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

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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 m^3/year per person and water stress at < 1,700 m^3/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 experienced 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:

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_{25}	Median	Q_{75}
Flow (mgd)	11.3	8.4	10.5	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	5.4	328
pH	7.9	7.5	7.6	8.2
Total organic N (mg/L)	6.4	2.3	5.4	10.9
Ammonium	18.2	9.5	18.8	25.6
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

	Annual volume (acre-feet/year)
Groundwater from Mexico	500
Surface water from Mexico	20,450
Wastewater effluent from NIWTP	13,338
Tributary inflows from U.S.	26,574
Mountain front recharge	15,904
Total inflows	73,575
Surface water outflow	21,960
Phreatophyte consumption	24,098
Groundwater outflow	8,524
Pumpage from wells	19,993
Total outflows	73,575

Source: ADWR 1989. 1 acre-foot = 1233m^3

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 x 10^6 m^3/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 (Stromberg 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-N/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 choleraw, 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 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?

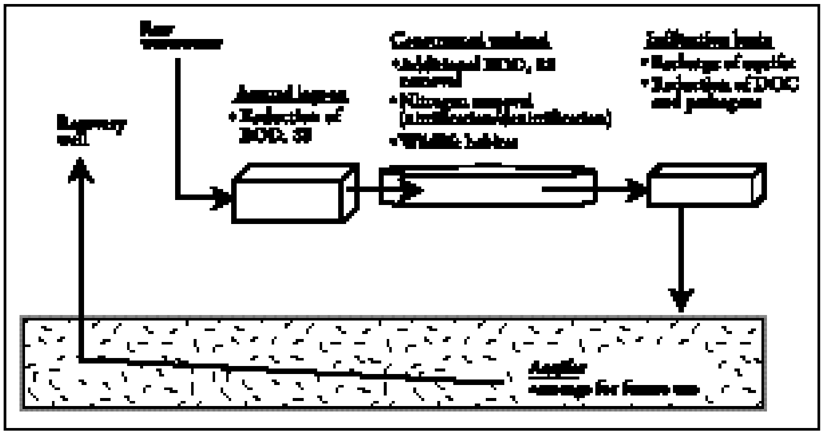
Because the Santa Cruz River below the NIWTP is effluent dominated, recharging effluent to underlying aquifers could result in a dry riverbed during the summer. Loss of streamflow would probably have a negative effect on the well-developed riparian zone below the NIWTP and damage the aquatic ecosystem. 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.

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 (Bouwer 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.

$$C_{106}H_{236}O_{110}N_{16}P + 910 \quad 106CO_2 + 16NH_4^+PO_4^{2-} + 86H_20 \qquad \text{equation 1}$$

$$NH_4^+ + 20_2 \quad NO_3^- + 2H^+ + 2H_2O \qquad \text{equation 2}$$

$$NO_3^- + 1.25CH_2O + H^+ \quad N_2 + 1.2CO_2 + 1.75H_2O \qquad \text{equation 3}$$

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

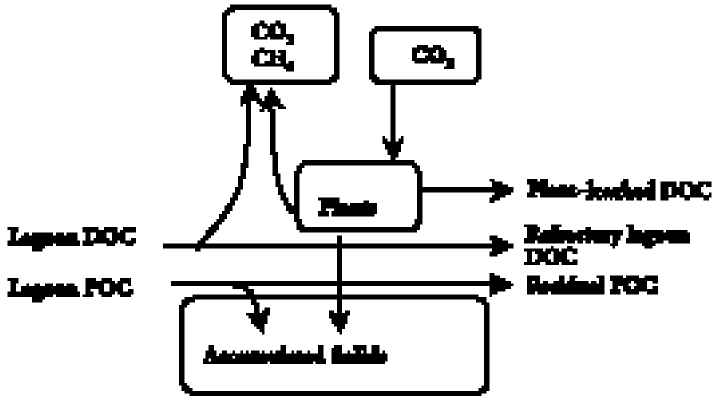
A wetland receiving lagoon effluent would have to accomplish both nitrification and denitrification. The key factor is oxygen supply; from the stoichiometry of nitrification shown above (equation 2), one can infer an oxygen requirement of 4.6 mg O_2/mg NH_4. The oxygen requirement to completely nitrify 20 NH_4-N/L (e.g., Nogales lagoon effluent) is therefore about 80 mg O_2/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 aerencyma 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 CH_2O 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 dentirification 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

