontium	Medium-Low	--	NA	--	None	None[a]
Sulfate	High	--	NA	--	--	--
Uranium	High	--	NA	None	None	None

[a]Avian benchmark not available. Risk based on mammalian receptors only.
[b]Analyte not detected in sediment. "Low" risk based on one-half the detection limit, but actual risk is probably "none."
-- = No hazard quotients available
NA = Not applicable to this area

The risk results from Area D are very similar to those from Area C with the exception that manganese concentrations in the surface water are much lower. Therefore, the principal COPCs for water in this area are selenium, sulfate, and uranium, with ammonium, nitrate, and strontium being of potential concern if sensitive species were present. Again, selenium may be of concern to wetland predators (possibly including the southwestern willow flycatcher) in this area.

The risk results from Area E are similar to those from Area B in terms of the COPCs identified; however, the HQs were higher in some cases (e.g., selenium). The principal COPCs for water in this area are selenium, sulfate, and uranium, with ammonium, nitrate, and strontium being of secondary importance. No aquatic or benthic organisms, or wetland plants are currently present at this area, and risks to wildlife and livestock potentially exposed through drinking water are low. The primary concern at this site is the potential for release of this water into the San Juan River by a surface flow event. These risk results indicate that without significant dilution, these waters and sediments could pose a risk to aquatic receptors in the river and (in the case of selenium) could increase the downstream loading of COPCs in the food chain.

No potential ecological risks were identified for exposures to radiation at the Shiprock site. Additional risk to wildlife receptors associated with the ingestion of salt crusts in Areas A, C, D, and E were found to be inconsequential. Comparisons of surface water concentrations to water quality standards for livestock indicate that nitrate and sulfate concentrations may be of concern at Areas A, C, D, and E, and sulfate may be of concern at Areas A and E with regard to the use of these areas for livestock.

6.3 Summary of Risk Assessments

Acceptable risk levels were, in most cases, defined by use of ground water as drinking water in a residential setting. Therefore these risks are potential risks only since ground water is not currently used for this purpose. Risks for sulfate could not be determined due to lack of an RfD. However, levels of sulfate in the floodplain and terrace systems far exceed levels that have been determined to produce no adverse effects. The five COCs identified in Table 6–54 represent the overwhelming percentage of human health risk for the Shiprock site. Sodium and magnesium on the floodplain were qualitative COPCs only; they were not included as COCs because reliable toxicological data required for their risk evaluation are not available. However, these constituents probably make up only a minor part of the overall risk. Any compliance strategy that results in a decrease in concentration of the retained COCs will also cause a decrease in the other constituents that represent a small fraction of the total risk.

Table 6–54 and Table 6–55 show the human health COPCs from the 1994 BLRA, rationales for retaining or deleting them, and the final list of COCs and COPCs for both the floodplain and the terrace respectively. Contaminants on the terrace are still considered to be COPCs because it is not anticipated that numerical remediation goals will be established for these constituents (see Section 7.0). All constituents listed as COCs either exceed UMTRA standards or acceptable risk levels, with the exception of sulfate.

Table 6–54. Floodplain Human Health Risk COPC Updates

COPCs From BLRA	UMTRA MCL (mg/L)	Comments and Rationale for Retaining or Deleting a COPC for Human Health Risk	COC Yes (Y) or No (N)
Antimony		Mostly below detection limits; similar to background concentrations; not retained	N
Arsenic	0.05	Mostly below detection limits; similar to background concentrations; not retained	N
Cadmium	0.01	Mostly below detection limits, similar to background concentrations; not retained	N
Magnesium		Exceeds background concentrations, qualitative COPC, not retained	N
Manganese		HQ[a]>1; Negative health effects probable	Y
Nitrate	44	HQ>1; UCL_{95} for data exceeds UMTRA MCL	Y
Selenium	0.01	UCL_{95} for data exceeds UMTRA MCL	Y
Sodium		Exceeds background concentrations; qualitative COPC; not retained	N
Strontium		Mostly below background concentrations; below risk-based concentration; not retained	N
Sulfate		Toxicity data are currently under evaluation by EPA but concentrations are high enough to be of probable concern	Y
Uranium	0.044	UCL_{95} greater than UMTRA MCL. Exceeds EPA's carcinogenic risk range	Y

[a]HQ = hazard quotient.

Table 6–55. Terrace Human Health Risk COPC Updates

COPCs From BLRA or Added From This Study	UMTRA MCL (mg/L)	Comments and Rationale for Retaining or Deleting a COPC for Human Health Risk	COPC Yes (Y) or No (N)
Ammonia as Ammonium (added)		Ammonium exceeds background concentrations. Screened out as ingestion of ammonium; retained because inhalation of ammonia could be of potential health risk under residential scenario.	Y
Manganese (added)		HQ[a]>1; Negative health effects probable	Y
Nitrate	44	HQ>1; UCL_{95} for data exceeds UMTRA MCL	Y
Selenium (added)	0.01	UCL_{95} for data exceeds UMTRA MCL	Y
Sulfate		Toxicity data are currently under evaluation by EPA but concentrations are high enough to be of probable concern	Y
Uranium	0.044	UCL_{95} greater than UMTRA MCL. Exceeds EPA's carcinogenic risk range	Y

[a]HQ = hazard quotient.

To evaluate ecological risks, surface water, sediment, and soil concentrations from six areas potentially affected by the millsite ground water were compared with data from reference areas and elevated levels of some analytes were found. These analytes were designated as ecological COPCs. A screening-level risk assessment based on calculated HQs was used to evaluate potential risks to ecological receptors at each of the six areas from exposures to these COPCs. Receptors included aquatic and benthic organisms, wetland and upland plants, livestock, and wetland and terrestrial wildlife. HQs were calculated based on both maximum and UCL_{95} concentrations. In addition, wildlife HQs were calculated for those areas where salt crusts have been observed on the surfaces of soils and sediments based on the exposure scenario that all incidentally ingested soil or sediment is in the form of this salt (data for the salt crust are limited to nitrate, sulfate, and uranium). Surface water data were also compared to radiological benchmarks and livestock drinking water standards.

Risks were considered low if the HQs were less than 10 and none if the HQs were less than unity. COPCs showing no risk are dropped from further consideration, and those with low risks are also dropped provided that the receptors showing the low risk do not include or represent potential risks to endangered or threatened species. Because conservatisms have been incorporated into the exposure models and toxicity benchmarks, HQs are expected to overestimate the actual risks posed by these COPCs. Therefore, HQs less than 10 are expected to be protective of populations and communities, but may not be protective of individuals in the cases where threatened or endangered species may be exposed. Table 6–56 summarizes the ecological COPCs that remain at each of the six evaluated areas. These constituents are considered to be of potential concern because their concentrations in environmental media indicate a potential exists for adverse toxicological effects to ecological receptors. Although there is no visible evidence to date that indicates toxic effects are occurring, the lack of evidence does not exclude the possibility that effects are occurring. Of the six areas evaluated, the Upland Terrace (Area F) can be eliminated from further consideration with regard to ecological risk. Surface water is the principal medium of concern at the other five areas, with selenium, uranium, and sulfate being the primary risk drivers (manganese is also an important risk driver on the Shiprock floodplain). Ammonium and strontium are minor contributors to ecological risk at these areas. Endangered fish may be exposed to COPCs at Areas A and B. Selenium is of concern at Areas A, B, C, and D because of its potential for magnification in the food chain, resulting in potentially toxic exposures to shorebirds and insectivorous birds, which may include the endangered southwestern willow flycatcher.

Table 6–56. Summary of Ecological Contaminants of Potential Concern at the Shiprock Millsite Based on the Ecological Risk Screening Results

Area A: Distributary Channel and Tributaries	Area B: San Juan River	Area C: Shiprock Floodplain	Area D: Bob Lee Wash	Area E: Many Devils Wash	Area F: Upland Terrace
Ammonium Manganese Nitrate Selenium Strontium Sulfate Uranium	Manganese Selenium Strontium Sulfate Uranium	Ammonium Manganese Nitrate Selenium Strontium Sulfate Uranium	Ammonium Nitrate Selenium Sulfate Uranium	Ammonium Nitrate Selenium Strontium Sulfate Uranium	(none)

A request was received from stakeholders who reviewed the first revision of this SOWP to consider ecological implications of flowing well 648 remaining on and being off. This ecological assessment only considers well 648 as continuing to flow. Two reasons are given for this decision. First, data collected for this exercise represent conditions that exist today, and suppositions made from these data are directly applicable to reality. If flow from the well were terminated, a large variety of ecological consequences could result, and the exercise of choosing the correct or most applicable scenario would be extremely difficult. Second, DOE will request that well 648 be left flowing in order to enhance the proposed compliance strategy for the floodplain. A large change in the contribution of ground water flowing into the floodplain would have significant consequences to the timing and cost of the proposed remediation.

7.0 Ground Water Compliance Strategy

7.1 Compliance Strategy Selection Process

The framework defined in the PEIS (DOE 1996b) governs selection of the strategy to achieve compliance with EPA ground water standards. Stakeholder review of the final PEIS is documented and supported by the Record of Decision (CFR v.62, No.18, 1997). Figure 7–1, Figure 7–2, and Figure 7–3 present summaries of the framework used to determine the appropriate ground water compliance strategies for the Shiprock site. The framework takes into consideration human health and environmental risk, stakeholder input, and cost. A step-by-step approach in the PEIS results in the selection of one of these three general compliance strategies:

- ***No remediation***—Compliance with the EPA ground water protection standards would be met without altering the ground water or cleaning it up in any way. This strategy could be applied for those constituents at or below MCLs or background levels or for those constituents above MCLs or background levels that qualify for supplemental standards or ACLs, as defined in Section 2.2, "EPA Ground Water Protection Standards."

- ***Natural flushing***—This strategy would allow natural ground water movement and geochemical processes to decrease contaminant concentrations to regulatory limits within 100 years. The natural flushing strategy can be applied where ground water compliance could be achieved within 100 years, where effective monitoring and institutional controls can be maintained, and where the ground water is not currently and is not projected to be a source for a public water system.

- ***Active ground water remediation***—This strategy would require engineered ground water remediation methods such as gradient manipulation, ground water extraction and treatment, land application, phytoremediation, and in situ ground water treatment to achieve compliance with EPA standards.

7.2 Shiprock PEIS Compliance Selection Framework Analysis

DOE is required by the PEIS to follow the ground water compliance selection framework presented in Figure 7–1, Figure 7–2, and Figure 7–3 (explained in Table 7–1, Table 7–2, and Table 7–3, respectively) in selecting the appropriate compliance strategies for the surficial aquifer and ground water systems at the Shiprock site. Because the Shiprock site is divided physiographically and hydrologically into two regions, the compliance strategies for each region, the floodplain aquifer and the terrace ground water system, are considered separately. In addition, the terrace system is subdivided into two areas, Terrace East and Terrace West. The floodplain aquifer consists of San Juan River alluvium and underlying weathered Mancos Shale. The terrace ground water system consists of river alluvium, scattered loess deposits, and weathered Mancos Shale. Unweathered Mancos Shale is not considered an aquifer in either region because of limited yield.

The compliance strategy proposed for the floodplain is described in Section 7.2.1 and the compliance strategies for the two areas in the terrace are described in Section 7.2.2. DOE believes that implementation of the proposed strategies for the floodplain and terrace regions will address the immediate need of cleaning up contaminated ground water and concurrently enable

DOE to monitor the terrace region and isolate the source or sources of residual moisture drainage from the disposal cell. This may consist of residual (or transient) moisture seeping at a decreasing rate from the disposal cell or may result from additional moisture that is recharging the disposal cell, or some combination of both.

7.2.1 Floodplain Strategy

The proposed compliance strategy for the floodplain surficial aquifer is active remediation in combination with natural flushing (Figure 7–1 and Table 7–1). This strategy consists of pumping water from the most contaminated part of the plume, piping the water up to the terrace to a lined evaporation pond in the radon cover borrow pit, and spray-evaporating the water. Without a source of residual moisture from the disposal cell, it is predicted that the floodplain will flush in 100 years. However, in the event that a residual moisture source from the disposal cell is present, based on plume maps, consistent water levels around the disposal cell, and estimated water flow from nested wells between the disposal cell and the floodplain, DOE will address any necessary changes to the strategy.

Concurrent with pumping contaminated ground water from the floodplain, monitoring of the floodplain and terrace will continue to determine the extent and nature of drainage from the disposal cell. As discussed in Section 7.6, DOE will monitor water levels and ground water chemistry for 5 years after pumping commences. At the end of this period, DOE will reevaluate the implemented compliance action. During the 5-year period, interim actions will have isolated contaminated ground water at seeps along the edge of the floodplain; the public and the environment will be protected from exposure.

7.2.2 Terrace Strategy

Prior to milling activities, the dry terrace area supported only sparse plant life as shown in early aerial photographs (Figure 3–1). No terrace ground water has been found in locations away from the former millsite or from irrigated areas, even after repeated drilling efforts to find background ground water in 1998 and 1999. The ground water in the eastern half of the terrace (Terrace East) is believed to be derived primarily from activities associated with uranium milling; the western half of the terrace (Terrace West) receives recharge primarily from irrigation. Because of the different sources of recharge and sources of contamination for these two areas, separate compliance strategies have been developed for each.

7.2.2.1 Compliance Strategy for Terrace East

The proposed compliance strategy for Terrace East is to pump the remaining relict milling-related water out of the surficial water system (the alluvium and weathered Mancos Shale) and allow this portion of the ground water system to revert to its original nature, thereby drying the seeps and curtailing surface expression of the ground water. The objective of this action is to eliminate the currently complete exposure pathways that exist at the washes and seeps. Cleanup standards such as MCLs are irrelevant based on this objective. However, a list of COPCs was developed to identify constituents that appeared in elevated concentrations around the tailings and raffinate pond areas and are attributable to millsite-related activities. The standard PEIS flow chart does not conveniently accommodate this strategy; the footnote in Figure 7–2 and explanation in Table 7–2 explain this modified approach.

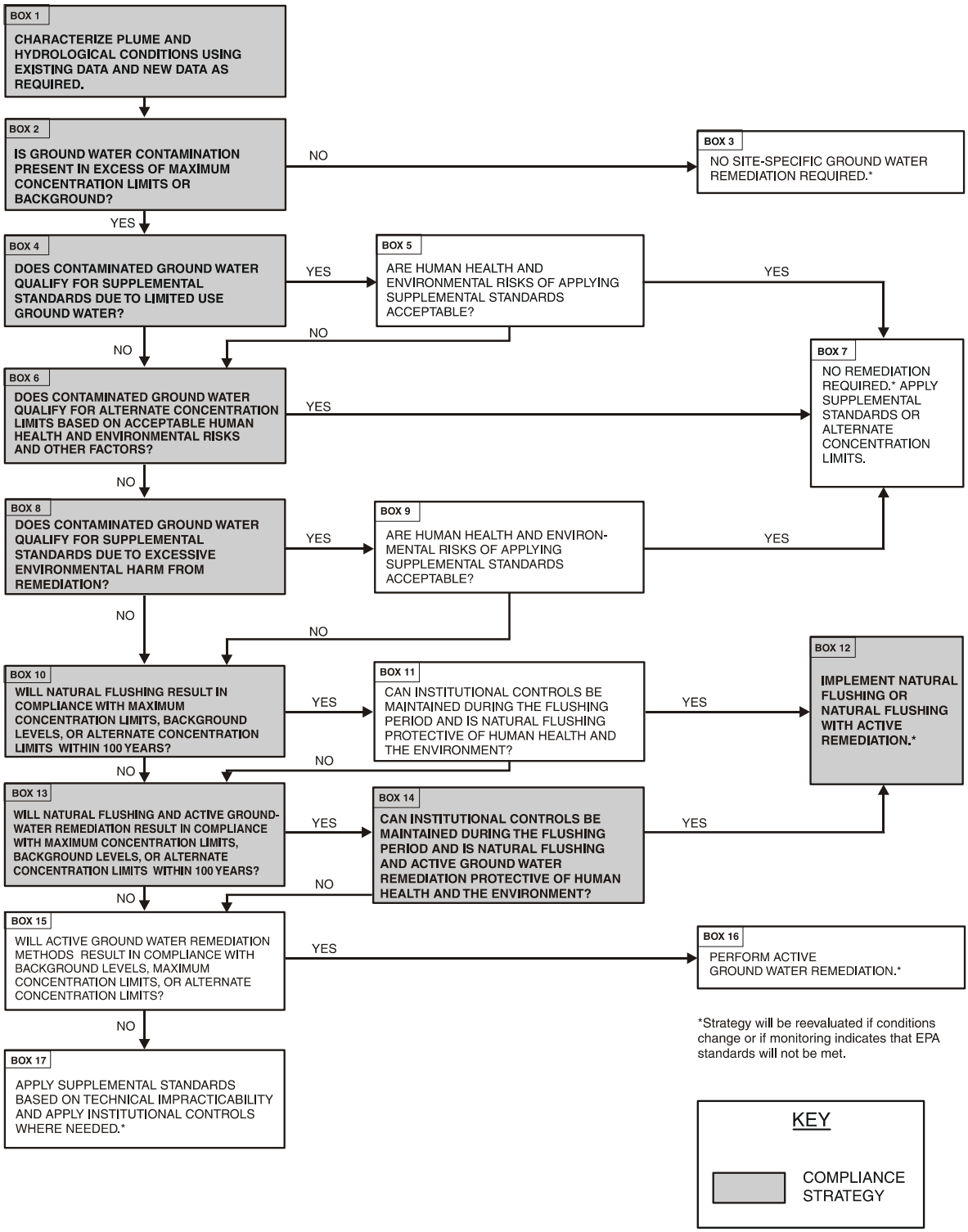

Figure 7–1. Ground Water Compliance Selection Framework for the Floodplain

Table 7–1. Explanation of the Compliance Strategy Selection Process for the Floodplain Alluvial Aquifer

