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Water Issues along the U.S. Mexican Border

Paul Westerhoff

Sustainable rural and urban prosperity along the border between the United States and Mexico depends on the availability of suitable water supplies. Agricultural irrigation requires large volumes of water during certain periods of the year, and the salt content must be low enough ($< 1,000$ mg/L) to prevent damage to plant roots. Industrial manufacturers, such as offshore assembly industries (maquiladoras), need water for commercial applications (e.g., cleaning, processing, and cooling water). Many border communities are experiencing rapid population growth, which will provide labor for intensive agriculture, maquiladora industries, and cross-border work, increasing the demand for industrial water resources. Unplanned urban settlements (colonias) are growing quickly and do not have adequate infrastructure or safe domestic water supplies. Increased population growth, irrigation return flows, and industrial/domestic wastewater flows impact downstream water users and ecosystems on both sides of the border. This monograph comprises summaries of five studies of the U.S. Mexican border region that aid in understanding border water issues and pose potential strategies for sustainable border development.

POPULATION GROWTH

Development in the border region has occurred primarily near areas with surface water supplies. The border region comprises an area that extends approximately 100 km on either side of the border. Figure 1 illustrates the U.S. Mexican border region and its river sys-

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Figure 1: Major River Systems of the U.S. Mexican Border

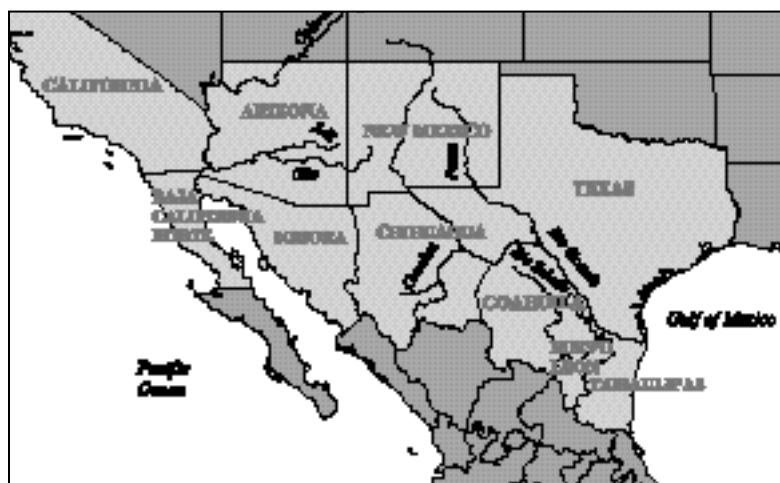
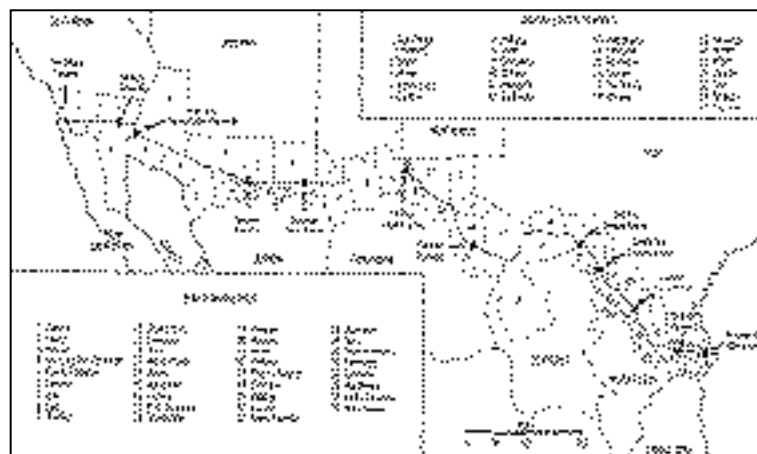


Figure 2: U.S. Counties and Mexican Municipalities of the Border



tens. Figure 2 presents U.S. counties and Mexican municipios in the border region. Cities with the highest populations and maquiladora densities are located along the Rio Grande and Tijuana Rivers, both of which flow year-round. In some areas, such as Nogales, Arizona,

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and Nogales, Sonora, rivers rarely flow due to the over-pumping of groundwater wells (e.g., Santa Cruz River). Such ephemeral rivers flow only during sustained winter rains or intense summer monsoons.

Tables 1 and 2 show the population and rate of population growth for U.S. counties and Mexican municipios along the border, respectively. In 1995, approximately 10 million people lived along the border, with 55% in the United States and 45% in Mexico (Infomexus 1996). If the current rates of population growth continue, the border population will double in 25 years. A portion of the population of Mexico is

Table 1: Total Population (1995) and Growth Rate (1990-1995) for U.S. Counties along the U.S. Mexican Border

State/County	Population*	Growth Rate (%)
California		
San Diego	2,644,132	1.1
Imperial	158,852	4.8
Arizona		
Yuma	152,863	4.4
Pima	780,158	3.2
Santa Cruz	96,878	4.8
Cochise	118,862	2.4
New Mexico		
Hidalgo	6,354	8.7
Lea	23,922	5.8
Dona Ana	158,849	3.2
Texas		
El Paso	678,819	2.8
Hudspeth	3,194	1.2
Culberson	3,245	-8.9
Jeff Davis	2,122	1.8
Presidio	7,656	2.9
Brewster	9,128	1.1
Terrell	1,299	-1.7
Val Verde	43,698	2.4
Kinney	3,289	1.8
Morick	43,766	4.5
Dimock	18,588	8.1
Webb	170,865	4.9
Zapata	11,875	3.2
Faith	51,442	4.9
Hidalgo	479,783	4.6
Cameron	589,378	3.5
Total Population	5,861,431	2.6 (Average)

Sources: Infomexus (1996); *U.S. Census Bureau (1995)

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Table 2: Total Population (1995) and Growth Rate (1990-1995) for Mexican Municipios along the U.S. Mexican Border

State/Municipio	Population*	Growth Rate(%)
Baja California		
Tijuana	991,392	5.8
Tecate	62,617	5.5
Mexicali	696,054	2.6
Sonora		
San Luis Rio Colorado	132,781	5.4
Puerto Peñasco	27,160	8.96
Caborca	64,617	1.6
Altar	7,116	1.8
San José	2,286	1.4
Nogales	139,491	5.8
San Cruz	1,487	-0.8
Casa Grande	29,158	1.8
Nogales	4,911	1.0
Guad. P.R. Collins	18,334	N/A
Agua Prieta	56,288	6.7
Chihuahua		
Janet	18,784	-0.2
Arriola	19,657	5.3
Juan	1,811,786	4.3
Guadalupe	9,611	1.1
Francisco G. Guerrero	8,923	1.1
Olinda	25,585	-0.2
María Guadalupe	2,591	-5.1
Cochila		
Chihuahua	9,870	9.9
Acuña	81,577	6.8
Juárez	8,342	2.2
Florino Nolasco	116,148	5.0
Nuevo	28,497	5.5
Guaraca	2,175	-1.8
Hidalgo	1,273	0.7
Nuevo León		
Andrés	18,278	6.96
Tamaulipas		
Nuevo Laredo	275,040	4.1
Guerrero	4,087	2.0
Mier	6,370	8.87
Miguel Alemán	22,363	6.88
Camargo	15,389	6.31
Gustavo Díaz Ordaz	15,685	-2.12
Rayón	997,053	5.1
Rio Bravo	108,376	1.1
Valle Hermoso	55,274	1.3
Matamoros	563,487	5.3
Total Population	4,759,488	2.1 (Average)

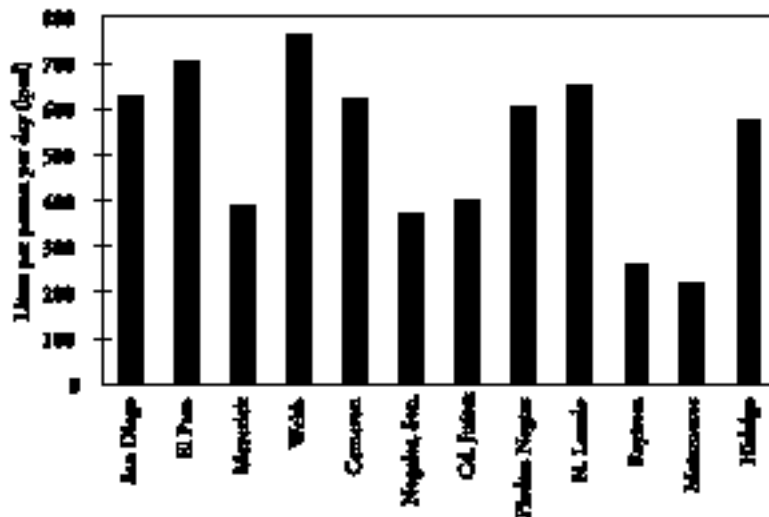
Sources: Infomex (1996); *Censo de Población y Vivienda (1995),

employed in maquiladora industries, but many more people have also moved into colonias along the border. Growing populations can have a detrimental impact on water quality, yet require a sustainable supply of safe drinking water.

WATER CONSUMPTION PATTERNS

The increase in population along the border may result in a shift in current water use from agricultural irrigation to domestic and industrial applications. Currently, irrigation water use is significantly larger than domestic requirements; however, the demand for municipal water treated for domestic and industrial applications is increasing (see Chapter 2). Furthermore, domestic drinking water requires higher water quality standards than does irrigation applications. The amount of water treated to provide for domestic, public, and commercial uses is generally reported in liters per capita (person) per day (lpcd). Figure 3 illustrates per capita water use for U.S. counties and Mexican municipios along the border. The average water consumption along the border in the United States (615 lpcd) is 41% greater than that in Mexican border municipios (435 lpcd). Water consumption is correlated with the standard of living, and as the standard of

Figure 3: Per Capita Water Use for U.S. Counties and Mexican Municipios along the Border



Source: adapted from Infomexus (1996).

living increases in Mexico, water consumption will increase as well. Applying the U.S. and Mexican water consumption rates to the total border population in 1995 would result in an annual water consumption rate (non-irrigation) of $1.3 \times 10^9 \text{ m}^3/\text{yr}$ (1.1 million acre-feet/yr) and $0.76 \times 10^9 \text{ m}^3/\text{yr}$ (0.61 million acre-feet/yr), respectively.

DOMESTIC WASTEWATER

Approximately one-half of all water treated for domestic and industrial applications is conveyed through sewage treatment plants. Sewage contains high concentrations of pathogens (virus, protozoan), organic matter (BOD material), and nutrients (nitrogen and phosphorous). Both sides of the border suffer from inadequate civil infrastructures with which to convey wastewater from households and industries to a centralized location for treatment. Where insufficient infrastructure exists, inadequate on-site wastewater treatment (e.g., surface disposal or septic tanks) can lead to surface or ground water contamination. In some border communities where sewage is collected, wastewater treatment occurs with low-tech, low-cost systems such as lagoons or wetlands. In other communities, sewage pipe infrastructure does not exist and sewage enters surface water supplies as runoff, impairing downstream water quality. With appropriate planning, treated wastewater from these systems can be safely used to irrigate nonconsumptive crops (Chapter 5) or used to augment depleted groundwater aquifers (Chapter 6). Therefore, treated wastewater should be considered a valuable commodity in the U.S. Mexican border region.

AGRICULTURE

In addition to the maquiladora industries, agricultural water demand dominates the annual water usage, especially along the Rio Grande River (Chapters 2 and 3). As water is applied to crops, plants directly utilize approximately 50% to 70% of the water through evapotranspiration or biomass production. While some nutrients are taken up by the plants, most of the dissolved solids in the water (e.g., sodium, calcium, chloride, sulfate) are concentrated in the remaining 30-50% of the water not utilized by the plants, and dissolved solid salts concentrate in the pore water beneath agricultural fields. Salts will migrate into the groundwater, precipitate in the soils, or leach into surface waters as irrigation return flows (Chapter 3). Irrigation practices therefore contribute significantly to increasing salinity, or total dissolved solids (TDS), in downstream groundwater and surface water

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aquifers. Many types of plant roots can not tolerate high salinity levels. Salinity guidelines also exist for salt concentrations in domestic and industrial source waters due to health effects (e.g., sulfate causes diarrhea), taste, and distribution issues (salts precipitate and clog pipes and equipment). Long-term salinity management will be an increasingly significant issue in the border region.

WATER QUALITY

While the availability of water for various uses along the border is a major issue for sustainability, and salinity-related water quality deterioration is a key concern for agriculture, water quality emerges as the most significant environmental health issue in colonias in the border region. Both chemical and microbial water quality issues pose health concerns. The World Health Organization has guidelines regarding a large suite of inorganic (e.g., nitrate, arsenic, mercury, fluoride, lead, copper, radionuclides) and organic (e.g., pesticides, fertilizers, petroleum products) contaminants. Guidelines have also been developed to protect against microbial pathogenic waterborne diseases (e.g., typhoid fever, hepatitis A, giardiasis, cryptosporidiosis, cholera) due to pathogenic virus and protozoa (Chapters 5 and 6). The frequency of waterborne disease outbreaks associated with microbial contaminants increases during warm weather conditions (Craun 1988), thus placing surface waters in the border region at a high risk during many months of the year. In many arid regions, pumping groundwater in close proximity to surface waters creates a direct connection between groundwater and surface water supplies. Chemical and microbial contaminants can readily move through such connections and present significant health risks associated with drinking groundwater. Many of the chemical contaminants pose long-term, chronic health risks whereas microbial pathogens pose significant short-term, acute health risks. Thus, management of risk from water quality contamination must include protection against both chemical and microbial contaminants.

In summary, many regions along the border have historically relied upon both groundwater and surface water supplies for drinking water. Some water supplies have become contaminated by a high salt content or other chemical contamination. Other aquifers are being pumped at rates greater than natural groundwater recharge, resulting in unsustainable mining of groundwater, and ultimately impacting riparian ecosystems. Therefore, some communities are shifting from groundwater to surface water supplies. During periods of prolonged

drought, regional surface water supplies will have to be balanced between irrigation requirements and domestic/industrial needs.

SUMMARY

The following chapters address in more detail general issues regarding the quantity and quality of water in the U.S. Mexican border region. Most of the chapters address the status of water quality in the border region as it relates to the availability of water for various uses. The following is a summary of the focus of each of the chapters in this monograph:

- Chapter 2 addresses the availability of water along the Rio Grande and the impact of upstream reservoir operation on downstream water quality (predominantly salinity).
- Chapter 3 establishes a linkage among surface water flows, water uses, and regional aquifer water quality in the El Paso Ciudad Juárez region.
- Chapter 4 presents results from a study of rainfall runoff that enters surface waters in the Tijuana River watershed and its impact on microbial contamination based upon land usage.
- Chapter 5 considers water quality, and details an innovative, low-cost system capable of safely discharging and reusing domestic wastewater in Ojinaga, Mexico, for irrigation of fast-growing trees that can provide fiber and energy.
- Chapter 6 describes a sustainable, low-cost natural treatment process for wastewater that protects human health from water quality contamination and minimizes detrimental impacts to riparian stream ecosystems.

Without considerable research, planning, and management of water along the border, serious conflicts regarding water resources and water quality will develop. It is especially important to understand the issues related to the availability of water in the border region, including the potential to reuse irrigation return flows and treated wastewater, as well as the impact of urban development on water quality.

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I

Water Issues along the Rio Grande Elephant Butte Reservoir: A Water Quality and Quantity Assessment

Charles D. Turner

ABSTRACT

Water issues are invariably contentious because of the value of water and its scarcity in the Rio Grande Basin. The management of water resources is mainly concerned with two important aspects quantity and quality, both of which vary over time. This article provides an historical overview, dating back 50 years, of releases from Caballo Reservoir and storage in Elephant Butte Reservoir. Historical diversions by water users and historical water quality issues, such as salinity, are presented graphically. A perspective of the past, current trends, and implications for the future are presented.

INTRODUCTION

The 100 years between 1900 and 2000 have brought tremendous change to the upper half of the Rio Grande Basin. In 1900, the Mexican government was contesting reduced flows in the Rio Grande (Rio Bravo) at Ciudad Juárez, Chihuahua, to the U.S. government. Mexican and U.S. farmers were unable to irrigate land that had been receiving water for decades. At the same time, 500 miles to the north, settlers in the San Luis Valley of southern Colorado near the headwaters of the Rio Grande were developing privately funded irrigation districts as fast as possible. In New Mexico, between Albuquerque and Las Cruces, spring flooding along the Rio Grande was an annual irritation for farmers. (The spring floods, however, were part of the process that provided a rich and diverse riparian habitat for

wildlife.) As a result, Elephant Butte Dam was proposed just prior to 1900. The proposed dam would store spring runoff near Truth or Consequences, New Mexico, thereby providing water for reliable summer irrigation releases and preventing downstream flooding. Private funds were not available to construct the dam, and the U.S. Bureau of Reclamation (USBR) did not yet exist. However, the U.S. Congress supported water development for agriculture because it viewed farming as the preeminent tool in settling the west.

The USBR, which was created in 1902, completed construction of Elephant Butte Dam in 1916 and continues to operate the facility for power generation and for water deliveries to the Elephant Butte Irrigation District (EBID) and the El Paso County Water Improvement District (EPCWID). The dam has a storage capacity of 2.065 million acre-feet¹ and a generation capacity of 28 MW of hydroelectric power. The USBR completed Caballo Dam 40 kilometers downstream of Elephant Butte in 1938 to capture and store winter power generation releases from the dam. Below Caballo Dam, all the way to El Paso, the Rio Grande is largely channelized, i.e., straightened and bounded by flood control levees. Most of the riparian habitat has been replaced with irrigated fields, including the habitat of the silvery minnow, an endangered species that reflects the troubled ecological status of the river. Elephant Butte Reservoir provides irrigation water for 178,000 acres in New Mexico and Texas, with return flows irrigating an additional 18,000 acres in Hudspeth County, Texas. Agriculture still controls the vast majority of the water along the Rio Grande, despite the fact that the economy is dominated by urban industries. Agriculture employed only 14,000 (1.7%) of the 820,000 person workforce and generated \$268 million (1.06%) of the \$25 billion income in the Upper Rio Grande basin in 1993 (Niemi and McGuckin 1997). Elephant Butte Dam produced approximately 102 million kilowatt hours of electricity in 1999, which at seven cents per kilowatt hour would be valued at \$7.1 million.² The economic power of municipalities is shown by the transfer of water into municipal and industrial uses. The El Paso water utility is switching from groundwater to sustainable Rio Grande surface water by purchasing rights from agricultural irrigation districts. At the turn of the century, most people lived in rural areas, whereas today most live in cities. Changes are being driven by the tremendous increase in the population of the region and the movement from rural to urban communities.

One other important variable has entered the water equation: salinity. For the lower part of the Upper Rio Grande basin, salinity has become nearly as important as water quantity. Salinity is measured by total dissolved solids (TDS). The headwaters of the Rio Grande

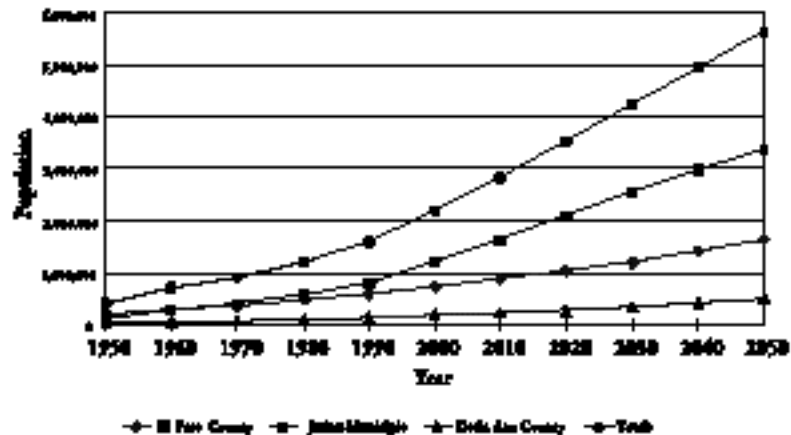
contain less than 50 mg/L of TDS, releases from Elephant Butte Reservoir contain approximately 300 mg/L, and flows in the Rio Grande at El Paso during the winter months contain 1,600 mg/L. The TDS drinking water limit in Texas is 1,000 mg/L and the recommended U.S. EPA secondary limit is 500 mg/L. TDS concentrations in the Rio Grande at El Paso are directly impacted by upstream irrigation return flows. As El Paso switches from groundwater to Rio Grande surface water, conflicts are certain to arise between irrigators and downstream users.

POPULATION

The El Paso Ciudad Juárez region population has grown from less than 60,000 in 1900 to over 2 million today. The population is projected to climb to approximately 6 million by 2050 as shown in Figure 1. El Paso alone, currently at 650,000, is projected to grow to approximately 1.8 million by 2050. The population of Doña Ana County, New Mexico, which includes Las Cruces, was 155,000 in 1994 (CERM 1998). This growth is occurring in a desert region that receives 20 cm of rainfall annually, yet all of these communities tout growth and promote it at every opportunity.

Sustainable development is a buzzword that attempts to distinguish quality growth, especially concerning sufficient water supplies. The cities of El Paso and Ciudad Juárez are intimately connected com-

Figure 1: Population Trends for the El Paso Ciudad Juárez Region



Source: CERM (1998).

mercially, culturally, and geographically. Maquiladoras tie both cities together economically. Culturally, both cities share a 400-year history, with many families in El Paso having strong family ties to Mexico. Geographically, the two cities are connected because they share the Rio Grande, as well as the region's two main aquifers, the Hueco and the Mesilla bolsones. As a result, Ciudad Juárez's projected population of 3.5 million for the year 2050 will have a direct impact on El Paso. Water usage rates between the two communities vary greatly, with Ciudad Juárez using approximately 340 liters per capita per day (lpcd) and El Paso using 643 lpcd in 1999. The El Paso Water Utilities Department hopes to reduce water use to 605 lpcd in the year 2000.³

RIO GRANDE COMPACT

The Rio Grande Compact affects the states of Colorado, New Mexico, Texas, and Chihuahua (Mexico). The compact deals with the portion of the Rio Grande drainage basin above Ft. Quitman, and forms the basis for water allocation by law. The Rio Grande Compact Commission was created to equitably allocate water to the three U.S. states and Chihuahua, Mexico.

In the 1890s, water shortages began to occur along the Rio Grande in the Mesilla and El Paso valleys. People near Ciudad Juárez began to complain to the Mexican government. In response, the Mexican government filed a claim against the United States, asserting that the water shortage was due to increased diversions from the Rio Grande in Colorado, New Mexico, and Texas. In 1906, a treaty with Mexico was reached that entitled Mexico to 60,000 acre-feet annually, to be delivered from Texas. To ensure delivery, Elephant Butte Dam, with a capacity of approximately 2 million acre-feet, was constructed in 1916. With water demands increasing in Colorado, New Mexico, and Texas, a commission was formed of appointees from each state and one federal representative to study the water supply in the area and to allocate water to the states involved. They concluded a compact in January 1929 (Clark 1987; Hill 1968).

Later, on 28 October 1935, Texas sued New Mexico, citing that New Mexico had violated the terms of the 1929 compact by impairing the water supply in Elephant Butte Reservoir through increased diversions and increased salt content in the river, and that these actions violated the rights of Texas water users as set out by the Supreme Court. As a result, then-president Franklin D. Roosevelt requested an investigation of the situation, and upon completion, the Rio Grande commissioners met to negotiate a new compact on 27 September 1937. These negotiations resulted in the formulation of the Rio

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Grande Compact Commission of 1938, which included a set of delivery schedules and requirements for the three states (Clark 1987). The delivery schedule stipulates that 790,000 acre-feet are to be released annually from Elephant Butte for use by New Mexico, Texas, and Chihuahua. In order to guarantee this release, during a year of typical runoff, Colorado is required to deliver one-third of the total Rio Grande flow to Elephant Butte. This amount increases to 50% in wet years and reduces to 20% in dry years.

New Mexico's delivery requirements are as follows: the state must contribute 60% of the Rio Grande flow passing the Otowi bridge above Albuquerque. Flow at this bridge serves as the index station for determining deliveries south of Elephant Butte, and is measured at the San Marcial gauging station above Elephant Butte Reservoir. The requirement is 80% in wet years with no requirement for water deliveries at the lower end of the scale (Clark 1987).

Texas is required to deliver 60,000 acre-feet to Mexico in accordance with the 1906 treaty. The quantity of water available for use in the El Paso County Water Improvement District (EPCWID) and the Hudspeth County Conservation and Reclamation District (HCCRD) of Texas are less precisely defined.

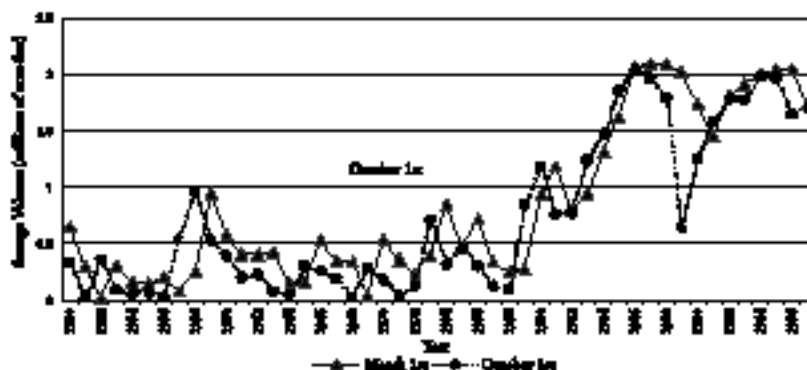
The states of Colorado and New Mexico do not need to strictly adhere to their delivery requirements each and every year. However, the compact does set limits for debits and credits. New Mexico may not be charged with more than 150,000 acre-feet in any one year, or a total of 200,000 accrued acre-feet. Similarly, Colorado has a debit limit of 100,000 acre-feet in any one year and an accrued debit of the same amount. In the case of over-deliveries, or credits, the maximum amount that either state may claim in any year is 150,000 acre-feet. Also, the amount of credit will be reduced by the amount of the spill

Table 1: Water Allocations from Elephant Butte Reservoir

Item	EPCWID	HCCRD	Mexico	Total
Water right acres	69,510	90,640	19,840	179,490
Percent of total	38.5%	50.5%	11%	100%
Delivered at head gates when 492,000 acre-feet are available for release	217,000 af	285,000 af	33,400 af	555,400 af
Delivered at head gates in years of full supply	360,00 af	475,000 af	60,000 af	895,000 af

Source: U.S. Department of Interior (1989).