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

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

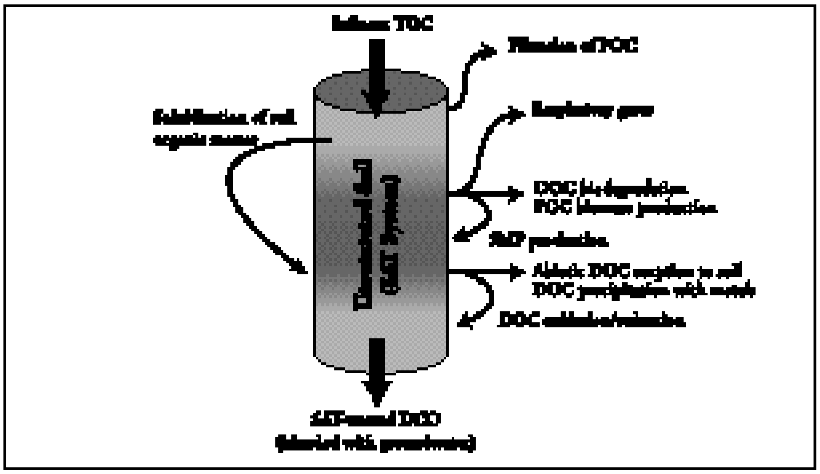


Table 3: Summary of Organic Carbon Removal Rates for SAT Systems

SAT System Description	Organic Carbon Removal	Reference
Laboratory-Scale Mechanistic Experiments		
Aerobic soil columns Anaerobic soil columns	28% DOC removal 25% DOC removal	Drewes and Jekel 1996.
Biologically active 1m soil columns Biologically-inhibited columns	43% DOC removal (initial DOC=13 mg/L) 17% removal of halogenated organics 10% DOC removed, suggesting biodegradation is more important than DOC sorption.	Quanrud et al. 1996.
Field-Scale SAT Applications		
Application of activated-sludge treated wastewater to an SAT system	50% decrease in TOC over 6.1 m of soil	Amy et al. 1993.
Application of trickling filter wastewater to an SAT system	77% decrease in DOC over 5 m of soil 85% decrease in DOC over 30 m of soil, although some groundwater dilution occurred	Quanrud et al. 1997.
Pore water samples beneath agricultural fields irrigated with wastewater	DOC > 20 mg/L in porewater suggesting leaching of soil organic matter, or unmeasurable DOC removal	Arnold et al. 1998.
Application of Secondary-treated wastewater to an SAT system	50% to 97% removal of non-halogenated organics, but lower removal of halogenated organics (e.g., DBPs)	Bouwer and Rice 1984.
Application of treated wastewater to an SAT system	Removal of halogenated organics (> 65%) was higher than DOC removal rates	Drewes et al. 1998; Wilson et al. 1995.
Application of treated wastewater containing surfactants (LAS) to an SAT system	Partial LAS removal occurs Residual LAS accounted for 30% (2.1 mg/L) of the DOC after SAT, and LAS may be a reason for observed DOC increase in groundwater	Field et al. 1992; Krueger et al. 1998; Zoller 1994.