Box (Figure 7–1)	Action or Question	Result or Decision
1	Characterize plume and hydrological conditions.	See site characterization results in Section 4, site conceptual model in Section 5, and risk assessments in Section 6. Move to Box 2.
2	Is ground water contamination present in excess of UMTRA MCLs or background?	Yes; nitrate, uranium, and selenium concentrations exceed the UMTRA MCLs. Sulfate and manganese concentrations are elevated compared to background. Move to Box 4.
4	Does contaminated ground water qualify for supplemental standards due to limited use ground water?	No, alluvial ground water does not currently meet any criteria for limited use. Move to Box 6.
6	Does contaminated ground water qualify for ACLs based on acceptable human health and environmental risks and other factors?	No, ACLs are not proposed at this time. Move to Box 8.
8	Does contaminated ground water qualify for supplemental standards due to excessive environmental harm from remediation?	No, it is unlikely that remedial action would cause excessive harm to the environment. Move to Box 10.
10	Will natural flushing result in compliance with UMTRA MCLs, background, or ACLs within 100 years?	No, ground water modeling shows that natural flushing alone will not reduce nitrate, uranium, and selenium to background or below MCLs within the 100-year time frame unless drainage of residual moisture is removed. Move to Box 13.
13	Will natural flushing and active ground water remediation result in compliance with MCLs, background levels, or ACLs within 100 years?	Yes, active remediation is required to remove contaminated ground water. Compliance with MCLs, background levels, or ACLs will be met within 100 years except for a small area at the base of the escarpment where drainage of residual moisture from the disposal cell is assumed. The exact nature and source of this contamination will be evaluated during the first 5 years of remedial action. Results of this evaluation will be used to consider future action(s) for the floodplain. Natural flushing will not be effective unless drainage of residual moisture is contained or removed. Move to Box 14.
14	Can institutional controls be maintained during the flushing period and is active ground water remediation protective of human health and the environment?	Yes, institutional controls will be maintained during the period of active ground water remediation. The strategy to address the drainage of residual moisture from the disposal cell will be formulated 5 years after remedial action starts and sufficient monitoring data are collected. Human health and the environment will be protected during this time. **Move to Box 12 – implement active remediation and natural flushing.**

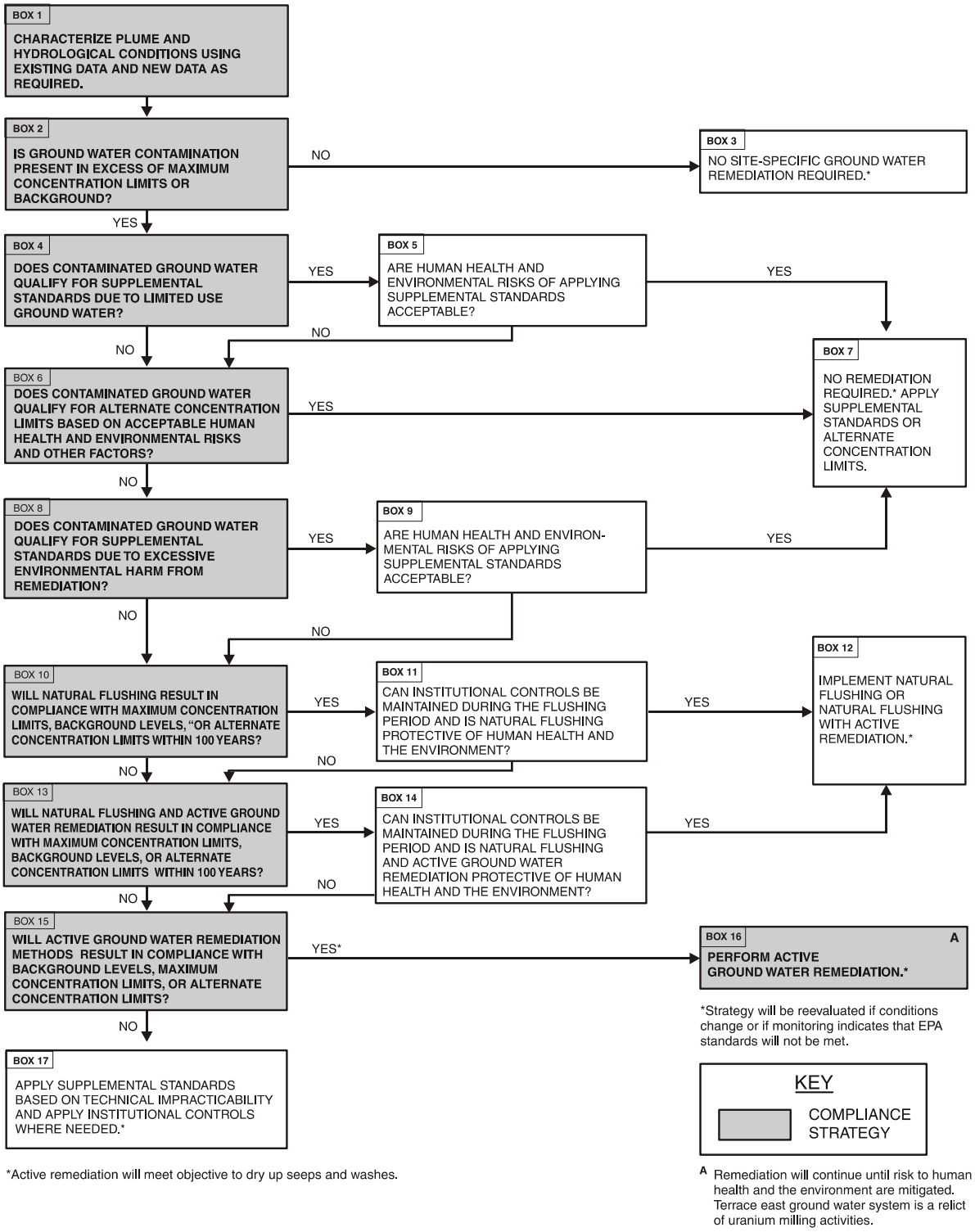

Figure 7–2. Ground Water Compliance Selection Framework for Terrace East

Table 7–2. Explanation of the Compliance Strategy Selection Process for Terrace East Ground Water

Box (Figure 7–2)	Action or Question	Result or Decision
1	Characterize plume and hydrological conditions.	See site characterization results in Section 4, site conceptual model in Section 5, and risk assessments in Section 6. Move to Box 2.
2	Is ground water contamination present in excess of UMTRA MCLs or background?	Yes, nitrate, uranium, and selenium concentrations exceed the UMTRA MCLs. Sulfate and ammonium concentrations are elevated compared to background. Move to Box 4.
4	Does contaminated ground water qualify for supplemental standards due to limited use ground water?	No, Terrace East ground water does not currently meet any criteria for limited use. Water is millsite related. Move to Box 6.
6	Does contaminated ground water qualify for ACLs based on acceptable human health and environmental risks and other factors?	No, ACL would not be protective of human health and the environment. Move to Box 8.
8	Does contaminated ground water qualify for supplemental standards due to excessive environmental harm from remediation?	No, remediation would not cause excessive environmental harm. Move to Box 10.
10	Will natural flushing result in compliance with UMTRA MCLs, background levels, or ACLs within 100 years?	No, natural flushing will not result in compliance with MCLs within 100 years. Move to Box 13.
13	Will natural flushing and active ground water remediation result in compliance with MCLs, background levels, or ACLs within 100 years?	No, leaching of Mancos Shale makes it unlikely that MCLs will ever be attained. Move to Box 15.
15	Will active ground water remediation result in compliance with MCLs, background levels, or ACLs?	Active ground water remediation will not meet standards but will achieve the objective of drying up the seeps and washes. Therefore, remediation standards are not applicable. Move to Box 16.
16	Perform active ground water remediation	Continued drainage of residual moisture from the disposal cell will recontaminate the Terrace East ground water system if it continues. Therefore, in addition to active remedial action, DOE will gather data for 5 years after active remedial action begins and will use these data to determine the nature and extent of contamination from the disposal cell. A decision for future action will be made at that time that will allow remedial action to be completed for the Terrace East ground water system.

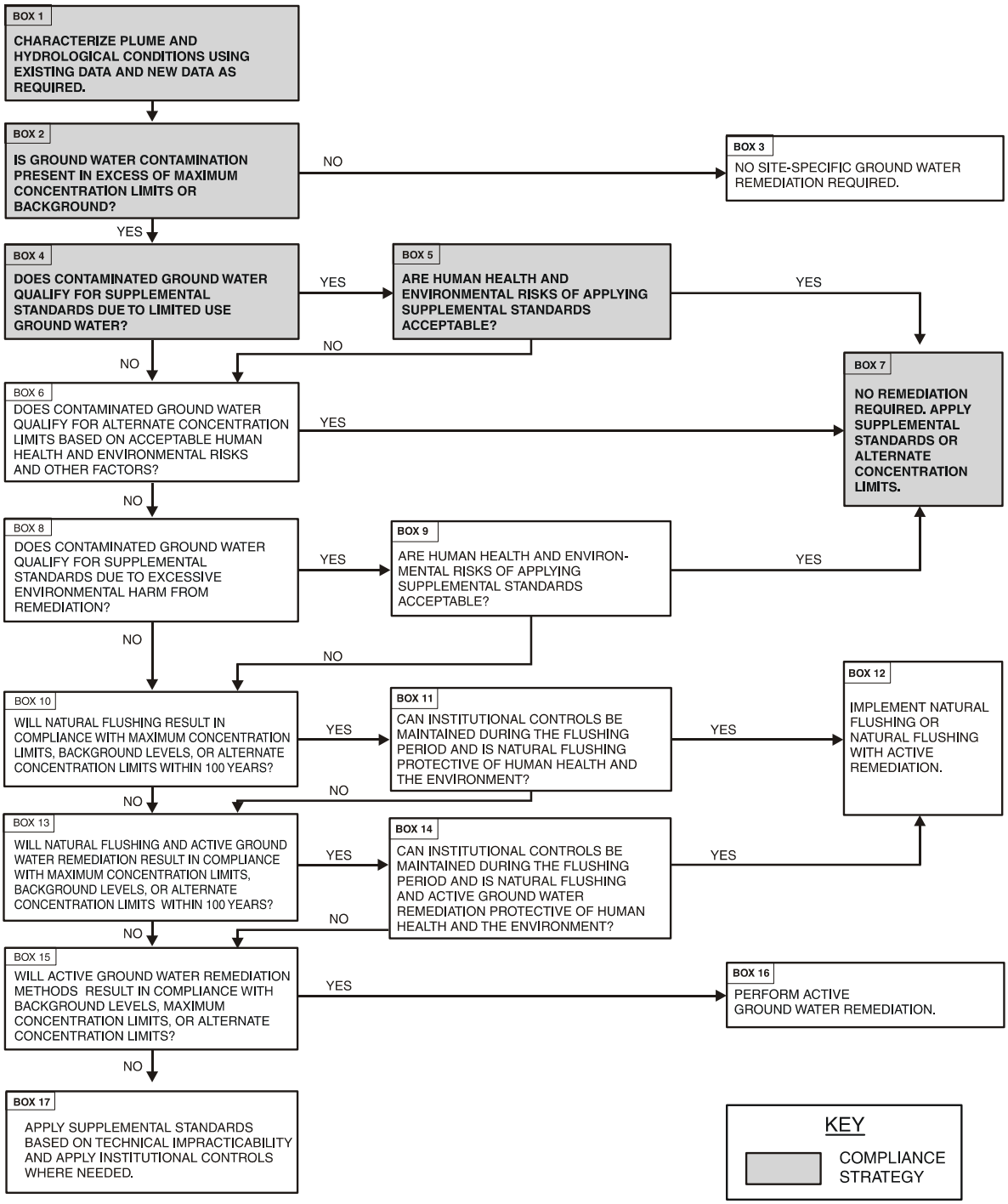

Figure 7–3. Ground Water Compliance Selection Framework for Terrace West

Table 7–3. Explanation of the Compliance Strategy Selection Process for Terrace West Ground Water

Box (Figure 7–3)	Action or Question	Result or Decision
1	Characterize plume and hydrological conditions.	See site characterization results in Section 4, site conceptual model in Section 5, and risk assessments in Section 6. Move to Box 2.
2	Is ground water contamination present in excess of UMTRA MCLs or background?	Yes, uranium, selenium, and sulfate are naturally elevated due to leaching of Mancos Shale by irrigation water; nitrate is also elevated. Move to Box 4.
4	Does contaminated ground water qualify for supplemental standards due to limited use ground water?	Yes, uranium, selenium, and sulfate are naturally elevated. Move to Box 5.
5	Are human health and environmental risks of applying supplemental standards acceptable?	Yes, other sources of drinking water are readily available. Where yield is sufficient for agriculture and livestock uses, water quality permits these uses. Move to Box 7.
7	No remediation required. Apply supplemental standards.	

When pumping of the Terrace East commences, the Terrace East ground water system will begin to be cut off from the Terrace West system. Figure 7–4 shows the approximate boundary between the two systems after approximately 7.5 years of pumping (based on the simulation in Figure 4–73).

Again, DOE believes that the terrace surficial ground water system may become saturated again after pumps are turned off because drainage of residual moisture from the disposal cell. As mentioned in the previous section, evaluation of this drainage will require monitoring for at least 5 years after remedial action starts. Monitoring results will be shared with stakeholders and regulators, and additional recommended remedial action strategies will be evaluated if necessary. In the meantime, contaminated ground water will be removed from the terrace, and risks to humans and the environment have been minimized by interim actions.

7.2.2.2 Compliance Strategy for Terrace West

It is proposed that supplemental standards be applied to Terrace West. Criteria for meeting this standard are (1) widespread ambient contamination not related to milling activities that cannot be cleaned up using treatment methods reasonably employed in public water supply systems, (2) concentrations of TDS that are in excess of 10,000 mg/L, or (3) demonstration that the surficial aquifer will not consistently produce 0.1 gpm (150 gallons per day).

Irrigation water will continue to provide a source of ground water recharge to Terrace West after it is cut off from the Terrace East system during active remediation. This may cause some flushing of contaminants from the system. However, as discussed in Section 4.4.8, it is highly probable that some constituents in the system—notably uranium, selenium, and sulfate—are derived from leaching of Mancos Shale and standards may never be obtained for this region. Because the source of these three constituents can be attributed to natural causes and not necessarily to milling processes, criterion 1 above, widespread ambient contamination not related to milling activities that cannot be cleaned up using treatment methods reasonable employed in public water supply systems, is adopted. Section 4.4.7 provides evidence for widespread ambient contamination unrelated to millsite activities. A cost analysis study done for ground water in the

Grand Junction, Colorado, area showed that treatment of that water in a similar geological and geographical setting is economically infeasible compared with the use of alternative water sources (DOE 1999f). Because other drinking water sources are readily available in the Shiprock area, it is unlikely that treatment of Terrace West water for drinking water purposes would be economical. However, in areas of Terrace West where water yield is sufficient, water quality is suitable for agriculture and livestock watering. Therefore, the application of supplemental standards to Terrace West ground water is protective of human health and the environment. Figure 7–3 and Table 7–3 provide an explanation of how the Terrace West compliance strategy was selected.

7.3 Risks Assessments

7.3.1 Human Health

Section 6.1, "Human Health Risks," presents an assessment of human health risks for the floodplain and terrace. Table 6–54 and Table 6–55 list the human health COPCs from the BLRA (DOE 1994), appropriate MCLs, and rationales for retaining or deleting the COPCs based on 1998, 1999, and 2000 data. The constituents in the right column of each table labeled "Y" are considered final COCs or COPCs for the Shiprock site. Unacceptable risks to humans would result only from the use of terrace or floodplain ground water as a primary source of drinking water. Use of any ground water or surface water for agricultural purposes would not present unacceptable risks to humans. Consumption of meat or milk from livestock that consumed water exclusively from seeps could pose a slightly unacceptable risk to humans, although these waters would not be acceptable for watering livestock because of risks to the animals themselves. Incidental exposure to floodplain or terrace surface water by children playing in those areas would not result in unacceptable risks. Interim actions have eliminated the only complete exposure pathway. The proposed compliance strategy will further reduce the future potential risks to humans.

7.3.2 Ecological Risk

Section 6.2, "Ecological Risk Assessment," presents the assessment of ecological risks for the floodplain and terrace. Ecological risks at the site were considered negligible in the SOWP, Revision 1, except where livestock or other animals might drink at seeps. Interim actions taken during summer 2000 in Bob Lee Wash, Many Devils Wash, and two seeps on the floodplain prevented access to these areas and removed the only complete exposure pathways. In this assessment, new data were used to revise the results of the 1994 BLRA. Data collected from 1998 through 2000 indicate that the areal extent of contamination associated with the millsite ground water exceeds the area considered in the BLRA; therefore, the ecological assessment area was also enlarged. This ecological assessment closely follows EPA guidance.