Figure 2: Elephant Butte Storage

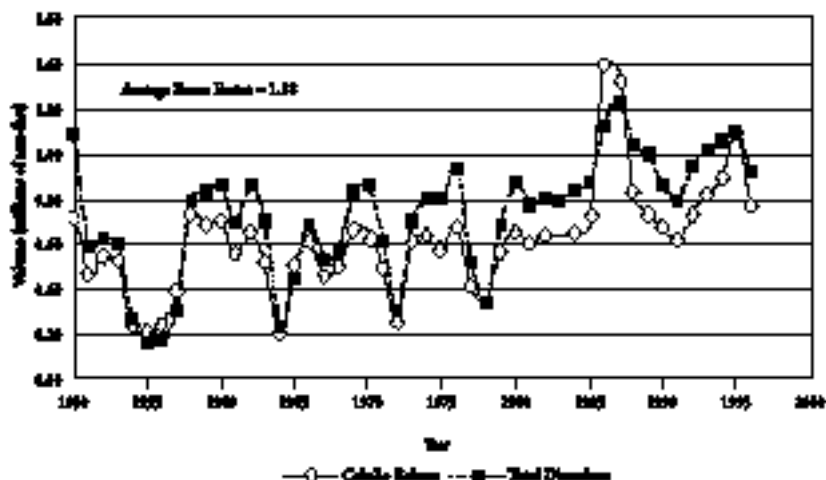


Source: USBR data.

from the reservoir, and no credits can be claimed in the year of any such spill.

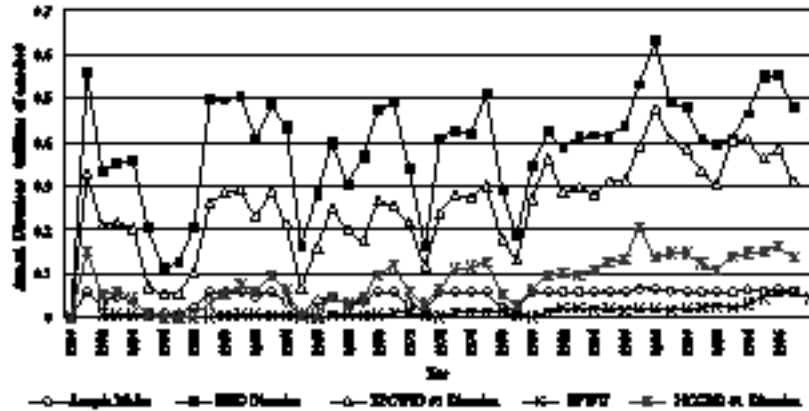
The agreements of the compact are much more specific than stated above and include many more provisions for the debit and credit of water. Despite continued controversy over water use, the Rio Grande Compact has accomplished what it was designed to do, namely, to equitably allocate water for use by each state (CERM 1998; Hill 1968).

Figure 3: Caballo Releases Versus Total Diversions



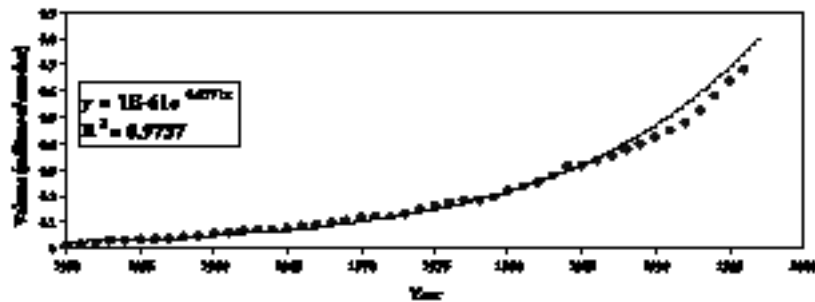
Source: USBR data

Figure 4: Annual Diversions from the Rio Grande below Elephant Butte Reservoir



Source: USBR data

Figure 5: Cumulative Rio Grande Water Usage for EPWU



Source: USBR data

WATER STORAGE, USE, AND SALINITY

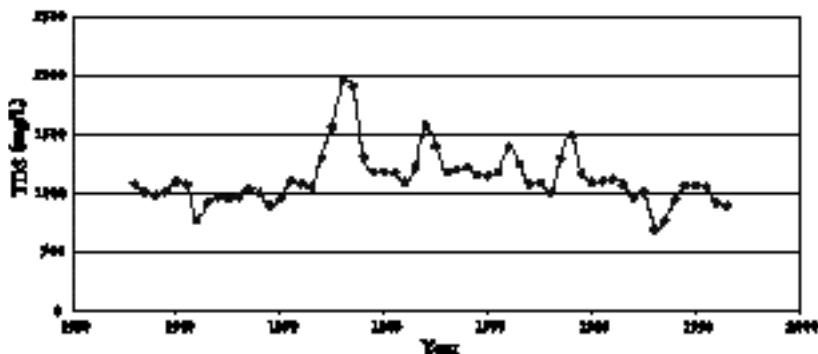
A summary of the water allocations from Elephant Butte Reservoir is shown in Table 1. The EBID is responsible for water deliveries in New Mexico from Elephant Butte Dam, and the EPCWID has similar duties in El Paso County in Texas (Moseley 1970). Water deliveries to Mexico are made at the Acequia Madre headgate in Ciudad Juárez. The difference in total deliveries between years of limited supply and full supply is approximately 360,000 acre-feet of water. (A year is considered limited in supply when only 492,000 acre-feet of water are available for release.) This is very significant, espe-

cially as municipal water supplies become more dependent on the water formerly used for irrigation.

Water storage levels in Elephant Butte Reservoir have varied dramatically since the 1950s, as shown in Figure 2. Beginning in 1979, the volume of water in storage on March 1 increased dramatically over the previous 30 years. This increase in storage translates to a high degree of certainty for full-supply deliveries to head gates during the irrigation season. When 200,000 acre-feet are in storage on March 1, head gate deliveries are usually close to the full-supply scenario shown in Table 1 and confirmed in Figure 3. Figure 3 includes the average reuse factor, which was determined by dividing Caballo Reservoir releases by total diversions. The average reuse factor between 1950 and 1996 was 1.18. The reuse factor since 1980 has been higher than the long-term average.

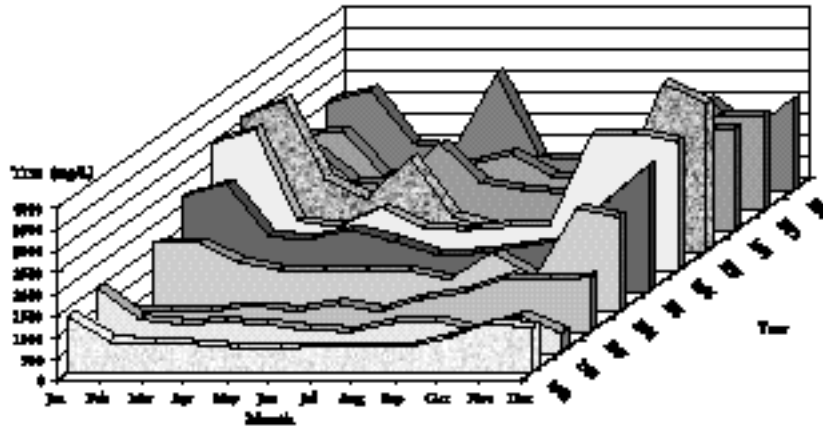
Diversions from the Rio Grande between Caballo Reservoir and HCCRD are shown in Figure 4. Diversions by EBID are followed by EPCWID, HCCRD, and Mexico at the Acequia Madre. Bringing up the rear is the El Paso Water Utility (EPWU). Although the amount of the diversion for EPWU seems insignificant by comparison, there has been a marked increase since the early 1990s. This increase reflects the EPWU's plan to increase utilization of Rio Grande water as a renewable resource. Figure 5 shows the exponential growth of El Paso's cumulative water use since 1950. The R² value for the exponential fit to the data in Figure 5 is 0.97. The exponential growth of Rio Grande water use clearly illustrates El Paso's increasing dependence on the Rio Grande for water. El Paso's total water demand increases every year despite decreasing per capita use, because the rate of population growth is higher than the rate of declining per

Figure 6: Yearly Average TDS in the Rio Grande at El Paso Station



Source: IBWC (1931).

Figure 7: Rio Grande TDS in Wet Years (1942, 1966, 1987) and Dry Years (1954-57, 1964, 1972, 1978)



Source: IBWC (1931).

capita use. In addition, El Paso is being forced to decrease its use of aquifer water due to declining water tables and salinization.

El Paso currently has 300,000 m³/d of surface water treatment capacity split evenly between two surface water treatment plants. These plants treat surface water during the irrigation season, which begins in late February and ends in early October. The EPWU wants to bring these plants on-line earlier in the year and shut them down later, eventually operating them throughout the year. In addition, the EPWU is currently working on the design of a new 300,000 m³/d surface water treatment facility that would be located near the New Mexico Texas border at Anthony, Texas. The city of Las Cruces, New Mexico, is also considering the use of Rio Grande water to supplement its groundwater resources. The trend is obvious. Municipalities are switching to the Rio Grande as a long-term sustainable resource as aquifers decline and become more saline.

Salinity is a contentious issue. Salinity reduces the economic usefulness of water in every instance, whether for agriculture or municipal use. Unfortunately, nearly every use of water increases salinity. Municipal use increases salinity 200 to 300 mg/L as it makes the transition from the water supply in the distribution system to treated wastewater. This has little impact since less than 10% of water usage is for municipal and industrial use in arid regions. Figure 4 shows that the EPWU has diverted approximately 50,000 acre-feet per year in recent years. In 1999, 58,000 acre-feet were taken from the Rio Grande. Total agricultural diversions are 10 to 20 times this amount.

Figure 8: Fields near Hatch, New Mexico



Irrigation has a dramatic impact on salinity because 85-90% of consumptive water use is for agriculture as shown in Figure 4. With each application, salts are concentrated, as approximately half of the water is lost through evapotranspiration, while additional salts are leached out of the soil profile.

The annual average TDS concentration in releases from Elephant Butte Dam is approximately 300 mg/L, while the annual average TDS concentration at the El Paso Station is approximately 1000 mg/L, as shown in Figure 6. The legal maximum TDS concentration in drinking water in Texas is 1000 mg/L. TDS concentrations for January through December for selected wet years (1942, 1986, and 1987) and dry years (1954, 1957, 1964, 1972, and 1978) are shown in sequence in Figure 7. The impact of successive dry years is shown when looking at the years 1954, 1955, 1956, and 1957, which are stacked one behind the other. The TDS concentrations during the winter months are extremely high during each of these years, and the TDS during summer months gradually increases for each succeeding year of drought. The TDS concentrations in winter months increase to nearly 4000 mg/L in 1956 and 1957 as a result of the drought that started in 1954. Fields such as those shown in Figure 8 near Hatch, New Mex-

ico, suffer decreased productivity when TDS increases and urban areas have to consider the use of membrane desalination.

CONCLUSIONS

- ¹ Population growth and the predominance of urban economics indicate that water transfers from agricultural use to urban use will continue.
- ¹ Water is available to support urban growth as long as it can be transferred from agricultural use to municipal and industrial uses.
- ¹ Yearly average TDS concentrations in the Rio Grande shown in Figure 7 are directly impacted by drought, but appear to remain constant overall. High winter season TDS concentrations limit water use during this period. Increased use and reuse of water in upstream metropolitan areas like Albuquerque, New Mexico, are likely to slowly increase the salt load delivered downstream.
- ¹ Periods of drought pose the largest challenge, not only because of reduced water quantities, but also because of the increased salinity in the available water supply.
- ¹ The need for urban areas to have surface water deliveries 12 months of the year will gradually change the long-term dynamics of the riparian system.
- ¹ The riparian ecosystem will continue to degrade because of the pressures of urban growth and more intensive water-use practices. Only major efforts and significant investment will change this.

El Paso has supported its growth by utilizing a large part of the available fresh groundwater in the Hueco Bolson. This aquifer is gradually being exhausted. Plans to inject treated Rio Grande water into this aquifer will make it into a storage reservoir for seasonal or drought use, depending on management decisions. As the Upper Rio Grande basin system becomes more and more intensively managed and developed, perturbations such as drought or climate change will have significant impacts. The extent and magnitude of these impacts should be studied in order to better understand the effects of our growth and management decisions on the environment in which we live.

NOTES

1. Land units and water volumes in the United States are commonly measured in terms of acres. One acre is equal to 43,560 square feet, which is equivalent to a square parcel of land approximately 209 feet on each side. The amount of water used to irrigate one acre of land to a depth of one foot is called an acre-foot. An acre-foot contains 325,851 gallons of water, or 1233.5 cubic meters. One cubic meter of water contains 264.2 U.S. gallons. An acre of cotton requires three to five acre-feet during a growing season. A family of four using 200 gpd of water during a year will use 0.9 acre-feet of water.
2. For more information regarding the Elephant Butte Dam see <<http://www.usbr.gov/power/data/sites/elephant>>.
3. For more information regarding El Paso usage rates see <<http://www.epwu.org/>>.

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III

Surface and Ground Water Interac - tions: El Paso Ciudad Juárez Region

John Walton and Gregory Ohlmacher

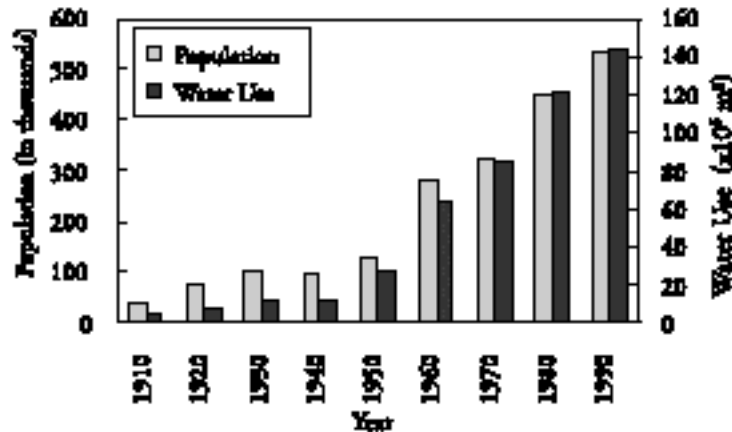
ABSTRACT

Ion concentrations and flow rates in the Rio Grande measured at El Paso reflect a strong interaction between surface and ground waters. The interaction occurs as irrigation waters are applied to fields with shallow groundwater, returning to the river in agricultural drains. Strong seasonal cycles are apparent in flow and water chemistry caused by irrigation patterns. Summer water release leads to higher flows and improved water quality. Winter flows primarily represent irrigation return flow and are of lower quality. Multiyear cycles reflecting climatic cycles are also apparent in the data. Empirical evidence is presented suggesting precipitation of calcium carbonate and calcium sulfate in fields.

INTRODUCTION

Growth of the El Paso Ciudad Juárez metropolitan area is increasing the demand on the available freshwater resources of the area. In the El Paso area, the increase in water use is directly correlated to the increase in population (Figure 1). Population pressures have increased water usage, even with the water conservation measures implemented in the 1980s. El Paso and Ciudad Juárez primarily use groundwater from intermontane-basin aquifers to supply their needs. However, during the past 10 years, El Paso has increased its usage of water from the Rio Grande. Based on 1994 data, El Paso obtained 56% of its water supply from intermontane-basin aquifers in the

Figure 1: Relationship between Population Growth and Water Use for El Paso, Texas



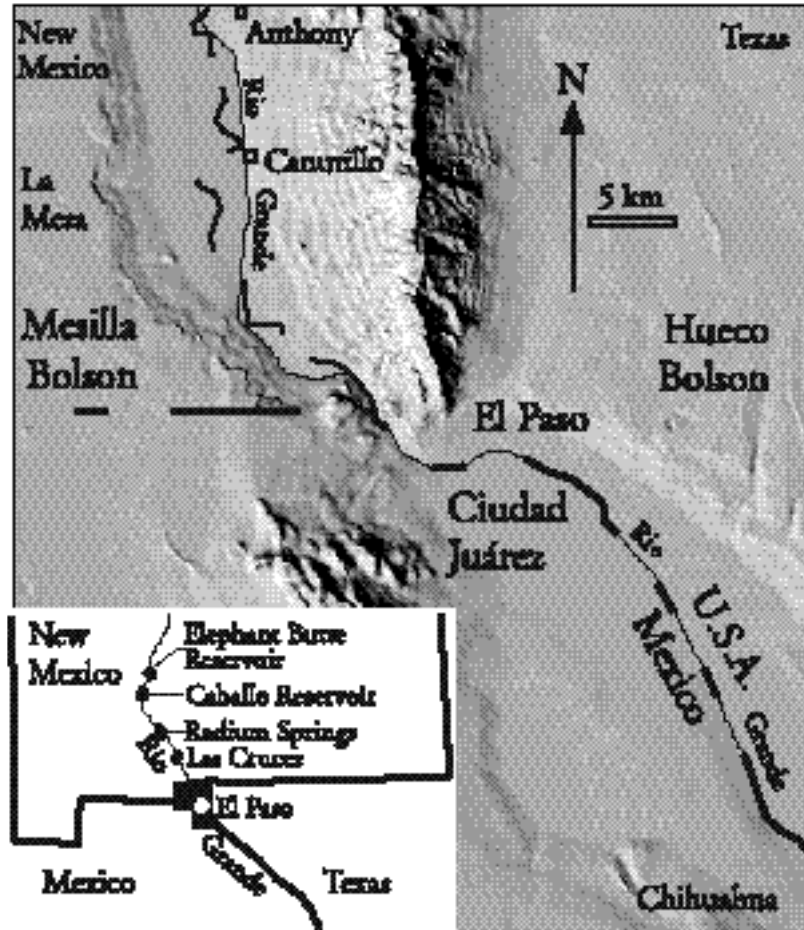
Source: Rebuck et al. (1996).

Hueco and Mesilla bolsones (Figure 2) and 44% came from the Rio Grande (Rebuck et al. 1996). Repeated usage of river water for irrigation between the headwaters and El Paso has degraded the quality of the water by increasing the salinity. During periods of high discharge, the water quality meets water standards and can be used by El Paso. However, during periods of low discharge, including the non-irrigation season (October March) and droughts, the salinity increases to the point that the water is no longer usable for domestic purposes.

Groundwater used for municipal purposes comes from aquifers in the Mesilla and Hueco bolsones. Four aquifers, referred to as the shallow, upper and lower intermediate, and deep aquifers, are recognized in the sediments of the Mesilla Bolson (Nickerson 1989). These aquifers are recharged by the Rio Grande, irrigation canals, water spread on agricultural fields, and groundwater flow from the La Mesa, New Mexico, region. The Rio Grande is a losing stream (a zone of groundwater recharge) where it enters the north end of the Mesilla Bolson near Radium Springs (Figure 2). Traditionally, the Rio Grande was a gaining stream (a zone of groundwater discharge) where it exited the south end of the Mesilla Bolson near El Paso. This pattern has become more complex and seasonally variable because of irrigation and municipal water usage.

The quality and quantity of recharge from surface water and groundwater flow control the quality of the water in the shallow aquifer beneath the Mesilla Bolson. A strong hydraulic connection

Figure 2: Shaded Relief Map of the Area Surrounding El Paso, Texas, and Ciudad Juárez, Chihuahua (Digital Elevated



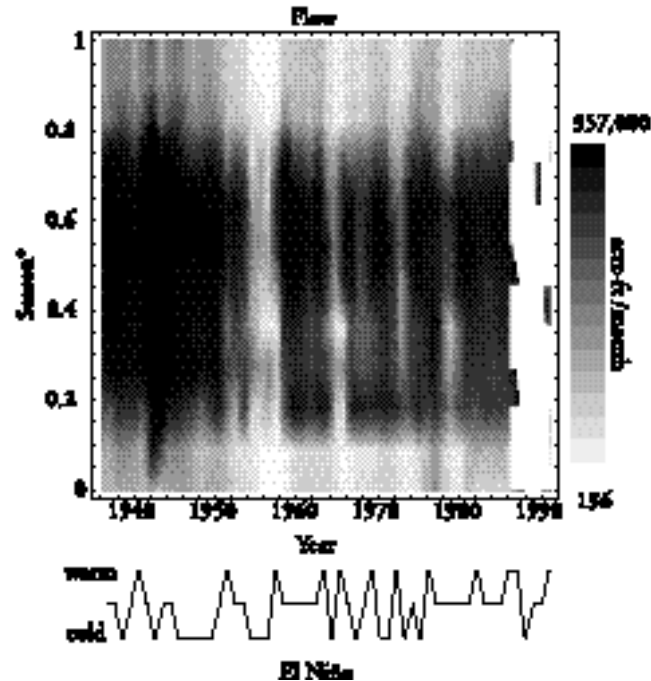
Note: The area of the shaded relief map is shown as a black box near El Paso on the inset map. The Mesilla Bolson stretches from Radium Springs, New Mexico, to El Paso, Texas.

exists between the surface water and the shallow aquifer. Water mass balances for the shallow aquifer show annual cycles of draw-down and rebound related to irrigation practices (Updegraff and Gelhar 1978). However, recharge to the shallow aquifer beneath the Mesilla Bolson appears to be keeping pace with water usage (Hernandez 1978; Peterson et al. 1984). Current water-use practices control salinity in the shallow aquifer. River water is applied to agricultural areas where the salts are concentrated by the high evapotranspira-

tion rates characteristic of this semiarid region. Additional salts are added to the water due to the weathering of minerals in the soil. The overall salinity of the shallow groundwater reflects the balance between applied irrigation water, evapotranspiration, and leakage of water from canals. Water currently leaking from irrigation canals is not wasted as is commonly believed rather, it plays an important role in reducing the salinity of groundwater in the shallow aquifer.

The brackish groundwater from the shallow aquifer is discharged into the irrigation drains and flows back into the Rio Grande. This leads to an increase in the salinity of the river water as it flows through the Mesilla and Hueco bolsones (Hernandez 1978). Additionally, pumping in the intermediate and deep Mesilla Bolson aquifers has affected groundwater flow by causing the downward migration of brackish groundwater from the shallow aquifer. The migration of this water will

Figure 3: Discharge in the Rio Grande at El Paso



*The seasons in Figures 3-6 are represented as numbers between 0 and 1, where 0 is January 1 and 1 is December 31.

Note: White regions on the right side of graph represent times when discharge data are lacking. The generalized El Niño temperature range is represented as a solid line below.

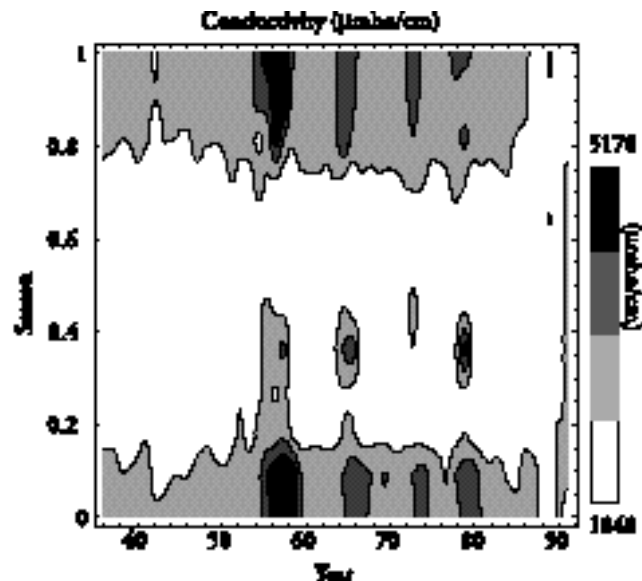
eventually cause degradation of the intermediate aquifers (Walton et al. 1999). This study documents and analyzes the relationship between the water quality, discharge, and source waters of the Rio Grande in the El Paso area.

WATER CHEMISTRY AND DISCHARGE TRENDS ON THE RIO GRANDE

The approximately 60-year record of discharge for the Rio Grande at El Paso is shown in Figure 3. Higher discharges are shown in the darker shades. Several trends emerge from the data. Discharges are generally greatest during the irrigation season when water is released from Elephant Butte and Caballo reservoirs (Figure 2). The irrigation season extends (approximately) from March to October and would have seasonal values of about 0.17 to 0.83 (Figure 3).

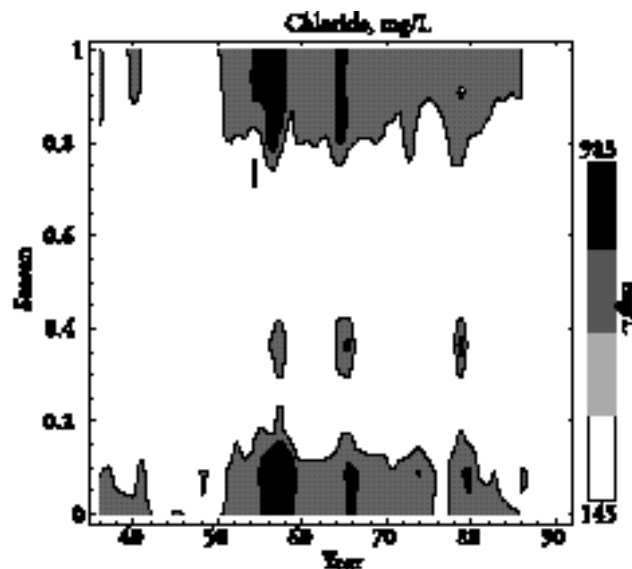
Multiyear variations in precipitation are also visible as vertical light and dark bands. Darker vertical bands represent wetter years, while lighter vertical bands represent drier years. The dark band in the early

Figure 4: Conductivity in the Rio Grande at El Paso by Year and by Season



Note: Conductivity can be used as a surrogate for salinity in the river. Higher conductivity values are related to higher salinity values.

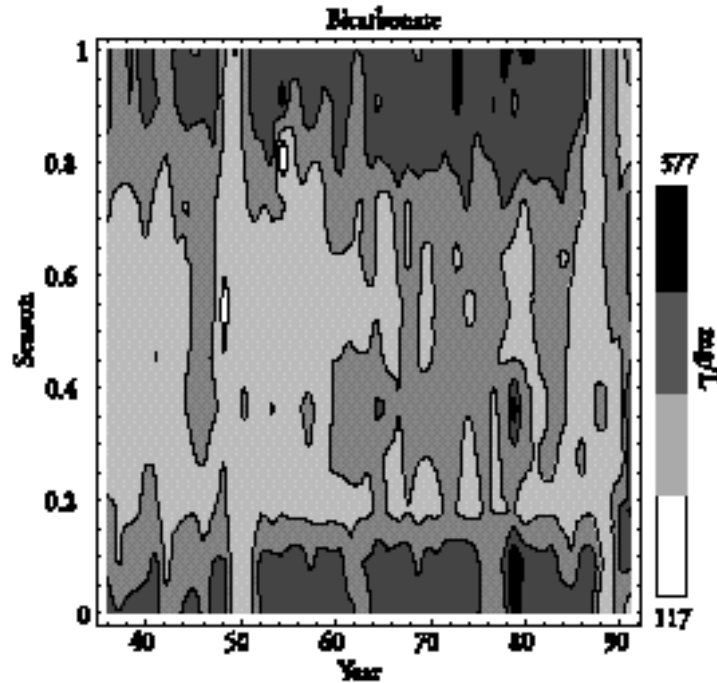
Figure 5: Chloride Concentrations in the Rio Grande at El Paso by Year and by Season



1940s represents a period of high precipitation. A severe drought is evident during the mid- to late-1950s. The droughts and wet periods shown in Figure 3 can be compared with the El Niño temperatures from the southern Pacific Ocean. El Niño temperatures are classified as warm, neutral, and cold and are shown as high, middle, and low values, respectively, as demonstrated by the line at the bottom of Figure 3. The multiyear variations of discharge appear to have complex relationships with the El Niño temperature variations. Some of the wet periods appear to follow transitions from warm to neutral or cold El Niño temperatures. Droughts tend to occur after transition from colder El Niño temperatures. However, exceptions to these patterns are observed.