Table 4: Pathogen Removal in Aerated Lagoons

System	Suggested Removal Mechanism	Bacteria (%)	Viruses (%)	Conditions	Reference
Aerated Lagoon	Aggregation	[illegible]		Domestic sewage	[illegible]
		[illegible]		Low-organic loading	[illegible]
		[illegible]		High-organic loading	[illegible]
		[illegible]		Coagulant addition	[illegible]
	Adsorption & predation		[illegible]	[illegible] bacteria	Kim and Unno (1996)
			[illegible]	Floc-forming bacteria	Kim and Unno (1996)
			[illegible]	[illegible]	Kim and Unno (1996)
			[illegible]	Protozoa present	[illegible]
			[illegible]	Protozoa present	[illegible]
Activated Sludge	Aggregation	[illegible]		High-organic loading	[illegible]
	Adsorption		[illegible]	No protozoa present	[illegible]
			[illegible]	Protozoa present	[illegible]
	[illegible]		[illegible]	High-organic loading	[illegible]
Stabilization Pond	Die-off		[illegible]	Winter	[illegible]
			[illegible]	Summer	[illegible]
			[illegible]	Annual	[illegible]
			[illegible]	[illegible]	[illegible]

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 m/yr and C* = 6 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

[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]		[illegible]	[illegible]	[illegible]
[illegible]	[illegible]		[illegible]	[illegible]
[illegible]	[illegible]		[illegible]	[illegible]
[illegible]	[illegible]		[illegible]	[illegible]
[illegible]	[illegible]		[illegible]	[illegible]

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

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

[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]		[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]			[illegible]		[illegible]	[illegible]
[illegible]	[illegible]			[illegible]	[illegible]	[illegible]
[illegible]		[illegible]	[illegible]	[illegible]	[illegible]	[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 Cryptosprodium 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 ($k = 0.29$ day^{-1}) than for a subsurface flow system ($k = 0.86$ 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 virudical 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.

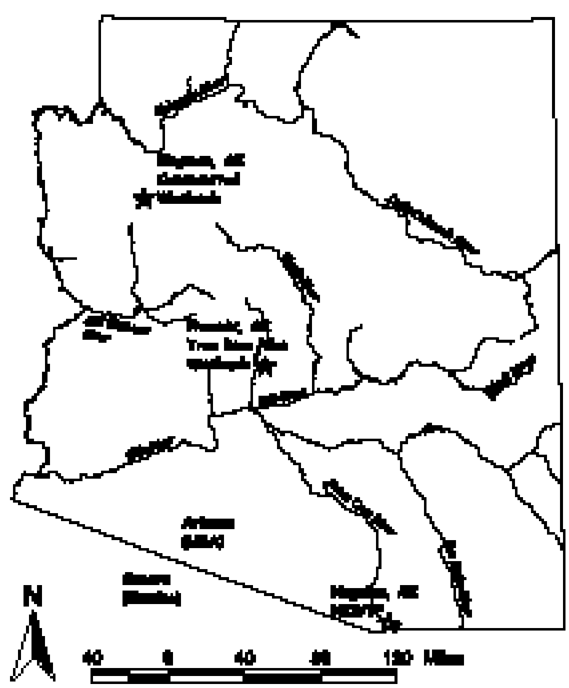
Overall, wetlands can provide efficient (> 90%), sustainable removal of bacteria and viruses. There is insufficient literature to evaluate protozoan cyst removal in wetlands. Further research in this area is needed.

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_m) ($d_m/d_p < 20$). The dm/dp 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