During 1999 and 2000, DOE worked with the U.S. Fish and Wildlife Service and Navajo Fish and Wildlife Department to gather additional data from a number of new locations. These data were used to perform a screening-level risk assessment based on HQs, further quantifying risks to the environment and especially to threatened and endangered species of birds and fish (see Section 6.2, "Ecological Risk Assessment"). Receptors included benthic and aquatic organisms, wetland and upland plants, livestock, and wetland and terrestrial wildlife. The site was divided

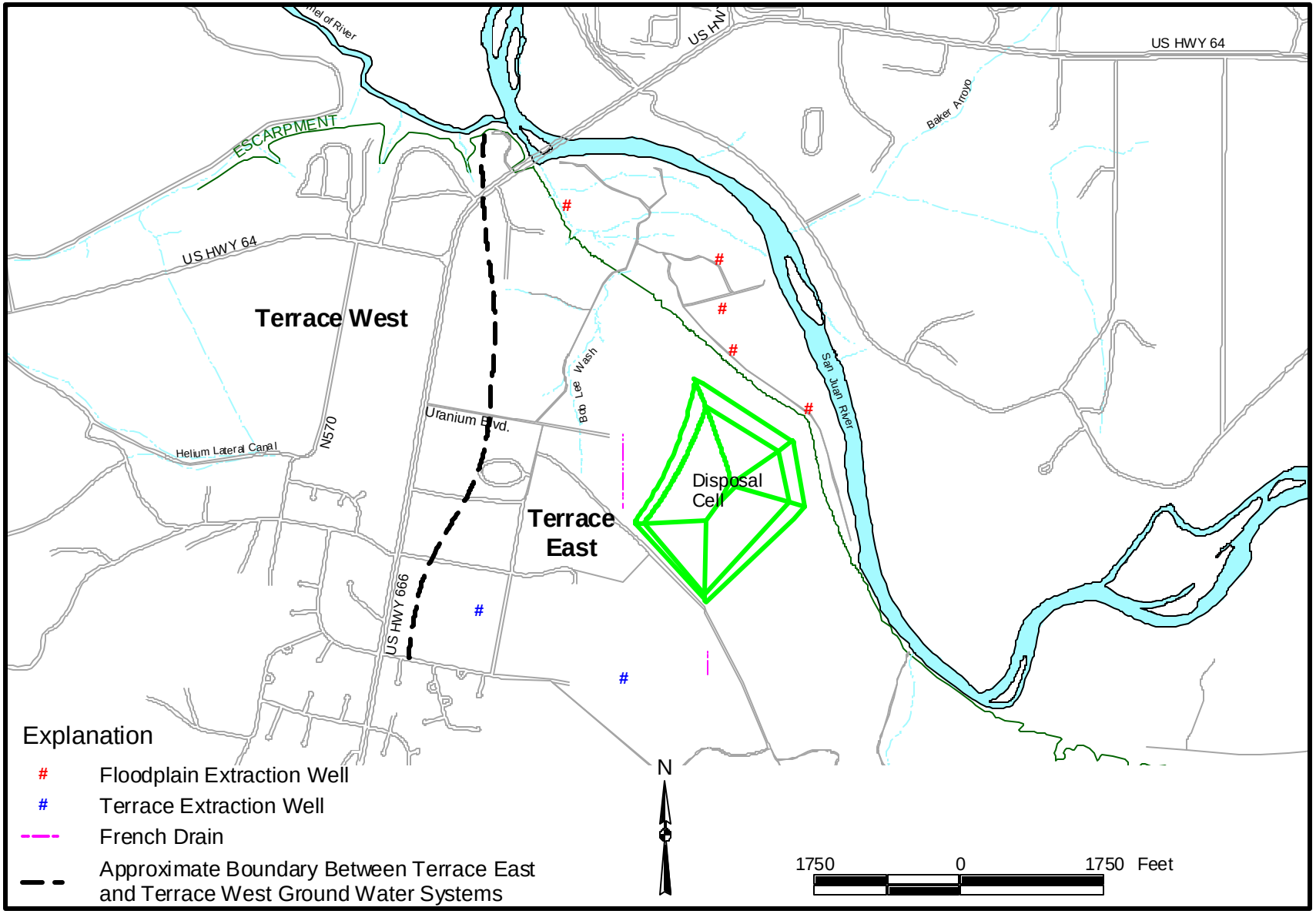

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Figure 7–4. Approximate Boundary of the Terrace East and Terrace West Ground Water Systems, and Proposed Extraction Well and French Drain Locations

into six areas that represent discrete environments or geographic regions (see Plate 5, areas A through F). The six areas are: (A) the distributary channel and its tributaries, (B) the San Juan River, (C) the floodplain, (D) Bob Lee Wash, (E) Many Devils Wash, and (F) the upland (millsite) terrace. Each area was considered individually in the identification of potential receptors, and ecological benchmarks were identified and compared to sample analyses. Because there is no visible evidence that animals or plants are adversely affected by the contaminants, all contaminants are considered COPCs. However, it is understood that unobserved toxic effects may be occurring. For each area of the site, COPCs were identified through comparison of the maximum measured constituent concentration from that area to the maximum concentration from a corresponding reference location (see Tables 6–16 to 6-21).

Results of the ERA are considered conservative. Summaries are provided at the end of Section 6.2 (Table 6–54) and in Section 6.3. No potential risks were identified for exposures to radionuclides in any of the areas. Any additional risks to wildlife associated with ingestion of salt crusts were found to be inconsequential. Area F, the upland or terrace area, was completely eliminated from further consideration for ecological risks. Surface water is the principal medium of concern in the other five areas; selenium, uranium, and sulfate are the primary risk drivers. Manganese is also an important risk driver on the floodplain, and strontium and ammonium are minor contributors to risk. Endangered fish may be exposed to ammonium, manganese, nitrate, selenium, strontium, sulfate, and uranium in the San Juan River and the distributary channel. Selenium is of concern in the floodplain and river system because of its potential for magnification in the food chain and could affect shorebirds and other insectivorous birds including the endangered southwestern willow flycatcher. Comparisons of analytical data to quality standards for livestock indicate that nitrate and sulfate concentrations may be of concern in Areas C, D, E, and to some extent, Area A. Areas D and E have undergone interim actions to prevent livestock or other animals from accessing surface water. Grazing is not being allowed in Area C (the floodplain). Area A will continue to be monitored, but concentrations are not necessarily associated with millsite activities.

Although several of the COPCs showed HQs greater than 1, most of these were less than 10. Because the uncertainties associated with these HQs were generally biased toward greater conservatism, these HQs do not necessarily indicate conditions of high risk or of potentially acute toxicity to ecological receptors currently using the areas. Based on results of the ERA, interim actions beyond those currently in place do not appear to be warranted. The current compliance strategy will remove the most contaminated ground water from these areas and that will progressively diminish the chance of unacceptable exposure to humans, animals, and plants.

7.4 Explanation of Floodplain Strategy-Active Remediation and Natural Flushing

7.4.1 Active Remediation

DOE plans to drill five extraction wells in the most contaminated part of the millsite floodplain, construct a network of piping from the wells up to the terrace to a lined evaporation pond in the radon cover borrow pit area, and spray-evaporate the contaminated water. The approximate locations of wells used in the model to optimize this pumping scheme are shown in Figure 7–4. Water will be withdrawn from the floodplain alluvial aquifer at a rate of about 80 gpm. Conceptual details of these active remediation options are described in Section 8.0 “Development and Evaluation of Active Remediation Alternatives.”

The COCs for the millsite floodplain are nitrate, uranium, selenium, manganese, and sulfate based on human health risks. An arcuate area containing high concentrations of these constituents extends northward from the base of the escarpment from a line connecting wells 735, 610, 614, and 615 toward well 1008 near the San Juan River (Figure 7–5). The shape of the contaminant plume results from a normal flow path toward the river being interrupted by the mounding effect of water from Bob Lee Wash (and artesian well 648) entering the floodplain and flowing northward and northwestward through the subsurface to the San Juan River. Bob Lee Wash water is thought to divert the normal northwest flow of ground water in the floodplain more directly north toward the San Juan River in the area of well 1008. High contaminant concentrations along this path may be the result of either residual milling-contaminated ground water, the flow of which was retarded because of flow from well 648, or from continued flow of contaminated ground water from the base of the escarpment below the disposal cell, or both.

Additional characterization was conducted in December 1999 to determine if a continued source of contamination existed in the millsite floodplain. During surface remedial action in 1986, verification of tailings removal was based on monitoring for radium-226. The floodplain was verified as clean by this criterion. Although not likely, other constituents may have been left in floodplain sediments. To test this, additional soil samples were collected from 24 locations at the surface and subsurface and a background sample was also collected. Figure 4–51 shows these locations. Samples were leached with 5-percent nitric acid and the leachate was analyzed. Although results showed floodplain concentrations of aliquots from the acid leach to be slightly above background results, concentrations were not considered elevated enough to account for the values found in the ground water plume. Contaminated ground water flowing from the terrace probably left a geochemical signature in the soils. No uranium mill tailings were found. It was concluded that a continued source does not exist in the millsite floodplain.

Modeling predicts that pumping one pore volume from the floodplain plume using the five-well extraction system will require 2.8 years at an extraction rate of 40 gpm. Approximately 10 pore volumes are required to clean up contaminated water to acceptable limits. Doubling this pumping rate to 80 gpm will produce the same result in half the time, or about 1.4 years per pore volume. Therefore, the model predicts that pumping will be required for about 14 years to reduce contaminant levels to acceptable limits. The model also predicts that a vestige of the plume will still exist at the base of the escarpment after this period of time because contaminated water continues to move from the disposal cell down into the millsite floodplain, but the remainder of the floodplain will be clean. No discharges to the San Juan River above proposed standards will occur.

Rationales for cleanup standards of COCs depend partly on background values and partly on the effects that weathered Mancos Shale bedrock has on ground water. Compliance standards for uranium and nitrate are their UMTRA standards of 0.044 mg/L and 44 mg/L, respectively. If monitoring indicates that leaching of Mancos in the irrigated area of Terrace West is contributing uranium contamination to the floodplain, the uranium cleanup standard may be reevaluated and an ACL proposed. For manganese, the cleanup objective is the maximum background concentration, which is currently 2.74 mg/L. This value may change if higher background values are found in future sampling.

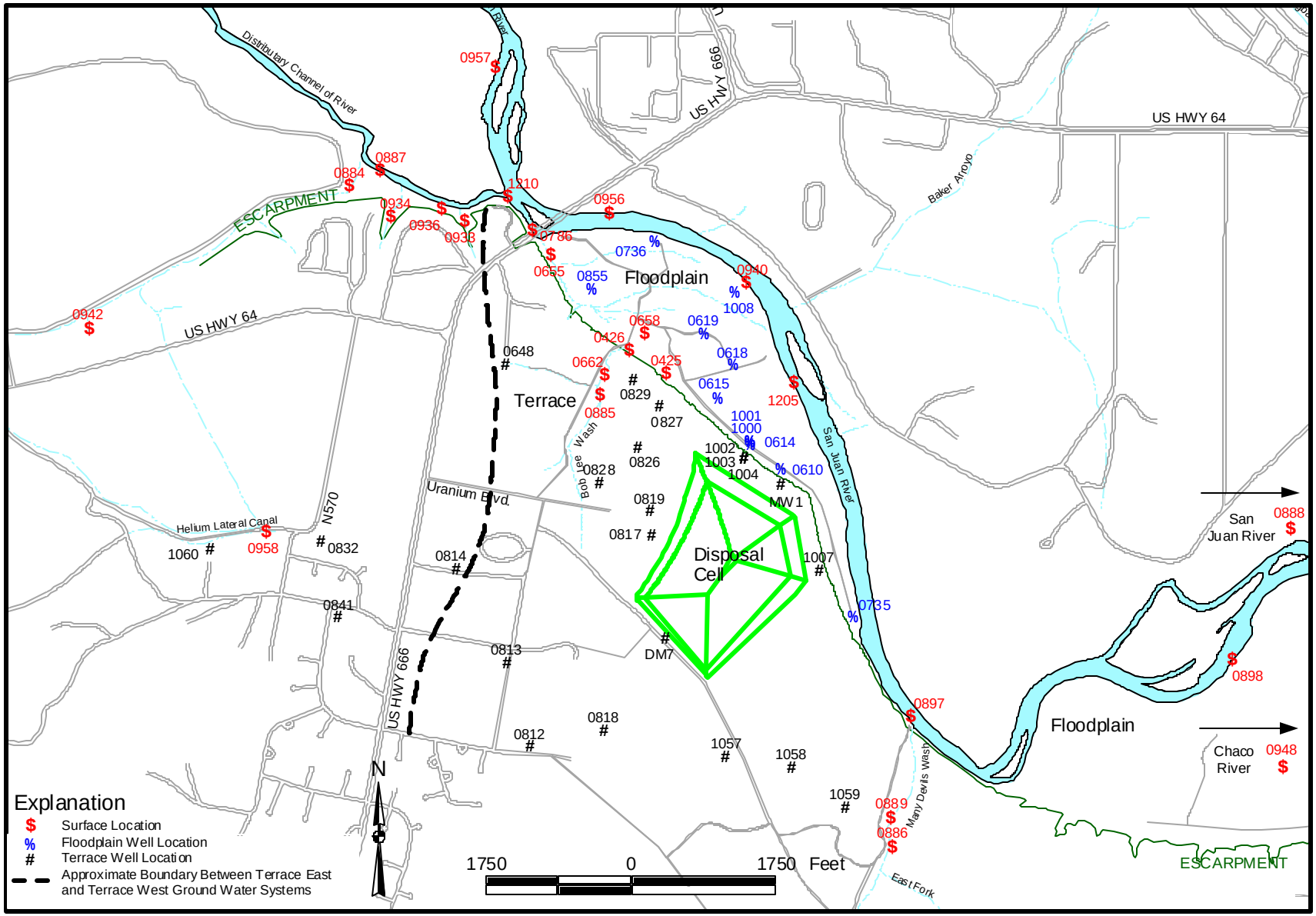

Figure 7–5. Surface and Well Sampling Locations for Future Monitoring at the Shiprock Site

The EPA cleanup goal for sulfate is presently uncertain because sulfate toxicity is undergoing review by that agency. Sulfate is another COC that will not diminish to low levels because it is constantly being added to the floodplain aquifer from the outflow of artesian well 648 and from sulfate leaching from weathered bedrock. Sulfate from well 648 flows down Bob Lee Wash to the floodplain and percolates into the alluvium, accounting for about 60 percent of the ground water in this part of the floodplain and exerting great control on the mass of sulfate. Sulfate concentration in water from well 648 ranges up to 2,340 mg/L, so concentrations are not expected to decrease below this value in the floodplain aquifer as long as the well is flowing. Well 648 is screened from depths of 1,482 to 1,777 ft below ground surface in the Morrison Formation. Ground water in the Morrison Formation is not contaminated from uranium-ore processing and contains naturally elevated sulfate concentrations. Even if the water from well 648 is prevented from flowing, background ground water maximum values will be used. The background sulfate concentrations currently range up to 1,920 mg/L; therefore, floodplain sulfate concentrations may never drop below this value.

The relatively high concentrations of selenium in the millsite floodplain aquifer makes it unlikely that the UMTRA standard of 0.01 mg/L can be met in 100 years or ever, and DOE defers to the higher value of 0.05 mg/L from the Safe Drinking Water Act. Several factors support the use of this higher standard. Section 4.4.8 discusses the widespread ambient concentrations of selenium detected in water associated with Mancos Shale terrain, especially as the shale weathers. Selenium is released from oxidation of sulfides and from the breakdown of sulfate in this formation. Since the 1940s, water associated with the Mancos Shale has been known to contain concentrations of selenium that can be harmful to livestock (Larkin and Byers 1941). The San Juan River naturally contains 0.002 mg/L selenium (one fifth of the UMTRA standard) because it flows through large areas of Mancos Shale (DOI 1999). Background values for selenium are admittedly low compared to average concentrations in the floodplain ground water within the site. This is thought to result from the lack of background water in the terrace above the floodplain background location and subsequent geochemical dominance of the San Juan River water. Concentrations of selenium in ground water from wells in areas considered to be flushed by irrigation over the last 40 years (e.g., terrace wells 837, 838, and 847) and millsite floodplain wells (e.g., 736, 855, 856, 629, and 630) flushed for over 30 years by well 648 flowing into the system generally have selenium concentrations greater than 0.01 mg/L. Because the effect of leaching weathered Mancos Shale will continue, the situation is not expected to change.

It is probable that the floodplain will receive additional moisture from the terrace after pumping ceases and will not flush in 100 years if a continued source of contamination exists and is not contained or greatly diminished. Anecodotal evidence suggests that large quantities of water were used during construction of the disposal cell for compaction, dust suppression, and possibly to slurry one pile to the next. Such large quantities of water require long periods of time to drain. To determine the nature and extent of this continued drainage, DOE will perform monitoring for the 5 years after remedial action begins (see Section 7.6).

7.4.2 Natural Flushing

Results of ground water contaminant transport modeling are presented in Section 4.5, “Numerical Ground Water Modeling.” Former modeling of the floodplain presented in the SOWP, Revision 1, considered the floodplain as a hydrologically bounded unit and showed that the system would flush in 100 years if no continuing source were present. Recent modeling

considers a 4-layer model connecting the terrace with the millsite floodplain and considers a continuing source from the terrace disposal cell. The model presented in this SOWP more closely approximates the field measurements, and predictions are thought to be more realistic.

Modeling predicts that pumping ground water from the plume from a limited number of extraction wells will require about 14 years before levels of COCs drop to acceptable levels. Water from the San Juan River will be drawn into the floodplain alluvium and will replace existing contaminated ground water. The model also predicts that the floodplain will not continue to flush after pumping ceases if the amount of contamination from the terrace continues at its current estimated level. If the volume of drainage of residual moisture from the disposal cell decreases over the next 5 to 15 years, the floodplain may continue to naturally flush. In the unlikely event that it does not, additional actions will be required. The recommended 5-year monitoring period should allow DOE to gather sufficient data to make this determination.

7.5 Explanation of Terrace Compliance Strategies

7.5.1 Terrace East—Active Remedial Action

The proposed compliance strategy for the Terrace East is active remediation until potential risks to humans and the environment have been eliminated. Contaminated ground water will be pumped to a lined evaporation pond and spray evaporated. A secondary result of this action will be to curtail the movement of residual contaminated ground water west of U.S. Highway 666. Ground water in the area west of U.S. Highway 666 will continue to flush and be diluted through irrigation, though it may not meet EPA cleanup standards because of ambient contamination related to leaching of Mancos Shale. Figure 7–2 and Table 7–2 show the strategy for Terrace East.

An artificial ground water system was produced during active milling when an estimated 300 million gallons of water used during processing percolated into the ground. DOE considers the water used during milling, which was pumped from the San Juan River, was purchased by the Federal Government in support of the milling operation. This volume of water has been slowly draining from the terrace for the past 30 years by seepage along the base of the escarpment, by movement into Bob Lee Wash and Many Devils Wash, and by seepage downward along other pathways into the millsite floodplain to the east and north. The predominant joint direction of N40E (Figure 4-6) may provide a preferred flow direction for any residual ground water remaining in the terrace. Approximately 38 million gallons are estimated to remain in the alluvial portion of the terrace system in the buried ancestral river channel southwest of the disposal cell (see Section 4.3, “Hydrology”).

A pump and evaporation system is proposed in the Terrace East area to eliminate potential risk from contaminated ground water surfacing in seeps, Bob Lee Wash, and Many Devils Wash. Modeling indicates that 5 to 7 years will be required to pump ground water levels down sufficiently to hydrologically isolate contaminated ground water from seeps in the washes and floodplain. This assumes the use of two extraction wells in the zone of maximum saturated thickness in the area of wells 813, 814, and 832 and the use of two French drains (as interceptor trenches). One French drain will be located between the disposal cell and upper Bob Lee Wash, and the other French drain south of the disposal cell near well 1057 will intercept ground water traveling toward Many Devils Wash. The total pumping rate of 8 gpm is expected for the 5- to 7-year effort. French drains east of Bob Lee Wash and northwest of Many Devils Wash are

designed to capture the small amounts of ground water traveling in relatively thin sequences of alluvium and weathered Mancos Shale and emerging in these washes. Water collected in these drains will be pumped to the evaporation pond along with water from the extraction wells. Details of all the active remedial alternatives are discussed in Section 8.0, "Development and Evaluation of Active Remediation Alternatives."

The ground water model also assumes a continued source of moisture from the disposal cell and therefore predicts that the seeps will not stay dry unless this source is contained or substantially reduced. As with the floodplain and concurrently with pumping in the Terrace East area, DOE will also conduct extensive monitoring to evaluate the nature and extent of the drainage of residual moisture from the disposal cell. DOE will consider additional appropriate actions, if necessary, after the monitoring data are evaluated.