Conductivity of Rio Grande water is related to the discharge of the river. Figure 4 shows the observed conductivity of the Rio Grande by season and year. Conductivity of the river water increases with the concentration of dissolved salts in the water and, as such, is an analog for salinity. Conductivity is lowest during the irrigation season and increases during the winter months. During the winter, discharge in the river consists predominantly of irrigation return flows, which are dominated by brackish groundwater from the shallow aquifer. Thus, discharge and water quality during most winters represent an integrated picture of the groundwater quantity and quality in the shallow

Figure 6: Bicarbonate in the Rio Grande at El Paso by Year and by Season

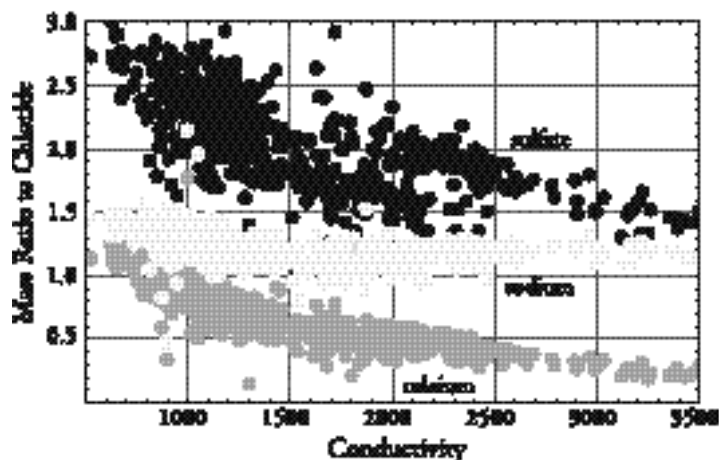


aquifer for the entire region between Elephant Butte Reservoir and El Paso. Most years the salinity of the water in the winter is too high for domestic use. The drought in the 1950s is visible as a darker vertical band and the wet period in the 1940s is visible as a lighter vertical band in Figure 4.

Chloride concentrations behave analogously to conductivity levels in the river. Figure 5 is the observed chloride concentration in Rio Grande water by season and year. Chloride concentrations are higher during the winter months and during droughts when more of the discharge in the river is from the irrigation drains. The drought in the 1950s is evident in Figure 5.

The variation in bicarbonate concentration (HCO_3^-) is more complex (Figure 6). Notice that the variability of bicarbonate is less than the variability of chloride. Seasonal variations in bicarbonate concentration are evident. Bicarbonate levels drop in February and March at the beginning of the irrigation season and increase in September and October at the end of the season. This may indicate a change in water source for the river from reservoir releases associated with irrigation to brackish groundwater discharge from the shallow aquifer.

Figure 7: Ion Mass Ratios ($\text{SO}_4:\text{Cl}$, $\text{Na}:\text{Cl}$, and $\text{Ca}:\text{Cl}$) as a Function of Conductivity



Note: Higher conductivity values indicate periods of lower discharge. Decreases in the ion mass ratio for calcium and sulfate for high conductivity values indicates the precipitation of calcite (CaCO_3) and gypsum

The multiyear variations associated with wet-dry climatic cycles are not as visible in the bicarbonate concentration data, suggesting concentrations are moderated by solubility controls.

Figure 7 plots the observed ion mass ratios for sulfate ($\text{SO}_4:\text{Cl}$), sodium ($\text{Na}:\text{Cl}$), and calcium ($\text{Ca}:\text{Cl}$) to chloride as a function of water conductivity for the recorded period. As discussed above, the higher conductivity waters are associated with periods of low discharge. In Figure 7, the higher conductivity values and, correspondingly, the lower discharge are to the right of the graph. Chloride is used as a tracer of evaporative concentration of the waters because it participates in few chemical reactions. Notice that the ratio of sodium to chloride in the waters is not heavily influenced by the conductivity of the water. In contrast, the ratio of calcium to chloride decreases by a factor of five and the ratio of sulfate to chloride decreases by a factor of two as the conductivity of the water increases in Figure 7.

The water chemistry trends suggest that calcium carbonate is precipitating in fields during periods of low discharge. Precipitation of calcium carbonate (CaCO_3) in concentrated waters (and potentially some dissolution during higher water years) would explain the observed calcium to chloride ratio and the attenuation of variation in bicarbonate ion. The sulfate trend suggests that, to a lesser extent,

precipitation of calcium sulfate as gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) occurs in fields. Precipitation of calcium carbonate and calcium sulfate decreases the amount of calcium in solution without affecting sodium concentrations, thereby increasing the sodium adsorption ratio (Richards 1954), which is a critical factor for the quality of irrigation water. Water with high sodium absorption ratios can cause the breakdown of certain clay minerals.

DISCUSSION

A number of management decisions are likely to influence the quantity and quality of water flowing in the Rio Grande in the future. Population growth upstream from El Paso, in New Mexico and Colorado, will inevitably lead to increased water demands. Even with water reuse and (or perhaps especially with upstream water reuse) the salinity of the water flowing downstream is likely to increase with time.

El Paso is currently considering placing an impermeable lining along the base of some canals from Elephant Butte and Caballo reservoirs to the lower Mesilla Valley. This would provide a year-round source of higher quality surface water to El Paso. However, reducing the canal leakage will tend to increase the salinity of the shallow groundwater and irrigation return flows. This could further degrade the quality of the Rio Grande and eventually damage the quality of the intermediate and deep Mesilla Bolson aquifers.

Another potential water management strategy for the Mesilla Valley, which has not been attempted, would be to apply excess irrigation water during high water years at the end of the growing season. This would tend to periodically flush the salts from the shallow groundwater. The shallow groundwater is the primary source of recharge for the Mesilla Bolson aquifers used extensively for water supply by the city of El Paso. Lowering ion concentrations in the shallow aquifer would serve to protect this valuable resource into the future as well as lower downstream concentrations of ions in the Rio Grande during low water years.

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IV

Quality of Urban Runoff in the Tijuana River Watershed

Richard M. Gersberg, Chris Brown, Victor Zambrano,
Karilyn Worthington, and Daniel Weis

ABSTRACT

A sampling program was conducted to assess the quality of runoff associated with a variety of land uses in the Tijuana River watershed, a binational river basin on the U.S. Mexican border. Generally, metal concentrations in samples collected during the first two to four hours of runoff (early storm) were higher than those in samples collected 24-36 hours into the rain event (late storm). A notable exception to this pattern was observed for the site on Tecate Creek, where levels of cadmium, chromium, copper, and nickel were higher in the late-storm sample. This is possibly due to the point source discharge of wastewater effluent from the Tecate Municipal Wastewater Treatment Plant just one mile upstream. At the industrial site, concentrations of lead and zinc in samples of early-storm runoff fell in the 85th percentile range (80th percentile for copper) of a U.S. industrial runoff dataset (Line et al. 1997). Other urban land-use sites (including residential and commercial) were generally comparable to the 90th percentile values for wet-weather runoff in an urban watershed of Los Angeles County. The resulting data suggest that nonpoint source pollution arising from a variety of land uses in the Tijuana River watershed will continue to enter the Tijuana Estuary and near-shore ocean during wet weather, arguing for basin-wide wastewater and stormwater management in this urban watershed.

INTRODUCTION

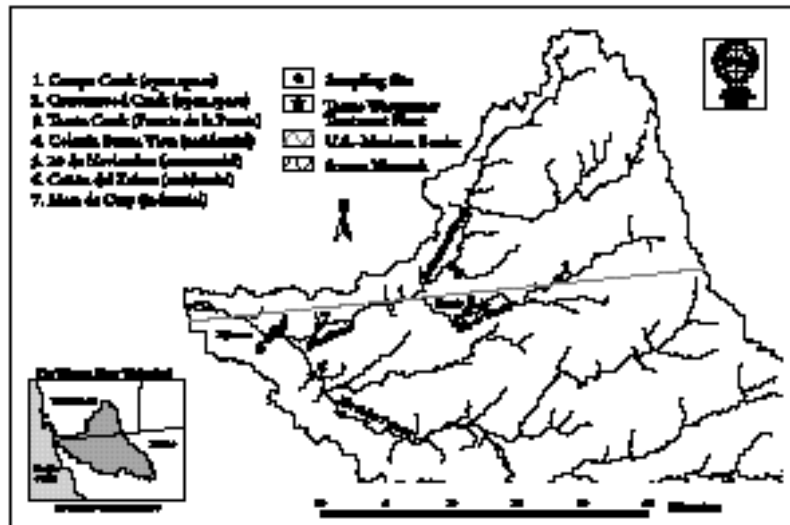
For decades, raw sewage from the city of Tijuana, Mexico, has flowed into the Tijuana River and across the international border into the Tijuana Estuary. This problem has worsened in recent years with the substantial growth of Tijuana's population, along with the intensive industrial development associated with the maquiladora program (in-bond manufacturing and assembly plants) in Mexico. Although discharges from the Tijuana River watershed account for only a small percentage of total gauged runoff to the Southern California coastal ocean, it contains the highest concentrations of suspended solids, Cd, Cu, Ni, Pb, Zn, and PCBs among the eight largest creeks and rivers in Southern California (SCOWRP 1992).

Many of the water quality problems in the Tijuana River watershed are due to diffuse, nonpoint sources of pollution and, as such, may be addressed more effectively along watershed lines. Managing natural resources on a watershed basis offers a geographic context within which the interactions of land, water, and human activity can be monitored, assessed, and understood. Nonpoint pollution processes, such as stormwater runoff, are inherently difficult to model due to the temporal and spatial complexity of pollutant loading and hydrological processes. Such modeling generally requires organizing and processing large amounts of spatially referenced data. To accomplish this, a geographic information system (GIS) may be linked with a hydrological urban runoff model and land-use-specific water quality data to estimate the mass emission of selected anthropogenic pollutants into the Tijuana Estuary. Unfortunately, no data currently exist on the water quality characteristics of runoff in the Tijuana River watershed. The objective of this study was to generate such land-use-specific water quality data for the Mexican portion of the Tijuana River watershed, including industrial, residential, commercial, and open-land sites, as well as Tecate Creek. In addition, we compare the water quality at these sites to that in the main river during rain events and to stormwater quality of similar land-use sites in the United States.

METHODOLOGY

Seven sites were sampled in the Tijuana River watershed: two in the United States (Campo and Cottonwood Creeks), and the remaining sites in Mexico (Figure 1). The specific geographical coordinates for these sites were determined with a global positioning system (GPS) receiver, and were then incorporated into a GIS for further analysis.

Figure 1: Water Quality Sampling Sites



Determination of Land-Use Types within Each Contributing Subbasin

To delineate which land-use types were present within the subbasins upstream from the sampling points, we used modeling algorithms in our GIS software, a digital elevation model (DEM), and the locational data of the sampling sites to generate boundaries of the subbasins. Specifically, the GRID module of the Arc/Info GIS was used to examine a DEM for the Tijuana River basin to model the direction and accumulation of surface water flows for each grid cell. The resulting flow direction and accumulation data formed the basis for a stream network map for the entire basin. Then, each sampling point's location was placed exactly on the stream network and a further series of algorithms was run that incorporated the accumulation data in order to generate the specific boundaries of the contributing subbasins above each of the sampling points. These subbasin boundaries were then overlaid with land-use data for the larger watershed to determine the composition of the land use within each subbasin. The locations of these sites in the watershed are shown in Figure 1 and are described below.

Open-Space Sites

Two sites, which drained mostly rural and undeveloped subbasins, were sampled to characterize this land use. The first site, Campo

The U.S. Mexican Border Environment

Creek in the United States, is just upstream of the city of Tecate, Mexico, and is 88% undeveloped. This reach of the river runs through a sparsely populated, predominantly agricultural rural area and flows into Tecate Creek, a main tributary of the Tijuana River. The other site, on Cottonwood Creek, is located directly under the State Highway 94 bridge and drains a region that is mostly undeveloped (92%), with limited agricultural activities.

Residential Sites

Two sites were sampled to characterize this type of land use. The first, Colonia Buena Vista, lies in an arroyo that drains a fairly large residential area of Tijuana near Otay Mesa. This site is nearly 100% residential land use. The Colonia Buena Vista extends along the length of the channelized Tijuana River and is comprised of two settlements of low-income families. The second site, located in Cañon del Zainas, is an arroyo that drains a large residential area of southwestern Tijuana, and then empties into the Río de las Palmas branch of the river. Residential land use comprises about 30% of the developed land-use area of this urbanizing subbasin.

Industrial Site

This site lies at the foot of Otay Mesa, and drains a rather small subbasin (Otay Mesa) that is almost entirely developed (98.5%) with maquiladoras industrial facilities comprised mostly of assembly and manufacturing plants. Maquiladora plants are foreign-owned facilities that had initially used lower-priced Mexican labor to assemble goods from imported components, however, currently, NAFTA has allowed many of these plants to be full-scale production facilities. Industries located at this site include Nypro, Matsushita, Hitachi, Tabuchi Electronics, Tocabi, Sanyo, Santomi, and Energy Labs.

Commercial Site

This site is on a large stormwater channel that drains a main commercial center close to the Tijuana River. The site is located on Avenida 20 de Noviembre in the colonia (neighborhood) of the same name, at the intersection of several important transit routes: Blvd. Díaz Ordaz, Blvd. Benítez, Blvd. Agua Caliente, Paseo de los Héroes, and Calle F. C. Sonora. Of the developed land-use area at this site, commercial use accounts for about 40% of the total.

Tecate Creek Site

Tecate Creek lies just below the urban area of Tecate, Mexico, (a city of 90,000 inhabitants) and is a major tributary of the Tijuana River.

Quality of Urban Runoff in the Tijuana River Watershed

Tecate has marginal sewage treatment and disposal infrastructure. This site is located approximately one mile downstream from the Tecate Municipal Wastewater Treatment Plant. It drains a very large subbasin of 39,660 hectares, which is 81.5% nondeveloped. The two major land-use categories are agriculture and residential, comprising 9.5% and 3.8%, respectively, of the total land-use area of this sub-basin.

Chemical and Biological Analyses

A total of seven storm events were sampled at Lindbergh Field, San Diego, with the dates of each storm and the total amount of precipitation given below. Three events were sampled during the 1996-1997 rainy season: 21-23 November 1996 (4.3 cm), 15-17 January 1997 (0.53 cm), and 10-12 March 1997 (< 0.25 cm). During the 1997-1998 rainy season, four storm events were sampled: 24-25 September 1997 (2.0 cm), 26-27 November 1997 (1.0 cm), 9-10 January 1998 (2.9 cm), and 14-15 February 1998 (2.9 cm).

Surface water grab samples were taken twice during each storm event once within the first two to four hours of the storm's inception (early-storm samples), and once again at an interval of 24-36 hours after the first sample was taken (late-storm samples). An exception to this regime was the commercial site (September 1997 and November 1997 events) and light industrial site (January 1998 and February 1998 events), which were sampled on four occasions during the above rain events.

All samples were handled, preserved, and analyzed according to the Standard Methods for the Examination of Water and Wastewater (APHA, AWWA, WPCF 1989). Stormwater samples were filtered (for dissolved metal analysis), and the filter was then digested for total metal analysis. Both particulate and dissolved metal concentrations were analyzed by graphite furnace atomic absorption (GFAA) using a Perkin-Elmer SIMAA 6000 AA with Zeeman corrector. The exception was for the analysis of Zn, where flame atomic absorption spectrophotometry (Perkin-Elmer Model 2380) was used because of the relatively high levels of this metal. Quality assurance/quality control (QA/QC) for metals analysis included duplicate analyses, blanks, and standard additions to stormwater samples consisting of both dissolved metal spikes, and particulate metal spikes (in the form of a soil standard addition). Statistical analyses comparing metal levels among land-use sites and for early- versus late-storm samples were done using the Student's T-test.

Figure 2: Cd Concentrations for Early- and Late-Storm Events

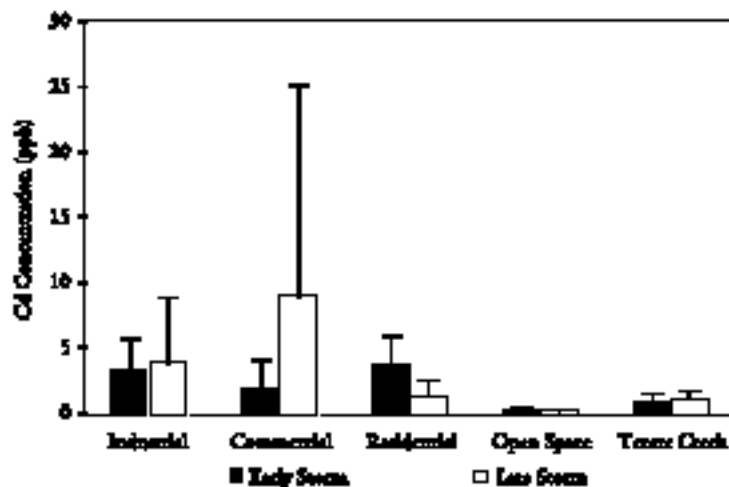
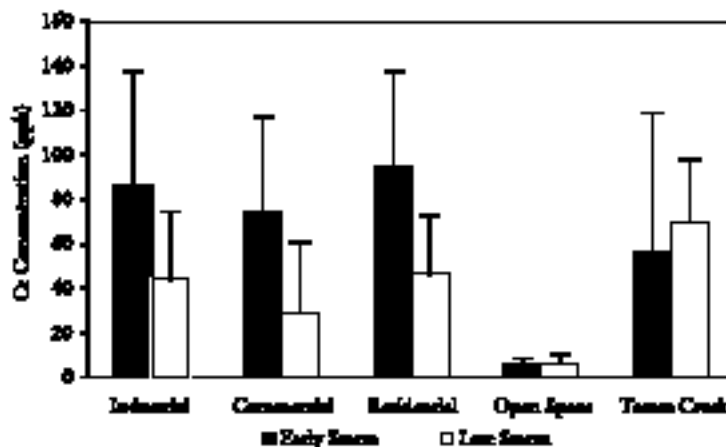


Figure 3: Cr Concentrations for Early- and Late-Storm Events



RESULTS AND DISCUSSION

This study was conducted to characterize the metal levels in stormwater runoff from a variety of land-use sites in the Tijuana River watershed. During the 1996-1997 rainy season, precipitation was relatively light, with only 17.5 cm recorded at Lindbergh Field, San Diego, from 1 October 1996 through 31 May 1997. From 1 October 1997 through 31 May 1998, precipitation was relatively high, with 42.7 cm recorded. The precipitation for each storm event sampled is

Figure 4: Cu Concentrations for Early- and Late-Storm Events

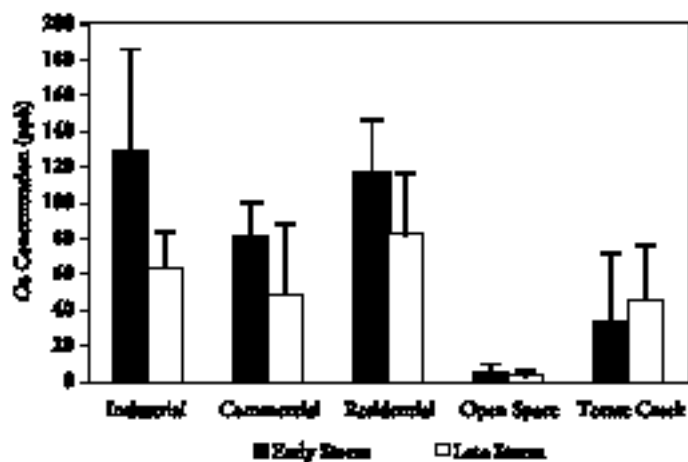
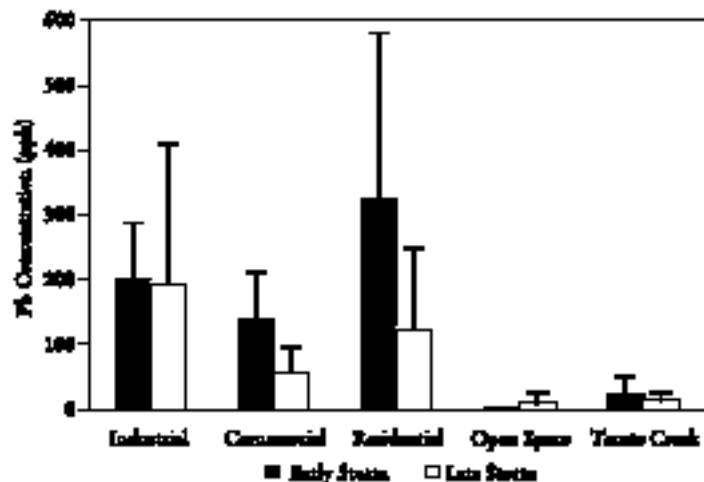


Figure 5: Pb Concentrations for Early- and Late-Storm Events



given in the methodology section. Due to logistical constraints associated with sampling in Mexico, there were no measurements made of stream flow volume when samples were taken.

Mean concentrations ($n = 7$ for industrial and residential sites, and $n = 5$ for residential and open-space sites) of total metals for early- and late-storm events are shown in Figures 2-7. Metal concentrations in stormwaters of the Mexican portion of the Tijuana River watershed vary considerably among the following circumstances: during the course of a storm event, from event to event at a given

Figure 6: Ni Concentrations for Early- and Late-Storm Events

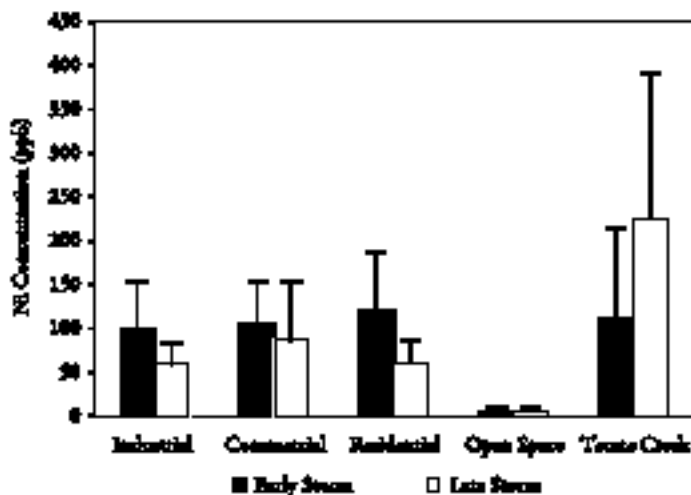
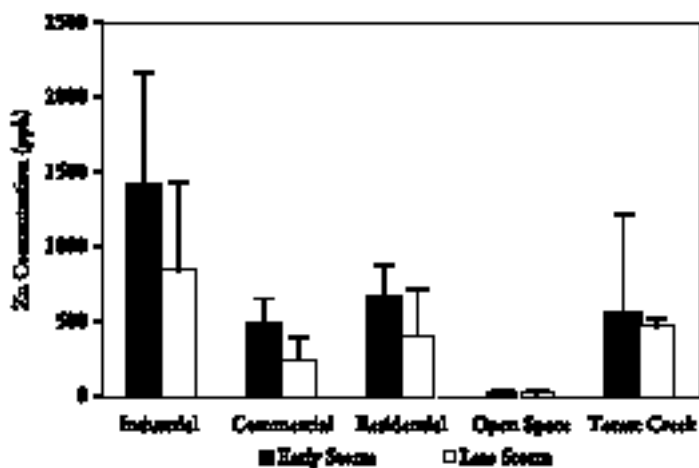


Figure 7: Zn Concentrations for Early- and Late-Storm Events



site, and from site to site depending on land use. Such differences in concentrations are a result of variations in rainfall characteristics, differing watershed features that affect runoff quantity and quality, and variability in urban activities. Generally, the early-storm metal concentrations for the sites in the Tijuana River watershed were higher than the late-storm values, this pattern reflecting what is generally termed the first-flush phenomenon. Levels of Cd, Cr, and Ni found in

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the residential sites were significantly higher ($p < 0.05$) in early-storm samples as opposed to late-storm samples. Additionally, for the industrial site, levels of Cu in the early-storm samples were significantly higher ($p < 0.05$) than in the late-storm samples. However, due to the relatively small number of samples (and the high variation of metal levels within a given storm event), other temporal differences were not found to be statistically significant.

The term first flush was coined to describe the phenomenon whereby high pollutant concentrations are observed during the initial stages of a runoff event, decreasing as the event progresses. The variation in quality during an event can be explained by relating it to the transport capacity of the runoff flow. The potential capacity to transport pollutants is significantly reduced once the hydrograph peak is reached. The result is that a significant fraction of the mass load of an insoluble pollutant can be removed by a relatively small fraction of the total runoff volume (Griffin et al. 1989). A notable exception to this

Table 1: Toxic Heavy Metal Concentrations (mg/L) for Nationwide Urban Runoff Program and Tijuana River Watershed

Land Use	NURP			Tijuana River		
	Total Cu	Total Pb	Total Zn	Total Cu	Total Pb	Total Zn
Industrial	0.072	0.113	0.494	0.099	0.197	1.158
Residential	0.095	0.159	0.598	0.101	0.231	0.549
Commercial	0.072	0.113	0.494	0.063	0.094	0.343
Open Space	0.055	0.140	0.498	0.061	0.066	0.016

*NURP values are 90th percentile event mean concentrations (U.S. EPA 1983).

first-flush pattern was observed for Tecate Creek, where total levels of Cd, Cu, Cr, and Ni in our late-storm samples were higher than for the early-storm samples. For Ni, the late-storm sample on Tecate Creek showed the highest value (226 ppb) among all of the sites (Figure 6). The presence of continuously elevated levels of these toxic heavy metals in Tecate Creek, a major tributary of the Tijuana River, is surprising due to the rather undeveloped nature of the watershed, but is probably due to the point source discharge from the Tecate Municipal Wastewater Treatment Plant just upstream of our sampling point. This fact, coupled with our finding of rather continuous loading of metals at this site (with no observable first-flush effect) suggests that Tecate Creek is a significant contributor to metal loading in the Lower Tijuana River watershed and estuary. In addition, the site on Tecate Creek consistently showed the highest level of total

dissolved solids (TDS) among all of the sites tested, with a mean TDS concentration of 1300 mg/L as compared to 782 mg/L, 752 mg/L, and 297 mg/L for open-space, residential, and industrial sites, respectively. Unfortunately, major anions and cations were not measured in this study.

To the best of our knowledge, the present study represents the first time that data have been published on the quality of stormwaters in the Mexican portion of the Tijuana River watershed. The EPA's Nationwide Urban Runoff Program (NURP) identified toxic metals as the most prevalent priority pollutants found in urban runoff in the United States (U.S. EPA 1983). The toxic metals lead, copper, and zinc were identified in 91% of the samples. Other inorganic pollutants detected were arsenic, chromium, cadmium, and nickel (U.S. EPA 1983). For the toxic heavy metals Pb, Zn, and Cu, the 90th percentile concentrations of the NURP data for specific land-use categories are shown in Table 1.