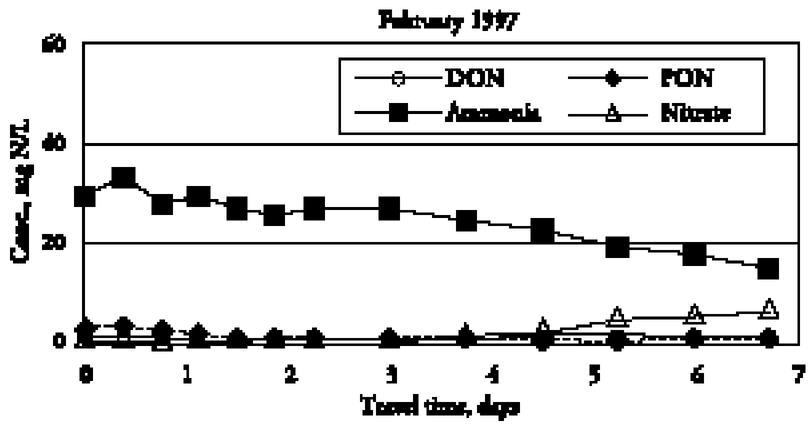
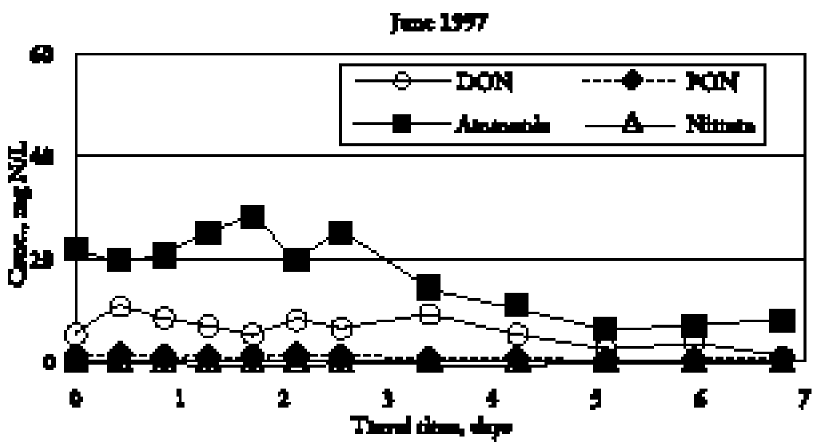


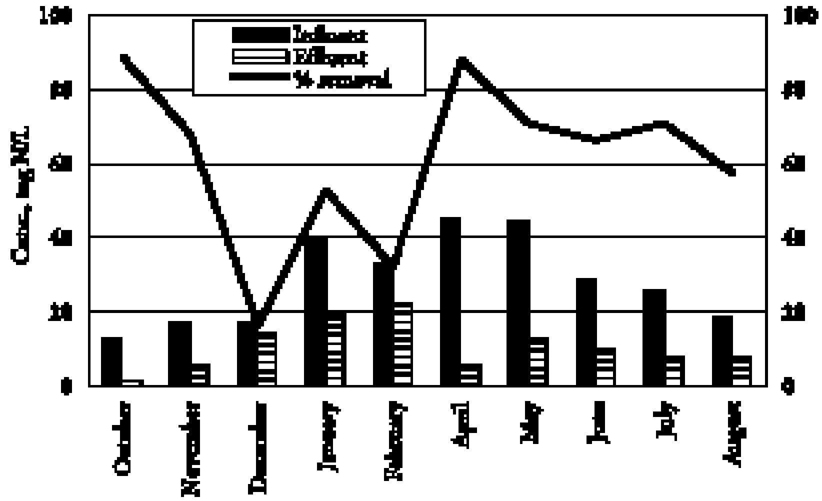
Figure 5b: Transformations in Nitrogen Species, June 1997



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.

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

$$\frac{d\,[NH_{4+}]}{dt} = k_1\,[org\text{-}N]/h - k_2\,[NH_3]/h \qquad \text{equation 6}$$

$$\frac{d\,[NO_3]}{dt} = k_2\,[NH_3]/h - k_3\,[NO_3]/h \qquad \text{equation 7}$$