7.5.2 Terrace West—Supplemental Standards With No Remediation

The proposed compliance strategy for the Terrace West is supplemental standards with no remediation. Contamination in the ground water west of U.S. Highway 666 is partly the result of millsite processing activities and partly from leaching of uranium, sulfate, and selenium from underlying Mancos Shale bedrock by irrigation water. Nitrate and ammonium, other COPCs that occur west of U.S. Highway 666, may also be derived from sources other than milling activities, such as fertilizers, weathering of Mancos Shale, and septic systems.

Uranium isotopes 234 and 238 were used in an effort to distinguish ground water originating from weathering of Mancos Shale from ore- or milling-related water. Results of these analyses in Section 4.4.7 suggests that ground water resulting from the leaching of Mancos Shale contains a Uranium-234/Uranium-238 ratio of 1.5 to 2, and the ratio for millsite-related water is closer to 1. This methodology has been used at other uranium millsites to distinguish milling-related from non-milling related ground water. The area of mixing from millsite-related contamination and other sources, particularly irrigation, is not well defined. The mixing area starts east of U.S. Highway 666 and generally includes the area west of Navajo Road 570 and continues northward to the edge of the terrace (Figure 7–5). Isotopic ratio data from ground water suggest the existence of this area of mixing. Therefore, it is assumed that the Terrace West part of the ground water system is influenced by Mancos Shale and that supplemental standards apply. Where the yield of the Terrace West system is sufficient for agricultural or livestock watering purposes, the water quality is suitable for those uses.

The distribution of nitrate west of U.S. Highway 666 may be a function of ground water movement to the northwest along the buried ancestral channel of the San Juan River, as discussed in Section 4.2.3. Concentrations of nitrate diminish toward the northwest, in response to flushing from irrigation and other human sources. The uranium isotopic ratios from ground water west of U.S. Highway 666 and other geochemical studies of ground water associated with Mancos Shale support the notion that this marine shale is being leached, and therefore, that COPCs in this region may never be reduced to MCL levels.

DOE plans to continue monitoring this area to ensure that COPC levels are not increasing and anticipates that some decrease in concentrations of nitrate will occur over time as irrigation continues to flush residual milling-related contamination. Neither the milling-related nor natural contamination leaching from the Mancos Shale poses an excessive risk to humans or wildlife at this time.

7.6 Future Monitoring Activities

The overall monitoring strategy for the site has two separate purposes. The first is to assess the progress of active remediation and natural flushing, and the second is to gather information that will allow DOE to evaluate the nature of drainage of residual moisture from the disposal cell.

7.6.1 Floodplain Monitoring Requirements

The millsite floodplain alluvial aquifer will be pumped for 5 years, and DOE will evaluate the effectiveness of the strategy during this period. The second part of this strategy, natural flushing, will depend on the outcome of evaluation of drainage of residual moisture. The following activities will be conducted during this 5-year period. After 5 years of monitoring, the floodplain monitoring strategy will be reassessed. Floodplain monitoring requirements are summarized in Table 7–4; surface sample and well sampling locations are shown in Figure 7–5.

7.6.1.1 Monitor Chemistry in the San Juan River

Samples will be collected four times per year for 5 years at surface locations 948 in the Chaco River; 888 and 898 upgradient (background) in the San Juan River; 897, 1205, and 940 on site in the San Juan River; 957 downgradient in the San Juan River; 956 on the north side of the San Juan River near the water intake structure; and 1210 at the junction of the San Juan River and the distributary channel. Samples will be analyzed for the floodplain COCs (manganese, nitrate, selenium, sulfate, and uranium), ammonium, and strontium. Results will be used to monitor compliance with surface water standards and continue to evaluate ecological risks.

7.6.1.2 Monitor Chemistry From Other Floodplain Surface Locations

Additional surface locations will be sampled four times per year for 5 years to evaluate water quality and ecological risk concerns. These locations are: 655 in the floodplain drainage channel, 658 in the floodplain wetland, and 887 in the distributary channel. Samples will be analyzed for floodplain COCs, ammonium, and strontium.

7.6.1.3 Monitor Water Levels and Analyze for List of Constituents in Observation Wells Associated with the Extraction Wells

Wells 610, 614, 615, 618, 619, 735, 736, 855, 1000, 1001, and 1008 are in the contaminant plume area near the extraction wells. They will be sampled four times per year for 5 years for the floodplain COCs, ammonium, strontium, and a number of major elements and other parameters to evaluate effectiveness of the pumping strategy. DOE may add other observation wells to evaluate the area of influence during pumping. Concentrations of these contaminants will be evaluated for changes over time.

7.6.1.4 Monitor Water Levels Versus Time and Analyze for List of Constituents from Extraction Wells

Extraction wells will be monitored continuously and sampled quarterly for 5 years to determine water levels and adjust flow rates. Samples will be analyzed for floodplain COCs and strontium. This information will be used to refine the estimate for pumping duration and to refine the transient model for the floodplain.

Table 7–4. Summary of Monitoring Requirements for the Floodplain

Location	Monitoring Purpose	Analytes	Frequency
Well 610	Compliance action levels	COCs: Mn, NO_3, Se, SO_4, U (and ammonium and Sr based on ecological risk concerns), Ca, Na, K, Mg, Cl, pH, Alk, ORP, and Conductivity	Quarterly for 5 years
Well 614	Compliance action levels		
Well 615	Compliance action levels		
Well 618	Compliance action levels		
Well 619	Compliance action levels		
Well 735	Compliance action levels		
Well 736	Compliance action levels		
Well 855	Compliance action levels		
Well 1000	Compliance action levels		
Well 1001	Compliance action levels		
Well 1008	Compliance action levels		
Surface 948	Chaco River, background		
Surface 888	San Juan River, background		
Surface 898	San Juan River, background		
Surface 897	San Juan River on site, risk		
Surface 1205	San Juan River on site, risk		
Surface 940	San Juan River on site, risk		
Surface 1210	San Juan River, risk		
Surface 957	San Juan River, downgradient, risk		
Surface 956	Intake on N side San Juan R., risk		
Surface 658	Floodplain wetland, risk		
Surface 655	Floodplain drainage channel, risk		
Surface 887	Distributary channel, risk		

7.6.2 Terrace Monitoring Requirements

A pump and evaporation system will operate for 5 years to remove contaminated water from the terrace ground water system. DOE will evaluate the effectiveness of the strategy to (1) reduce risks to humans and the environment and (2) to define drainage of residual moisture from the disposal cell. Terrace monitoring requirements are summarized in Table 7–5; surface sample and well sampling locations are shown in Figure 7–5. After 5 years of monitoring, the monitoring strategy will be reassessed.

7.6.2.1 Monitor Water Levels and Chemistry in Wells Around the Disposal Cell and West of U.S. Highway 666

Wells 648, 812, 813, 814, 817, 818, 819, 826, 827, 828, 829, 832, 841, 1002, 1003, 1004, 1007, 1057, 1058, 1059, 1060, MW1, and DM7 (Figure 7–5) will be sampled and water levels measured four times per year. Samples will be analyzed for the terrace COPCs (ammonium, manganese, nitrate, selenium, sulfate, and uranium). Water levels are anticipated to decline, especially during the latter part of the 5-year pumping period. The chemistry of the ground water samples will be analyzed for 5 years, at which time the compliance strategy will be reevaluated.

7.6.2.2 Monitor Chemistry in Seeps at the Base of the Terrace, in Bob Lee and Many Devils Washes, and in Seeps West of U.S. Highway 666

Surface locations will be sampled four times per year for 5 years and analyzed for the terrace COPCs. Samples from Many Devils Wash and locations west of U.S. Highway 666 will also be analyzed for strontium based on ecological concerns. Sampling locations are: 886 and 889 in

Table 7–5. Summary of Monitoring Requirements for the Terrace

Location	Monitoring Purpose	Analytes	Frequency
Well 648	Cleanup stds for floodplain	COPCs: NH_4, Mn, NO_3, Se, SO_4, and U Other Analytes: Ca, Na, K, Mg, Cl, pH, Alk, ORP, and Conductivity (Sample surface locations 786, 884,886,889,933, 934, 936, and 942 for COPCs plus Sr for ecological risk concerns)	Quarterly for 5 years
Well 812	Water level and chem		
Well 813	Water level and chem		
Well 814	Water level and chem		
Well 817	Water level and chem		
Well 818	Water level and chem		
Well 819	Water level and chem		
Well 826	Water level and chem		
Well 827	Water level and chem		
Well 828	Water level and chem		
Well 829	Water level and chem		
Well 832	Water level and chem		
Well 841	Water level and chem		
Well 1002	Water level and chem		
Well 1003	Water level and chem		
Well 1004	Water level and chem		
Well 1007	Water level and chem		
Well 1057	Water level and chem		
Well 1058	Water level and chem		
Well 1059	Water level and chem		
Well 1060	Water level and chem		
Well MW1	Water level and chem		
Well DM7	Water level and chem		
Surface 886	Chem for risks		
Surface 889	Chem for risks		
Surface 662	Chem for risks		
Surface 885	Chem for risks		
Surface 786	Chem for risks		
Surface 933	Chem for risks		
Surface 936	Chem for risks		
Surface 934	Chem for risks		
Surface 942	Chem for risks		
Surface 884	Chem for risks		
Surface 958	Chem for risks		

Many Devils Wash, 662 and 885 in Bob Lee Wash, 786 under U.S. Highway 666 bridge, 933 in 1st Wash, 936 between 1st and 2nd Washes, 934 in 2nd Wash, 942 in the ponds associated with old gravel pit operations, 884 from the irrigation return flow ditch, and 958 in the Helium Lateral Canal. The analyses will be used to monitor the levels of these constituents surfacing at locations where no interim actions have been proposed; this will address the ecological risk concerns expressed for these areas by the USFWS.

7.6.2.3 Monitor Flow Rates in Washes and Seeps Along the Base of the Escarpment, in Bob Lee Wash, and (if possible) in Many Devils Wash

A system for measuring the flow from seeps draining into Bob Lee Wash and seeps 425 and 426 along the base of the escarpment will be established. Baseline data collection will begin in 2001. Afterward, data will be collected four times per year during normal water sampling. Flows in the washes and from the escarpment seeps are anticipated to decline toward the end of the 5-year period.

7.6.2.4 Monitor Storm Event Flows in Many Devils Wash

An automated water sampler will be evaluated as requested from the USFWS that will collect water samples during a storm event. If accepted, this device will be placed in Many Devils Wash near its confluence with the San Juan River. A surface water sample (or samples) will be collected automatically when water levels exceed specified thresholds in the wash. Samples will be returned to the GJO and analyzed for COPCs and strontium.

7.6.2.5 Monitor Volume Versus Time From Extraction Wells

The estimated location of extraction wells is shown in Figure 8–1. Flow versus time will be monitored and used to further refine the water balance for the terrace, calibrate the transient flow model, and estimate the required time to pump for remediation.

7.6.2.6 Gather Data From the Weather Station

A weather station will be established near the disposal cell to measure rainfall, humidity, barometric pressure, wind speed, and several other parameters. This information will be used along with other data to further define the water balance in and around the disposal cell.

7.6.2.7 Measure Runoff From the Disposal Cell During Heavy Rains

A system to measure runoff from the disposal cell during major precipitation events will be established. The exact nature of this system has not been determined, and permission from the NRC to install it will be necessary.

7.6.2.8 Evaluate Lysimeter Installation at the Site

The most reliable way to determine the amount of water percolating beneath the root zone is to directly measure it. A proposed lysimeter field would answer important questions about the water balance in the terrace system and provide an improved estimate of the amount of water entering the disposal cell. Several lysimeter models and their associated costs will be evaluated, and construction will proceed in FY2001 if funds are available. Data collection will commence as soon as possible.

7.7 Completed Interim Actions

As discussed in Section 4.4, contaminated ground water from the terrace discharges to the surface in several locations. These surface expressions of ground water represent the only currently complete exposure pathways to ground water at the site and the greatest potential for

risk to occur. To address these potential risks in the near term, interim actions have been taken to eliminate these exposure pathways. Areas affected by the interim actions are upper Bob Lee Wash, lower Many Devils Wash, and seeps 425 and 426.

Table 7–6 presents the concentrations of each of the terrace COPCs in surface water samples from the two washes, based on the February 2000 sampling round. It also gives the concentrations of each of these constituents from a June 1998 sampling of artesian well 648, which now partially outflows onto the floodplain through Bob Lee Wash.

Table 7–6. Surface Water Contamination in Bob Lee and Many Devils Washes

Constituent	Sample Location			
	885	886	889	Well 648
	Bob Lee Wash	Many Devils Wash	Many Devils Wash	Outflow to Bob Lee Wash
Ammonium	0.0596	0.0655	0.117	0.585
Manganese	0.0568	0.0022	0.005	0.0886
Nitrate	134	2,930	3,520	<0.0265
Selenium	0.0293	1.89	2.32	<0.001
Sulfate	4,990	16,500	20,100	2,000
Uranium	1.71	0.144	0.171	<0.001

Concentrations in mg/L

7.7.1 Bob Lee Wash

Bob Lee Wash is west and northwest of the disposal cell and the NECA facility. Outflow from artesian well 648 previously discharged into Bob Lee Wash before the wash drains onto the floodplain. Since construction of a small pond in fall 1999 the well discharge no longer flows in the outflow ditch east of the pond. Instead, the well water in the small pond percolates down through terrace material and weathered Mancos Shale creating a new pathway to lower Bob Lee Wash. The flow in Bob Lee Wash is ephemeral upstream of the well 648 outflow and perennial downstream of the outflow. The surface water in Bob Lee Wash, as measured at sample location 885 (which is upstream of the well 648 outflow), exceeds the UMTRA MCLs for nitrate, selenium, and uranium, and is very high in sulfate.

The long-term remediation strategy for Bob Lee Wash is to dry up the terrace system to eliminate the source of contaminated ground water. Interim actions completed in summer 2000 for Bob Lee Wash consisted of the following actions:

- Installation of a fence around the perimeter of the upper part of Bob Lee Wash to keep livestock from entering and to minimize human access.

- Placement of riprap in low areas of the main drainage where water has ponded.
  —A woven geotextile was first placed on the surface in the ponded areas to stabilize the soil under riprap loading. Small aggregate was placed over the geotextile, and a geogrid was placed over the aggregate to provide a barrier to prevent small animal access to the water. Large riprap was then placed over the geogrid.

Dilution of surface water by water from well 648 should result in a final mixture in lower Bob Lee Wash that will only marginally exceed concentrations for sulfate that have been determined to be "safe" in drinking water (EPA 1999). Table 7–7 presents the composition of water from artesian well 648 and surface water at sample location 662 (based on the February 2000 sampling round), which is downstream of where outflow from well 648 enters Bob Lee Wash.

Table 7–7. Surface Water Contamination in Bob Lee Wash

Constituent	Sample Location	
	Well 648	662 (surface location below well 648 outflow)
Ammonium	0.585	0.0672
Manganese	0.0886	<0.0108
Nitrate	<0.0265	18.2
Selenium	<0.001	0.0138
Sulfate	2,000	2,570
Uranium	<0.001	0.077

Concentrations in mg/L

The composition of outflow water from well 648 and of surface water from location 662 are similar. This shows that the contaminated seep water in upper Bob Lee Wash is sufficiently diluted by well 648 outflow water by the time it reaches location 662 and that it should not present a major risk to human and ecological receptors (see Section 6.0 for further discussion).

7.7.2 Many Devils Wash

Many Devils Wash is southeast of the disposal cell and the NECA gravel pit. South of sample location 886, the wash is ephemeral. North of location 886, the wash seems to contain water year-round or nearly so, but the flow is low except during and immediately after storms. Many Devils Wash drains into the San Juan River at a location where no floodplain exists.

The surface water in Many Devils Wash, as measured at sample locations 886 and 889, is highly contaminated, exceeding the UMTRA MCLs for nitrate and selenium by 2 or more orders of magnitude and the MCL for uranium by 5 to 12 times. Sulfate levels in samples from Many Devils Wash are also extremely high; sulfate concentrations at both sample locations exceeded 21,000 mg/L.

The long-term remediation strategy for Many Devils Wash is to dry up the terrace system to eliminate the source of contaminated ground water. Interim actions completed in summer 2000 for Many Devils Wash consisted of the following actions:

- Installation of a fence in the main wash at the confluence of the East Fork, a fence along the west side of the wash on the terrace above, and a fence along the east side of the wash at wash access points. The fencing prevents livestock from entering the wash area. A fenced corridor was placed on the siltstone bed at the knickpoint to allow livestock to cross the wash.

- Installation of a drain pipe in a shallow trench cut through the siltstone bed at the knickpoint to prevent livestock from drinking the contaminated water while using the fenced corridor.

- Placement of riprap in the bottom of the wash in all areas above and below the knickpoint where water has ponded.
  —A woven geotextile was first placed on the surface in the ponded areas to stabilize the soil under riprap loading. Small aggregate was placed over the geotextile, and a geogrid was placed over the aggregate to provide a barrier to prevent small animal access to the water. Large riprap was then placed over the geogrid.

7.7.3 Seeps 425 and 426

The long-term remediation strategy for the escarpment seeps 425 and 426 is to dry up the Terrace East ground water system, the source of contaminated ground water feeding the seeps. Interim actions completed at seeps 425 and 426 consisted of constructing a fence around both seeps and placing netting over the top of each fenced area to prevent birds from accessing the seep water.

7.7.4 Notes on Interim Actions

The areas where interim actions were conducted were investigated for the presence of threatened and endangered species and cultural sites. These investigations found no cultural sites or threatened and endangered species immediately adjacent to the interim action areas.

The interim actions in Bob Lee and Many Devils Washes will slightly increase the flow of contaminated water into the San Juan River, particularly from Many Devils Wash. However, its effect is expected to be insignificant considering the small flows in the washes (less than 5 gpm combined) compared with the diluting effect of the average flow rate of about 1,000 cfs, or 450,000 gpm, of the San Juan River.

At the present time, no treatment of the water collected from either of the interim actions is contemplated. The remediation program for the UMTRA Ground Water Project site at New Rifle, Colorado, includes a laboratory and pilot study of the effectiveness of ZVI on the COCs at that site, which include nitrate, ammonium, vanadium, and uranium. ZVI is known to be ineffective for remediation of sulfate. However, if the New Rifle studies indicate that use of ZVI can substantially reduce levels of nitrate or other Shiprock COPCs, it would be relatively simple to add a ZVI treatment stage, either in the form of a passive barrier (such as has been incorporated at the Durango, Colorado, UMTRA Project site) or a small reactor, to reduce levels of those COPCs in the ground water prior to discharge into the San Juan River.