The mean levels of each metal for each land use in the Tijuana River watershed, obtained by averaging the early- and late-storm values for all of the rain events, are also shown in Table 1. While storm event sampling is not entirely equivalent to the event mean concentrations (EMC) given by the NURP data, it does provide a useful benchmark for comparison purposes. When mean values for our early-storm and late-storm events (Table 1) are compared to the 90th percentile NURP values, there was a general agreement (within a factor of 2.5) among all land uses except open space.

Comparing metal levels between land-use types, the industrial land-use site showed the highest level for Zn only, with a mean level

Table 2: Percentage of Total Metals in Stormwater Runoff
Associated with Filterable Particulate Matter

Metal	Industrial		Commercial		Residential		Open Space	
	Early	Late	Early	Late	Early	Late	Early	Late
Cadmium	67.5	71.1	63.1	55.9	86.3	82.3	33.8	33.6
Chromium	95.9	94.2	95.8	90.8	95.9	95.1	98.1	94.8
Copper	73.5	68.1	69.9	57.7	88.5	65.5	88.8	96.9
Lead	89.3	85.3	82.3	82.6	95.7	96.4	20.6	38.3
Nickel	86.3	81.6	96.8	96.8	67.5	91.4	33.2	76.8
Zinc	83.6	73.9	82.5	88.6	86.4	89.2	42.7	67.9

of 1150 ppb. Zn levels at the industrial site were significantly higher ($p < 0.05$) than at any of the other sites tested. However, for Cr, Cu, and Pb, the residential land use showed the highest levels, with mean levels of 73 ppb, 101 ppb, and 231 ppb, respectively. This is somewhat surprising since there are a number of electronics firms at the

industrial site that might be expected to be significant sources of metals such as copper. When metal levels were statistically compared among the land-use sites, only open space was significantly different ($p < 0.05$) from all other land-use sites, for all metals tested. For the open-space sites, the levels of Pb and Zn in the Tijuana River watershed were less than 5% ($< 10\%$ for Cu) of the 90th percentile NURP values. The reason for this significant discrepancy is unknown, but it does suggest that the relative contribution to the loading of metals from the nonurbanized area of Tijuana is probably even less than one might surmise from NURP data alone.

In order to lend additional perspective to the data, in an attempt to assess the degree to which the lack of an industrial pretreatment program in Mexico influences the quality of stormwaters, we can compare these data to that of Line et al. (1997), which measured the water quality of first-flush runoff from 29 industrial sites in the United States. These authors found mean levels of Pb, Zn, and Cu were 82 ppb, 593 ppb, and 116 ppb, respectively. Corresponding levels of these same metals in our early-storm runoff samples for the industrial land-use site in Mexico were 199 ppb, 1412 ppb, and 129 ppb, and all were in the 80th percentile (or above) of the U.S. industrial runoff dataset.

Table 2 shows the percent of heavy metals associated with the filterable particulate fraction of the stormwater runoff for industrial, urban, and open-space land uses. It can be seen that the values for the particulate-associated fraction are rather high for all the metals at most of the sites. Only at the open-space site does the percentage of particulate-associated metals fall below 50% for Pb and Zn. For Cd, the early-storm value for the commercial land use is also below 50%. The high percentage of particulate-associated metals implies that management strategies that involve erosion control in the upper watershed may also function to reduce metal contamination of the downstream estuary.

Placchi (1998) measured metal levels under base flow conditions during dry weather from 30 April 1997 through 25 June 1997 in the Upper Tijuana River watershed. The open-space and industrial sites investigated in Placchi's study were identical to the wet-weather sampling sites of the present study. Placchi found concentrations of Cu, Pb, and Zn were 0.015 mg/L, 0.009 mg/L, and 0.082 mg/L, respectively, at the open-space sites during base flow conditions. In contrast, our wet-weather values (Table 1) were markedly lower, particularly for Cu (0.004 mg/L) and Zn (0.016 mg/L). This pattern was attributed to rather constant, low-level sources of these pollutants in these subbasins, which were then diluted by runoff during rain

events. A similar pattern was shown by Cu at the industrial site, where the concentration during dry weather flow was 50% higher than during a rain event. However, at the same site, both Pb and Zn levels were significantly elevated for wet-weather as opposed to dry-weather flows (30-fold for Pb and 7-fold for Zn).

Figure 8: Storm Event, 7-8 December 1997

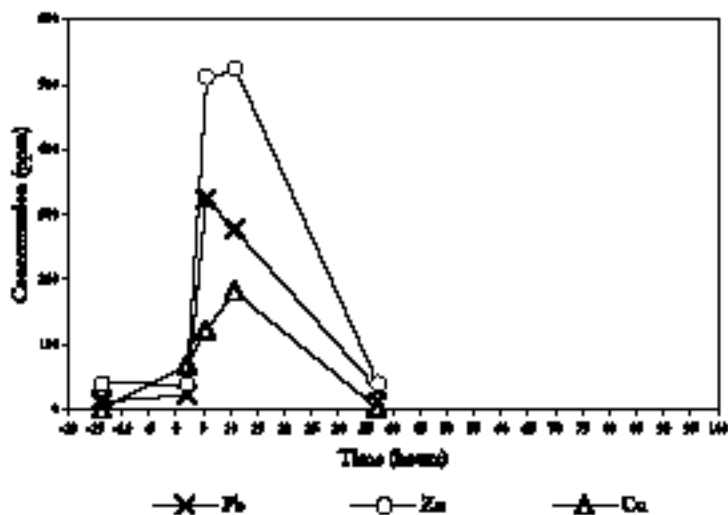


Figure 9: Storm Event, 24-27 March 1998

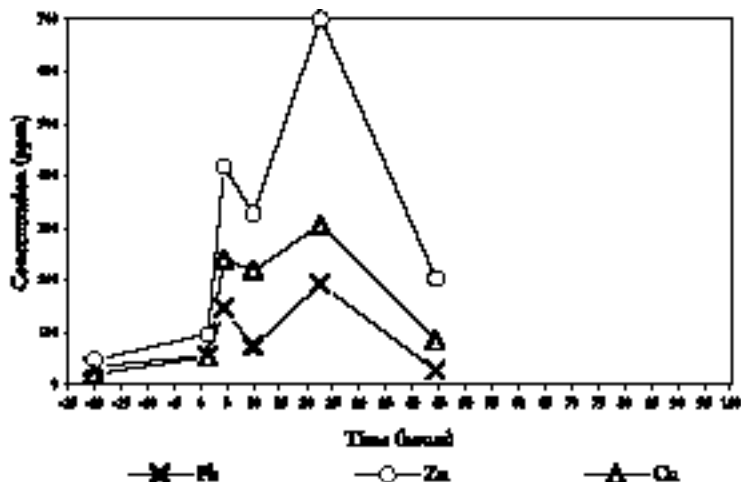
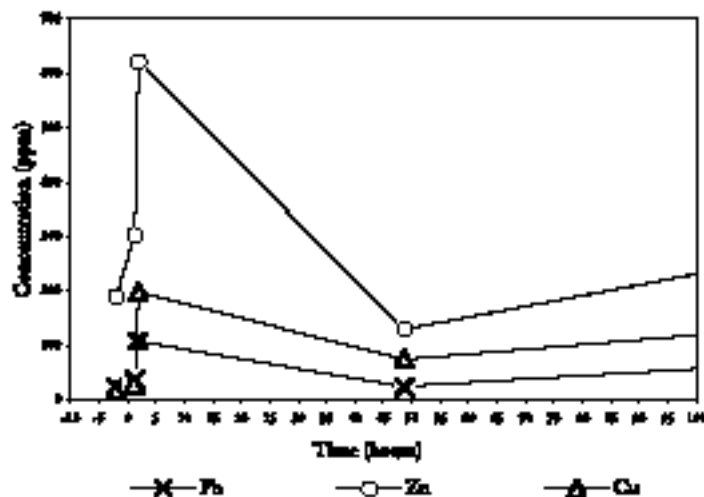


Figure 10: Storm Event, 11-15 April 1998



In order to estimate metal loadings to the downstream estuary, a land-use runoff model was developed using a GIS database for the Tijuana River watershed, coupled with an empirical runoff model. Several assumptions were made for this modeling effort. First, since water quality data were only available for four land-use classifications in the Mexican portion of the watershed, the land-use database was reclassified with all land-use types (and areas) being assigned to either open-space, residential, commercial, or industrial designations. Second, since annual precipitation was not available for the Mexican portion of the watershed, we used the annual precipitation values for Lindbergh Field in San Diego for our regional estimate. Third, since much of the runoff in the Tijuana River watershed is impounded by reservoirs, only four subbasins were considered as contributing to the downstream flow of the estuary. These subbasins are the Rio Tijuana (24,708 ha, encompassing much of the urban city of Tijuana), Lower Cottonwood Creek (35,147 ha), Campo Creek (43,159 ha), and El Florido (28,343 ha). Using land-use specific runoff coefficients (Driscoll et al. 1990), we could then calculate the volume of runoff from each land-use area of each of these four subbasins, which, when multiplied by our mean land-use-specific metal concentration, yielded an estimate of the metal loading from the watershed on an annual basis.

The results of this analysis for the two years from October 1996 through May 1998 (mean annual precipitation of 30.1 cm) yields modeled values for annual metal loading of 6,660 kg/yr for Pb, 3,140 kg/yr

for Cu, and 22,740 kg/yr for Zn. From September 1986 through August 1988 (mean annual precipitation of 39.6 cm) the Southern California Coastal Water Research Project measured the metal loading from the Tijuana River (SCCWRP 1992). They found that annual loads of Pb, Cu, and Zn were 24,867 kg/yr, 10,468 kg/yr, and 28,964 kg/yr, respectively. Comparison of our modeled estimates with their measured data shows that while both estimates of annual loading for Zn are relatively similar (within about 20%), in the case of Pb and Cu our model underpredicts SCCWRP's measured loading values by nearly 70%. For Pb, this discrepancy may be rather easily explained by the phase-out of leaded gasoline that occurred in the United States in the early 1980s and was subsequently initiated in Mexico in 1991. Apparently, concentrations of Pb in runoff have declined markedly since the late 1980s when SCCWRP did their study. Indeed, the mean Pb levels for all of the urban land uses that we investigated were in the range of 0.98-0.231 mg/L, while SCCWRP's flow-weighted mean lead concentration was 0.988 mg/L for the Tijuana River.

In order to assess the nature of the temporal variations in water quality, sampling was conducted at Dairy Mart Road during three storm events: 7-8 December 1997, 24-27 March 1998, and 11-15 April 1998. This process determined how the levels of total metals varied as a function of flow throughout the storm events. Figures 8-10 show the pattern of change throughout the three storm events for Pb, Zn, and Cu. For all the metals tested, there is a pronounced first-flush phenomenon (paralleling what we observed at most of the individual land-use sites), with metal levels typically increasing markedly (compared to base flow level) after the onset of the rain event to some peak level, and then decreasing thereafter to near the original baseline level.

Since the Tijuana River watershed is a semiarid environment, it is useful to compare our results to those of similar coastal chaparral basins of Southern California. A study on water quality under wet-weather conditions at three sites in Ballona Creek (which drains into Santa Monica Bay) showed that the 90th percentile values for Pb, Zn, and Cu were 1,329 ppb, 2,055 ppb, and 247 ppb, respectively (Stenstrom and Strecker 1993). With a minor exception (Cu during storm two), these values for Ballona Creek were higher than levels we measured for the Tijuana River. The fact that these values for an urban environment in coastal Los Angeles County generally exceeded those for our sites in the Tijuana River watershed suggests that stormwater contamination is ubiquitous in urban environments.

Quality of Urban Runoff in the Tijuana River Watershed

even where industrial pretreatment and stormwater permitting regulations are in place.

CONCLUSIONS

This study represents the first published data on the quality of stormwater in the Mexican portion of the Tijuana River watershed. Results show that levels of Cu, Pb, and Zn at both industrial and urban land-use sites in the Tijuana watershed were generally comparable to the 90th percentile NURP values. Levels in the main Tijuana River were not significantly elevated above those in urban wet-weather flows in Ballona Creek in Los Angeles County. The data document the effect of the discharge from the Tecate Municipal Wastewater Treatment Plant on metal levels in Tecate Creek, and suggest that this major tributary to the Tijuana River is a significant source of pollutants in the watershed. Nonpoint source pollution arising from a variety of land uses in the watershed will continue to contaminate the Tijuana Estuary and near-shore coastal ocean during wet weather and, therefore, highlights the need for comprehensive wastewater and stormwater management in the urbanized portions of the watershed.

ACKNOWLEDGEMENTS

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V

Sustainable Use of Wastewater for Small Communities: A Model System for Short Rotation Woody Crop Pro- duction

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ABSTRACT

The U.S. Mexican border region has experienced rapid population growth in the last 30 years, resulting in natural resource degradation and increasing threats to public health. A primary concern is the threat posed by water pollution, especially through improper treatment and disposal of human wastes. Binational efforts are underway to find solutions, particularly through the development of alternatives to high-cost, conventional waste treatment systems. Alternative systems should reduce contaminants as well as innovatively recycle nutrients and carbon present in the waste materials. This pilot study, which is a full integration of applied research and technology transfer, demonstrates a wastewater land application system integrated with the production of fast growing trees for fiber and energy. In the first year of this study, trees reached an average height of 1.78 m and ground line diameter of 21.8 mm, with an average survival of 88%. The installation of a full-scale system of this type is economically justified based on cost-benefit analysis. This pilot study could serve as a model system for effective waste treatment in other communities in the border area.

INTRODUCTION

The Rio Grande, or Rio Bravo, flows nearly 2,000 kilometers through the Texas-Mexico border region. Paradoxically, it is both the region's lifeblood and a source of environmental contamination. Communities depend on the river for drinking water, farming, industry, and recreation. Yet, decades of rapid population growth, a lack of infrastructure, and poor environmental management have led to an escalating contamination risk. Thus, the once-clean river is now polluted with industrial organic compounds, heavy metals, sewage waste, agricultural run-off and pesticides, and high levels of salts and sediments (Sharp 1998).

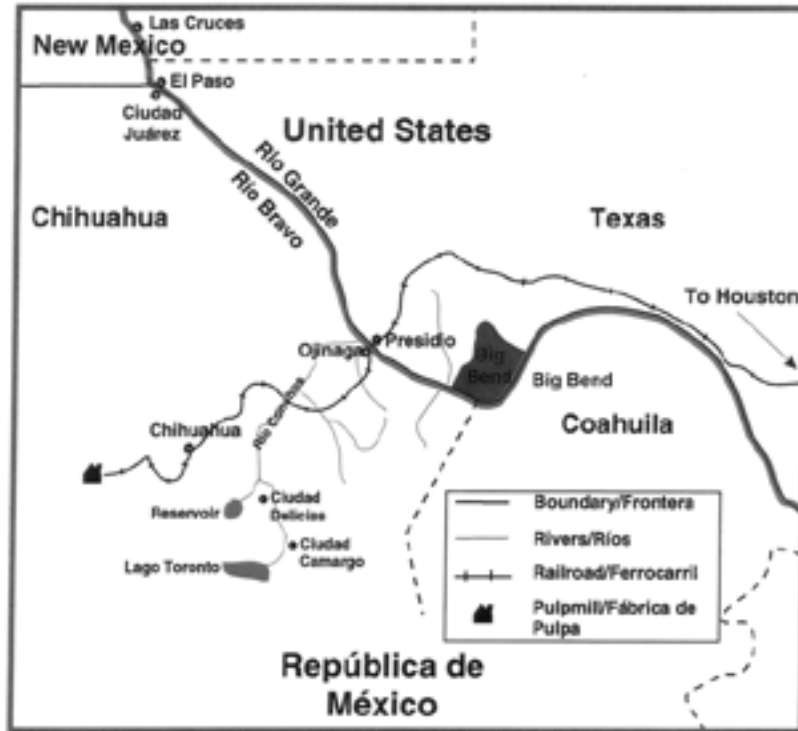
Binational efforts to address these environmental problems were initiated in 1983, with the establishment of the La Paz Agreement. The EPA followed in 1996 with the Border XXI Program an extensive plan designed to decentralize the management of environmental issues in order to increase public participation and to encourage better communication and collaboration among pertinent agencies (U.S. EPA 1996). The program's Border Framework Document identified several key areas of concern, but one of the principal areas was the alleviation of water pollution by developing and rehabilitating infrastructure for drinking water, wastewater collection and wastewater treatment (U.S. EPA 1996).

Many cities in the border region have wastewater treatment systems that provide only minimal treatment or are inadequate to handle the large amounts of sludge and wastewater generated. Furthermore, some communities lack a treatment system altogether. Conventional wastewater treatment systems, while effective, are generally too expensive to install and maintain for many small communities. Instead, innovative, low-cost systems are needed as viable alternatives to the current lack of effective waste treatment in the border communities. These alternatives must not only be relatively inexpensive and capable of safely recycling sludge and wastewater, but they should also provide opportunities for economic development (Bastian et al. 1992). This chapter explores one alternative wastewater treatment system being studied in Ojinaga, Mexico.

PROFILE OF OJINAGA, CHIHUAHUA

Ojinaga, located on the West Texas Mexico border, is situated at the confluence of the Rio Grande and Rio Conchos, about 500 kilometers southeast of El Paso Ciudad Juárez (Figure 1). Ojinaga, with a

Figure 1: West Texas Mexico Border Area along the Rio Grande



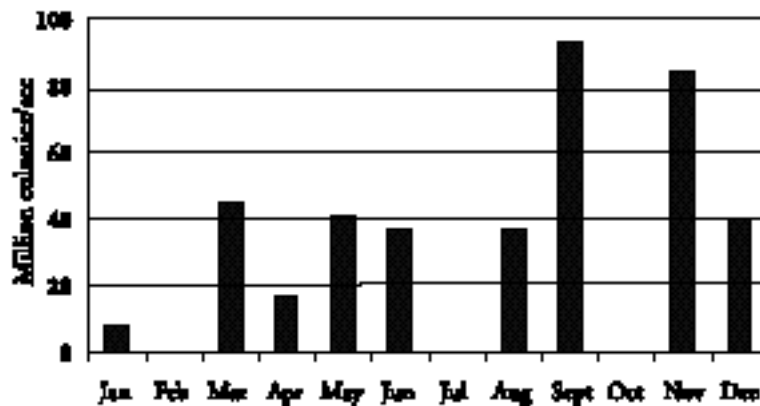
population of approximately 24,000, is located across the river from its sister city, Presidio, Texas, (population 3,500). Unlike other border communities, the population of Ojinaga has decreased, dropping from about 26,000 in 1980 to 23,600 in 1995 (U.S. EPA 1996). This decline has been attributed to a lack of economic opportunities in the community, small landholding size, marginal farmland, flooding, and migration to the United States (Prieto Barrera 1995; Nuñez 1997).

Ojinaga's climate is hot (with a maximum temperature of 50°C, and a minimum temperature of 10°C) and dry (the average annual rainfall is 235 mm), yet the community has an irrigation infrastructure that supports approximately 12,000 ha of agricultural land. Although Ojinaga produces an array of crops, including alfalfa, cotton, corn, wheat, melons, onions, pecans, and forages, less than one-half of the farmland is currently in production. This is due, in part, to small landholdings, averaging 5 ha, making agricultural production uncompetitive (Nuñez 1997). Furthermore, poor farm management has resulted in soil salinization, creating unsuitable conditions for the economical production of agronomic crops on some lands (CNA 1998).

Unlike other border communities, industry does not play a major role in Ojinaga's economy. There are few maquiladoras, which contribute less than agriculture in terms of income and employment opportunities in the community. More importantly, these maquiladoras are not a point source for contamination to either groundwater or river systems (Pando 1996).

In terms of infrastructure, Ojinaga currently lacks an efficient waste treatment system. For over 30 years, the municipal sewage had been piped into a 1.5 ha (45,000 m³) unlined, anaerobic settling lagoon. The lagoon separated the solids from the waste stream and provided some reduction in waste strength. After the lagoon filled with settled solids, a new 2 ha (60,000 m³) anaerobic lagoon was constructed in 1995. Currently, about one-half of Ojinaga households are connected to the municipal wastewater system. The Junta Municipal de Agua y Saneamiento (JMAS) hopes to have 95% of the households connected to the system within a few years. Wastewater from Ojinaga is almost exclusively domestic in origin, and current flow rates are expected to double from 70 L/s to 150 L/s when the entire community is connected to the system (Flores 1994). With these projections, it is anticipated that the new lagoon will fill with collected solids within five years. These solids may have utility as a soil amendment if the organic content is high and the salt content is not deleterious.

Figure 2: Average Monthly *E. coli* Contamination of the Rio Grande below Ojinaga Effluent Discharge Point, Adjusted for River Flow (taken January 1996 through June 1998)



Source: Waggoner (1998).

Presently, effluent from the sewage lagoon is used for irrigation of adjacent pasturage, with the excess water discharged into the Rio Grande. Measurements of Rio Grande water quality near Ojinaga indicate that even this relatively small discharge of effluent into the river significantly increases fecal coliform levels. Water flow below Ojinaga averaged $7.9 \text{ m}^3/\text{s}$ from January 1996 through June 1998 (Waggoner 1998). Over this time period, the community effluent ($0.07 \text{ m}^3/\text{s}$) represented less than 1% of total flow (min. = 0.3%; max. = 4.0%). While the contribution from the effluent to river flow is negligible, fecal coliform contamination increased from an average of 110 colonies/100 mL (se = 28) above the Rio Conchos to 579 colonies/100 mL (se = 101) below the Rio Conchos (Waggoner 1998). Correcting for differences in flow over the course of the year indicates high contamination in the fall when flow is moderate, but irrigation demands are low (Figure 2). During this sampling period, the fecal coliform counts exceeded the Texas Natural Resources Conservation Commission (TNRCC) river quality standards (200 colonies/100 mL) over 50% of the time compared to less than 20% above the effluent outlet. This contamination represents a serious health hazard for downstream irrigation water users, or anyone coming in contact with river water, including children who play or fish in the river. More importantly, fecal coliform contamination indicates the likelihood that more serious biological hazards, such as hepatitis or cholera, may be present.

In order to address the wastewater situation in Ojinaga, a Search Conference and Participative Design (SCPD) workshop was held in Ojinaga in May 1995. Participative design emphasizes specific organizational principles and community-based, participative, democratic processes as the keys to sustainable human and natural resources development. This methodology asserts that projects imposed upon communities, or those that simply assume community support of the project, are destined to fail because members of the community have played little or no part in the project's design. This emphasis on self-determination is critical to the realization of technologically appropriate, nondependent, and sustainable development of human and natural resources (Cabana et al. 1995). Thus, through a community referencing system, members of the Ojinaga community, along with other organizations concerned about Ojinaga's future development, were invited to attend the SCPD workshop.

Within a historical, social, economic, and environmental context, the participants identified some key needs, challenges, and possible prescriptions for solving the wastewater problem. These were:

1. Given the economic conditions of the community, the future wastewater system must have low capital and operating costs; must help to revitalize the economy of the community, and must be able to generate revenues to repay loans and investors.
2. The system must be technologically appropriate and easy to maintain by the community, rather than a costly, high-tech, conventional waste treatment facility that would be an economic and maintenance burden.
3. The system should improve the quality of any water discharged into the Rio Grande and meet the environmental standards of both Mexico and the United States.

Given these criteria, a land application wastewater treatment system that reused the water for pulpwood production was favorably received. Furthermore, the community committed to work toward bringing such a proposal to reality. Since the conclusion of the workshop, the community has collaborated with a binational, multidisciplinary team consisting of Ojinaga community leaders and experts from both Mexican and U.S. agencies (Lujan 1996; Prieto Barrera 1995). With full community support, a pilot study was initiated in 1996 to integrate pulpwood production with wastewater remediation and economic development.

LAND APPLICATION OF WASTEWATER

Land application of municipal wastewater and sludge for remediation, coupled with nutrient and organic matter recycling by vegetation, is not a new concept, and has been practiced in different countries, including Australia, the United States, and Israel. Solids and wastewater have been applied to forest plantations, disturbed lands such as mine spoil sites, edible and nonedible crops, rangelands, and recreational areas, including parks and golf courses (Sopper and Kardos 1973; Sopper et al. 1982; Bastian and Ryan 1986; Cole et al. 1986; Luecke and de la Parra 1994; Myers and Polglase 1996).

Land application systems include various designs, such as the application of wastes to the soil surface using Slow Rate, Rapid Infiltration, and Overland Flow treatment systems, and to the subsurface, using leaching fields and absorption beds (WPCF 1990). Site characteristics, such as soil properties, ground topography (slope and relief), local hydrology, groundwater depth and quality, land use, climatic factors (temperature, precipitation, evapotranspiration, wind, and length of growing season), and expected waste loading rates, as well as consideration of possible social and economic constraints determine the suitability of a particular system (Reed and Crites

1984; WPCF 1990). The land application concept should be distinguished from water reuse, where wastewater is reused after complete multistep treatment. In land application systems, the application of wastewater to the land is an integral part of the waste treatment system, occurring after minimal upstream pretreatment.

The underlying principle of land treatment systems is that the soil environment treats and remediates applied wastes through dynamic physical, chemical, and biological processes (Zasoski and Edmonds 1986). Physically, the soil acts as a buffer between wastewater/sludge particulates and surface and ground water systems. As the wastewater infiltrates the soil profile, waste particles are trapped by the soil. Managing the quantity and frequency of waste loading permits adequate drying, thereby avoiding pooling and soil clogging, which result in anaerobic conditions (Thomas 1973). This system may be particularly well suited to arid and semiarid lands where rainfall is less likely to interfere with land application schemes.

The chemical nature of the soil environment is critical for the reactions necessary for waste remediation. Soil colloids and organic matter adsorb and exchange ions present in the soil water solution. When waste is applied to soils higher in colloids and organic matter, the soil acts as a chemical filter by removing ions from the soil water solution. This feature is especially attractive in land treatment systems used in conjunction with cropping systems, since nutrients are assimilated for plant production (Ellis 1973).

Biologically, the soil-plant system is a reservoir of diverse microorganisms that thrive and multiply under favorable environmental conditions (specific pH, temperature, moisture and oxygen levels, and adequate energy source). Applying organic matter at controlled rates, coupled with a favorable environment, results in increased microbial activity and subsequent decomposition of organic compounds in the waste.

Like other microbes, the survival of human pathogens is a function of the complete soil environment (Foster and Engelbrecht 1973; Reddy et al. 1981). Similarly, antagonistic microorganisms, which occupy specific niches within the soil system, utilize a variety of mechanisms to effectively compete with introduced microbes. While there is the potential for harmful organisms to persist in the soil, the physical and biological processes are not conducive to the long-term survival of these pathogens. Generally, over 90% of the pathogen population die within 30 days of application to soil (Smith 1996). The overall effect of microbial activity in land application systems is that the indigenous microbes facilitate the recycling and transformation of wastewater constituents without extraordinary measures (Miller

1973). This means that wastewater can be applied to soil without prior disinfection with chlorine and still be safe for humans with minimal precautions.