(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

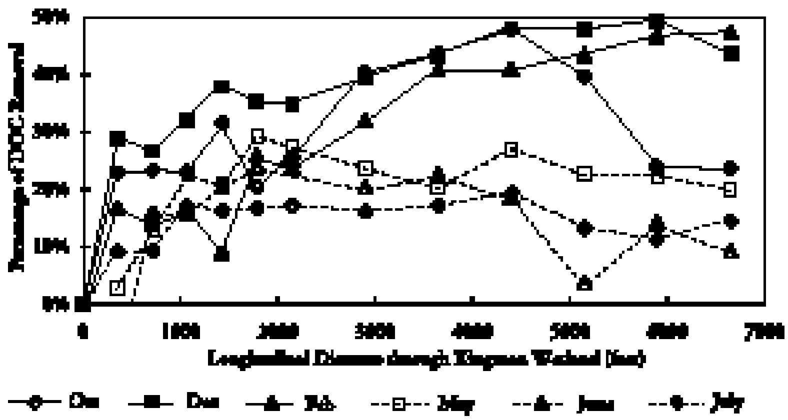
Table 7: Coefficients Developed for the Wetland DOC Model

	[illegible]	[illegible]	[illegible]
Winter	0.01	[illegible]	0.19
Summer	0.14	[illegible]	[illegible]

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 occassions. 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 mg NO_3-N/L.

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.

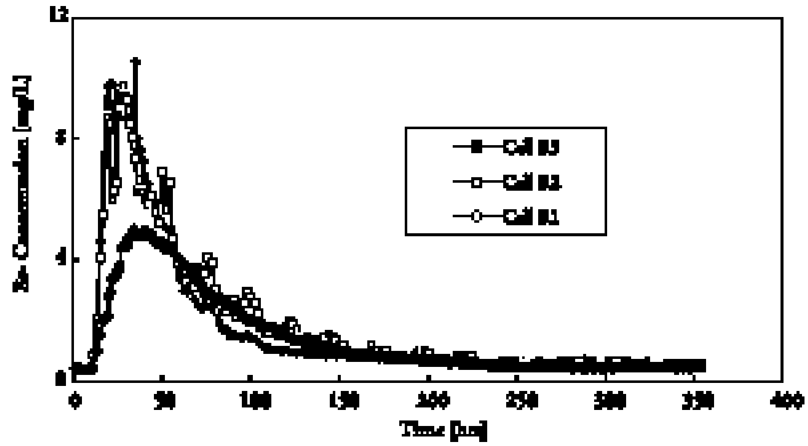
This model was calibrated with data from the Kingman wetland from October 1996 to August 1997 using formal optimization methods (Gerke et al. in review). Average seasonal coefficients were used to validate the model using data from 1998 and early 1999 (Table 7). Analysis using these data showed that 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:

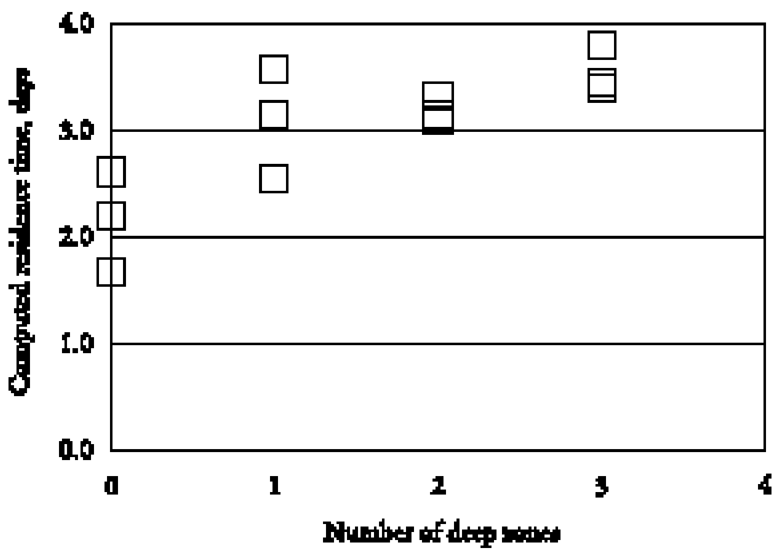
- The ratio of plant carbon to effluent nitrogen loading (C:N ratio) was about 7:1. This is barely above the 5:1 ratio determined to be optimal in wetland microcosms (Ingersoll and Baker 1997). The same experiment shows that the denitrification rate coefficient was directly proportional to organic carbon supply.
- 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: Whitmer 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: Whitmer (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.