7.8 Institutional Controls

DOE will need Navajo Nation cooperation to restrict use of contaminated ground water during the remedial action period. Restrictions may include a drilling moratorium, permit restrictions, or other administrative measures. Several alternate sources of water are available near the Shiprock site.

7.8.1 Floodplain Controls

Several controls are in place to prevent access by people and animals to potentially harmful contaminated ground water in the Shiprock millsite floodplain. The southwest boundary of the floodplain is a near-vertical escarpment 50 to 60 ft high that separates the floodplain from the terrace. The narrow southern end of the floodplain is fenced just north of well 735, and a locked gate is maintained across the road at the bottom of Bob Lee Wash where it enters the floodplain. Northwest of Bob Lee Wash, the escarpment continues to the end of the millsite floodplain where the San Juan River is against the escarpment at the U.S. Highway 666 bridge. No grazing is allowed on the floodplain. Grazing permittees were compensated for loss of grazing rights during the first 5 years of cleanup. Access to the floodplain is controlled by the Navajo Nation and DOE.

DOE proposes to enter into an agreement with the Navajo Nation to limit access to drilling wells and for use of ground water in the floodplain until remediation is completed.

DOE proposed to enter into an agreement with the Navajo Water Code Administration to ensure that well 648 is left open and continues to flow in it's present course or in its former course directly into Bob Lee Wash down to the floodplain. The investment in the proposed remediation for the floodplain will depend on well 648 continuing to produce. If well 648 is terminated, a different set of conditions will exist and changes to the remediation strategy may be required.

7.8.2 Terrace East Controls

The disposal cell is fenced on three sides, and warning signs are posted indicating radioactive materials are stored in the area. The cell is open to the east and north for a short distance to the escarpment edge. The 50- to 60-ft escarpment above the millsite floodplain is an effective barrier against entry from the floodplain below. Southeast of the disposal cell, the adjacent NECA gravel pit is fenced eastward nearly to Many Devils Wash. South of the cell, the radon cover borrow pit is fenced around its perimeter and posted with "keep out" signs. North and northwest of the cell, the NECA yard and pond area are fenced and posted. Interim actions completed during the summer of 2000 in Many Devils Wash and Bob Lee Wash, including placement of fencing around the washes and riprap in the bottoms of the washes, limit access by humans or animals to water in these drainages. Construction of fencing and netting around escarpment seeps 425 and 426 was also part of the interim actions. Signs are posted indicating contaminated water and restricting access.

7.8.3 Terrace West Controls

No institutional controls are necessary. The only use of terrace system ground water is from well 847, which is operated by Shiprock High School to irrigate their grounds. The quality of ground water from this well is suitable for this purpose.

7.9 Anticipated Future Land Use

The Navajo Nation and the town of Shiprock have plans for the use of land over portions of the contaminant plume in the terrace area west of the NECA facility. These plans include

- Moving the present fairground facilities to a new location about 4 mi to the south.
- Constructing a hotel and several other businesses in the area of the former fairgrounds.
- Constructing a new Bureau of Indian Affairs office and a multipurpose cultural center, which will include sports fields, in the area south of the senior citizens center.
- Constructing a new Diné College facility in the tract east of the Shiprock High School.

For more information about future land use, see Section 3.3, "Present and Anticipated Land and Water Use." None of these uses are anticipated to be adversely affected by the proposed compliance strategies.

8.0 Development and Evaluation of Active Remediation Alternatives

As presented in Section 7.0, "Ground Water Compliance Strategy," the selected ground water compliance strategy for the Shiprock site is a combination of active remediation, natural flushing, and supplemental standards. The remediation mechanism for the floodplain will be natural flushing in combination with active remediation of the most contaminated areas. Institutional controls will prohibit ground water use until ground water quality improves sufficiently to allow some future use. The Terrace East will be remediated by pumping of the ground water system until ground water no longer surfaces at the seeps and washes. This will reduce further contamination of the floodplain and surface waters and eliminate the only complete exposure pathways at the Shiprock site, thereby protecting human health and the environment. The Terrace West ground water system qualifies for supplemental standards because of widespread ambient contamination, as discussed in Section 4.4.8. Drinking water in the Terrace West area is provided by a high quality alternative source. In portions of Terrace West where ground water yield is adequate for agricultural or livestock watering, quality of the ground water is suitable for those purposes.

The purpose of this section is to develop and evaluate different active remediation alternatives for the floodplain and Terrace East and to recommend a treatment process for remediation of site ground water contamination. The alternatives evaluation process will follow the model that was used for the Tuba City and Monument Valley sites of the Navajo Nation in Arizona. This alternatives evaluation assumes that one treatment process will be used for ground water extracted from the floodplain and from the Terrace East area.

As described in Section 4.0, "Site Characterization Results," attempts to locate naturally occurring ground water in a background terrace location have been unsuccessful. All indications are that all the ground water in the Terrace East and Terrace West areas are the result of milling and irrigation activities, respectively. While much of the contamination in the Terrace East system is likely milling related, it is probable that a significant amount of uranium, selenium, and sulfate in the ground water is from leaching of Mancos Shale. Widespread ambient contamination is one criterion for applying supplemental standards to a ground water system if it is protective of human health and the environment. Application of supplemental standards is protective in the Terrace West area. However, Terrace East water is a significant contaminant source of millsite floodplain ground water, seeps, and washes. Removal of this water would restore the Terrace East area to its pre-milling condition and conform to the intent of the UMTRA Project.

The estimated volume of ground water in the floodplain is about 150 million gallons (460 acre-feet). The planned extraction could require pumping six pore volumes of this water, or about 900 million gallons (270 acre-feet).

The chemical compositions of the Terrace East ground water system and the floodplain aquifer are sufficiently similar that any treatment process that works effectively for the Terrace East system should also be effective for the floodplain. Therefore, the same treatment unit can and should be used for both the Terrace East and the millsite floodplain. For the remainder of this section any mention of the terrace or terrace system refers to the Terrace East area; also, any mention of the floodplain refers to the millsite floodplain area.

Section 8.1 presents an overview of the process used to evaluate and screen technologies and alternatives for remediation of the terrace and floodplain, including a detailed explanation of the evaluation criteria. Section 8.2 develops a list of potential technologies that could be used for remediation, evaluates the technologies as they might be applied, and screens out technologies that are not feasible. Section 8.3 lists technologies that passed the initial screening, combines the technologies into alternatives, and develops the parameters that will be used for the detailed evaluation, which is presented in Section 8.4. Section 8.5 presents the proposed alternative for active remediation, along with discussions of how the proposed method may be deployed and the uncertainties and limitations of the proposed alternative.

8.1 Process for Development and Evaluation of Technologies and Alternatives

This section presents an overview of the process used to select proposed alternatives for remediation of contaminated ground water in the terrace and floodplain systems at the Shiprock site. It also includes descriptions of the criteria used to evaluate technologies and alternatives.

8.1.1 Overview of the Process

The process used to select proposed alternatives for remediation of contaminated ground water includes

- Develop, evaluate, and screen technologies that could be used for remediation.
- Combine the technologies into alternatives and evaluate the alternatives.
- Select an alternative as a proposed remediation method.

Technologies considered could be used for extraction of ground water, disposal of ground water, or treatment of ground water. The initial screening of technologies, generally qualitative in nature, considered whether the particular technology was appropriate for use at Shiprock, given the types, quantities, and locations of the contaminated water and the concentrations of contaminants at the site. This initial screening did not consider cost or implementability except in the most general sense. The technologies that were considered appropriate for detailed review, based on the initial screening, were then combined into alternatives for extraction, treatment, and disposal.

The next step in the process was the evaluation of the alternatives to determine the preferred alternatives for extraction, treatment, and disposal. The evaluation of alternatives used the same criteria as the evaluation of technologies (i.e., effectiveness, implementability, and cost) but was conducted in more detail and included a detailed cost estimate for each alternative. The final step in the evaluation of alternatives was a comparative analysis of the alternatives considering the evaluation criteria.

The last part of the process presents the proposed treatment process for remediation of the terrace and floodplain systems and describes the limitations of the proposed approach.

8.1.2 Evaluation Criteria

Each remediation alternative was evaluated for its effectiveness, implementability, and cost. The proposed alternative is the one that represents the best mix of all three criteria. The evaluation criteria were developed from standard engineering practice for assessing the feasibility of any large-scale project. A discussion of each evaluation criterion is provided in the following sections.

8.1.2.1 Effectiveness

The effectiveness evaluation criterion considers a number of factors, including

- Remediation time frame.
- Conformance with ground water system restoration standards and goals.
- Short-term effects (i.e., effects of remediation on workers, the community, and the environment).
- Disposal of treatment residuals.

Remediation Time Frame

DOE has established 20 years as a goal for remediation of the contaminated ground water under the UMTRA Ground Water Project. Hydrologic modeling at Shiprock suggests that remediation of the floodplain to meet MCLs can be accomplished in just under 14 years; remediation of the terrace is expected to take less time than the floodplain. Because the estimated flow rate of contaminated water from the terrace is less than 10 gpm, and initial modeling shows the floodplain would be pumped at approximately 80 gpm, the treatment system will not be significantly affected by completion of the terrace remediation. For cost estimation purposes, a treatment duration of 15 years has been used, and the design flow rate has been set at 100 gpm. This will allow for some flexibility in operation and also for uncertainty in modeling.

Conformance with Aquifer-Restoration Standards and Goals

The treatment standards for UMTRA ground water projects are the MCLs (specified in 40 CFR 192.04, Table 1), ACLs, or background. These govern the quality of the water in the aquifer after remediation, although not necessarily the quality of effluent from a treatment process. Because the remediation strategy for the terrace specifies pumping to dry up seeps and washes (i.e., eliminate exposure pathways), aquifer restoration standards are irrelevant to the terrace remediation. As described in Section 7.0, “Ground Water Compliance Strategy,” the terrace system will be pumped until yields in monitor wells 812, 813, 818, and 1059 are reduced to levels that no longer feed seeps in Bob Lee and Many Devils Washes (see Section 7.6.2). No attempt will be made to remediate the terrace system to meet MCLs. The quality of any effluent from the treatment system will be dictated primarily by disposal requirements.

Floodplain ground water will be required to meet MCLs. Hydrologic modeling indicates that the combination of natural flushing and active remediation will reduce the concentrations in the floodplain to below MCLs within 15 years. However, if a continuing influent source is present, that source will need to be removed or treated; otherwise, concentrations will increase until they exceed MCLs after active remediation is stopped. Reinjection of treated water into the floodplain from the treatment process is not presently contemplated.

Although treatment standards do not apply to the terrace ground water, knowledge of the composition of the ground water in the terrace is required for the design of the treatment system. Table 8–1 presents the composition of the ground water in the terrace and floodplain, based on analytical data from the March 1999 sampling. The average concentrations of COPCs in the terrace ground water system are computed by averaging the contaminant compositions of samples from the 24 wells in that system that exceeded the MCL for at least one COPC. The composition of the floodplain ground water is calculated by averaging the compositions of samples from the 21 wells in the floodplain that exceeded the MCL for at least one COC, and the background composition of the ground water based on the average composition of samples from wells 850, 851, and 852. These wells are located on the floodplain upstream of the former millsite and are not true background water for the terrace, but no true background ground water in the terrace exists.

Table 8–1. Average Composition of Shiprock Contaminant Plumes

Constituent[a]	Terrace	Floodplain	Background
Ammonium	36.7	17.39[b]	0.084
Manganese	1.535	3.90	1.512
Nitrate	1,388	834	0.241
Selenium	0.761	0.140	<0.001
Sulfate	7,178	6,845	1,527
TDS	14,057	7,847[c]	2,380[c]
Uranium	0.331	0.865	0.017

[a]Concentrations are in mg/L.
[b]Ammonium is not a COC for the floodplain.
[c]June 1999 sampling event.

Short-Term Effects

Short-term effects consider the effects of the remediation program on the community, workers, and the environment. The Shiprock site is mainly within the developed areas of the town of Shiprock, the largest community in the Navajo Nation, and is directly adjacent to residences, businesses, and recreational facilities. U.S. Highways 64 and 666 pass within a few hundred yards of the disposal cell at the site. These highways are heavily traveled by tourists and residents throughout the year. All users of the highways are also classified as part of the community.

Evaluating the effects to workers entails considering the risks to people employed to the treatment system and to those employed to operate and maintain the system during its operational life, as well as individuals supporting the remedial action, such as samplers and equipment operators disposing of treatment residuals.

The evaluation of short-term effects also considers environmental effects. Environmental effects include potential environmental harm caused by deployment of a technology or alternative and

whether the potential harm of remediation outweighs the benefits to be derived from restoration of the ground water systems.

Disposal of Treatment Residuals

Active treatment processes produce a significant amount of residual waste. This waste may include dissolved solids from the ground water, as well as the residuals from any other chemicals that may have been added during the treatment process (e.g., antiscalants or softening agents). These residuals must be contained during the remediation process and must be disposed of either during or at the end of remediation.

8.1.2.2 Implementability

Implementability is an assessment of the feasibility of building, operating, and maintaining a remediation system.

The following aspects of feasibility will be discussed in this SOWP:

- Ease of construction.
- Ease of operation and maintenance.
- Expected reliability.
- Ability to handle changes in influent composition.
- Ability to handle increases in extraction capacity.

Ease of Construction

The Shiprock site is part of the largest community in the Navajo Nation and is only 30 mi west of Farmington, New Mexico, which contains a significant petroleum processing and support industry. Skilled construction labor should be readily available in the area. Therefore, little advantage exists, other than cost (which is evaluated separately), for treatment systems that are easier to construct.

Consideration of construction also requires examining the uncertainty associated with construction, such as the potential for schedule delays caused by technical problems.

Ease of Operation and Maintenance

In general, systems that are more complex require a higher level of skill to operate and maintain. Complexity can be either process complexity or mechanical complexity, and each type has its particular demands on the skills of the operations and maintenance staff. The proximity of the Shiprock site to the petroleum processing facilities in Farmington mean that skilled operating and maintenance personnel should be readily available for the Shiprock remediation project. Systems that are easy to operate and maintain will be preferred over systems that are more challenging.

Expected Reliability

Reliability includes both the physical reliability of the equipment making up the system and the process reliability, which considers the potential for variability in process performance on both a day-to-day basis and a year-to-year basis. Evaluation of the potential reliability of a treatment system must consider the technical and operational complexity and required level of training for operators.

Ability To Handle Changes in Influent Composition

The concentrations of contaminants in the terrace ground water system and the floodplain aquifer are expected to change as dewatering progresses. The composition may also change if currently unknown hot spots (small areas of highly contaminated water) are identified as extraction progresses. Some technologies are better suited to handle such variations than others, and this ability will be considered in evaluating technologies for use at the Shiprock site.

Ability To Handle Variations in Extraction Capacity

Uncertainties of hydrologic modeling mean that it is possible that the treatment system may be required to handle higher, or possibly lower, flow rates from the floodplain aquifer and terrace system than the present model suggests. The ability of a remediation system to handle such changes must be considered in evaluating technologies for use at the Shiprock site.

8.1.2.3 Cost

Once the initial screening of technologies has eliminated those that are not suitable for technical reasons, cost estimates for treatment processes that pass the initial screening process will be developed. Capital costs (both direct and indirect) and operating and maintenance (O&M) costs are calculated for each process. The accuracy of the cost estimates for evaluation of the alternatives is defined at an accuracy of +50 percent to -30 percent. Total cost of the remediation over the life of the project is determined by combining the initial capital cost for the treatment system with the estimated O&M costs over the project duration, using a net present worth analysis. By discounting all costs to a common base year, the costs for expenditures in different years can be compared on the basis of a single figure (i.e., the net present worth). The Office of Management and Budget (OMB) recommends calculating net present worth using a real interest rate (i.e., a rate that does not consider inflation) to discount out-year costs that have not been adjusted for inflation.

Where possible, direct capital costs are developed from invoice costs of similar systems. If that information is not available, generic unit costs, vendor information, and conventional cost-estimating guides have been used. O&M costs are based on labor costs, energy costs, material and equipment costs, and maintenance costs.

8.2 Evaluation of Technologies

8.2.1 Technologies Considered for Remediation

During the process of alternatives evaluation for the Tuba City and Monument Valley sites, technologies for ground water extraction, effluent discharge, and treatment were evaluated. This

process is described in the final SOWPs for those sites. Where applicable, the lessons learned during development of treatment processes for the Tuba City and Monument Valley sites were also applied to the Shiprock site.

Table 8–2 presents a comparison of the average plume compositions in the terrace system at Shiprock with those in the alluvial aquifers at the Tuba City and Monument Valley sites, using spring/summer 1999 analytical data for all sites. Shiprock ground water is obviously more highly contaminated than the aquifers at Tuba City and Monument Valley.

Table 8–2. Average Composition of Shiprock, Tuba City, and Monument Valley Contaminant Plumes

Constituent	Shiprock Terrace	Tuba City	Monument Valley
Ammonium	36.7	38.9	86.8
Manganese	1.535	n/a	
Nitrate	1,388	974	242
Selenium	0.761	0.033	0.003
Sulfate	7,178	2,120	846
Total Dissolved Solids	14,057	5,134	1,688
Uranium	0.331	0.286	0.010

All results are in mg/L.

Given the higher levels of contaminants at the Shiprock site, remediation technologies that were viable for the other sites may not be appropriate for Shiprock. For example, at the Monument Valley site, which is contaminated with moderate concentrations of nitrate and sulfate, remediation by plant farming is under consideration. This technology would not be feasible at Shiprock, because the contaminant concentrations at Shiprock would be toxic to plants. Also, the higher contaminant levels at Shiprock will affect the economic evaluations, so that technologies that were economically viable at the other sites may not be so at Shiprock.

8.2.2 Extraction Technologies

Two types of extraction-well systems were considered: Conventional vertical wells and horizontal wells. French drain systems also were considered as an alternative to horizontal wells and as a supplement to vertical and horizontal wells.

8.2.2.1 Conventional Vertical Wells

Vertical wells are the most commonly used ground water extraction devices, so the bulk of field experience and knowledge relates to conventional vertical wells. Installation of vertical wells is relatively straightforward in most cases. Tests of newly installed vertical wells at the Monument Valley and Tuba City UMTRA sites have demonstrated that vertical wells can provide highly satisfactory yields when combined with proper well design, construction, and development. Vertical extraction wells can be readily converted to injection wells if necessary, or vice versa, and can also be easily decommissioned when necessary. The theoretical performance of a vertical well can be simulated analytically or numerically during the design process using readily available and accepted mathematical formulations, but no comparable knowledge base exists for other technologies.