Trees play a useful role in a remediation program. Biologically, tree roots support a variety of organisms that decompose organic matter and absorb and metabolize nutrients. Furthermore, trees require less maintenance than other crops, thereby reducing the health risks to humans. Economically, the wood produced can be sold on the open market or used in the community to sustain the project. The selection of tree species should be based on the anticipated nitrogen loading rates as well as the water-use requirements. For example, in semi-arid climates, trees tolerant of high evaporative demands, yet capable of high nitrogen utilization, would be preferred.

ENVIRONMENTAL, ECONOMIC, AND SOCIAL BENEFITS OF LAND APPLICATION SYSTEMS

Land Restoration

Currently, there are over 2,000 ha of irrigable farmland in the Ojinaga region that have been removed from production because of high soil salinity. This land is unsuitable for economic production of most agronomic or horticultural crops. However, the land may be suitable for production of woody crops that are less sensitive to salt than agronomic crops. This would restore the land to beneficial use, and after one or two rotations (about 14 years) with good management, the land would likely be suitable for agronomic crop production again, if desired.

Clean Water (70/70 20/20)

Currently, Mexico requires a 70 mg/L BOD and 70 mg/L total suspended solids (TSS) standard, while the United States requires a 20/20 standard for wastewater treatment. A system that returns no water to the Rio Grande, other than through groundwater infiltration, would not have to consider this guideline. The loss of flow to the Rio Grande with the land application system would be minor (less than 1%), and a greater benefit would be the tremendous reduction in biological contamination. Organisms, such as fecal coliforms, viruses, and parasites, would be confined to the land where they would pose little threat to surface water systems. Furthermore, system design would ensure worker protection standards equal to or exceeding federal guidelines.

Reduction of Demand on Natural Forest Systems

There are over 7.6 million ha of forest land in the state of Chihuahua, of which over 3.7 million ha are classified as commercial forests. In 1998, the forest communities were permitted to cut 2.4 million m³, or 70% of the mean annual increment of 0.9 m³/ha/yr for the state (Iglesias 1997). It is projected that fiber plantations in Ojinaga could produce nearly 29 m³/ha/yr or 200 m³/ha with a seven-year rotation. Harvesting only 100 ha/yr of short rotation woody crops would meet 1% of the entire state's fiber demand. In addition to providing a sustainable, economical approach to waste handling, the wastewater treatment project could serve as a catalyst for the development of a tree farm cooperative in the Ojinaga community and surrounding areas. A successful cooperative could conceivably bring much of the abandoned agricultural land in Ojinaga back into production, and could replace a significant portion of the wood harvested in the Sierra Madre mountains of Chihuahua. The production of wood fiber in Ojinaga could also provide valuable time for the ecological restoration of a region long exploited for its timber. Furthermore, this land could be used continuously on a sustainable basis, further reducing logging pressures to natural forest systems.

Efficient Water Use and High Productivity

Wastewater can be utilized more efficiently in arid climates as a result of high evapotranspiration and solar energy rates available for plant growth. Additionally, the low rainfall reduces the uncontrolled water loss and drainage problems that are associated with humid regions. The combination of high evaporative demand and low rainfall leads to more efficient use of the nutrient-enriched wastewater for crop production.

Better Control in Nutrient Management

Because there is little rainfall in arid climates, the applied water is practically the only source of irrigation. This should minimize excessive loss of nutrients, due to uncontrolled runoff or deep percolation. Furthermore, crop water consumption can be estimated and water applications managed, such that the available water and its nutrients are efficiently used by the plant. However, even in a tree farm production system, it is imperative that soil salinity be managed to prevent further salinization or loss of production. This can be accomplished through proper irrigation scheduling and leaching of excess salts below the root zone of the trees.

Sustainable Rural Development

Currently, between 20-40% of the land irrigated by the Ojinaga water district is no longer under cultivation due to salinization. By using species of pulp trees with high salt tolerance (e.g., *Eucalyptus* spp.), it should be possible to reclaim much of this land. By bringing land into production not currently under cultivation, the development of a pulpwood plantation would create much needed jobs. Approximately the equivalent of one full-time job would be created for each 2 ha of land returned to production (Reiche et al. 1991). Moreover, by adopting technology appropriate to the local economy, such development should be sustainable in the long run. The jobs created would be skilled to semi-skilled, and would include field work jobs (such as tree planting) as well as more skilled harvesting jobs (e.g., chain saw operators). Furthermore, many jobs would be outdoors, which is attractive to some people, and would pay competitive wages.

Model System for Arid Regions of the World

Many small communities are pursuing alternative systems for wastewater treatment. Economics and safety are the primary factors driving this movement. Several communities in New Mexico have constructed wetlands for wastewater treatment (Tessneer 1998). The advantage of a constructed wetland is ease and cost of operation. The disadvantage is that it generates no future revenue stream. Thus, a tree plantation system with a market for the wood products would be favorable over a constructed wetland system. Currently, the city of Las Cruces, New Mexico, is designing a land application tree system for its West Mesa Industrial Park based upon the Ojinaga model (Watson 1998). As with Las Cruces, the Ojinaga system could serve as a model for other small communities throughout the arid and semiarid world. In the case of Ojinaga, the target market is short fiber pulpwood, but in other communities the market could easily be fuelwood, specialty hardwoods, Christmas trees, or pine for pulpwood or saw timber. The tree plantations could even serve as recreational areas. Thus, the state of Chihuahua could become a leader in system innovation.

CHALLENGES OF LAND APPLICATION SYSTEMS

Although land application systems provide many benefits, there are also some constraints. The four frequently voiced objections are: (1) human pathogens, (2) organic compounds, (3) nitrogen contamination, and (4) metals and trace elements (Bastian et al. 1982; Kowal 1986). Pathogens can pose a health threat to both humans and ani-

imals through contamination of surface water and groundwater and subsequent crop contamination. However, the survival of most pathogens, including bacteria, viruses, and protozoans, is greatly reduced by exposure to sunlight, high temperatures, and drying (Kowal 1986). Helminths (worms) have more adaptive resistance and can persist in the soil for longer periods, from a few days to several years, depending on the species (Burge and Marsh 1978; Feacham et al. 1980). However, adults, eggs, and cysts are not likely to be problematic where primary treatment of the wastes precedes land application (Kowal 1986; Zasoski and Edmonds 1986). Treatment as minimal as a settling pond, like that used in Ojinaga, in which the sludge and organic matter separate from the effluent, removes most protozoans and helminths. Smith (1996) reported the time to kill 90% (T_{90}) of human parasites ranged from 2 to 10 d for protozoans, to 6 d for viruses, to 17 d for *Ascaris*, to less than 30 d for *E. coli*. Thus, the major risk to humans would be all but eliminated after 30 days. Nevertheless, caution should be exercised during land application processes to limit public access and to allow periods of drying out to facilitate pathogen die-off (Foster and Engelbrecht 1973). Generally, pathogens pose little health risk when applied to nonedible crops. Chlorination would be effective in reducing pathogen numbers, but the treated wastewater would have to be dechlorinated prior to land application to minimize subsequent damage to the trees. Both chlorination and dechlorination are expensive and unnecessary if simple precautions are taken.

Groundwater contamination by toxic organic compounds from industrial wastes and household wastewater is another potential threat. Although most organics are eventually biodegradable, many are resistant to decomposition because of their chemical complexity. Subsequently, they could eventually leach into the groundwater (Kowal 1986). The best management strategy for these materials is to enforce laws requiring industry to remove these materials from their waste streams before they enter the municipal system. Likewise, implementing toxic waste minimization programs by providing alternative depositories for household chemical wastes, such as pesticides and automotive lubricants, and educating people about the proper use of the municipal sewage system are important strategies. Beyond this, wastewater streams containing toxic organic compounds at low levels should be applied at low rates, thus providing optimal conditions for degradation.

High levels of nitrogen, which are typical of domestic wastewater, can pose a threat of nitrate (NO_3^-) contamination to the groundwater, since NO_3^- is mobile within the soil system and susceptible to leach-

ing. However, nitrogen loading can be managed to avoid leaching. Waste application can be based upon the amount of mineralized nitrogen (plant-available forms of nitrogen) that the tree crop needs at a particular growth stage. Typical loading rates for land application systems supply 0.3-2.4 kg N/ha/d or 110-876 kg N/ha/yr (U.S. EPA 1992). Using this method, most of the NO_3^- -N and ammonium-N (NH_4^+ -N) should be available for plant assimilation (some NH_4^+ -N may volatilize from the system) (Brockway et al. 1986; Sommers and Barbarick 1986). In addition, microbially mediated pathways of nitrification/denitrification can affect further nitrogen removal.

Heavy metals and trace elements are of concern in terms of drinking water and groundwater quality and possible assimilation into edible plant parts. Of the heavy metals, only cadmium is significantly absorbed by plant roots (U.S. EPA 1984; Sommers and Barbarick 1986). Lead and mercury can be problematic, although they are insoluble and immobile in plant root systems. Furthermore, most metals become less soluble as pH increases, with the exception of anionic metals (Logan and Chaney 1983). Generally, soils with a neutral to alkaline pH immobilize toxic metals as precipitates, which are not available to plants and not susceptible to leaching into the groundwater (Jewell 1982; Zasoski and Edmonds 1986). Under moderate waste loading rates, it would take decades to accumulate lead and mercury in soils to dangerous levels (Kowal 1986). To avoid potential risks, applications can be controlled and limited by determining the maximum cumulative amounts acceptable for each element applied over a period of years, and then managing the loads accordingly (Sommers and Barbarick 1986).

These concerns are valid and can pose a possible health threat through the contamination of surface water, groundwater, and subsequent crops. Yet, overall, the potential health threats posed by land application systems are no greater than conventional waste treatment systems, if land application systems are properly managed (Kowal 1986). Sustainable, safe management practices must be based on a thorough understanding of land application design, the soil-plant system, the surrounding environment, and the risks associated with handling wastes.

In addition to the above concerns encountered in using land application systems throughout the world, there are some unique challenges with these systems in arid and semiarid regions. Any land application system using water high in salts must be managed to minimize salt buildup in the plant rooting zone. Excess salts can decrease crop productivity and, in severe cases, destroy productive farmland. The salt concentration in the soil is a function of the salt

concentration in the applied irrigation water and the leaching fraction (the ratio of drainage water to irrigation water). Agricultural systems use a leaching fraction to flush salts below the rooting zone but not into the groundwater. In reusing wastewater for crop production, where the soil and the plants are used as a treatment unit, the level of salt and nitrogen accumulation in the groundwater will show the effectiveness of the management of the land treatment system. Therefore, the groundwater should be monitored throughout the life of the project. Increased nitrogen levels in the groundwater can present a health hazard in places where shallow wells are used to obtain potable water.

Another challenge is organic matter induced soil deterioration. High organic matter in untreated wastewater can plug soil pores and create a reducing environment, rendering the soil unfit for agricultural uses. Properly designed primary wastewater treatment results in the separation of a significant part of the organic material from the wastewater before it is applied to the land.

Wastewater application in arid areas is coupled with consumption of the wastewater by vegetation based on the evapotranspiration. Thus, the most effective way to utilize the wastewater is to use species that have the longest possible growing season, including native plants that may initiate growth earlier than non-native types, and to incorporate perennial or winter-type forage crops that can be intercropped between the trees, thereby utilizing the wastewater during periods of tree dormancy. Land application treatment systems in humid areas, where the water is almost inconsequential, are well studied. However, arid regions have received less attention, with the bulk of the research centering in Australia and Israel (Myers et al. 1995; Myers and Polglase 1996; Myers et al. 1997). Thus, there is a need to determine site-specific factors and management approaches, which are most effective in waste remediation and utilization in arid regions.

OJINAGA PILOT STUDY

Materials and Methods

Municipal sewage in Ojinaga is piped directly into an anaerobic lagoon, which provides primary treatment of the sewage. Sludge samples were obtained from both the old and new lagoons to determine quality and utility. A 1.2 ha site, adjacent to an oxbow lake, was selected downstream from the lagoon. Soils were sampled to a depth of 1 m to characterize texture and to model the effects of soil type on tree growth and water use. Routine sampling and analysis of the

wastewater effluent and influent, along with water from an oxbow lake, the Rio Grande, and the Rio Conchos also were implemented. The depth to groundwater at the site is approximately 3 m, and the slope is < 5%. Wells were installed to monitor the groundwater (in particular, the levels of nitrate and chloride). Baseline samples from the monitoring wells were taken and analyzed prior to starting irrigation with full-strength wastewater.

One goal of the study has been to identify tree species and clones that exhibit the greatest biomass growth and ion uptake. Based on previous studies (Yadav 1980; Donaldson and Standiford 1983; Stewart et al. 1986; Mather 1993) and Eucalyptus field trials conducted by the INIFAP experiment station in Ojinaga (Núñez 1995a, b, c; Tena Vega 1998), Eucalyptus camaldulensis was selected for its cold tolerance and fast growth. Three Eucalyptus camaldulensis clones from Simpson Timber Co., California, were chosen for inclusion in the study: SC5 (505), 4016, and 4019. Two other tree species were also selected: hybrid Populus (poplar) and Robinia pseudoacacia (black locust). Populus is native to the Rio Grande and is found in other river valleys in hot dry areas (Bongarten 1996). Three clones were purchased from Broadacres Nursery, Oregon: TD 15-029 (P. trichocarpa × P. deltoides) TD 50-197 (P. trichocarpa × P. deltoides) and OP 367 (P. deltoides × P. nigra). Robinia pseudoacacia, known for its hardiness, has been used in stream bed stabilization and mineland reclamation (Myatt 1997). Open-pollinated Robinia pseudoacacia plants were obtained from the Oklahoma Department of Agriculture.

In April 1997, the site was plowed, disked, and shaped into 54 separate test plots about 7 m × 7 m in size each. Containerized Eucalyptus camaldulensis, bareroot Robinia pseudoacacia seedlings, and Populus cuttings (20 cm in length) were transplanted at 2 m × 2 m spacing. At seven months, four representative trees of each of the seven tree sources (28 total trees) were selected based on mean tree diameter, excavated and fractionated into leaves, stems, trunks, and roots. Tissues were dried, weighed, and analyzed for chemical constituents. At eight months and at 20 months after planting, the survival, height, and diameter of the trees were measured.

During the first growing season, plots were manually flood irrigated with water from the oxbow lake to establish the trees before implementing irrigation regimes using full-strength wastewater effluent. The choice of flood irrigation versus other types of irrigation systems was based on the premise that flood irrigation technology was familiar to Ojinaga farmers, whereas other systems would be less familiar and more costly. Weeds were controlled mechanically and chemically

Table 1: Selected Water Quality Indicators (min max range)
Ojinaga, Chihuahua, December 1996 March 1998

Test Parameter	Rio Conchos	Rio Grande	City Well Water (12/96)	Orshaw Lake at the Exp. Site	Wastewater Effluent
pH of water	7.3-8.4	7.3-8.4	7.73	7.6-8.6	7.3-7.7
Electrical Conductivity ($\mu S/cm$)	1.3-2.3	1.4-3.3	2.51	1.9-3.4	2.7-3.1
Total Dissolved Solids (mg/L)	791-1460	1192-2160	1892	2167-2670	1948-2217
Sulfur Absorption Ratio (SAR)	3.67-6.40	5.17-6.77	5.5	6.89-8.85	5.49-6.96
Partial Calcium (Ca^{2+}) (mg/L)	140	190-390	-	4-800	(2.8-6.0) $\times 10^6$
BOD (mg/L)	-	30	-	13.2-14.6	28.6-32.5
COD (mg/L)	-	90	-	67-89	100-117
Nitrate/Nitrite as N (mg/L)	0.42-8.66	-0.85	1.8	-0.85-1.88	-0.85-8.16
Ammonium as N (mg/L)	0.87-8.88	0.88	-	0.86-6.1	7.6-12.3
Water Kjeldahl N (mg/L)	0.9-2.5	1.87	-	5.4-16.0	14.1-37
Sulfate (mg/L)	328-513	457-538	-	795-1079	729-1067
Chloride (mg/L)	98.6-342.3	440-690	191	240-384	201-383

with Fusilade. After the first growing season, the plots were irrigated at three regimes with wastewater effluent based on potential evapotranspiration (PET) data. The first irrigation regime was based on the PET plus 36% additional water for leaching. This resulted in the application of excess water throughout the year. This was done in consideration of the possibility that just enough trees will be planted to treat the wastewater, and that farmers using river-fed flood irrigation tend to use excess water. A second irrigation regime was based upon the PET plus approximately 20% additional water for leaching. This regime would tend to subject the trees to mild water stress as salts may accumulate in the upper soil profile. The third schedule was to examine deficit irrigation, supplying 8% less water than PET. This irrigation regime would tend to maximize the accumulation of salts in the soil. Water application rates, plant growth, wastewater quality, weather data (rainfall, insolation, and temperatures), soil nutrient analysis to a depth of 1 m, and the quality of the leachate below the root zone were analyzed.

In addition to the field research, the economic feasibility of a full-scale project was investigated. The standard method for evaluating the economic impact of a project extending over several years is to

Table 2: Analysis of Sludge Samples Obtained in Ojinaga, Chihuahua, December 1996

Parameter	New Lagoon Sludge	Old Lagoon Sludge
pH	7.29	7.73
EC (dS/m)	2.35	6.54
TDS* (mg/L)	1947	-
SAR*	1.2	-
Magnesium (meq/L)	1.6	-
Calcium (meq/L)	21.5	-
Sodium (meq/L)	43	-
Nitrate-N (mg/L)	0.0	246 mg/kg
Ammonia-N (mg/L)	-	0.3%
Kjeldahl-N(mg/L)	72.0	-
Chloride (mg/L)	177	355
Phosphorus (mg/kg)	29.0	-
Potassium (mg/kg)	78.0	-
Sulfur (mg/L)	27.0	-
Coliforms ¹ (col/100ml)	-	<2FC* count/g

*TDS (total dissolved solids); SAR (sodium absorption ratio); FC (fecal coliform)
¹Coliform data from analysis taken June 1997.

calculate the net present value (NPV) of the project. Evaluation of the economic return to short fiber production in Ojinaga is not a simple matter. Both the environmental benefits, as well as the financial profitability, must be considered. Placing a value on environmental benefits is notoriously difficult, however, and this was not done directly in this study. Rather, environmental benefits were evaluated indirectly by assuming that a specific environmental standard must be met, and that the preferred method for meeting this standard is the least costly method. In particular, it was assumed that the Ojinaga waste treatment system must meet the standards for effluent established by SEMARNAP (Secretaría de Medio Ambiente, Recursos Naturales y Pesca). Further, it was assumed that the alternative method for meeting these standards is a conventional sewage treatment facility similar to the facility proposed by the Comisión Nacional del Agua (1994). Thus, the return to fiber production includes both the tree plantation and the avoidance of costs incurred in constructing and operating a traditional sewage treatment plant. This approach ignores the envi-

Table 3: Comparison of the Concentration of Metals in Sludge Samples Taken from the Former and Present Sewage Lagoons near the Ojinaga Pilot Study Site and the U.S. EPA Part 503 Sewage Sludge Annual Pollutant Loading Rate Regulations

Metal	Old Lagoon Sludge (g/Mg)	New Lagoon Sludge (g/Mg)	U.S.EPA Annual Pollutant Loading Rate (g/ha/yr)*
Cd	0.0	7.6	1,900
Cr	14.5	8.9	150,000
Cu	0.0	192.0	75,000
Hg	1.7	10.3	15,000
Ni	10.0	0.0	29,000
Pb	75.0	81.0	15,000
Zn	0.0	573.0	140,000

*Source: U.S. EPA 1994.

ronmental benefits arising from fiber production not associated with water quality, such as habitat creation and reduction in air pollution.

Sustainable economic development includes the creation of local financing. Information was obtained from Banco Nacional de Crédito Rural (BANRURAL) officials in Ojinaga concerning yields and the cost of production of various crops. In addition, investigators also discussed terms under which credit might be made available for long-term financing of wood production beyond the pilot project. Interest rates in Mexico are moderately high by international standards, but favorable rates are available for small farmers and also for financing of exports. The Ferrocarril Nacionales de México railroad has experience in shipping timber, and has railcars suitable for shipping logs, chips, or pulp to U.S. or Mexican markets. To determine the extent of the domestic (Mexican) market for short fiber pulpwood, the Copamex facility (one of the largest manufacturer of paper products in Mexico) in Anahuac, Chihuahua, was visited.

Results

Water Analysis

All of the water sources have high pH (pH > 7.3), electrical conductivities (EC) between 1.3 and 3.4 dS/m, and sodium absorption ratios (SAR) between 3.7 and 8.8 (Table 1). These high EC and SAR val-

Table 4: Tree Survival, Height, and Diameter Growth, Measured 8 and 20 Months After Planting (First and Second Growing Seasons)

Genus/Species	Survival (%) Yr 1	Survival (%) Yr 2	Height (m) Yr 1	Height (m) Yr 2	Diameter Yr 1		Diam. Yr 2
					Growth: Basal Area (mm)	16 Achive: Basal Area (mm)	
<i>Acacia greggii</i> 4816	99	97	2.19	3.36	36.5	97	67.69
<i>Acacia greggii</i> 4673	98	94	2.58	3.34	34.8	99	49.66
<i>Acacia greggii</i> 8523	98	98	2.88	3.16	36.1	98	44.96
<i>Populus 339</i>	71	67	1.73	4.75	38.6	69	90.97
<i>Populus 197</i>	19	34	1.36	3.98	16.5	51	28.46
<i>Populus 347</i>	93	88	3.17	4.41	37.8	47	54.49
<i>Arbutus</i>	99	94	1.35	3.38	19.2	41	19.87

Note: Both survival measurements are based on initial stocking levels at the time of planting.

ues indicate that the water is marginal for traditional agriculture (Miller and Donahue 1990); however, short rotation woody crop production is feasible with this water. The wastewater had a total Kjeldahl nitrogen (NH_4^+ -N and organic-N) of 14.37 mg N/L of wastewater. Most of the nitrogen was in the NH_4^+ form, with low NO_3^- levels, suggesting that leaching of nitrates would not be a problem. At an application rate of $2.0 \text{ m}^3/\text{m}^2$, the loading rate of N as NH_4^+ would be approximately 250 kg N/ha/yr .

There were 2.3×10^5 colonies of fecal coliform bacteria/100 ml of effluent. International guidelines suggest a maximum geometric mean concentration of 1,000 fecal coliforms/100 ml for wastewater applied for edible crops (Kowal 1986). For crops raised intensively with a short rotation, such as vegetable crops, these levels are clearly unacceptable. In contrast, woody crop production is extensive, and in comparison to vegetables, the rotation is long. Most importantly, these trees are not being raised for food, and represent no secondary health hazard.

Biosolids (Sludge) Analysis

Dried sludge from the original lagoon had high pH and EC (Table 2), but it had only 5.8% organic matter with $0.25 \text{ kg NO}_3^- \text{-N/Mg dry matter}$. The sludge consists primarily of soil particles carried by wind or water, precipitated calcium carbonate, and caliche, which is calcium carbonate typically found in the desert soils of the area. Thus, this sludge is worthless as a crop nutrient source. The sludge contained small amounts of lead and mercury in insufficient amounts to pose problems for land application. The EPA limit for annual loading rate

Figure 3: Biomass (in grams) at Seven Months of *Eucalyptus camaldulensis*, Hybrid *Populus*, and *Robinia pseudoacacia*

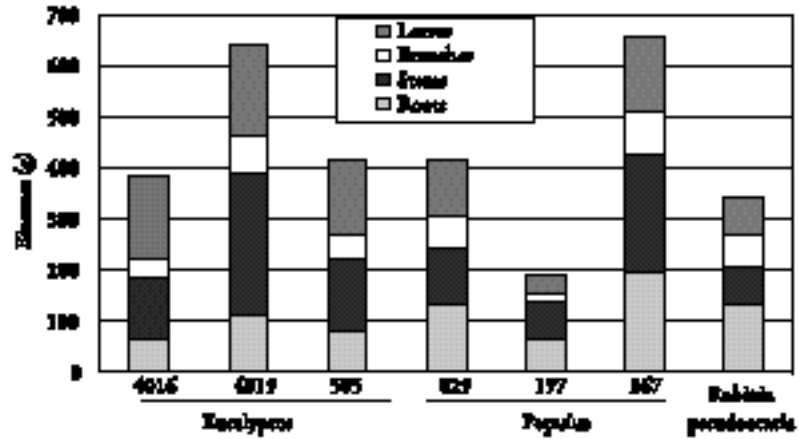


Table 5: Nutrient Analysis of *Eucalyptus camaldulensis*, Hybrid *Populus*, and *Robinia pseudoacacia* at Seven Months

Plant Tissue		N (%)	Ca (%)	Mg (%)	Cl (%)	Na (%)
Leaves	<i>Eucalyptus</i>	2.79	1.17	0.21	0.79	0.36
	<i>Populus</i>	2.57	2.55	0.40	1.47	0.28
	<i>Robinia</i>	2.48	3.57	0.40	0.69	0.02
Woody	<i>Eucalyptus</i>	0.63	0.69	0.10	0.45	0.09
	<i>Populus</i>	0.70	0.61	0.10	0.13	0.40
	<i>Robinia</i>	1.52	0.86	0.09	0.18	0.04
Roots	<i>Eucalyptus</i>	0.73	0.31	0.10	0.55	0.11
	<i>Populus</i>	0.90	0.63	0.09	0.22	0.07
	<i>Robinia</i>	1.53	0.57	0.08	0.33	0.04

for both Pb and Hg is 15,000 g/ha/yr (U.S. EPA 1994). The sludge from the old lagoon contained 75.0 g Pb/Mg and 1.7 g Hg/Mg, and the new lagoon contained 81.0 g Pb/Mg and 10.3 g Hg/Mg sludge, on a dry weight basis. Using the EPA annual loading rate criteria, the concentration of every metal analyzed was well below the limit specified (Table 3). The sludge poses little environmental threat. Unfortunately, the sludge from this lagoon also has no economic value.

Soil Analysis

The soils at the experimental site are nonuniform. The site is situated in the floodplain of the Rio Grande, immediately next to the old river channel, and soils (fluvents) of varying texture have been deposited over time as a result of flood events. The experimental site measures 290 m (roughly east-west) by 45 m, and straddles a variety of soil deposits. At one end of the site, the soils are predominately silty clays and silty clay loams, with plant available water (AW) contents ranging from 12-15 cm water in the top meter of soil. In the central area of the site, there are layers of loam, sandy loam, and sand, with lower AW in the top meter. Moreover, some of these layers are extremely gravelly (> 60% gravel by volume), very gravelly (35-60% gravel by volume), and gravelly (15-35% gravel by volume), which further lowers the AW. In this part of the site, AW ranges from a low of 4 cm water/m soil with more typical values running from 6-12 cm water/m soil. At the other end of the site, the gravelly layers disappear, and a mixture of clay, clay loam, silty loam, loam, and sandy loam layers are found. The AW here ranges from 10-15 cm water/m soil.