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

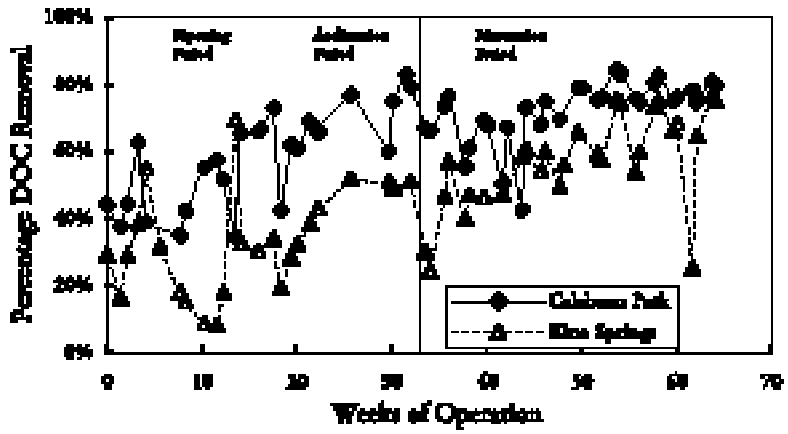
If denitrification rates are limited by organic carbon during winter, a logical method to improve N removal would be to increase the supply of organic carbon during the winter. This could be done by adding an external supply of carbon during the winter (e.g., hay) or by modifying the wetland design to 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 m^2 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; additional 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.

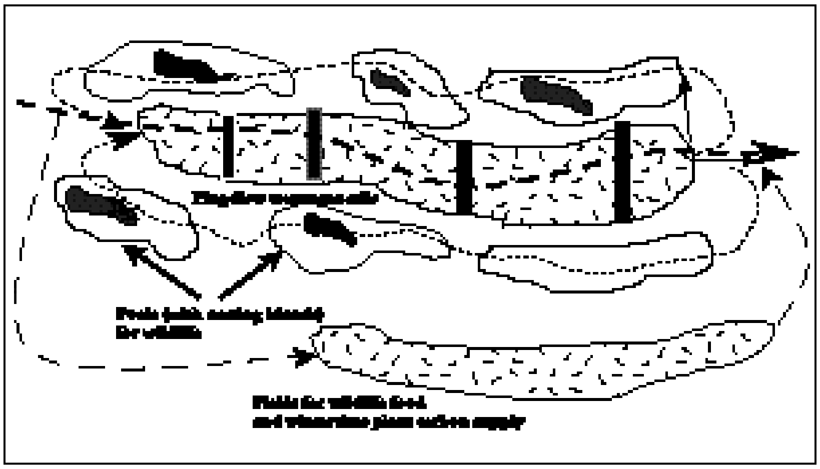
Since the KS soil had a larger fraction of fine-grained material (41%) compared against the CP soil (23%), less water infiltrated through the KS soil. DOC loading (DOC concentration multiplied by infiltration volume) was therefore lower for the KS soil column. Average DOC removal was 2 mg/L for the CP 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 $m^{-1}(mg/L)^{-1}$ in the wastewater being applied to the soil columns to an average of 4.2 $m^{-1}(mg/L)^{-1}$ 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 waste-water, 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 waste-water 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.

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 tis 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 conductivies, 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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REFERENCES

Acra, A., M. Jurdi, H. Mu allem, Y. Karahagopian, and Z. Zaffoul. 1990. Water disinfection by solar radiation: Assessment and application. Ottawa, Ont., Canada: International Development Research Centre.

Amiel, A. J., M. Magaritz, D. Ronen, and O. Lindstrand. 1990. Dissolved organic-carbon in the unsaturated zone under land irrigated by waste-water effluent. Res. J. Wat. Pollut. Con. Fed. 62 (7): 861 66.