8.2.2.2 Horizontal Wells

Horizontal well technology was originally developed in the oil and gas industry and has been applied during recent years to environmental engineering. The technique uses directional drilling methods. Typically, boreholes are initially advanced in the vertical orientation and later are turned to a horizontal orientation. Although the initial cost of installing a horizontal well is relatively high, a cost saving may result from lower O&M costs because fewer wells are required due to the greater screened length possible with a horizontal well.

The implementation of horizontal-well technology is considered expensive and risky compared with conventional vertical wells. The long lengths of well screen required increases the difficulties of well completion and development. Other difficulties could evolve later in the project as the ground water cleanup proceeds because few options are available for sealing off the restored parts of the ground water system.

8.2.2.3 French Drains

An alternative to drilling horizontal wells that would serve a similar purpose is the installation of a French drain system. This system consists of a shallow gravel-filled trench installed in the shallow ground water system to enhance flow and recovery of the ground water. These drains are useful for recovering ground water from systems that have low flow rates and/thin saturated thicknesses. They can be easily installed using conventional construction methods.

8.2.2.4 Choice of Extraction Technology

Much of the terrace ground water system south and southwest of the disposal cell has a saturated thickness in alluvial material of 2 ft or less. This is an unfavorable characteristic for productivity of vertical wells. The small area in the ancestral river channel portion of the terrace system where the saturated thickness is greater than 6 ft (Figure 4–15) would be the optimum location for efficient vertical wells. Installation of vertical wells is less expensive than horizontal wells.

The use of horizontal wells would appear to be a favored technology because of the small saturated thickness in the terrace ground water system. However, the coarse composition (gravel and cobbles) of this lower alluvial saturated zone material would pose difficulties in controlling drill directions and maintaining the drill in the thin saturated zone. Also, drilling of horizontal wells is much more expensive than conventional vertical wells.

French drains are relatively easy to install and would be effective where the terrace ground water is at shallow depths (less than 20 ft), such as near the disposal cell. In these areas a few French drains could serve the function of many vertical wells.

Advantages of vertical wells and French drains for conditions in the terrace ground water system offset potential advantages of horizontal wells. **Therefore, a combination of vertical wells and French drains were recommended for use as part of the extraction process for the terrace system at the Shiprock site.**

A large saturated thickness of contaminated ground water occurs in the millsite floodplain aquifer. This, along with the small area underlain by the arc-shaped contaminant plume, makes the use of a small number of vertical wells as the most feasible extraction method.

8.2.3 Effluent Discharge Technologies

This section describes how effluent from the treatment plant could be discharged.

8.2.3.1 No Discharge

No discharge would be required if evaporation were used to treat the ground water. Ground water may be evaporated under conditions in which the contaminants are contained and concentrated for later disposal. Because evaporation produces no effluent, it is both a treatment and a disposal technology. The advantages and disadvantages of evaporation as a treatment option are discussed in Section 8.3, "Evaluation of Alternatives."

8.2.3.2 Discharge to Surface Water

Under this option, treated ground water would be discharged to the San Juan River, either directly or by way of Bob Lee Wash or Many Devils Wash. The feasibility of this option would depend on the quality of the treated water. Discharge to the San Juan River would require a permit from the Navajo Nation.

8.2.3.3 Injection of Treated Water

Because remediation of the terrace ground water system at the Shiprock site will consist of pumping the ground water to dry up surface water at seeps and in washes, injection of treated water would be counterproductive. Injection of treated water into the floodplain is not necessary because of the high volume of natural infiltration from the San Juan River. Therefore, the Shiprock remediation will not use injection of treated water.

8.2.3.4 Effluent Discharge Technologies Recommended for Detailed Evaluation

Evaporation would be used as the effluent-discharge technology if the economic evaluation shows it is the most attractive remediation technology. Discharge to surface water could be used if another treatment technology is selected that produces an effluent acceptable for introduction into the San Juan River. The final selection of treatment discharge technology must be deferred until the treatment alternatives evaluation is completed.

8.2.4 Treatment Technologies

Many treatment processes were identified as potentially applicable for cleaning up the contaminated ground water in the terrace system and the floodplain aquifer at the Shiprock site. The processes can be categorized as follows:

- Evaporation systems.
- Distillation systems.
- Through-medium processes such as ion exchange.
- Biological processes.

- Chemical treatment processes.
- Membrane separation processes.

This section provides a review of the potential applicability of these treatment processes to the Shiprock site and eliminates those that are obviously unsuitable. The processes that are not eliminated in this first screening will be evaluated in greater detail in Section 8.3, "Evaluation of Alternatives."

8.2.4.1 Evaporation Systems

Solar evaporation, in which contaminated water is fed into large lined or unlined outdoor ponds at influent rates that match the rate of natural evaporation, is an established method for reducing the volume of contaminated surface or ground water in arid and semiarid regions of the United States. Nonvolatile contaminants such as nitrate, sulfate, uranium, and other components of TDS, which are the only constituents of concern at Shiprock, will not evaporate and instead will concentrate as a sludge that must be removed for disposal. Solar evaporation systems are constrained by climatic effects, notably temperature (solar radiation), humidity, and wind.

Pond evaporation rates at the Shiprock site are estimated at about 49 in. per year, and precipitation at Shiprock averages about 7 in. per year. Thus, an evaporation system at Shiprock would be expected to be effective for most of the year. The surface area required to achieve complete evaporation would be considerable, however. Preliminary calculations suggest that a solar evaporation pond for the Shiprock site would require a surface area of about 1 acre for every 2 gpm of influent. Thus, treating the design influent rate of 100 gpm would require a solar evaporation pond with a surface area of about 50 acres.

The effectiveness of solar evaporation systems can be significantly enhanced by adding spray systems in which water is sprayed as a fine mist into the air above the solar pond. The fine mist droplets evaporate more readily than does the bulk water at the pond surface. Use of a spray system can substantially reduce the size of the pond required. However, addition of a spray system considerably increases the complexity of the system and requires more maintenance and operator attention than simple solar evaporation.

In general terms, evaporation is a low-cost remediation option for large quantities of contaminated water in arid climates. Spray-enhanced solar evaporation is in widespread use for remediation of contaminated ground water at UMTRA Title II sites, so there is a significant base of experience for design, installation, and operation of such systems. **Because there is no requirement to recover treated water from either ground water system, evaporation was selected for detailed evaluation as a treatment alternative.**

8.2.4.2 Distillation Systems

In a simple distillation process, water is vaporized by heating it to its boiling point. The water vapors are then condensed and recovered as clean water. Nonvolatile contaminants such as nitrates, sulfates, uranium, and other components of TDS will not evaporate. Instead, they will concentrate in the evaporation chamber and must be removed at an appropriate rate. If no volatile contaminants are present, the condensed water will be of high quality and can be used for

virtually any purpose. The concentrate, or brine, may be taken off site for disposal; alternately, it may be evaporated to dryness and the residue can then be disposed of as a solid.

Distillation is an expensive treatment technology to implement because of the significant capital costs of distillation equipment. However, distillation does recover almost all the water, and the product water is of high quality. Because the Shiprock ground water does not contain volatile contaminants, the condensate from a distillation system would be of such high quality that the concentrations of contaminants would be orders of magnitude below regulatory standards for drinking water.

Distillation was chosen as the primary treatment technology at the Tuba City site. The distillation treatment system will be put into service at Tuba City in 2001. Thus, DOE will have accumulated experience with the design, installation, and operation of distillation treatment systems by the time remediation begins at Shiprock. Distillation recovers more treated water than any other technology, and the treated water is of higher quality than that produced by any other technology. **Accordingly, distillation was selected for detailed evaluation as a treatment alternative at the Shiprock site.**

8.2.4.3 Through-Medium Processes

In a through-medium process, a flow stream is passed through a column or reactor containing an insoluble adsorptive or exchange medium. Synthetic ion exchange resins, which are manufactured to have high affinities for certain types of ions, are widely used in through-medium processes for removal of uranium and many other dissolved ionic contaminants.

Ion exchange processes are generally impractical for liquids with TDS concentrations higher than about 1,500 mg/L. The TDS level in the terrace system will average nearly 10 times this amount, and the TDS level in the floodplain is even higher. At such high concentrations, the on-stream time of an ion exchange unit treating the Shiprock ground water would be poor. Because of the need for frequent regeneration, chemical consumption would be high, and the volume of regenerant liquid would be excessive. Thus, ion exchange processes appear to be a poor choice as a remediation technology for Shiprock.

Another type of through-medium process uses zero-valent iron (ZVI) to remove a wide variety of contaminants from ground water. A passive ZVI barrier has been installed at the Durango, Colorado, UMTRA Project site for removal of radionuclides and metallic contaminants. ZVI is an effective process for removal of heavy metals, uranium, and nitrate. However, its efficacy for removal of ammonium has not been demonstrated, and it is ineffective for sulfate removal. **Because ammonium and sulfate are both COPCs for Shiprock, ZVI is not an appropriate primary treatment technology for the Shiprock site, and it was not retained for detailed evaluation.**

8.2.4.4 Biological Processes

Biological processes use bacteria to convert hazardous compounds to other forms that are less hazardous or more amenable to disposal. These processes may be conducted either in situ by injecting the bacteria and/or the carbon nutrient source into the aquifer or ex situ by pumping the water into an aboveground treatment pond or reactor. In situ biological processes were reviewed during the Innovative Treatment Remediation Demonstration (ITRD) process and were rejected

for further consideration in the UMTRA Ground Water Project. Therefore, this section will deal only with ex situ processes.

Nitrate, one of the principal regulated COCs in the Shiprock ground waters, is amenable to treatment with biological processes. Biological denitrification can reduce nitrate levels in water to less than the MCL or to the background level. The primary by-product of denitrification is nitrogen gas (N_2), along with small amounts of nitrous oxide (N_2O). Because nitrogen gas is relatively inert, denitrification generates a treatment residual that does not require handling and disposal, and it has no significant effect on the environment.

Denitrification may be conducted either in a pond or in a biological reactor or series of reactors. A pond-based denitrification process at Shiprock could operate only seasonally because the denitrification reaction loses effectiveness when the water temperature drops below about 50 °F, and it would be impractical to maintain the temperature of a large outdoor pond at 50 °F during the winter months. The treated water would require posttreatment to remove residual organics before it would be suitable for discharge to the San Juan River. Therefore, at Shiprock the biological denitrification process is best suited for indoor reactors, rather than an outdoor pond.

The average sulfate concentration in the terrace system is about 7,000 mg/L. Bacteria that have an affinity for nitrate also have an affinity for sulfate, and desulfurization will take place in parallel with denitrification. While biological denitrification generates nitrogen gas that does not require special handling or disposal and has no significant effect on the environment, biological desulfurization produces hydrogen sulfide (H_2S) as a by-product. Hydrogen sulfide is malodorous, explosive, and extremely toxic. From the bacteriological standpoint, denitrification is the preferred reaction path. However, given the high sulfate levels present in the Shiprock ground water system, it is virtually inconceivable that denitrification can proceed to the extent required to reduce nitrate levels to below 44 mg/L without significant desulfurization.

Laboratory studies have indicated that sulfate-reducing bacteria can be effective at reducing concentrations of uranium in uranium-bearing ground waters by reducing the soluble hexavalent form to the tetravalent form that is amenable to precipitation. This process has not yet been implemented in a full-scale water-treatment process, so potential barriers to full-scale operation have not yet been explored. Biological processes do not address selenium, which would need to be removed using some other process.

Although removal of sulfate, manganese, selenium, and uranium by biological processes is problematic, those constituents can be easily treated using a membrane process such as nanofiltration or reverse osmosis. Such a combined process was investigated for the Tuba City site but was deemed economically unfeasible because the need for constant operator attention to the membrane process resulted in impractically high O&M costs, despite the low capital cost of the system. However, the much shorter treatment duration of the Shiprock system means that low capital cost is a more important consideration for the Shiprock site than was the case for Tuba City or Monument Valley. **Therefore, biological denitrification was retained for detailed evaluation as part of an integrated treatment process for remediation of the contaminated ground water at Shiprock.**

8.2.4.5 Chemical Treatment

Chemical treatment is typically defined as a system using precipitation, coagulation and flocculation, gravity settling, and filtration processes and generally includes addition of chemicals for pH adjustment and formation of precipitates. Such systems are effective for removal of contaminants such as uranium, radium, and sulfate. However, conventional chemical treatment processes are not effective for removal of nitrate, which would have to be addressed by some other technology.

Nitrate could be removed using an ex situ biological denitrification process downstream of the chemical process. The removal of sulfates in the chemical process by precipitation of barium sulfate obviates the need for a biological desulfurization step, thus eliminating the need to dispose of hydrogen sulfide formed as a by-product of biological desulfurization.

The alternatives analysis performed during the preparation of the SOWP for the Tuba City remediation project included a detailed analysis of a combined process using biological denitrification along with a chemical process for removal of sulfate and uranium. That analysis assumed that a DOE-owned 100-gpm chemical treatment facility, which was then in operation at the Monticello, Utah, Superfund site, would be used for the Tuba City remediation. Even with that assumption, the cost analysis for this system at Tuba City concluded that it was a poor choice for that site largely because of high operating costs resulting from the cost of barium chemicals necessary to remediate the sulfate levels at Tuba City. The sulfate concentrations at Shiprock are about 3 times higher than those at Tuba City. Also, the Monticello chemical treatment facility has been claimed by another site and is not available for use at Shiprock, so a new chemical treatment unit would have to be designed and fabricated. **Therefore, chemical treatment does not appear to be viable and was not chosen for detailed evaluation as a treatment alternative.**

8.2.4.6 Membrane Separation Processes

Membrane separation includes all processes in which extremely fine or molecular-level filters are employed. The fine filter, operated under pressure, allows clean water to pass through the element as a clean stream, or permeate, on the downstream side of the element, while the contaminants collect as a concentrate stream, or brine, on the upstream side. The most commonly employed membrane separation processes, in increasing order of effectiveness in removal of dissolved ionic species, are ultra-filtration, nanofiltration, and reverse osmosis (RO). As a general rule, the more completely a membrane separation process removes contaminants from an aqueous stream, the more brine is produced.

The high levels of nitrate in the Shiprock ground water render any membrane separation process technically unfeasible as a stand-alone system. The nitrate ion has a small molecular diameter and is difficult to remove through filtration. RO, the most effective of the membrane separation processes, can remove nitrate ions at 70- to 90-percent efficiency. The average nitrate concentration in the terrace system (see Table 8–1) is 1,388 mg/L. Meeting the treatment standard of 44 mg/L would require a nitrate removal efficiency of almost 97 percent, which is beyond the capability of any membrane process. Thus, the nitrate removal efficiency of membrane separation processes appears to be inadequate for the requirements of the Shiprock project.

However, membrane separation processes are effective for removal of manganese, selenium, sulfate, and uranium, the other principal contaminants at Shiprock. As noted in Section 8.2.4.4, "Biological Processes," biodenitrification is an effective and proven technology for treatment of nitrate-contaminated ground water. Biological denitrification, coupled with membrane separation for removal of selenium, sulfate, uranium, and other contaminants with larger molecular diameters than nitrate, has the potential to be an effective treatment process for Shiprock. **Accordingly, the membrane separation process was retained for detailed evaluation, as part of a combined process incorporating biological denitrification, as a treatment alternative for remediation of Shiprock ground water.**

8.3 Evaluation of Alternatives

This section combines technologies evaluated in the previous section into extraction alternatives and treatment alternatives for the Shiprock ground water.

8.3.1 Extraction Alternative

Remediation of the terrace will use vertical extraction wells in combination with French drains. Although only one alternative is presented, this section includes a discussion of its effectiveness and implementability. Administrative issues associated with implementing the extraction alternative would be minimal because the ground water on the terrace is a result of milling activities.

Remediation of the floodplain will use conventional vertical wells.

8.3.1.1 Extraction Technology—Vertical Extraction Wells

The extraction system consists of vertical wells extending across the saturated zone in the terrace system and across the most highly contaminated area of the floodplain. French drains would be installed near Bob Lee Wash and west of Many Devils Wash to enhance ground water recovery in these areas of the terrace and accelerate drying of the seeps and washes. The initial flow rate is projected to be about 2 gpm per well on the terrace and 8 to 16 gpm per well on the floodplain. Over time, the yield per well on the terrace is expected to decline as the ground water levels decline.

8.3.1.2 Effectiveness

Vertical wells are by far the most commonly used technology for ground water extraction. DOE–GJO has conducted several studies of well drilling and construction methods at different UMTRA Project sites and has refined the techniques for constructing wells with high yield per foot of screened area, which is important because of the small thickness of the saturated zone in the terrace system at Shiprock.

Figure 8–1 shows the proposed locations of extraction wells and French drains to achieve drying of the terrace ground water system and extraction of contaminated water from the floodplain at the required rate. Two wells and two French drains are required for the terrace and five wells are required for the floodplain, producing a maximum total extraction rate of 90 gpm when all wells and drains are operating. Actual pumping rates would be determined in the field after the wells were emplaced.