These variations in texture and the resulting AW can affect tree growth. On loamy soils, the *Populus 367* clone had an average height of 2.8 m, while on soils containing layers of gravelly to extremely gravelly sand, these clones were only 1.9 m in height. In contrast, *Robinia pseudoacacia* plants had a height of only 1.1 m on loamy soils, but a height of 1.7 m on gravelly soils. Thus, the *Populus 367* clone grew taller in soils with higher AW than in soils with lower AW, whereas the *Robinia pseudoacacia* grew taller in soils with lower AW than in soils with higher AW. This response agrees with the preference of *Robinia pseudoacacia* for lighter, well-aerated soils (Bongarten 1996), in contrast to *Populus*, which is native to riparian areas (Little 1950).

Table 6: Cumulative Difference (in U.S. dollars) between Net Present Values from Fiber Production and Conventional Waste Treatment Plant at Different Loan Rates*

Interest Rate	Year 1	Year 7	Year 14	Year 21	Year 28
25.0%	1,660,071	1,675,881	1,667,633	1,708,408	1,708,055
10.0%	1,886,445	2,086,229	2,160,698	2,710,808	2,730,992
0.0%	2,075,090	2,617,191	3,110,583	6,345,209	7,443,606

*Note: Assuming 424 hectares under cultivation, a price of \$20 per ton, and a yield of 160 tons per hectare at harvest.

Table 7: Employment from Fiber Production

Activity	Employment per ha/yr	Employment for 424 ha farm/yr
Establishment (Year 1)	150 labor-hours	63,600 labor-hours
Growth & Maintenance (Years 2–6)	50 labor-hours	21,200 labor-hours
Harvest (Year 7)	250 labor-hours	106,000 labor-hours

Tree Growth

During the first and second growing season, the three Eucalyptus clones had high survival rates (Table 4). In contrast, survival of Populus was clone-dependent, with the 367 clone having the highest survival rate (95% for year one and 88% for year two) and clone 197 having the lowest survival rate (53% for year 1 and 24% for year 2). Robinia had high survival rates the first year (93%), declining to 73% in the second year. Eucalyptus clones had good height and diameter growth both years. However, during the first winter, temperatures dropped to -10°C , resulting in damage to all of the Eucalyptus clones. Clones 4016 and 4019 died back to ground level, whereas the SC5 clone had damage only to the leaves. Nevertheless, during the second growing season, the 4016 and 4019 clones outgrew the SC5 variety in height, although the SC5 maintained a slightly greater breast height diameter. In the second winter, freeze damage was minimal for all three Eucalyptus clones, with only slight foliar dieback on scattered trees.

Growth of the Populus clones was clone dependent, with clone 367 growing best. During year two, clone 367 outperformed all other clones and species for both height and diameter, growing over 4 m in the second year to an average height of 6.4 m. This average height was over 1 m taller than the other species and nearly 2 m taller than the second best Populus clone (029). Robinia growth was highly variable both years, because it was an open-pollinated seed source and was sensitive to heavy soils. However, a number of trees grew well and show promise for the development of Robinia clones adapted to conditions in Ojinaga.

Biomass/Ion Uptake Data

Eucalyptus camaldulensis (4019) and Populus clone (367) produced the most biomass the first growing season (Figure 3). However, Eucalyptus camaldulensis had the greatest proportion of biomass in woody tissue. All three Eucalyptus clones had the lowest percentage

of biomass in root tissue (19%), while Robinia had the highest percentage (40%). There appeared to be no relation between root biomass and survival among and within species. However, the greater survival and growth of Robinia in xeric plots might be explained by the greater root production relative to shoot biomass.

There were also differences in the ability of the species to accumulate salts (Table 5). Robinia had higher nitrogen contents in both roots and stem tissues, but foliage nitrogen levels were comparable to the other species. However, Robinia had higher accumulation of calcium in the foliage, almost to the total exclusion of sodium. This trend followed for woody tissue as well. There were no differences in magnesium accumulation. Populus had higher chloride accumulation in the foliage, but generally lower levels in woody tissue. Long-term accumulation could impact soil restoration or species performance if excessive levels develop.

Economic Analysis

The Copamex mill has the capacity to process 146,000 metric tons of wood fiber per year but currently only processes 128,000 metric tons per year. Of that quantity, 50% is short fiber from hardwood species and 50% is softwood fiber from pine. The rail lines connecting Ojinaga to Chihuahua allow direct transshipment into the sorting yard of the Anahuac facility. Currently, the short fiber is imported from the United States. Copamex officials are interested in developing domestic sources of short fiber, and are willing to purchase all output from Ojinaga. Indeed, Copamex has investigated the possibility of developing large-scale production of Eucalyptus and other species near Ojinaga.

Table 6 presents the differences in estimated net present value returns for forest production and a traditional waste treatment facility for various interest rates and over different time periods. The calculations in the table assume that 424 ha are planted in Eucalyptus and Populus in the initial year, and then harvested in seven-year cycles. It is assumed that production at harvest is 160 tons/ha and that a price of \$20/ton is received. Regardless of interest rate or time period considered, waste treatment could be achieved at lower cost with fiber production than with a traditional sewage treatment facility. Fiber production requires less initial capital expenditure and has a lower operating cost than a traditional waste treatment plant. Of particular interest is the last line of Table 6, which is the actual budgetary savings to Ojinaga from fiber production. In the initial year, lower capital costs of fiber production saves Ojinaga \$2 million. The cumulative savings over 28 years is more than \$7.4 million, compared to

constructing and operating a conventional wastewater treatment system.

An important goal of sustainable development is the creation of employment opportunities. This is an especially important issue for Ojinaga given the job losses and accompanying decline in population experienced by the city in recent years. Potential employment arising from biomass production is broken down into three categories: fiber farm establishment, which occurs during the initial year and includes site preparation, planting, and irrigation; growth and maintenance, which occurs during years two through six and involves primarily irrigation; and harvest, which occurs in year seven and includes harvesting activity (Table 7). For each set of activities, 2,000 labor-hours per year were included for administration, including management, organization, and secretarial support.

RECOMMENDATIONS

The foundation has been laid for continued research on the application of tree plantations to the treatment of municipal wastewater by way of land application in arid climates. Many small communities in the border region lack suitable wastewater treatment facilities. Furthermore, these communities lack incentives to implement waste management programs. An approach that creates financial benefits for these communities has the best chance of effecting change. The experience gained in Ojinaga provides the basis for a sustainable model of waste treatment. Moreover, with appropriate infrastructure development, Ojinaga could serve as a valuable training center for other border communities. The Ojinaga model has short fiber pulpwood as the target outcome, based on the need in Chihuahua and the availability of rail transport. However, other communities could produce fuel wood (for cooking or co-firing), pines for different products (including Christmas trees), specialty hardwoods, or even amenity plantings for recreation.

A key to success is the identification of tree species suitable for the target outcome. The three species used in this model are suitable for pulpwood production. However, differences in growth rate, cold-hardiness, drought tolerance, and salt tolerance indicate a need for continued development of suitable plant material. From this study, only one clone each of Eucalyptus and Populus are suitable for long-term use. There are selections of Robinia that could be valuable on droughty soils. However, propagation techniques must be developed. There are other species (Liquidambar or Platanus) and cultivars of native cottonwood that may be better suited to Ojinaga, with its cal-

careous soils, high evaporative demand, and long growing season. Moreover, an evergreen softwood species such as *Pinus eldarica* may prove superior for winter water use, when deciduous species are dormant.

A successful land application system includes not only an economically viable system, but also an environmentally safe system. Both are required for sustainability. Continued monitoring of effluent can prevent endangering the treatment process by contamination with heavy metals or toxic organic compounds, while monitoring the groundwater can minimize the risk of compromising groundwater quality by overloading the system. Both will require continued community involvement to prevent inappropriate dumping of toxic chemicals.

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VI

A Low-Tech Treatment Strategy for Treating and Reusing Wastewater in Arid Lands

Lawrence A. Baker and Paul Westerhoff

ABSTRACT

Many cities in arid regions with limited renewable surface or ground water supplies will need to consider reusing treated municipal wastewater for long-term sustainability. On the U.S. Mexican border, a low-tech treatment train composed of aerated lagoons, constructed treatment wetlands, and a soil aquifer treatment system was evaluated as a method to treat and store wastewater for later use. Literature review and field and lab experiments show that such a system could provide water with low nitrate and DOC that would probably require only disinfection prior to municipal use. Results were used to propose design and operational guidelines for a low-cost, low-tech treatment system. The potential impact of recharging water, which is now being discharged directly into the Santa Cruz River at Nogales, on the flow of the river was also evaluated.

THE NEED TO REUSE WASTEWATER: WATER SHORTAGES ALONG THE U.S. MEXICAN BORDER

Potable water is a scarce commodity in many parts of the world, particularly in arid regions. The problem of water scarcity involves both the quantity and quality of supply. Water quantity is often measured in terms of scarcity at $< 1000 \text{ m}^3/\text{year}$ per person and water stress at $< 1,700 \text{ m}^3/\text{year}$. These values include requirements for household, agricultural, and industrial use, including energy production. Using these criteria, 28 countries with a total population of 335 million expe-

perienced water scarcity or stress in 1990. By 2025, 46 to 52 countries with a total population of 2.8 to 3.3 billion people will experience water stress or scarcity. Increasing population, rising per capita consumption, and declining water availability will exacerbate the problem in the future (Postel 1997). Poor quality of what little water is available further amplifies the problem of obtaining potable water in arid lands. Worldwide, about 20 million children die of waterborne diseases each year (Tchobanoglous and Schroeder 1985). Diarrheal diseases are a major cause of infant mortality in the Mexican border region; rates of hepatitis A are seven times higher in Nogales, Arizona, than in the general population (Sanchez 1995; Varady and Mack 1995).

Limited water supply and poor water quality are key concerns along the U.S. Mexican border because the region is arid, the population is growing rapidly, and the area is economically depressed. The problem of providing an adequate supply of potable water has generated a growing interest in the reuse of municipal wastewater effluent. Treated wastewater can be reused either directly or indirectly. The most common direct use of treated wastewater is irrigation of non-food crops or turfgrass (e.g., parks and golf courses) or industrial applications (e.g., cooling water). Treated wastewater is rarely used directly as municipal water supply. Indirect reuse generally means that the water is stored for some period of time prior to reuse. An increasingly common practice in the arid southwestern United States is the storage of reclaimed wastewater in underground aquifers.

In most ongoing or planned projects in the United States, wastewater is treated at a fairly high level prior to reuse. In larger cities in the United States, treatment prior to recharge would typically consist of conventional secondary treatment followed by nitrification and denitrification (NDN) or advanced secondary treatment with simultaneous NDN. Further treatment may include sand filtration to reduce suspended solids, thereby reducing the potential for clogging during recharge; reverse osmosis to reduce total dissolved solids (salts); and chlorination and/or UV disinfection to kill pathogens prior to recharge. These treatment systems generally have high capital costs, extensive high-tech maintenance requirements, and high energy requirements. They are therefore not well suited for use in small municipalities in the United States or in less developed countries.

The goal of this project, which was funded by the Southwest Center for Environmental Research and Policy (SCERP), was to evaluate the technical feasibility of a low-tech, low-cost system to treat and reuse wastewater on the U.S. Mexican border. A low-tech approach was considered for the following reasons:

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1. Most towns in the border region are poor and small. These factors tend to favor wastewater treatment plants that are inexpensive to construct and require limited technical expertise or money to operate.
2. In the U.S. Mexican border region, undeveloped land is often available at a reasonable cost. Low-tech treatment systems are land-intensive and, therefore, not appropriate for highly urbanized areas in which undeveloped land is unavailable or expensive.
3. Low-tech systems, particularly wetlands, have ancillary benefits that may be valuable to border communities. Well-designed treatment wetlands may provide valuable wildlife habitat along the U.S. Mexican border area, where most natural wetland habitats have been destroyed.
4. Geological characteristics in the border region are often suitable for artificial recharge and recovery. Aquifers are often deep, and large unsaturated vadose zones provide in-situ treatment during recharge. Aquifer sediments are often porous with high water storage capacities and permeabilities, making it feasible to store and recover recharged water.

THE NOGALES INTERNATIONAL WASTEWATER TREATMENT PLANT

The specific area of concern in this project was the sister cities of Nogales, Arizona, and Nogales, Sonora. The area has limited water supply, some of which is polluted. The main wastewater treatment facility is the Nogales International Wastewater Treatment Plant

Table 1: Concentrations and Removal Efficiencies for Key Water Quality Constituents at the NIWTP

Constituent	Average	Q ₁	Median	Q ₃
Flow (mgd)	11.3	8.4	10.3	14.1
BOD ₅ (mg/L)	18	8	16	26
(% removal)	86%	72%	87%	93%
Suspended solids (mg/L)	24	9	17	55
(% removal)	94%	89%	95%	97%
Fecal coliform	102	2.4	3.4	328
pH	7.9	7.3	7.6	8.3
Total organic N (mg/L)	6.4	2.3	3.4	10.9
Ammonium	18.2	9.3	18.8	23.6
Nitrate	1.1	0	0.69	2.2

Source: U.S. EPA NPDES data files.

(NIWTP), which receives an average flow of 11 million gallons per day (mgd) of sewage, approximately two-thirds of which originates in Mexico and flows across the border for treatment in the United States. Treatment processes at the NIWTP include aerated lagoons, sand filtration, and chlorination. Treated effluent is discharged to the

Table 2: Water Budget for Santa Cruz County 1987

	<i>Actual volume (acre-feet/year)</i>
<i>Groundwater from Mexico</i>	<i>500</i>
<i>Surface water from Mexico</i>	<i>20,458</i>
<i>Wastewater effluent from NIWTP</i>	<i>13,338</i>
<i>Tributary inflows from U.S.</i>	<i>26,574</i>
<i>Mountain front recharge</i>	<i>13,904</i>
<i>Total inflows</i>	<i>75,975</i>
<i>Surface water outflow</i>	<i>21,960</i>
<i>Phreatophyte consumption</i>	<i>24,098</i>
<i>Groundwater outflow</i>	<i>8,524</i>
<i>Pumpage from wells</i>	<i>19,993</i>
<i>Total outflows</i>	<i>75,975</i>

Source: ADWR 1989. 1 acre-foot = 1233m³

Santa Cruz River, which flows north through Arizona. For most of the year the discharged wastewater effluent comprises approximately 100% of the streamflow in the Santa Cruz River, which is then characterized as an effluent-dominated stream. High-flow events in the Santa Cruz River occur during summer monsoons.

Although the NIWTP generally met 1996 NPDES water quality standards (Table 1), stricter standards were anticipated for suspended solids and chlorine residual and new limits were expected for chronic toxicity (particularly for ammonia), total phosphorus, and viruses (Camp Dresser and McKee, Inc. 1997). Under these new standards, effluent from the NIWTP would not be suitable for indirect municipal reuse without further treatment prior to recharge.

The volume of effluent produced by the NIWTP amounted to about $11 \times 10^6 \text{ m}^3/\text{yr}$ in 1987. The potential amount of water produced by wastewater reuse would therefore be 12% of the total water inflow to Santa Cruz County and about 67% of current groundwater pumping (Table 2). Wastewater is therefore a significant resource that could augment future water supplies.

CONCEPTUAL DEVELOPMENT

The first step in this study was to identify treatment objectives and sustainability issues for a low-tech treatment system. The second step was to review literature on treatment processes to formulate a series of research questions for further study.

Treatment Objectives

In reviewing water quality data from the NIWTP, three key constituents appeared to be of greatest concern: nitrogen (several species), dissolved organic material, and pathogens. A low-tech system designed to augment or replace the existing lagoon system would probably have to meet both discharge standards and recharge standards. Two other considerations in designing a low-tech treatment system would be habitat quality, particularly for the wetland component, and the impact of water reuse on the ecology of the Santa Cruz River.

Nitrogen

Discharge standard. The key concern for discharging this effluent to the Santa Cruz River is ammonia toxicity (Strömberg et al. 1993). Nitrogen in the form of free ammonia (NH_3) is highly toxic to fish. Arizona's water quality standard for free ammonia in receiving waters is 0.1 mg/L. At a pH of 8.8 (the 90th percentile for pH in NIWTP effluent) (see Table 1), this translates to a total ammonium concentration of 0.4 mg/L. Even at the average pH of 7.9, the total ammonium concentration would have to be less than 2.6 mg/L, far less than the level currently being discharged.

Recharge standard. The treatment objective for recharging effluent to the aquifer for later use as municipal water supply is total nitrogen concentration. The goal here would be to meet Arizona's recharge standard for total N of 10 mg N/L for effluent being recharged to aquifers. The Arizona standard is based on the premise that all N in recharge would eventually be converted to nitrate. Keeping total N < 10 mg N/L would therefore ensure that wastewater reaching the aquifer would meet the EPA's Maximum Contaminant Limit (MCL) for nitrate of 10 mg NO_3^- /L.

Organic Material

Discharge standard. For wastewater that is discharged to a river, the treatment objective is to protect aquatic biota in receiving waters from anoxic conditions caused by biodegradation of organic matter in the wastewater effluent. As seen in Table 1, the BOD-5 in the effluent of

the NIWTP generally meets the existing standard for biological oxygen demand (BOD), so further reductions would not be necessary.

Recharge standard. For wastewater that is recharged to aquifers and stored for municipal water supply, the potential for forming disinfection by-products (DBPs), upon subsequent groundwater pumpage and chlorine disinfection, such as trihalomethanes (THMs) and haloacetic acids (HAAs) would be a key concern. DBPs form when dissolved organic matter reacts with chlorine or other disinfectants during treatment of municipal water supplies. DBP formation is directly correlated to concentrations of dissolved organic carbon (DOC). The current MCLs are 80 mg/L for THMs and 60 mg/L for HAA-5 (the sum of five haloacetic acids).

Concentrations of DBPs could generally be kept below current MCLs if DOC levels in recharged effluent were kept below 3.5 mg/L. The effluent from the Nogales lagoon had an average DOC of 15 mg/L, so the proposed low-tech system would have to remove more than 10 mg/L DOC before the effluent reaches the aquifer. Controlling bulk organic carbon concentrations (i.e., DOC levels) will likely also control potential contamination from trace synthetic organic chemicals (e.g., solvents or pharmaceutical compounds).

Pathogens
Three classes of waterborne pathogens pose potentially serious health effects for humans: bacteria (e.g., *Campylobacter jejuni*, *Vibrio cholerae*, *Shigella* sp., *Salmonella typhi*), viruses (e.g., hepatitis A, Norwalk, rotavirus, poliovirus, coxsackie), and protozoa (e.g., *Giardia lamblia*, *Cryptosporidium parvum*, *Entamoeba histolytica*) (U.S. EPA 1993; Craun 1985).

Discharge standard. The current fecal coliform standard for effluent at the NIWTP of 200/100 mL (monthly average) and 800/100 mL (daily average) was generally met (see Table 1), so no improvement in coliform removal would be needed. Coliform are indicator organisms used as surrogates for microbial contamination. A virus standard may be added (Camp, Dresser and McKee, Inc. 1997).

Recharge standard. Treatment objectives for microbial pathogens after soil aquifer treatment (SAT) are similar to those for other municipal water supplies. The U.S. EPA (1997) has recommended the following treatment goals for groundwater recovery and treatment: zero coliforms per 100 mL, inactivation of 99.99% (4 log) of viruses, and 99.9% (3 log) inactivation of *Giardia*. Inactivation of pathogens (e.g., by chlorination) is recommended following withdrawal of recharged

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effluent from aquifers. A chlorine residual > 0.2 mg/L should also be maintained in the water distribution system.

Ecological Sustainability along the Santa Cruz River

Beyond meeting treatment objectives, there are several sustainability issues regarding the use of effluent and the use of wetlands as wildlife habitats.

Recharge or Discharge?

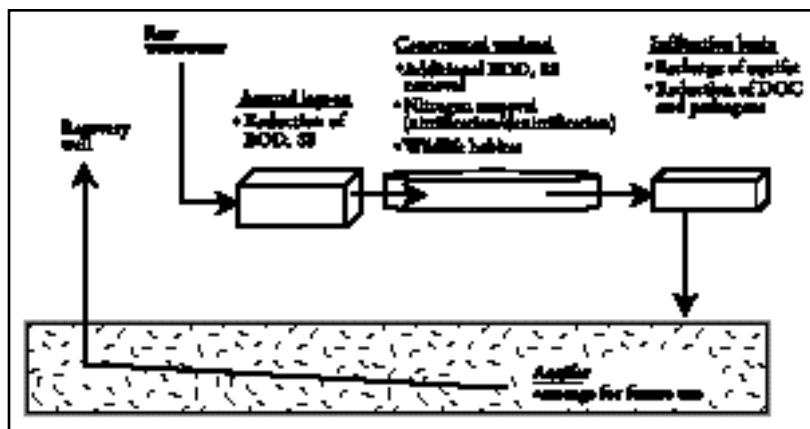
Because the Santa Cruz River below the NIWTP is effluent dominated, recharging effluent to underlying aquifers could result in a dry riverbed during the summer. Loss of streamflow would probably have a negative effect on the well-developed riparian zone below the NIWTP and damage the aquatic ecosystem. If recharging wastewater to aquifers located away from the Santa Cruz River were shown to reduce downstream flows, the impact might be avoided by limiting effluent recharge during low-flow period. Recharging effluent during high-flow periods would probably have no negative impact. It may even have a positive impact on the riparian ecosystem, since effluent recharged to the aquifer during high-flow periods may raise the groundwater level near the river, conceivably resulting in more flow to the river during the low-flow period as water leaked from the mounded groundwater system into the Santa Cruz River. A reasonable ecological objective (though not necessarily a regulatory one) would be to allocate the distribution of recharge and discharge to the river to preserve the riparian ecosystem.

Wetland Habitat Quality

A second sustainability issue pertains to the use of a treatment wetland as a wildlife habitat. Engineered wetlands are often designed to achieve plug-flow hydraulics in order to achieve efficient treatment. True plug-flow hydraulics means that the treatment wetland can be as small as possible to meet specified treatment objectives. The practical ramifications of this are low construction and land acquisition costs and low evaporation. The latter is particularly important in the desert environment.

However, a plug-flow wetland with densely planted emergent species has limited value for wildlife. Open ponds, seasonally flooded fringe areas, riparian trees and shrubs, and other components of a natural wetland greatly enhance the utilization of a wetland by wildlife. Thus, a key question regarding the use of wetland treatment systems in the arid west is how can both treatment objectives (e.g.,

Figure 1: Schematic Representation of a Low-Tech, Low-Cost System to Treat and Reuse Wastewater



nitrogen removal) and wildlife objectives (high species diversity, etc.) be achieved?

TREATMENT PROCESSES

The working hypothesis for this study is that a low-tech treatment system could provide a level of treatment sufficient to meet municipal drinking water requirements with minimal additional treatment, primarily chlorination. The research focused on the key treatment objectives discussed above: nitrogen removal, minimization of DBP precursors, reduction of pathogens, and ecological sustainability.

Several combinations of low-tech treatment operations would meet these treatment objectives. However, because many smaller cities in the border region already have aerated lagoons, the study focused on a treatment train that expanded the capability of a traditional lagoon system. One treatment system that would probably meet these requirements consists of aerated lagoons, constructed wetlands, and soil aquifer treatment systems (Figure 1). A description of each component of this treatment train follows.

Aerated Lagoons

Mechanical mixers inject air into the sewage to provide oxygen for aerobic degradation of dissolved organic matter (DOM) and limited oxidation of ammonia to nitrate. Some removal of suspended material occurs by sedimentation.

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Constructed Treatment Wetlands

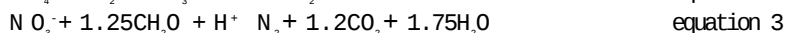
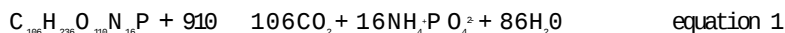
Sedimentation and filtration remove most of the suspended solids that remain in effluent from the aerated lagoons. Further BOD removal occurs by both anaerobic and aerobic processes. Nitrification occurs in aerobic zones and denitrification occurs in anaerobic zones, fueled by plant carbon, converting nitrate to harmless nitrogen gas (N_2)

Soil Aquifer Treatment System (SAT)

Effluent from the wetland is applied to infiltration basins that transmit water to the underlying aquifer. In the upper few meters of the soil, microbial activity degrades organic matter (BOD and DOC) and converts the remaining ammonium to nitrate. Pathogens are removed by straining, adsorption, and biological processes. The effluent moves downward, recharging depleted aquifers (Bower et al. 1980). This treatment has been called soil aquifer treatment (SAT) (Pyne 1995). Specific processes and treatment efficiencies for each component of the proposed treatment system are summarized below.

Nitrogen Removal

Most nitrogen in raw wastewater is ammonium or organic nitrogen. Ammonium is released by the degradation of organic N (equation 1, a modification of the classical Redfield Equation). Nitrification converts ammonium to nitrate (equation 2), consuming oxygen. Denitrification converts nitrate to nitrogen gas under anaerobic conditions (equation 3), consuming organic matter (represented as CH_2O). Most net removal of nitrogen from wastewater occurs by denitrification.



The role of each component of the proposed low-tech treatment system in transforming and removing N is discussed below.

Aerated Lagoons

Organic nitrogen is converted to ammonium during the process of BOD degradation, but lagoons are generally not designed to accomplish nitrification, so there is little nitrate in the effluent from most lagoons. With little conversion from ammonium to nitrate, there is little opportunity for denitrification. Because of this, aerated lagoons typically remove little nitrogen, as seen in relatively high effluent TN concentrations for the NIWTP (see Table 1).

Constructed Treatment Wetland

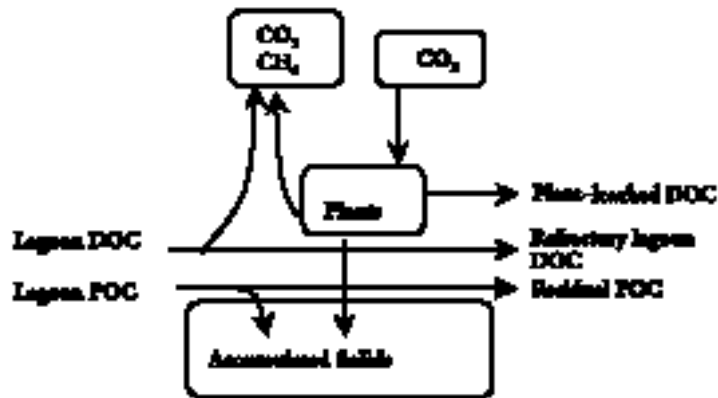
A wetland receiving lagoon effluent would have to accomplish both nitrification and denitrification. The key factor is oxygen supply; from the stoichiometry of nitrification shown above (equation 2), one can infer an oxygen requirement of 4.6 mg O₂/mg NH₄. The oxygen requirement to completely nitrify 20 NH₄-N/L (e.g., Nogales lagoon effluent) is therefore about 80 mg O₂/L. Since the saturation concentration of oxygen at 25°C is only 8 mg/L, oxygen must be supplied to the water from the air. Plants transfer oxygen through porous aerenchyma tissues to the root zone (Schulthorpe 1967). Oxygen mass balance studies have shown that the amount of oxygen that is transferred in this manner is low relative to the oxygen requirement for nitrification of wastewater at loading rates commonly applied to treatment wetlands (Crites 1994; Kadlec and Knight 1996). The main source of oxygen in effluent passing through wetlands is probably simple diffusion to the water surface and alga growth. Thus, wetlands, at least those with dense emergent plants, are not inherently efficient at nitrifying wastewater. Design features that have been used to increase nitrification rates in low-tech treatment systems include open ponds, rock filters, and cascades between cells (Crites 1994; Manthe and Ash 1993; Hammer and Knight 1994; Horne 1995).