Amy, G., L. G. Wilson, A. Conroy, J. Chahbandour, W. W. Zhai, and M. Siddiqui. 1993. Fate of chlorination by-products and nitrogen species during effluent recharge and soil aquifer treatment (SAT). Wat. Environ. Res. 65 (6): 726 34.

Bales, R. C., C. P. Gerba, G. H. Groundin, and S. Jensen. 1989. Bacteriophage transport in sandy soil and fractured tuff. Appl. Environ. Microb. 55 (8): 2061 67.

Bales, R. C., S. Hinkle, T. Kroeger, and K. Stocking. 1991. Bacteriophage adsorption during transport through porous media: Chemical pertubations and reversibility. Environ. Sci. Tech. 25 (Dec.): 2088 95.

Bales, R. C., S. M. LI, K. M. Maguire, M. T. Yahya, and C. P. Gerba. 1993. MS2 and poliovirus transport in porous media: Hydrophobic effects and chemical pertubations. Wat. Resources Res. 29 (4): 957 63.

Barzily, A., and Y. Kott. 1989. Survival of pathogenic bacteria in elevated temperature oxidation ponds. Wat. Sci. Tech. 21 (1): 109 14.

Bhamidimarri, R., A. Shilton, I. Armstrong, P. Jacobson, and D. Scarlett. 1991. Constructed wetlands for wastewater treatment: The New Zealand experience. Wat. Sci. Tech. 24 (5): 247 53.

Blanc, R., and A. Nasser. 1996. Effect of effluent quality and temperature on the persistence of viruses in soil. Wat. Sci. Tech. 33 (10 11): 237 42.

Bouwer, H., and R. C. Rice. 1984. Renovation of wastewater at the 23rd Avenue rapid infiltration project. J. Wat. Pollut. Con. Fed. 56

(Jan.): 76 83.

Bouwer, H., R. C. Rice, J. C. Lance, and R. G. Gilbert. 1980. Rapid infiltration research at the Flushing Meadows Project, Arizona. J. W at. Pollut. Con. Fed. 52 (10): 2457 70.

Brix, H. 1997. Do macrophytes play a role in constructed treatment wetlands? W at. Sci. Tech. 35 (5): 11 17.

Buchberger, S. G., and G. B. Shaw. 1995. An approach toward rational design of constructed wetlands for wastewater treatment. Ecol. Eng. 4 (4): 249 75.

Camp, Dresser and McKee, Inc. 1997. Draft progress report Element 8: Development of alternatives, task order no. 10 Nogales facilties plan. Prepared for the International Boundary Waters Commission. Tucson: Camp, Dresser and McKee, Inc.

Chang, J. C. H., S. F. Ossoff, D. C. Lobe, M. H. Dorfman, C. M. Dumais, R. G. Qualls, and J. D. Johnson. 1985. UV inactivation of pathogenic and indictor microorganisms. Appl. Environ. Microb. 49 (6): 1361 65.

Chen, S. L., G.T. Wang, and S. K. Xue. 1999. Modeling BOD removal in constructed wetlands with mixing cell method. J. Environ. Eng. ASCE 125 (1): 64 71.

Clarke, N. A., R. E. Stevenson, S. L. Chang, and P. W. Kabler. 1961. Removal of enteric viruses from sewage by activated sludge treatment. Amer. J. Public Health (AJPH) 51: 118 29.

Craun, G. F. 1985. A summary of waterborne illness transmitted through contaminated groundwater. J. Environ. Health 48 (Nov/Dec): 122 27.

Crites, R. W. 1994. Design criteria and practice for constructed wetlands. W at. Sci. Tech. 29 (4): 1 6.

Curds, C. R., and G. J. Fey. 1969. The effect of ciliated protozoa on the fate of Eschericia Coli in the activated-sludge process. W at. Res. 3: 853 67.

Drewery, W., and R. Eliassen. 1968. Virus movement in groundwater. J. Wat. Pollut. Con. Fed. 40: R257 R271.