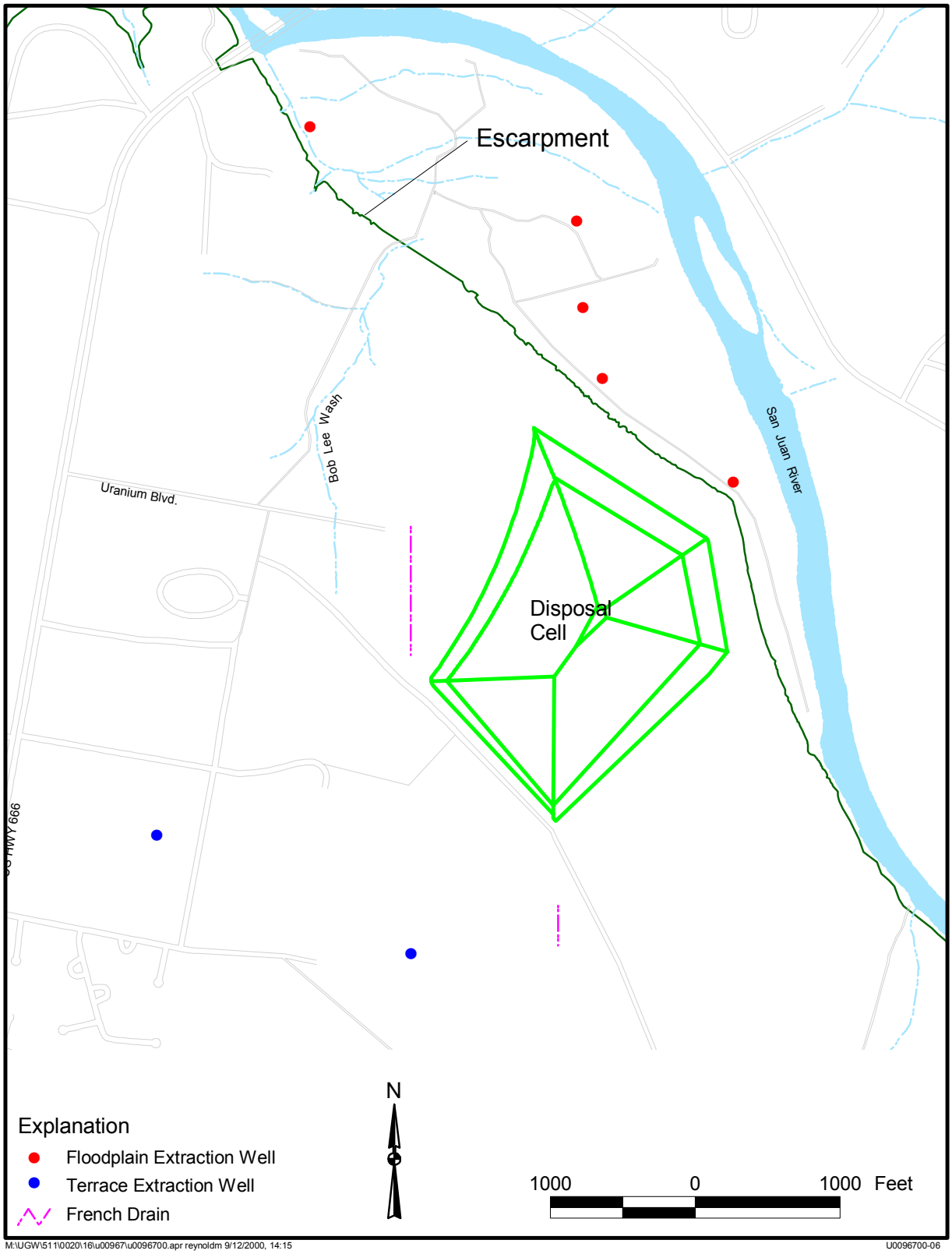

Figure 8–1. Proposed Extraction Wells and French Drain Systems for the Floodplain Alluvial Aquifer and the Terrace Ground Water System at the Shiprock Site

8.3.1.3 Implementability

Construction of the well field and French drains would be relatively straightforward and could be accomplished using readily available technology. The technical obstacles to constructing a remediation well field are relatively few. Potential obstacles include how to obtain the maximum possible ground water withdrawal rate from each well, how to control sand pumping, and how to control the pumping rates in a large well field. These obstacles can probably be overcome through careful well-design, construction, and development techniques. Test borings will be required to determine the depth and optimum location for the French drains. Also, the French drains will be constructed in an orientation perpendicular to the ground water flow path.

8.3.1.4 Cost

The total capital cost for this extraction alternative has not been estimated at this time. Because it is common to all treatment alternatives, the cost of the extraction system is not relevant to the process of treatment selection.

8.3.2 Treatment Alternatives

The treatment alternatives to be evaluated in this section are

- Treatment Alternative 1—Solar Evaporation with Spray Enhancement
- Treatment Alternative 2—Distillation
- Treatment Alternative 3—Membrane Separation and Biological Denitrification

Cost estimates for all treatment processes will be compared based on a net present worth, calculated over the total project life of 15 years, using the OMB standard discount rate of 7 percent.

8.3.2.1 Treatment Alternative 1—Solar Evaporation With Spray Enhancement

Effectiveness

All COCs and COPCs in the Shiprock ground water are nonvolatile (with the limited exception of ammonium, for which a pH-sensitive volatile component exists) and will be retained in the brine as the water evaporates for disposal at the end of the project. The evaporation rate for a spray nozzle designed for continuing operation under high solids loading levels is about 5 to 10 percent water loss per pass through the nozzle.

A disadvantage of a spray system, as opposed to a simple evaporation system, is that the water in the simple evaporation pond may be evaporated to dryness. A spray system, however, can only be operated as long as the pond contents remain liquid. Once the liquid in the pond reaches a certain concentration of solids, the efficiency of the spray system begins to drop dramatically. The concentration of solids at this point is still low enough that disposal is impractical without further concentration. The sludge mixture must be evaporated further by solar evaporation before disposal. The spray pond can be converted to a simple solar evaporation pond for the sludge at

the end of the project, although the time required for the final concentration may be significant because of the small surface area of the pond.

Spray systems usually cannot be operated when wind speeds exceed 15 knots (17 mi per hour). At such times the sprays would be shut off and the pond would operate as a solar evaporation pond. Winds at the Shiprock site are such that the spray system may be expected to be out of service for significant periods.

Evaporation meets the requirements of 40 CFR 192 and is protective of human health and the environment. The only residual produced is the concentrated sludge. The volume of sludge is minimized with this treatment option because evaporation does not require any pretreatment, so no additional chemicals are required.

Implementability

Addition of a spray system to an evaporation pond adds complexity and requires a significantly higher degree of oversight than a simple solar evaporation system. The spray evaporation system at the Homestake site in Grants, New Mexico, operates for 8 months of the year, shutting down from November through February. It also operates only during the daytime. The climate at Shiprock is similar to that of Grants, so the modeling of the spray evaporation system for Shiprock assumes that it will also operate for eight months of the year. The concentration of radionuclides in the water would increase significantly as the pond contents become more concentrated. Operating such a system without continuous monitoring is not practical because of the potential for loss of radionuclide containment.

Operating the system would require the following principal functions: embankment inspection and maintenance, liner inspection and repair, water level monitoring, circulation pump monitoring and maintenance, spray system monitoring and maintenance, and monitoring for leaks. All these functions could be performed by a single operator during the day shift, and the first three functions could be performed with periodic inspections. The need for inspections could be minimized by installing and maintaining adequate fencing to keep livestock and wildlife away from the pond.

Monitoring for leaks consists primarily of monitoring the water levels in the sumps of the leak-detection system, which can be done remotely using a telemetry system. Leak-detection sump pump status could also be monitored remotely using telemetry. Maintenance and repair of pumps and spray nozzles is an on-site function, but round-the-clock presence of maintenance personnel is not required because the spray system will not be operated continuously.

The principal environmental compliance issue associated with maintaining a large, lined pond is uncontrolled release through overflow or leaks. Use of a double-lined pond and an interliner leak-collection system would control subsurface releases. Such engineering controls are highly reliable. Overflow of the pond is unlikely because the water level in a large pond changes relatively slowly and the pond will be monitored on a regular basis by operating personnel.

A large, open body of water in an arid region attracts birds and insects, creating a potential exposure pathway for contamination. As the pond contents become more concentrated, the concentration of uranium, metals, and metalloids (e.g., selenium) in the pond water will increase. Birds and insects may be attracted to the pond and exposed to high levels of contaminants. The

risk increases with a spray system in which contaminants become airborne. Thus, the ability to control waterfowl and insect access to heavily contaminated water would be a concern.

Waste disposal will not be an ongoing function for the evaporation system because the bulk of the concentrated sludge can be disposed of at the end of the remedial action. Final disposal will entail stabilizing and removing about 12,000 tons of sludge from the pond and transporting the mass to an authorized disposal site. The pond liner system will also be removed and disposed of at an authorized disposal site at the end of its service life.

Cost

The estimated size of the spray evaporation system that would be required to treat 100 gpm, with the spray system operating only during the day shift, is about 1.4 acres, measured at the base. The required depth of the pond is about 30 ft. The capital cost of a spray evaporation system of this size is estimated at $1.45 million, and annual operating costs will be about $680,000. The net present worth of this treatment alternative, projected over the total estimated time of 15 years, is $7.66 million.

8.3.2.2 Treatment Alternative 2—Distillation

Effectiveness

Evaporation and water recovery using distillation is an established and proven technology for treatment of contaminated water. Table 8–3 presents data that were developed during pilot testing, at the Tuba City site, of a distillation system similar to the one proposed for Shiprock.

Table 8–3. Performance of Distillation System

Parameter	Influent	Effluent
Ammonium	61.9	2.09
Nitrate	819	2.48
Sulfate	2,440	0.824
TDS	4,900	37
Uranium	0.146	<0.0011

Concentrations are in mg/L

Most of the TDS in the effluent consisted of ammonium. The ammonium levels in the water from the wells used for the tests at Tuba City were higher than those in the ground water at Shiprock, and the Tuba City tests established that ammonium concentrations in the effluent can be minimized by control of the evaporator pH. On the basis of these data, the treated water produced by the distillation system should consistently contain less than 50 mg/L of dissolved solids. It typically will meet or exceed drinking water standards with no further treatment required, and will be suitable for any discharge purpose. The concentrated brine, which contains essentially all the dissolved solids, radionuclides, and other nonvolatile contaminants from the original feed, is expected to average about 10 percent of the total feed.

Pretreatment for the feed water is expected to consist of the addition of sulfuric acid for removal of carbonate and of an acrylic polymer to control crystal growth and shape. The pretreatment

chemicals will concentrate in the brine, where they are expected to increase the volume of the residual solids in the evaporation pond by about 5 percent.

Distillation meets the requirements of 40 CFR 192 and is protective of human health and the environment. The treated effluent is of high quality, and the volume of the concentrated brine is only slightly higher than that produced by the evaporation process.

Implementability

Commercial distillation units are self-contained and include all instrumentation required for monitoring and controlling the operation. The units are designed for outdoor operation with no building required other than the control building for the operators.

The Tuba City distillation system has experienced numerous startup problems. Some of these are attributable to the difficulties associated with executing a complex construction project in a relatively remote area. Shiprock is less isolated than Tuba City, and a construction labor force experienced with industrial work exists in the Shiprock area, so such problems should be greatly reduced. Other problems are related to the process, and are currently in the process of being worked out. By the time a Shiprock distillation system would come on line, the distillation process should be reliable. In general, commercial distillation systems are reliable and require a low level of oversight and only scheduled maintenance during their operating life. Operation of the distillation system will require a minimum of managerial and technical supervision. The acid pretreatment system can operate unattended, although periodic replenishing of the acid will be required. The cost estimate for the operation of the distillation system allows for two full-time employees 7 days a week on day shift for operation and maintenance.

For optimal operation, the distillation system should be operated as nearly continuously as possible. However, it is expected that the flow rate produced by the extraction system will be variable. To dampen variations in the extraction rate and produce a constant flow rate of feed to the distillation unit, a feed tank of approximately 10,000-gallon capacity would be erected at the site immediately adjacent to the treatment unit. Water from the extraction system would flow into the feed tank, and the distillation unit would take its feed from the tank; the level in the feed tank would be allowed to vary as needed.

Concentrated brine is continuously generated by the distillation process. The brine as discharged from the distillation unit is expected to contain no more than about 10-percent suspended solids, a solids-loading low enough that disposal is impractical without further concentration. The brine must be evaporated to dryness. Preliminary calculations indicate that use of a small spray-enhanced solar evaporation pond would be more cost-effective than a larger solar evaporation pond for this purpose. For a discussion of the implementability of solar ponds, see the "Implementability" subsection under Section 8.3.2.1, "Treatment Alternative 1—Solar Evaporation with Spray Enhancement."

Cost

The capital cost of the distillation system, including the evaporation pond and required ancillary equipment, is estimated at $3.98 million. Annual O&M costs would be about $0.82 million. The present worth cost of this treatment alternative, projected over the estimated project duration of 15 years, is $11.42 million.

8.3.2.3 Treatment Alternative 3—Biological Denitrification With Membrane Separation

Effectiveness

During spring 1998, data were collected on the effectiveness of the RO and nanofiltration processes in support of the remediation of extraction water from the Monticello, Utah, Superfund site. These data were collected from two pilot tests using the RO process and one test using the nanofiltration process. The feed water was taken from the feed pond for the Monticello wastewater treatment plant. Table 8–4 presents a summary of the results of these tests. The values given are the percent removal for each constituent, comparing the total quantity in the feed with the total quantity in the product water (permeate). The first RO test was optimized for maximum rejection of contaminants and generated about 25-percent brine; the second RO test was optimized for minimum brine generation and produced about 13-percent brine; the nanofiltration test produced about 20-percent brine.

Table 8–4. Removal Efficiency for RO and Nanofiltration Processes

Constituent	RO (first test)	RO (second test)	Nanofiltration
Calcium	99.6	99.6	97.6
Chloride	98.3	98.5	84.2
Magnesium	99.6	99.6	97.8
Nitrate	92.8	76.8	61.9
Selenium	no data	no data	97.0
Sodium	98.2	98.4	87.8
Sulfate	97.5	98.9	97.4
Uranium	99.4	99.4	98.3

Values are in percent.

Thus, both membrane separation processes are highly effective against most of the dissolved constituents that are present in the Shiprock ground water. The primary exceptions are nitrate and ammonium. (Ammonium was not present in detectable quantities in the contaminated water at Monticello, so no data were collected on ammonium removal in the study presented in Table 8–4.) Neither RO nor nanofiltration is sufficiently effective at removing nitrate to meet the MCL of 44 mg/L given the high concentrations of nitrate in the Shiprock ground water. The proposed process does not use membrane separation to remove nitrate. The nanofiltration process removes other dissolved solids, including sulfate, so that the feed to the biological treatment system contains primarily nitrate. Ammonium can be removed using ammonium stripping, a proven technology.

Assuming that the performance of the full-scale nanofiltration system is comparable to that of the pilot unit tested in 1998, the composition of the nanofiltration process effluent, which is the feed to the biodenitrification system, can be predicted. Table 8–5 presents the results of this prediction. The concentrations of all species, but most noticeably nitrate, in the effluent are higher than might be expected because only about 80 percent of the total feed reports to the effluent stream, with the rest going to brine.

Table 8–5. Predicted Effluent Concentration from Nanofiltration System

Constituent	Feed (mg/L)	Removal Efficiency (%)	Product (mg/L)
Calcium	462	97.6	13.9
Chloride	590	84.2	117
Magnesium	965	97.8	27.0
Manganese	1,535	96.4	0.070
Nitrate	1,388	61.9	665
Selenium	0.761	97.0	0.028
Sodium	1,937	87.8	297
Sulfate	7,178	97.4	236
Uranium	0.331	98.3	0.0072

Studies of the biological denitrification and desulfurization processes indicate that desulfurization will not proceed unless the sulfate loading is 300 mg/L or higher. The predicted sulfate concentration in the nanofiltration effluent is below this threshold, indicating that desulfurization, with its potentially serious consequences, is not likely to be a concern.

Extensive data have been gathered on the efficacy of the biological denitrification process at DOE's Weldon Spring facility near St. Louis, Missouri. The treatment cycle implemented at Weldon Spring produces an effluent containing less than 44 mg/L of nitrate, the MCL for that constituent, from a feed containing about 2,200 mg/L nitrate. The predicted nitrate concentration in the effluent from the nanofiltration unit, as shown in Table 8–5, is 665 mg/L. Thus the denitrification process is capable of meeting MCLs at influent concentrations higher than those that will be treated at Shiprock.

The effluent from the denitrification reactor would be discharged to an RO system to remove residual solids. The permeate from the RO system, which would constitute about 70 percent of the total influent to the treatment system, would be discharged as clean water. The brine, or concentrate, would be directed to a spray-evaporation pond, where it would combine with the brine from the nanofiltration process.

This treatment alternative produces an effluent that meets or exceeds the requirements of 40 CFR 192 and is protective of human health and the environment. Nanofiltration can achieve nearly complete removal of uranium, sulfate, and other dissolved solids from the raw water. Biological denitrification can achieve removal of nitrate from the treatment plant effluent sufficient to meet or exceed the regulatory treatment standard, and the RO polishing step will ensure the quality of the product water that is discharged.

Implementability

Nanofiltration and RO systems are commercially available as packaged treatment systems. One such system was operated at the Monticello Superfund site during 1998 and 1999. This unit experienced numerous startup problems, but eventually proved reliable and effective. The experience gained with that unit could also be used at Shiprock. The systems typically are well instrumented and require a minimum of operator attention. There is a low potential for schedule delays in the construction of the system at the Shiprock site. However, specialists will be needed to oversee construction of the system.

Specially trained personnel will be needed to operate the system. An extensive training program will be needed if Navajo Nation residents are to operate this alternative without extensive oversight by DOE technical contractors. A moderate degree of management oversight will be required to ensure that the plant operates safely and efficiently.

The combined process is expected to generate about 10 percent more sludge than the evaporation process. However, the two membrane processes will generate a much higher quantity of brine (reject water) than distillation, on the order of 30 percent of the total feed. This volume of brine will require a significantly larger spray-evaporation pond than the pond required for the distillation system. For a discussion of the implementability issues associated with the construction and operation of spray evaporation ponds, see the "Implementability" section under "Treatment Alternative 1—Solar Evaporation with Spray Enhancement."

The denitrification system consists of a pair of sequencing batched reactors (SBRs) in which the denitrification reaction will take place. The reactors will be operated in a "fill and draw" system in which one reactor is filling while the other is undergoing the denitrification process and preparing for discharge at the end of the treatment cycle. The system will require significant design work but will not be particularly difficult to construct.

Operation of the denitrification facility will take close operator attention. Denitrification is a batch process with a number of process steps that must be carefully controlled. For instance, the pH will drop rapidly once the denitrification process is under way and acidic ions are liberated. The pH of the ground water is around 6.5. If the pH in the reactors drops below about 6, denitrification will stop, and once it has stopped, it cannot be restarted easily. Also, at the end of the nitrate treatment cycle, it may be necessary to aerate the treated water to get the pH into a neutral (7 to 8) range and to strip residual organics that contribute to chemical oxygen demand.

The design presented in this SOWP is based on information from a system vendor who estimated that the denitrification process would require about 16 hours to reach completion. On the basis of this residence time, the SBRs must have a capacity of around 200,000 gallons each. The cost estimate assumes that the SBRs will have approximately this volume. However, sources at the Weldon Spring facility indicate that the ponds there require 3 to 5 days to complete denitrification. This would increase the size of the denitrification reactors at Shiprock to more than 1 million gallons.

The design upon which the cost estimate is based assumes that SBRs can be used. However, the treatment system should not be designed and installed without first testing this assumption on a laboratory or pilot scale. If biological denitrification were chosen as part of the remediation technology at Shiprock, a testing program should be completed before the final design is begun.