By contrast, wetlands are inherently suitable for denitrification, because wetland plants provide the organic carbon needed to fuel the process (equation 3, where CH₂O is provided by decomposing wetland plants) and a decomposing plant mat provides an anaerobic environment in close proximity to the overlying (oxic) water. Several researchers have shown that denitrification rates in wetlands are limited by carbon supply (Gersberg et al. 1983; Ingersoll and Baker 1997; Baker 1998). Ingersoll and Baker (1997) shows that denitrification rates in wetland microcosms supplied with nitrate-enriched water and chopped-up cattails were closely related to the input C:N ratio, and that the denitrification rate could be predicted from the carbon supply. For wetlands receiving nitrified effluent or other high-nitrate waters, denitrification rates up to 50 kg/ha per day have been observed (Horne 1995; Baker 1998).

Nitrogen removal in treatment wetlands is generally based upon simple box models in which N removal is represented by a single, overall reaction rate based upon the input and output of total nitrogen concentrations across the wetland (Kadlec and Knight 1996), where

$$K_N = [\ln[(TN_{out} - C^*)/(TN_{in} - C^*)]/t]h \quad \text{equation 4}$$

Figure 2: Mechanism of Organic Carbon Transformations in Constructed Wetlands



where $TN_{in, out}$ = total nitrogen concentration in the inlet and outlet, respectively; C^* = background TN concentration, which Kadlec and Knight found to be around 1.4 mg/L; t = travel time; and h = depth of the wetland. Using measured values of TN_{in} and TN_{out} for many treatment wetlands, Kadlec and Knight (1996) reports an average K_N value of 22 m/year for surface flow wetlands. However, K_N values are quite variable among wetlands, with a range from 1 m/year to > 60 m/year.

SAT System

Little N removal is likely to occur during recharge of effluent through the vadose zone, although further nitrification occurs (Kopchynski et al. 1996; Wilson et al. 1995). During infiltration, ammonium (NH_4^+) in recharged effluent adsorbs to soils. When recharged basins are periodically dried to reestablish hydraulic permeability, oxygen diffuses into the subsurface and oxidizes NH_4^+ to nitrate. The extent of NH_4^+ sorption to soils is a direct function of the cation exchange capacity of the soils. Subsequent wetting cycles flush nitrate into the groundwater.

Organic Matter Removal

A key function of wastewater treatment systems is the removal of organic matter. In the proposed low-tech system, organic matter would be removed in all three components of the system.

Figure 3: Mechanism for Organic Carbon Transformations in SAT Systems

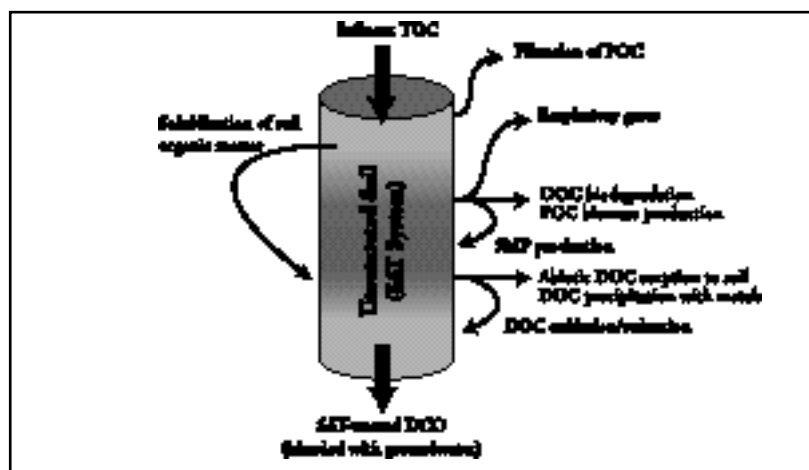


Table 3: Summary of Organic Carbon Removal Rates for SAT Systems

SAT System Description	Organic Carbon Removal	Reference
Laboratory-Scale Mechanistic Experiments		
Aerobic soil columns	28% DOC removal	Drewes and Jolal 1995, Quesada et al. 1996
Anaerobic soil columns	25% DOC removal	
Biologically active 1m soil columns	43% DOC removal (added DOC=12 mg/L) 17% removal of indigenous organics	
Biologically inhibited columns	19% DOC removal, suggesting biodegradation is more important than DOC sorption	
Field-Scale SAT Applications		
Application of activated-sludge treated wastewater to an SAT system	58% decrease in TOC over 6.1 m of soil	Army et al. 1995
Application of trickling filter wastewater to an SAT system	77% decrease in DOC over 5 m of soil 35% decrease in DOC over 38 m of soil, although some groundwater diffusion occurred	Quesada et al. 1997
Evapotranspiration beneath agriculture fields irrigated with wastewater	DOC > 20 mg/L in percolate suggesting leaching of soil organic matter, or unaccountable DOC removal	Amiel et al. 1998
Application of Secondary-treated wastewater to an SAT system	54% to 59% removal of non-indigenous organics, low removal of indigenous organics (e.g., DHP)	Drewes and Nève 1984
Application of treated wastewater to an SAT system	Removal of indigenous organics (> 65%) was higher than DOC removal rates	Drewes et al. 1996; Wilson et al. 1997
Application of treated wastewater containing surfactants (LAS) to an SAT system	Partial LAS removal occurs Residual LAS accounted for 38% (2.1 mg/L) of the DOC after SAT, and LAS may be a reason for observed DOC increase in groundwater	Held et al. 1993, Krugger et al. 1996; Zeller 1996

Table 4: Pathogen Removal in Aerated Lagoons

System	Supposed Natural Mechanism	Bacteria (%)	Virus (%)	Conditions	Reference
Anoxic Lagoon	Aggregation	95-97 97.3-97.9 99.9		Domestic sewage Low-organic loading High-organic loading Constant addition	Maslow and Palmer (1966) Mallory and Malina (1970) Mallory and Malina (1970) Black and Smith (1986)
	Adsorption & predation		98 98 98 47-98.8 98-99	Microcosm bacteria Hep-destroying bacteria Protozoa/infusoria passive Protona present Protona present	Kim and Ueno (1996) Kim and Ueno (1996) Kim and Ueno (1996) Goh et al. (1988) Schmidt et al. (1989)
Activated Sludge	Aggregation	98-99		High-organic loading	Wheeler and Odeh (1974)
	Adsorption		98 98	No protona present Protona present	Odeh and Fay (1988) Odeh and Fay (1988)
Subsidence Pond	Not removed		71-99.9	High-organic loading	Wheeler and Odeh (1974)
	Die-off		95.8-97 94.4-95.8 97-99 91-99	Winter Summer Autumn Non-summer	Goldschmidt et al. (1990) Goldschmidt et al. (1990) Fernandez et al. (1990) Fernandez et al. (1990)

Aerated Lagoons

BOD is removed in lagoons by settling and biological degradation. Aeration, long detention times, and a nutrient-rich medium result in high BOD removal rates (e.g., an average of 94% treatment efficiency in the NIWTP lagoon as shown in Table 1).

Constructed Treatment Wetlands

Wetlands are efficient at removing organic matter via aerobic and anaerobic processes (Figure 2) (Rich and King 1999; Segers 1998; Polprasert et al. 1998a, b; Bhamidimarri et al. 1991). Macrophytes filter particulate organic matter (POM) and improve BOD removal (Brix 1997; Tanner 1996). However, the decay of wetland plants contributes DOC. In wetlands receiving highly treated effluent, DOC in the effluent may actually be higher than the DOC in the influent (Horne 1995). Several process models for constructed wetlands have been proposed (Buchberger and Shaw 1995; Chen et al. 1999), but the simple, first-order, area-based pollutant reduction models (TOC, BOD, or COD removal) analogous to equation 4 are generally used for design. Typical rate constants for BOD removal in wastewater treatment wetlands are $k = 47.5 \text{ m/yr}$ and $C^* = 6 \text{ mg/L}$ (Kadlec and Knight 1996; Knight et al. 1999). Wetlands regularly achieve 80% to >90% BOD removal and effluent levels < 20 mg/L (Verhoeven and Meuleman 1999; Moreno et al. 1994; Zachritz and Fuller 1993; Bhamidimarri et al. 1991; Greenway and Woolley 1999).

Table 5: Pathogen Removal in Constructed Wetlands

Suggested Fertilizer Mechanism	Bacteria (%)	Fungi (%)	System	Reference
All nitrogenous		74 82-88	Mineral Marine substrate	Gooding et al. (1987) Lafitte and Gauthier (1978)
Phos- nitrogenous	79-89 83-93.4 85.59 92.36 97.59		PCW SBCW PCW+SBCW Bamboo plants Flowering plants	Bandhoo et al. (1994) Bandhoo et al. (1992) Bandhoo et al. (1994) Zai and Wang (1993) Zai and Wang (1992)
Phos- nitro- phos	$\bar{x} = 0.35 \text{ g}^{-1}$ $\bar{s} = 0.04 \text{ g}^{-1}$		PCW SBCW	Bandhoo et al. (1994) Gooding et al. (1987)
Other fertilization	$\bar{x} = 0.928 \text{ g}^{-1}$ $\bar{s} = 0.051 \text{ g}^{-1}$		Organic waste Organic waste	Gooding et al. (1988) Chakraborty et al. (1986)
Not available	84-85.3 90		3 plant types Compost, peat	Ward et al. (1994) Gooding et al. (1988)

SFCW (surface flow constructed wetland); SSFCW (subsurface flow constructed wetlands).

Table 6: Pathogen Removal across Soils Systems Representative of Infiltration Systems

[illegible]

SAT System

SAT treatment efficiently removes total organic carbon (TOC). Transformations of organic matter that occur as effluents move through soils include filtration of POC, biodegradation of DOC, sorption and

precipitation of DOC, partial oxidation/reduction reactions, and co-metabolism (McCarty et al. 1981; Drewes and Jekel 1996) (Figure 3). DOC transformations in SAT systems generally occur within the first few meters of soil (Wilson et al. 1995). The near-surface *schmutzedecke* layer and continued fixed- and free-living bacterial biofilms create mixed zones of aerobic and anaerobic conditions. Research summarized in Table 3 shows that organic carbon removal efficiencies in SAT systems vary from ~25% to ~90%.

Pathogen Removal

Pathogen removal would also occur in all three components of the proposed treatment system. Pathogen removal rates in each component are presented in Tables 4 (lagoons), 5 (wetland treatment systems), and 6 (SAT systems).

Aerated Lagoons

Microbial pathogens are removed in aerated lagoons by aggregation, adsorption, predation, and die-off (Table 4). Larger bacteria and protozoa aggregate into flocs that settle outside of the water column (Mezrioui and Baleux 1994; Melbart and Malina 1974). High salt levels (Zita and Hermansson 1994) and high organic loadings (Melbart and Malina 1974; Finch and Smith 1986) enhance flocculation. Viruses are removed from the solution by adsorption into settling flocs (Clarke et al. 1961; Schneiter et al. 1984; Gerba et al. 1980). Adsorption does not inactivate viruses, and virus desorption can occur with changing water quality conditions (Glass and O Brian 1980; Moore et al. 1975; Kim et al. 1995; Kim and Unno 1996; Zita and Hermansson 1994). Pathogens are also inactivated by predation from protozoa and other larger microbes. Predation is important for the permanent removal of viruses and bacteria (Curds and Fey 1969; Kim and Unno 1996). Non-pathogenic microorganisms can also secrete compounds that act as a virucide (Barzily and Kott 1989).

Removal rates for bacteria and viruses in biological treatment systems are substantial. While living outside their host organisms (e.g., humans), pathogen die-off rates increase with increasing pH, sunlight intensity, and large deviations from neutral pH (Geldreich et al. 1964; Sarikaya and Saatci 1987; Hoglund et al. 1998; Fallowfield et al. 1996; Gibbs et al. 1995; Mayo 1995).

High removal rates have also been reported for protozoan cysts. Removal efficiencies of 99% and 99.9% have been reported for *Giardia lamblia* in waste stabilization ponds and conventional wastewater treatment plants, respectively (Grimason et al. 1996; Mayer and

Palmer 1996), with somewhat lower removal rates for *Cryptosporidium* sp. (Mayer and Palmer 1996).

Constructed Treatment Wetlands

Important pathogen removal mechanisms include aggregation of microbes, adsorption to soil, adsorption to plants, solar disinfection, predation, and chemical inactivation (Table 5). Many of these processes are similar to the removal mechanisms that occur in aerated lagoons. Pathogen concentrations in wetlands can increase as a result of bacterial regrowth, wildlife defecation, and stormwater runoff (U.S. EPA 1983; Martin and Johnson 1995; Gersberg et al. 1995), yet, overall, constructed wetlands achieve high removal efficiencies of bacteria, protozoa, and viruses.

Microbial interaction with soil and plants is an important removal mechanism. Microbial removal rates in newly planted wetlands are not statistically different from removal rates in mature wetlands with large plants (Tanner et al. 1995; Shi and Wang 1991), suggesting that interaction with the soil alone is sufficient to achieve greater than 99% removal of fecal coliform after seven days. Gersberg et al. (1987) reports a lower inactivation rate constant for a surface flow wetland ($k = 0.29 \text{ day}^{-1}$) than for a subsurface flow system ($k = 0.86 \text{ day}^{-1}$), confirming the importance of soil interactions. However, Gearheart et al. (1989) observed higher removal of a bacteriophage when municipal wastewater was applied to a vegetated wetland than an unvegetated wetland, suggesting that plant interactions also contribute to pathogen removal. Solar radiation also inactivates pathogens (Wegelin et al. 1994; Acra et al. 1990). Compared to bacteria, viruses required three times more radiation and protozoan cysts required 15 times more radiation to be inactivated (Chang et al. 1985). The presence of particulates can reduce the importance of solar inactivation, while the presence of DOC can improve solar inactivation through the production of virucidal agents (Wegelin et al. 1994). Gersberg et al. (1987) measured virus decay rates in stagnant and flowing water. A lower rate constant ($k = 0.012$ to 0.028 hour^{-1}) was observed in stagnant water compared to flowing water ($k = 0.44$ to 0.502 hour^{-1}). Based on these observations, the authors conclude that predation and chemical inactivation contributed little to overall virus removal.

Overall, wetlands can provide efficient (> 90%), sustainable removal of bacteria and viruses. There is insufficient literature to evaluate protozoan cyst removal in wetlands. Further research in this area is needed.

SAT System

Infiltration systems readily remove biological particles (e.g., bacteria, protozoa, viruses) (Table 6). Basic mechanisms responsible for pathogen removal are filtration and inactivation (i.e., die-off). Particle filtration involves both transport and attachment processes (Bales et al. 1993; Fontes et al. 1991). Straining occurs when the diameter of the pathogen (d_p) is large relative to soil pore size (d_n) ($d_n/d_p < 20$). The d_n/d_p ratio is on the order of 1000 for viruses and less than 100 for bacteria (McDowell-Boyer et al. 1986). Consequently, straining is negligible for viruses, but is important for larger bacteria and protozoa.

Very small particles (e.g., viruses) must be transported to the surface via sedimentation, interception, or diffusion. The surface chemistry of both the pathogen and the soil surface determine whether the pathogen sticks to the media surface or is repelled (Harvey and Garabedian 1991). The surface charge of both clay particles and viruses depends on pH; pH therefore affects attachment and detach-

Figure 4: Locations of Sites Used in this Study



ment (Drewery and Eliassen 1968; Pieper et al. 1997; Zerda et al. 1985; Taylor et al. 1981; Bales et al. 1989). Hydrophobic attraction between nonpolar lipid-containing portions of the pathogen and soil surface can be important, especially at elevated pH levels (e.g., pH > 9) (Bales et al. 1991, 1993). Cations present in water can enhance pathogen attachment by forming salt bridges between negatively charged pathogen and soil surfaces (Bales et al. 1991; Lance and Gerba 1984a, b; Fontes et al. 1991). Electrostatic repulsion between pathogen and soil surfaces decreases with increasing ionic strength, so higher salinity enhances the removal of small pathogens (Pieper et al. 1997; Lance and Gerba 1984a, b; Ryan and Gschwend 1994).

Soil composition and in-situ conditions affect the extent of pathogen removal (Blanc and Nasser 1996). Clays have a much higher surface area than sand and adsorb more viruses (Schaub and Sorber 1977; Jin 1997). However, clays have low permeability and, therefore, clay-based soils are generally not suitable for SAT systems. Fine sands have been reported to remove pathogens faster, over a shorter distance, than coarse sand (0.56 mm) (Farooq and Al-Youssef 1993). Virus removal is also better in saturated soils than in unsaturated soils, possibly because flow velocities are lower and the liquid film thickness is smaller under unsaturated conditions (Powelson and Gerba 1994; Lance and Gerba 1984a, b). Organic matter readily sorbs to soil surfaces, decreasing pathogen attachment potential (Jin 1997; Jansons et al. 1989; Pieper et al. 1997; Johnson and Logan 1996). The presence of microbial biofilms, and the associated predation of pathogens, generally improves overall pathogen removal (Schaub et al. 1982; Hurst et al. 1980; Powelson et al. 1993; Weiss et al. 1995).

SAT systems demonstrate very efficient removal of bacteria and viruses. However, increased pathogen loadings or changes in water chemistry can lead to detachment and pathogen migration, resulting in the detectable presence of pathogens in groundwater (Wellings et al. 1975; Yanko et al. 1999; Pedley and Howard 1997). Therefore, disinfection (e.g., chlorine or UV irradiation) is encouraged upon groundwater withdrawal for human consumption.

LABORATORY AND FIELD STUDIES

The conceptualization outlined above led to the following questions, which required further research:

1. How can nitrogen removal processes be predicted more accurately, particularly for wetlands in the U.S. Mexican border region?

Figure 5a: Transformations in Nitrogen Species, February 1997

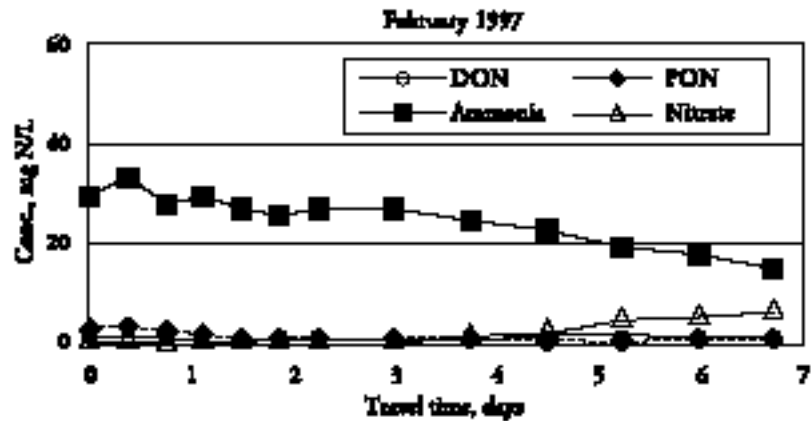
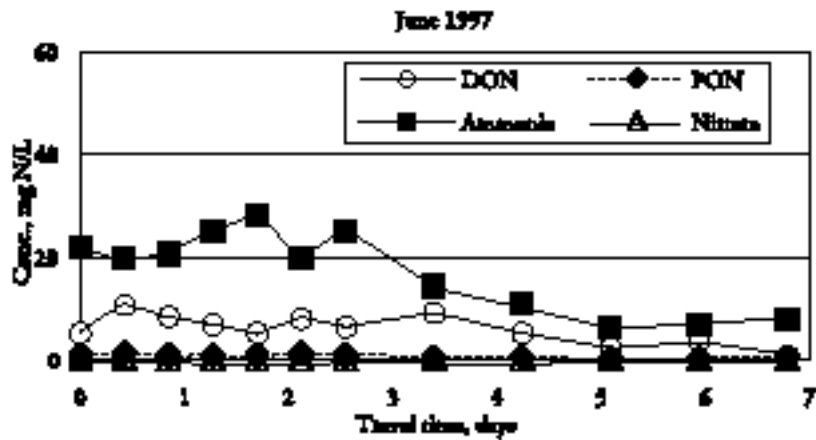


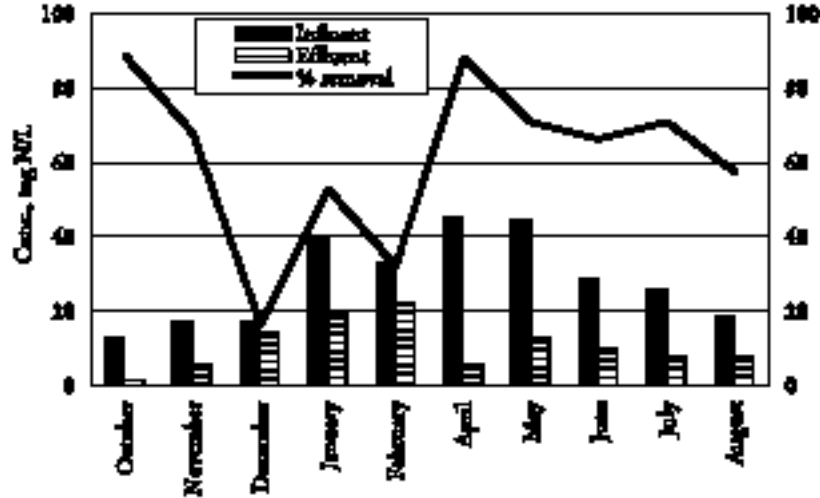
Figure 5b: Transformations in Nitrogen Species, June 1997



Source for Figures 5a, b: Gerke (1997).

2. What are the sources and fate of DOC in wetlands and SAT systems?
3. How does wetland design influence the hydraulic characteristics of wetlands?
4. What impact would diverting wastewater from small, effluent-dominated rivers have on the hydrology of these rivers?
5. How can treatment wetlands in this region be designed to optimize both pollutant removal and wildlife habitat?

Figure 6: Total N in the Inflow and Outflow of the Kingman Wetland



Source: Adapted from Gerke et al. (in review).

Research Sites

Research for this study was conducted at several sites: (1) a full-scale treatment wetland in Kingman, Arizona; (2) 12 wetland research cells at the Tres Rios Wetland Demonstration Project in Phoenix, Arizona; (3) the Environmental Engineering laboratory at Arizona State University; and (4) the region surrounding the NIWTP (Figure 4).

A constructed wetland in Kingman, Arizona, served as a nearly ideal surrogate for wetland treatment systems along the U.S. Mexican border for three reasons:

- (a) The climate in Kingman is nearly identical to the climate along the Arizona portion of the U.S. Mexican border.

$$\frac{d [\text{org N}]}{dt} = -k_1 [\text{org N}]/h \quad \text{equation 5}$$

$$\frac{d [\text{NH}_4^+]}{dt} = k_1 [\text{org N}]/h - k_2 [\text{NH}_3]/h \quad \text{equation 6}$$

$$\frac{d [\text{NO}_3^-]}{dt} = k_2 [\text{NH}_3]/h - k_3 [\text{NO}_3^-]/h \quad \text{equation 7}$$

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- (b) As with many border cities, Kingman has had a lagoon in place for many years, and had built the wetland to improve the quality of effluent.
- (c) An important treatment objective of the Kingman wetland was to meet a 10 mg/L total N standard so the effluent could be recharged. Cities along the U.S. Mexican border that want to recharge wastewater would also have to meet this standard.

The Kingman wetland is a free-surface wastewater treatment wetland that consists of three long cells (700 m x 50 m) connected in series. The shallow zones (~0.2 m deep) were planted with *Scirpus* in 1994; however, the wetland was a mixture of *Scirpus* and *Typha* by the time our study began (1996). Each cell is transected by two internal deep zones (1 m in depth) and includes an open pool near the outlet. The difference in plant density at the Kingman wetland between the time of construction and after three years of operation shows that the vegetation in treatment wetlands in this region

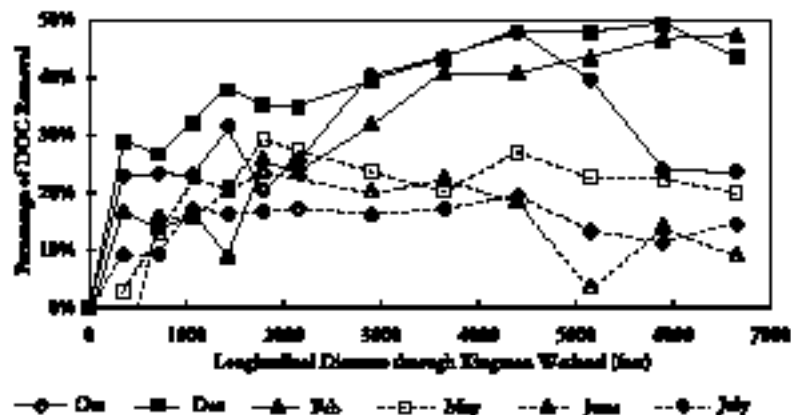
Table 7: Coefficients Developed for the Wetland DOC Model

	K_{dep}	K_{den}	K_{so}
Winter	0.01	0.006	0.19
Summer	0.14	0.046	4.49

reaches maturity very quickly. This wetland was the focus of our studies of nitrogen and carbon dynamics.

Wetland hydraulics was studied in the research cells at the Tres Rios Wetland Demonstration Project near the 91st Avenue Wastewater Treatment Plant in Phoenix. Several design configurations at this site allowed us to compare the effects of varying numbers of deep zones on the hydraulic characteristics of wetlands. Laboratory infiltration columns were used to study the fate and transport of DOC moving through the vadose zone. These columns were filled with soils from several potential recharge sites in Nogales and operated with treated wastewater effluent collected from the NIWTP. Finally, the impact of recharging wastewater from the NIWTP was studied using hydrologic modeling and hydrogeologic data from the Arizona Department of Water Resources.

Figure 7: DOC Removal in Wastewater Passing through the Kingman Wetland



Source: Pinney et al. (2000).

Nitrogen and Carbon Transformations in the Kingman Wetland

Transformations of nitrogen and carbon in the treatment wetland at Kingman, Arizona, were studied for one year. The goals of this study were (1) to develop a model of sequential nitrogen transformations appropriate for treatment wetlands in the border region, (2) to identify rate-limiting factors for nitrogen removal, (3) to develop ideas to improve nitrogen removal, and (4) to develop a conceptual model of DOC transformations in wetlands.

Samples were collected at 13 locations along the longitudinal axis of the Kingman wetland on 10 occasions over the course of a year, and were analyzed for nitrogen species (dissolved organic N, particulate organic N, ammonium, and nitrate), DOC, POC, and temperature on every sampling event. BOD and suspended solids were analyzed on several occasions. Details regarding the wetland design and experimental measurements are presented in Gerke (1997), Gerke et al. (in review), and Pinney et al. (2000).

Nitrogen Transformations

Nitrogen was removed in the Kingman wetland with an average efficiency of 66%. TN removal was generally less during the winter than the summer (Figures 5a, b). Most of the nitrogen in the lagoon effluent was in the form of organic nitrogen or ammonium. As the lagoon effluent moved through the wetland, concentrations of particulate organic nitrogen (PON) declined quickly in the first wetland cell (Figures 5a, b). Mineralization of PON produced ammonium (equation 1), consequently, ammonium concentrations often increased slightly in

the first cell, then subsequently declined, presumably from nitrification. Nitrate was then removed from the system by denitrification. During the winter, conversion from ammonium to nitrate resulted in elevated nitrate levels within the wetland. During the summer, denitrification was so rapid that nitrate concentrations consistently remained $< 1 \text{ mg NO}_3\text{-NL}$.

This series of reactions lent itself to the development of a sequential model in the following form:

In this model, organic N in the effluent degrades to yield ammonium (rate constant k_1 , equation 5). Mineralization produces ammonium, which, in turn, is lost by nitrification (rate constant k_2 , equation 6). Nitrate concentrations are represented as a balance between gains of nitrate from nitrification and losses of nitrate from denitrification. Although this model does not explicitly recognize plant uptake and subsequent degradation, the contribution of these processes to the long-term N balance of a heavily N-loaded treatment wetland is generally small.

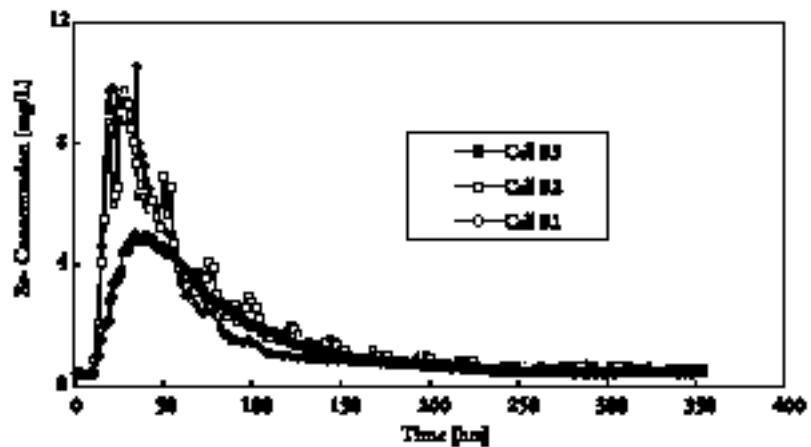
This model was calibrated with data from the Kingman wetland from October 1996 to August 1997 using formal optimization methods (Gerke et al. in review). Average seasonal coefficients were used to validate the model using data from 1998 and early 1999 (Table 7). Analysis using these data showed that the model captures the general seasonal trends in TN removal and effluent ammonium very accurately (Gerke et al. in review). Effluent nitrate concentrations were not quite as accurate, mainly because calibration of the denitrification rate coefficient (equation 7) was not as successful as calibration of the other coefficients.

Despite several limitations, the calibrated model is useful as a design tool for similar constructed wetlands in the border region. Theoretically, the calibrated model would be applicable to a range of influent types from non-nitrified to fully nitrified wastes but verification of the model for nitrified wastes needs to be conducted in wetlands that receive nitrified effluents.

The study also reveals potential limitations for nitrogen removal. In particular, it appears that denitrification is limited by organic carbon supply in the winter. Several points support this assertion:

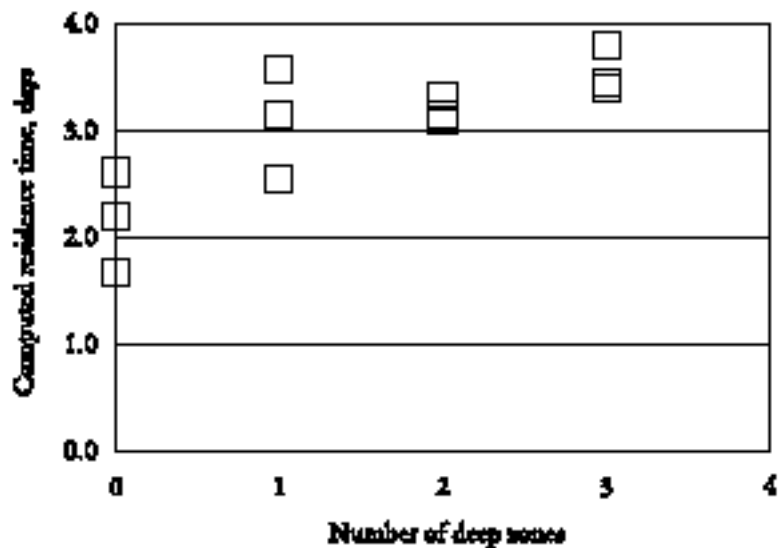
1. The ratio of plant carbon to effluent nitrogen loading (C:N ratio) was about 7:1. This is barely above the 5:1 ratio determined to be optimal in wetland microcosms (Ingersoll and Baker 1997). The same experiment shows that the denitrification rate coefficient was directly proportional to organic carbon supply.
1. The order of magnitude difference between winter and summer denitrification rate coefficients is much higher than would be

Figure 8: Bromide Recovery Curves for the Wetland Research Cells at the Tres Rios Wetland Demonstration Project



Note: The background bromide concentration is ~0.25 mg/L.
Source: Whimier and Baker (in review).

Figure 9: Computed (Actual) Residence Time Versus Number of Deep Zones for the Research Wetland Cells at the Tres Rios Site



Note: Theoretical hydraulic retention times were 4.6 to 5.1 days.
Source: Whimier (1998)

expected by a metabolic temperature effect over the 14° C range in water temperature throughout the year. This suggests that something other than temperature (e.g., organic carbon supply) controls the denitrification rate.

¹ In late winter, the wetland effluent BOD_5 was < 5 mg/L. Measured biodegradable dissolved organic carbon (BDOC) in the wetland effluent was only 3 mg/L in March. These measurements indicate that there was very little degradable organic carbon in the effluent that could support denitrification.

¹ Effluent nitrate concentrations were much higher (up to 8 mg $N O_3^-$) in the winter than in the summer (< 1 mg NO_3^-).

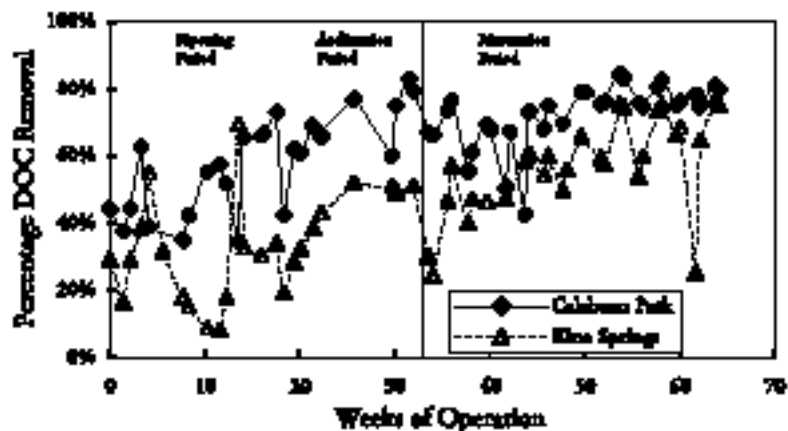
If denitrification rates are limited by organic carbon during winter, a logical method to improve N removal would be to increase the supply of organic carbon during the winter. This could be done by adding an external supply of carbon during the winter (e.g., hay) or by modifying the wetland design to provide additional organic carbon to enhance denitrification rates.

Sources and Sinks of DOC

As noted earlier, it would be desirable to keep DOC concentrations in wetland effluent low in order to minimize DBP formation, yet high enough to facilitate denitrification. DOC removal in the Kingman wetland changed along its length (Figure 7). On the average, DOC declined by 27% with higher net removal occurring in the winter (~45%) and lower net removal occurring in the summer (~15%). The chemical nature of DOC changed throughout the year. Ultraviolet adsorption at 254 nm (UVA) normalized to DOC concentration (specific UV adsorption, or SUVA) is a relative measure of the aromatic carbon content of DOC; SUVA values increase in proportion to the degree of aromaticity. SUVA increased along the length of the wetland by an average of 44%, with the largest increases (85% to 135%) occurring during the summer when the wetland plants were actively growing. Increasing SUVA values could have occurred by two mechanisms: preferential removal of aliphatic (i.e., biodegradable) DOC from the lagoon effluent, and leaching of aromatic lignin-derived DOC from the wetland plants.

These data suggest that DOC in the effluent from the treatment wetland has two sources: the lagoon effluent (wetland influent), and degradation of plants in the wetland itself (see Figure 2). To develop a model incorporating each of these variables, we split the real world wetland into components in the lab. Biological degradation rates (0.15/day) of lagoon DOC were determined in biological laboratory reactors (Pinney et al. 2000). Production of DOC from wetland

Figure 10: DOC in the Effluent of Laboratory SAT Columns



Note: Columns were filled with soils from two candidate recharge sites near Nogales (Kino Springs and Calabasas Park). Treated lagoon water was infiltrated through the columns for 70 weeks. Source: Westerhoff and Pin-

plants was determined in laboratory microcosms, comprised of water and sediment phases, that were fed known quantities of dried, coarsely chopped cattails each week. In this experiment the fraction of submerged wetland plant carbon (e.g., *Typha*) that became DOC was 5-7% of the amount added. Pinney et al. (2000) developed a general process model that represented both lagoon-DOC biodegradation and leaching of plant-derived material. The model shows that concentrations of DOC in the wetland effluent depend upon the amount of DOC in the lagoon effluent, the flux of DOC provided by decomposing plants in the wetland, degradation rates for both types of DOC, and the residence time of the system. The model predicts that minimum DOC concentrations occur at intermediate hydraulic residence times, about 5-10 days. Higher DOC concentrations at shorter hydraulic residence times (HRT) occur because lagoon-derived DOC has not had adequate time to degrade, while higher DOC concentrations at longer HRTs occur because plant-derived DOC accumulates.

Hydraulic Characterization of the Tres Rios Wetland Research Cells

Unlike engineered concrete-tank structures (e.g., circular clarifiers) where flow characteristics can be carefully controlled, achieving desirable flow characteristics in wetlands can be problematic. Plug flow is theoretically the most efficient hydraulic system for achieving pollutant removal (Tchobanoglous and Schroeder 1985). Achieving true plug flow means that a parcel of water moves through the system with no mixing. In reality, no reactor truly achieves plug flow because dispersion spreads the plug out as it moves through the system, but wetlands can be designed to approximate plug-flow conditions. First, Crites (1994) recommends length:width ratios up to 4:1. Second, the inclusion of deep zones that transect the longitudinal axis of the wetland help to remix water that has become channelized in shallow zones with emergent vegetation. This channelization causes short-circuiting in shallow zones; deep zones tend to remix water and maintain plug-flow conditions. Third, separating the wetland into distinct cells also helps to maintain plug-flow conditions. Even if the individual wetland cells were completely mixed, near-plug-flow conditions for the overall system would be approximated with four or more cells in series (Tchobanoglous and Schroeder 1985).

The effect of deep zones on wetland hydraulics was examined in the experimental wetland cells at the Tres Rios Wetland Demonstration Project (Whitmer 1998). Each cell was 1200 m² with a 2:1 length:width ratio. There were four design configurations: no internal deep zones, one deep zone, two deep zones, and three deep zones. There were three cells having each configuration, for a total of 12 cells. All had hydraulic loading rates of 8.3 cm/day, resulting in HRTs of 4.1 days (for zero deep zones) to 5.1 days (for three deep zones).

Bromide tracer was added to each of the experimental cells and bromide was analyzed in the effluent over a two-week period. Bromide recovery curves (Figure 8) were analyzed to determine hydraulic characteristics of each cell. Bromide recovery was poor, presumably due to plant uptake (Whitmer and Baker in review), but tracer curves still allowed the computation of effective HRTs (the point at which 50% of the recovered bromide passed through the outlet weir). The effective HRT varied with the number of deep zones (Figure 9). For the cells with no deep zones, the average measured HRT was only 2.2 days (compared with a theoretical HRT of 4.1 days). The measured HRT increased to 3.6 days (compared with a theoretical HRT of 5.1 days) for cells with three deep zones. Most of the increase in measured HRT came from the inclusion of one deep zone; addi-

tional deep zones had lesser effects. In summary, a conclusion drawn from the hydraulic tests is that the inclusion of at least one or two deep zones improves the hydraulic performance of treatment wetlands, but additional deep zones do little to improve hydraulic performance.

Laboratory Simulation of Soil Aquifer Treatment

DOC Removal During Infiltration

Two soils were collected below the root zone near the U.S. Mexican border in Nogales, Arizona, at sites that are viable recharge locations near the Santa Cruz River: Calabasas Park (CP soil) and Kino Springs (KS soil). The soils were sieved to remove particles greater than 4.75 mm, and both were characterized as sandy loam based upon the U.S. Department of Agriculture soil classification. KS soil had a higher organic carbon content (4.5 mgC/gsoil) than the CP soil (2.8 mgC/gsoil). Each soil was packed into separate laboratory plexiglass columns (82 cm long x 7.6 cm diam.) at a porosity similar to field conditions. Lagoon effluent from the NIWTP collected biweekly was passed through the soil columns by gravity (1 cm hydraulic head) and collected twice over a seven-day period. During the second seven-day period the columns were allowed to dry. This approach simulated the wet-dry cycling commonly used in full-scale SAT systems to maintain high permeability during wetting cycles.

The DOC of column influents (lagoon effluent from the NIWTP) ranged from 12 to 20 mg/L. Net DOC removal represented a balance between removal of lagoon-derived DOC and leaching of initial soil organic carbon. During 65 weeks of operation the performance of the simulated SAT soil columns were marked by three periods of performance for DOC removal: a ripening period (weeks 0 to 12), an acclimation period (weeks 13 to 35), and a maturation period (from week 36 onward) (Figure 10).

During the ripening period (weeks 0 to 12) infiltration rates declined from 14 L/week to 2 L/week as soils became saturated and a biological *schmutzdecke* biofilm developed. DOC concentrations exiting the columns were between 4 and 8 mg/L, representing average removals of 25–40%. Net DOC removal was lower in the KS soil column than in the CP soil column, probably because the KS soil had a higher initial organic carbon content and may therefore have leached more DOC into the solution than did the CP column.

During the acclimation period (weeks 13 through 35) infiltration rates gradually declined to less than 0.5 L/week. DOC removal stabilized at 66% removal for the CP soil and 40% removal for the KS soil.

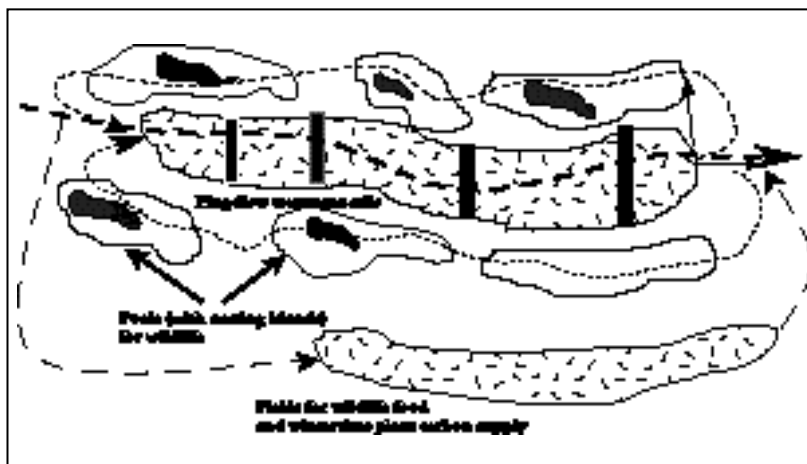
Since the KS soil had a larger fraction of fine-grained material (41%) compared against the CP soil (23%), less water infiltrated through the KS soil. DOC loading (DOC concentration multiplied by infiltration volume) was therefore lower for the KS soil column. Average DOC removal was 2 mg/L for the CP soil column, but only 0.9 mg/L for the KS soil. The significant difference in observed DOC removal performance between the two soils was attributed to variations in the fraction of organic carbon and percentage of fines.

The maturation period (weeks 36 through 65) was characterized by very low, yet steady, infiltration rates and constant DOC removal. Average DOC concentrations exiting the CP and KS soil columns were 3.7 and 5.8 mg/L, respectively. SUVA increased from between 1 and 2 $\text{m}^1(\text{mg/L})^{-1}$ in the wastewater being applied to the soil columns to an average of 4.2 $\text{m}^1(\text{mg/L})^{-1}$ in both of the CP and KS soil column effluents. The increase in SUVA represented a preferential removal of non-aromatic DOC, such as carbohydrates and other low molecular weight compounds.

To simulate withdrawal of groundwater and chlorination for potable use, water exiting the soil columns was collected and chlorinated to determine trihalomethane formation potentials (THMFP). In this experiment, THMFP was determined using a high level of free chlorine (5:1 chlorine to DOC) and a seven-day reaction period. Wastewater entering the column had an average THMFP of 508 g/L. THMFP levels in effluent that had infiltrated through the soil columns were much lower: 220 g THMFP/L for the CP column and 344 mg THMFP/L for the KS column. However, the reactivity of DOC (reactivity = g THMFP/mg DOC) of effluent from the soil columns (70 g THMFP/g DOC for the CP column and 68 g THMFP/mg DOC for the KS column) was higher than the reactivity of the influent (34 g THMFP/mg DOC). The reactivity of the DOC passing through the soil column was therefore about twice as high as the reactivity of DOC in the influent. In field applications, significantly longer flowpaths and in-situ treatment time would be available for continued removal of DOC. Therefore, the results of this work present a worst-case scenario for SAT treatment.

In the above work, lagoon effluent was passed through the soil columns. The soil columns effectively removed labile (biodegradable) DOC. If the lagoon effluent had been passed through a wetland treatment system before reaching the soil infiltration columns, the nature of the DOC entering the soil columns would have been different. As shown above, constructed wetlands are capable of transforming the DOC pool, resulting in less biologically degradable DOC in the effluent than in the influent (Pinney et al. 2000). Increasing the non-labile

Figure 11: Schematic of a Wetland Complex Designed to Treat Wastewater and Provide Wildlife Habitat



Note: The complex includes a plug-flow wetland, which treats the wastewater, surrounded by pools and associated riparian vegetation. A small flow would be maintained through these pool to prevent stagnation. A vegetated channel would be dry during the summer, providing nesting habitat and food. During the winter, the channel would be used as part of the treatment system, providing additional carbon for denitrification

fraction of DOC has been shown to decrease net removal of DOC during SAT (Drewes and Jekel 1998). Thus, in the proposed low-tech treatment train, much of the labile DOC in the lagoon effluent would already have been removed in the wetland before it reached the SAT system. We would therefore expect to observe lower net DOC removal during soil infiltration in the full-scale treatment system than was observed in this experiment, but the net result would still be DOC in the range of 3-5 mg/L in the infiltrated water.

Hydrologic Analysis For the Use of Effluent at the NIWTP

For the effluent-dominated rivers of the Southwest, recharging wastewater effluent to aquifers rather than discharging it to waterways could reduce river flows during the summer low-flow periods. To address this question, a hydrologic analysis was conducted to determine whether recharging wastewater to sites located upstream of the NIWTP, rather than discharging it directly to the Santa Cruz River, would reduce river flows downstream (McSparran 1998). Hydrologic modeling conducted with MODFLOW showed that groundwater levels in the Santa Cruz River below the NIWTP were declining and

would continue to decline due to overpumping, even if the NIWTP continued to discharge to the river. MODFLOW was used to simulate the effect of diverting effluent from the NIWTP to one of three sites, each in a different alluvial subbasin: the Kino Springs subbasin, the Highway 82 subbasin, and the Buena Vista subbasin. Simulations show that diverting NIWTP effluent from river discharge (the current condition) to aquifer recharge (proposed) at any of these sites has no effect on downstream monitoring wells, indicating that there would be no impact on the flow of the Santa Cruz River. Water levels in the aquifers upstream from the NIWTP would increase as the result of artificial recharge (McSparran 1998).

Quality and quantity of input data limited the accuracy of the hydrologic modeling. Specifically, McSparran (1998) found that MODFLOW simulations were very sensitive to the values used for hydraulic conductivity and recommended that field studies be conducted to more accurately determine hydraulic properties of the aquifers in the region. A second major limitation was the reliability of pumping information, particularly downstream from the NIWTP. Finally, surface elevations used in the model were derived from well elevation data; much better surface contours could be developed using Digital Elevation Model (DEM) data, which was not available at the time.

The model developed by McSparran could be used for other water resource issues. For example, it could be used to predict the effect of new pumping wells, determine the best location for pumping wells, and predict the effect of more complex effluent discharge/recharge scenarios.

Design Considerations for a Low-Tech System to Treat and Reuse Wastewater

Results from this study point to several considerations in the design and operation of components for a low-tech system to treat and reuse wastewater.

Aerated Lagoons

The major role of aerated lagoons is to provide time (and oxygen) sufficient to oxidize organic matter (BOD). For systems adding wetlands and SAT systems to existing lagoons, modification of lagoon operation to maximize nitrification would be an important consideration. Generally this would involve improving the aeration system.

Constructed Wetlands

Nitrification appears to be a key limiting factor in N removal in wetlands, particularly in the summer. Nitrification is probably limited by oxygen transport, so methods to increase nitrification rates generally involve increasing oxygen transport to the water. There are three approaches to doing this: (1) increase the surface area of oxygen transport, (2) decrease the depth of the water, or (3) use photosynthesis to supply oxygen. For nitrogen removal, the required hydraulic residence time of a wetland receiving lagoon effluent would be considerably reduced if the influent to the wetland were already nitrified. Using the sequential model of nitrogen transformations discussed earlier in this chapter, Gerke et al. (in review) predicted that nitrification of the lagoon influent would improve total N removal efficiency of the Kingman wetland from 32.66% in the winter and from 64.97% in the summer.

For many wetlands, denitrification is likely to be limited by carbon supply, at least in the winter. Since the carbon used in denitrification is provided by wetland plants, increasing the supply of plant material would probably increase the denitrification rate and improve overall N removal efficiency. The supply of plant material could be increased, expanding the size of the wetland; however, this would create additional water loss through evaporation and might increase DOC concentrations. A different approach would be to grow grasses and other vegetation in dryland channels adjacent to the main wetland. These channels would be dry during the summer (with occasional irrigation to supply the plants with required water) and then flooded during the winter, becoming part of the treatment system. The vegetation would then be a source of carbon for denitrification. This system would not increase overall water loss by very much, since evaporation rates in the winter are low.

A broader consideration in the design of wetland treatment systems is the integration of wildlife and treatment functions. Maintenance of plug-flow conditions is highly desirable for efficient water quality improvement. However, plug-flow wetlands with dense emergent vegetation have limited wildlife value, serving mainly as a source of food (high-protein seeds) and as a nesting site for a few species. Figure 11 shows how auxiliary channels and wetland ponds might be integrated into a treatment wetland. The auxiliary channels would be dry during the summer, providing nesting sites for ground-nesting birds. In addition, they would be planted with vegetation that would provide food and cover for wildlife in the summer (e.g., Japanese millet). Then the channels would become part of the treatment system in the winter, providing additional carbon supply and surface area.

A Low-Tech Treatment Strategy for Treating and Reusing Wastewater

Wetland ponds surrounding the main wetland channel would include nesting islands, submergent vegetation for dabbling ducks, and shoreline vegetation (e.g., willows, cottonwoods) to attract shrub and tree-nesting birds. A small amount of effluent from the treatment channel would be recirculated through the pond system to prevent water in the ponds from becoming stagnant. This water would go back through the treatment channel before being discharged.

Such a system would provide efficient treatment with minimum evaporation, while providing a high-quality riparian habitat, now rare in the arid border region. On a sufficiently large scale (hundreds of acres), such a wetland could become a major feature of the landscape of the region. This is particularly true for the Nogales area, with its proximity to the famous Arizona sky island bird sanctuaries and a major migration flyway. Such a wetland might also generate significant income from ecotourism.

Recharge and SAT

The decision to recharge wastewater should be made with consideration of its impact on the regional hydrology. Of particular concern in the arid border region is the potential impact of recharging effluent to aquifers rather than discharging it to rivers where it sustains riparian ecosystems. McSparran (1998) demonstrates the application of a regional hydrologic model for making such decisions in the Nogales area. Although the conclusions are somewhat tentative due to limited data, it appears that recharging all of the effluent from the NIWTP would have little effect on flows in the Santa Cruz River. McSparran also predicts that current rates of well pumping downstream from the NIWTP would cause further declines in the aquifer over time. For the Nogales region, as well as other border areas, water management decisions should be made with the support of hydrologic models. Although simple in concept, such models require extensive data input. The development of data bases on well pumping, aquifer and surface elevations, hydraulic conductivities, and other aspects of the system are the limiting factor in using hydrologic models.

With respect to SAT systems, the key design feature is to allow sufficient travel time through the vadose zone to provide adequate treatment of DOC, nitrate, and pathogens. A reasonable suggestion is to provide one year of travel time through the vadose zone and aquifer prior to withdrawal. Chemical disinfection at the well head would be required before the water would be safe for municipal use. However, persistence of high DOC levels could produce DBP levels that exceed health guidelines and regulations. Therefore, it would be

advantageous to blend recharged water with native groundwater (low DOC) prior to recovery and disinfection.

CONCLUSIONS

A low-tech system to treat and reuse wastewater is feasible, at least with respect to treatment of BOD, nitrogen, DOC, and pathogens. Traditional aerated lagoons provide effective removal of BOD and suspended solids. Wetlands provide additional removal of these constituents, but the main role of wetlands is nitrogen removal. A sequential model of nitrogen transformations for wetlands, calibrated to the Kingman wetland, is more appropriate for sizing wetlands in this region than single-parameter models calibrated in other regions. Increasing the rate of oxygen transfer to the water would increase nitrification rates and, therefore, would increase overall nitrogen removal rates in the summer. In the winter, increasing the supply of organic carbon would increase denitrification rates. Pathogen reduction occurs in both the lagoon wetland and the SAT system, yet the SAT system is necessary to reduce pathogen concentrations to very low levels. Mechanisms of pathogen removal include filtration, adsorption, and predation. The SAT system would also reduce DOC to acceptable levels via long-term sustainable biodegradation processes. Recovered water would have to be disinfected upon withdrawal for municipal use.

An integrated approach to providing wetland treatment and wildlife habitat has been developed. A wetland complex, as illustrated in Figure 11, located in a region where wetland and riparian habitats have largely been destroyed, may have substantial ecological and economic benefits that should be evaluated further.

Hydrologic analysis should be incorporated into a feasibility analysis of any effluent recharge system. Hydrologic modeling indicates that recharge of effluent from the NIWTP would not affect groundwater levels in wells below the NIWTP along the Santa Cruz River, suggesting that flows would not be seriously altered. The use of hydrologic modeling would be very useful for a wide range of water management issues along the border. The key limiting factor is data input for these models.

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