The cost estimate assumes that two operators per shift, with round-the-clock operator presence, will be required for continuous operation. One operator will work primarily on the membrane separation units, and the other operator will concentrate on the SBRs. A high degree of management oversight will be required to ensure that the plant operates safely and efficiently. The chemicals necessary for operation of the chemical treatment plant are not available near the site. The nearest source of commercial quantities of chemicals is Albuquerque, New Mexico.

Increasing the capacity of the treatment system will require installing additional membrane separation units and building additional SBRs.

Cost

The capital cost of the membrane-biological treatment system as described, with a capacity of 80 gpm, is estimated at $2.2 million, and annual operating costs will be about $1.1 million. The net present worth of this treatment alternative, for the total estimated remediation time of 15 years, is $11.8 million.

8.4 Comparative Evaluation of Alternatives

The following section compares the three alternative treatment technologies and recommends a proposed treatment alternative for implementation at the Shiprock site. The treatment alternatives are compared with one another on the basis of the evaluation criteria presented in the introduction to this section. To differentiate, where necessary, between the spray evaporation system proposed as Treatment Alternative 1 and the spray evaporation system used for final concentration of the brine generated by the distillation and membrane/biological treatment systems, Treatment Alternative 1 will be referred to as the ground water evaporation system, and the brine spray-evaporation system for Treatment Alternatives 2 and 3 will be referred to as the brine evaporation system.

8.4.1 Comparative Effectiveness

8.4.1.1 Remediation Time Frame

The three treatment alternatives cannot be differentiated based on this criterion. None of the treatment alternatives uses existing equipment or designs, and it is possible to design a system, using any one of the three treatment processes, that will meet the required project time schedule. The project time frame is affected by the extraction system, but this is common to all three treatment alternatives. There are no plans to reinject treated water into either ground water system. Treated water will be returned to the San Juan River or used for other purposes. Thus, there is no distinction between “consumptive” and “nonconsumptive” treatment systems. Therefore, this criterion will be dropped from the list of criteria used for evaluating the treatment alternatives for the Shiprock site.

8.4.1.2 Conformance with Project Treatment Standards (40 CFR 192) and Goals

As described in Section 8.1.2, “Evaluation Criteria,” the goal of remediation is to pump the terrace system to dry up seeps and washes and to reduce the concentrations in the floodplain to below MCLs. All the technologies will achieve this goal, none better than any of the others because the aquifer is not affected by the choice of treatment technology. Because there is no distinction between the three technologies based on this criterion, it will be dropped from the list of criteria used for evaluating the treatment alternatives for the Shiprock site.

8.4.1.3 Short-Term Effects

All the treatment alternatives are relatively benign in terms of their effect on workers, the community, and the environment. The greatest potential for releases of radionuclides or other hazardous substances is from the spray-evaporation ponds, which are used in all three alternatives. A larger pond is more vulnerable to such accidental releases than a small pond,

because releases are most likely to come from the sprays around the perimeter of the pond. Assuming that the distillation system is similar to that being used at Tuba City, leaks from the system itself will not result in releases because the system operates under vacuum. The greatest threat from the biological denitrification system is the accidental production of hydrogen sulfide from desulfurization. There is a slight chance for releases or other damage from the chemical addition used in the distillation and biological denitrification with membrane separation processes, but careful attention to design for chemical containment should minimize this likelihood. Considering that the membrane/denitrification system is relatively complicated to construct and operate, uses a relatively large pond, and has several other pathways for releases, it ranks below spray evaporation. Thus, distillation is first choice of the three treatment alternatives based on this criterion, spray evaporation is second, and biological denitrification with membrane separation is third.

8.4.1.4 Disposal of Treatment Residuals

All treatment processes produce a concentrated sludge that contains the dissolved and suspended solids that were removed from the ground water during treatment. The Shiprock ground water contains the equivalent of about 4,068 tons of sludge per year, based on the average TDS concentration. Sludge production may vary over the lifetime of the project. The initial rate of sludge production will be relatively high because the extraction rate will be highest at the beginning of the remediation project, and it will decline toward the end of the remediation cycle as the yield of the extraction wells declines and as influent concentrations decline.

Distillation (Treatment Alternative 2) will produce about 5 percent more sludge than spray evaporation (Treatment Alternative 1) because of the small amount of sulfuric acid and antiscalant that will be added to the distillation process. Biological denitrification with membrane separation (Treatment Alternative 3) will produce more sludge than distillation, and about 10 percent more than spray evaporation, because of the greater amount of chemicals required and the biological sludge that will be generated.

The other major treatment residual will be the pond liners, pumps, piping, and other materials that will be disposed of at the end of the remediation program. This is a comparatively small quantity. Because the brine evaporation ponds are smaller than the ground water evaporation pond, they will generate proportionately less of this material. The distillation pond will generate about half as much as the pond required for biological denitrification with membrane separation. Used piping, process equipment, filter elements, and other materials, that are discarded during treatment or are left over from the treatment systems at the end of the remediation should qualify for free release and disposal at any commercial landfill or reuse elsewhere if the need exists. For this reason, estimates of the volume of such materials have not been made.

Overall, spray evaporation is first choice of the three treatment alternatives based on this criterion, distillation is second, and biological denitrification with membrane separation is third.

8.4.2 Comparative Implementability

8.4.2.1 Ease of Construction

All three treatment systems use spray evaporation ponds, and the differences in size of the three ponds will have little effect on their relative ease of construction. The Tuba City distillation

system proved to be more difficult to construct than the alternatives evaluation for that site indicated. However, it should still be a less complicated construction project than the biological denitrification with membrane separation system. The nanofiltration and RO units will be self-contained, but the SBRs will be custom designed and built, and the relatively large number of interconnections between process units will significantly increase the complexity of construction for this process. Thus, spray evaporation is first choice of the three treatment alternatives based on this criterion, distillation is second, and biological denitrification with membrane separation is third.

8.4.2.2 Ease of Operation and Maintenance

The distillation system is a packaged system designed to require minimal operator interface beyond routine monitoring and is expected to be relatively easy to operate. The system will shut off automatically in the event of problems and will relay the required information to the system monitor. The cost estimate for the distillation system assumes only a single day-shift operator for operations and maintenance, though the operator for the evaporation pond and the extraction system will be available to supplement this operator on the rare occasions that additional labor is expected to be needed. These positions are specialty jobs, and individuals filling them will require extensive training. Maintenance of the distillation system is expected to be infrequent but will not be inexpensive because special parts and services that may only be available from the vendor or manufacturer will be required for repair and maintenance of these units.

Operation of the spray-evaporation ponds will require monitoring the function of the spray nozzles and pumping systems to eliminate pluggages and leaks, as well as the pond monitoring operations described in "Implementability" for Section 8.3.2.1, "Treatment Alternative 1—Solar Evaporation With Spray Enhancement." The larger size of the ground water evaporation pond means that it will require more monitoring and maintenance than the smaller brine evaporation system.

The nanofiltration and RO units are packaged systems. Based on experience with a similar unit at Monticello, the operation of these units is expected to require one full-time operator. The SBRs are expected to require the attention of another operator. The process is not expected to operate overnight unattended, so the cost estimate for biological denitrification with membrane separation includes round-the-clock operator coverage.

Overall, spray evaporation is first choice of the three treatment alternatives based on this criterion, distillation is second, and biological denitrification with membrane separation is third.

8.4.2.3 Expected Reliability

Based on discussions with personnel at the Homestake site, the spray evaporation systems are expected to be very reliable, with minimal downtime. And by its nature, a spray evaporation unit can tolerate a certain amount of downtime because the pond can continue to be fed for a period of time if the spray system is out of order. The Tuba City distillation system has not been in service long enough for its reliability to be known. Based on discussions with users of other similar systems, the distillation system proposed for Shiprock is expected to require less than 10-percent downtime for routine maintenance. The biological denitrification with membrane separation system includes more unit operations than either of the other two processes, including interfaces between the continuous membrane processes and the batch denitrification process, and

may be expected to be less reliable because of the increased complexity. Thus, spray evaporation is first choice of the three treatment alternatives based on this criterion, distillation is second, and biological denitrification with membrane separation is third.

8.4.2.4 Ability To Handle Changes in Influent Composition

Evaporation, as a process, is not affected by changes in influent composition. Changes in influent concentration will affect the rate of brine generation in the distillation system. However, the contaminants in the ground water are not volatile, so the distillation system is expected to be reasonably tolerant of changes in influent. The membrane processes are also highly nonselective, and the SBRs will be instrumented to permit them to handle significant changes in nitrate loading. One potential problem would be a significant increase in sulfate loading in the feed, or in the nanofiltration effluent, that could cause an increase in the sulfate in the SBR feed, allowing desulfurization to take place. Sulfate loading will have to be closely watched during remediation. Thus, spray evaporation is first choice of the three treatment alternatives based on this criterion, distillation is second, and biological denitrification with membrane separation is third.

8.4.2.5 Ability To Handle Variations in Extraction Capacity

The turndown capability of the spray evaporation systems is almost infinite because the spray system can be operated for as many, or as few, hours per day as required to maintain liquid inventory.

The Tuba City distillation unit, with a design flow capacity of 100 gpm, can be operated at feed rates as low as 25 gpm. The turndown capability of each of the two cells in that unit is about 50 percent, and if necessary, one of the cells can be turned off completely. The turndown capability of the Shiprock unit would be comparable. Feed rates less than 25 gpm could be accommodated by putting the unit into hot standby while the feed tank is filled, although this mode would compromise the energy efficiency of the process.

Membrane systems typically lose efficiency when operated at feed rates significantly below their design rate. The SBRs are batch systems and can be operated at essentially any turndown by allowing more time between batch operations.

Overall, spray evaporation is first choice of the three treatment alternatives based on this criterion, distillation is second, and biological denitrification with membrane separation is third.

8.4.3 Comparative Cost

The estimated capital cost, annual O&M cost, and total project cost, expressed as the net present worth (NPW) for each of the individual treatment processes were presented in Section 8.3, “Evaluation of Alternatives,” and are presented in Table 8–6. These cost estimates are for the treatment systems only. Costs for the extraction system have not been estimated because extraction is common to all treatment alternatives and does not affect the comparative evaluation. All costs are in millions of dollars.

Table 8–6. Costs of Treatment Processes

Treatment Alternative	Capital	O&M	NPW
Spray Evaporation	$1.45	$0.68	$7.66
Distillation	$3.98	$0.82	$11.43
Biological Denitrification With Membrane Separation	$2.19	$1.06	$11.84

8.4.4 Comparative Summary

The preceding discussion presented 11 evaluation criteria, of which two were dropped, and compared the treatment alternatives for each of the remaining criteria. Table 8–7 lists each of these evaluation criteria and gives the relative rating on each criterion for the three treatment alternatives, with 1 being the highest and 3 the lowest.

Table 8–7. Comparative Ranking of Treatment Alternatives

Criterion	Rating on Each Criterion		
	1	2	3
Short-term effects	Distillation	Evaporation[a]	Bio/Membrane[b]
Disposal of treatment residuals	Evaporation	Distillation	Bio/Membrane
Ease of construction	Evaporation	Distillation	Bio/Membrane
Ease of operation and maintenance	Evaporation	Distillation	Bio/Membrane
Expected reliability	Evaporation	Distillation	Bio/Membrane
Ability to handle changes in influent composition	Evaporation	Distillation	Bio/Membrane
Ability to handle variations in extraction capacity	Evaporation	Distillation	Bio/Membrane
Comparative cost—initial capital outlay	Evaporation	Bio/Membrane	Distillation
Comparative cost-present worth	Evaporation	Distillation	Bio/Membrane

[a]Evaporation = Spray evaporation
[b]Bio/Membrane = Biological denitrification with membrane separation

Determination of Proposed Treatment Process

Treatment Alternative 1—Spray Evaporation is the first choice of the three treatment alternatives. It is first choice for six of the subjective evaluation criteria, and is a close second in the seventh. It also offers the lowest initial capital outlay and the lowest total project cost. The only caveat is that if production of treated water becomes a requirement of the project, spray evaporation would have to be judged unacceptable. Therefore, it is essential that this determination be made before design of the treatment system begins.

Treatment Alternative 2—Distillation is first choice for Short-Term Effects, and is second choice for all other criteria except Initial Capital Outlay, where it is a distant third. Treatment Alternative 3—Biological Denitrification with Membrane is last choice for all evaluation criteria except Initial Capital Outlay, where it is about 50 percent higher than Spray Evaporation but about 50 percent lower than Distillation.

There is little incentive to select a treatment process that produces treated water unless there is a clear technical or economic advantage for such a process. Because this is not the case, spray

evaporation not only offers by far the lowest capital and total project costs, it is also first choice for almost every one of the subjective evaluation criteria. Therefore, given the current assumptions about the Shiprock remediation, Treatment Alternative 1-Spray Evaporation, which is technically preferable to the other treatment processes and also offers the lowest cost, is the preferred treatment technology for the Shiprock ground water remediation project.

8.5 Proposed Remediation Process

8.5.1 Description of the Proposed Remediation Process

The proposed remediation process consists of the extraction and treatment systems. This section presents a discussion of each system.

8.5.1.1 Proposed Extraction System

The objective of the proposed pumping alternative is to extract contaminated ground water from the terrace and floodplain and deliver it to the treatment system at rates that will satisfy the remediation goals.

The extraction system for the terrace consists of two vertical extraction wells and two French drains. The wells would be up to 60 ft in depth and screened lengths would be approximately 10 ft. The expected flow rate per well is 2 gpm, giving the extraction system a peak capacity of 8 gpm once all wells are in service. The extraction system for the floodplain consists of 5 vertical wells, with an average depth of about 15 ft and a screened length of about 10 ft. Figure 8–1 presents a proposed arrangement of the wells and French drains.

A typical extraction-well design for the Shiprock site consists of a 10-in.-diameter borehole completed with 6-in.-diameter stainless steel wire-wrapped well screen and a blank PVC riser. The section of the well containing the well screen will be completed with an appropriately sized sand pack. The final design of the well and the size of the pump will be optimized based on field conditions.

French drains would be constructed with perforated pipe set horizontally, bedded in well-graded (no fines) gravel, and plumbed to a vertical drop pipe and sump. The vertical drop pipe and sump would be positioned at the lowest elevation along the horizontal pipe. Ground water will be intercepted along the length of the perforated pipe and conveyed to the sump. A pump located in the area of the sump will lift the intercepted ground water to the land surface. Each pump will operate continuously and stabilize at an extraction rate of approximately 2 gpm.

8.5.1.2 Proposed Spray-Evaporation Treatment System

The output of the extraction wells will be piped directly to the spray evaporation pond. The preliminary design of this pond suggests base dimensions of about 350 ft in length and 170 ft in width, giving a base area of about 1.4 acres. The depth of the pond will be approximately 30 ft. The pond will contain a grid of about 100 fog-type spray nozzles, fed from a single 250 horsepower circulating pump (with installed spare). The spray system will be operated during the day shift only and will be shut down at times of high wind and during the low-evaporation, late fall and winter months of November through February.

The pond will be double-lined with appropriate geosynthetics and geocomposite materials, and will incorporate an interliner leak-collection sump with level controls and a sump pump. Although wildlife in this populated area is not common, and other sources of clean water are available for waterfowl and migratory birds in the immediate vicinity, appropriate wildlife and bird control measures will be incorporated in the design.

8.5.2 Summary

The proposed system will meet the project goals of pumping the terrace ground water system to remove the risk to health and environment posed by contaminated seeps and washes, and reducing concentrations in the floodplain to meet UMTRA MCLs. The contaminated water from both systems will be extracted and evaporated. All hazardous constituents will be retained in the pond and removed for disposal at a remote location at the conclusion of the remediation project.

8.5.3 Limitations of Proposed Alternative

The Shiprock remediation is problematic for a number of reasons that have been detailed elsewhere in this SOWP. Unresolved technical and political issues include the following:

- Ground water in a background terrace location has not been found, but there are strong indications that Mancos Shale leaching has contributed to ground water contamination.

- DOE is in the process of determining the sufficiency of water rights that would allow use of evaporation treatment at the site.

A successful remediation will require resolution of all of these issues. Pumping of the terrace ground water is predicated on the assumption that contaminated water from the milling operations is slowly migrating toward the edges of the terrace, creating contaminated seeps along the escarpment at the edges of the terrace and in washes that incise the terrace. Pumping the saturated zone in the buried ancestral river channel area of the terrace system is expected to lower water levels and curtail the discharge of water to these seeps, and eliminate the risk. If the pumping fails to dry up the seeps, the remediation program as currently outlined will not be successful. If that is the case, additional remedial measures would be considered.

The feasibility of using a pump-and-treat process to reduce the concentrations of contaminants in the floodplain to meet MCLs depends on the extent of a continuing contaminant source, if present. If no continuing source of contamination is present, hydrologic modeling indicates that the contaminant concentrations in the floodplain can be reduced to below MCLs within the 15-year time frame. However, if there is a significant continuing source, it may not be possible to meet MCLs, and once the pumping ends, the concentrations will rebound.

Technical criteria will need to be established to evaluate the success of the remediation. These criteria will be developed in the GCAP. The GCAP will define the logic that will be used to evaluate the success or failure of the remedial actions. It will also describe what action might be taken if the remediation fails to have any effect on the seeps at the terrace periphery or reducing contamination on the floodplain at the base of the seeps.

Even more than for most remediation projects, the success of the Shiprock project is driven by the extraction process. The main factors that influence the effectiveness of ground water

extraction systems are hydraulic inefficiencies, heterogeneity of the ground water system, and sorption of contaminants to the subsurface material. Hydraulic inefficiencies account for the diffusion of contaminants into low-permeability sediments and hydrodynamic isolation (stagnation points) within a well field. Heterogeneities (e.g., changes in the hydraulic conductivity and effective porosity) will affect the ability to extract ground water from all areas of the ground water system. The sorption of contaminants to the subsurface material retards the movement of the contaminants in the ground water. The longer a contaminant sorbs to the ground water system matrix, the more ground water must be extracted to remove the contaminant.

If pumping of the terrace and floodplain fails to achieve the desired risk elimination, other methods of protecting human health would have to be pursued. A provision in 40 CFR 192 allows the use of ACLs that would be set at higher concentrations than the current cleanup goals but would still be protective of human health. If the results of the pumping on the floodplain establish the existence of a continuing source beyond reasonable doubt, the hypothesis of a continuing source will have to be addressed.

End of current text

9.0 References

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