Drewes, J. E., P. Fox, and D. Ziegler. 1998. Impact of drinking water sources on refractory DOC in water reuse systems. Paper presented at the Third International Symposium on Artificial Recharge of Groundwater, 21 25 September at Amsterdam.

Drewes, J. E., and M. Jekel. 1996. Simulation of groundwater recharge with advanced treated wastewater. W at. Sci. Tech. 33 (10 11): 409 18.

Drewes, J. E., and M. Jekel. 1998. Behavior of DOC and AOX using advanced treated wastewater for groundwater recharge. W at. Res. 32 (10): 3125 33.

Fallowfield, H. J., N. J. Cromar, and L. M. Evison. 1996. Coliform die-off rate constants in a high rate algal pond and the effect of operational and environmental variables. Wat. Sci. Tech. 34 (11): 141 47.

Farooq, S., and A. K. Al-Youssef. 1993. Slow sand filtration of secondary effluent. J. Environ. Eng. ASCE 119 (4): 615 30.

Fernandez, A., C. Tejedor, and A. Chordi. 1992. Effect of different factors on the die-off of fecal bacteria in a stabilization pond purification plant. Wat. Res. 26 (Aug.): 1093 98.

Field, J. A., L. B. Barber, E. M. Thurman, B. L. Moore, D. L. Lawrence, and D. A. Peake. 1992. Fate of alkylbenzenesulfonates and dialkyltetralinsulfonates in sewage-contaminated groundwater. Environ. Sci. Tech. 26 (June): 1140 48.

Finch, G. R., and D. W. Smith. 1986. Batch coagulation of a lagoon for fecal coliform reductions. Wat. Res. 20 (Jan.): 105 12.

Fontes, D. E., G. M. Hornberger, A. L. Mills, and J. Herman. 1991. Physical and chemical factors influencing transport of microorganisms through porous media. Appl. Environ. Microb. 57 (9): 2473 81.

Gearheart, R. A., F. Klopp, and G. Allen. 1989. Constructed free surface wetlands to treat and receive wastewater, pilot project to full scale. In Constructed wetlands for wastewater treatment: Municipal, industrial, and agricultural, edited by D. A. Hammer. Chelsea, Mich.: Lewis Publishers.

Geldreich, E. E., H. F. Clark, and C. B. Huff. 1964. A study of pollution indicators in a wastewater stabilization pond. J. Wat. Poll. Con. Assoc. 36: 1373 79.

Gerba, C. P., S. M. Goyal, C. J. Hurst, and R. L. LaBelle. 1980. Type and strain dependence of enterovirus adsorption to activated sludge, soils, and estuarine sediments. Wat. Res. 14 (9): 1197 98.

Gerke, S., L. A. Baker, and Y. Xu. In review. Nitrogen transformations in a wetland receiving lagoon effluent: sequential model and implications for water reuse. Wat. Res.

Gerke, S. R. 1997. Nitrogen transformations in the Kingman, Arizona, wetland treatment system: Case study of wetlands in a low-tech water reuse system. Master s thesis, Arizona State University.

Gersberg, R. M., B. Elkins, S. Lyon, and C. Goldman. 1986. Role of aquatic plants in wastewater treatment by artifical wetlands. Wat. Res. 20 (March): 363 68.

Gersberg, R. M., B. V. Elkins, and C. R. Goldman. 1983. Nitrogen removal in artificial wetlands. Wat. Res. 17 (9): 1009 14.

Gersberg, R. M., S. R. Lyon, R. Brenner, and B. V. Elkins. 1987. Fate of viruses in artificial wetland. Appl. Environ. Microb. 53 (4):

731 36.

Gersberg, R. M., M. Matkovits, D. Dodge, T. McPherson, and J. M. Boland. 1995. Experimental opening of a coastal California lagoon: Effect on bacteriological quality of recreational ocean waters. J. Environ. Health 58 (Sept.): 24 28.

Gibbs, R. A., C. J. Hu, G. E. Ho, P.A. Phillips, and I. Unkovich. 1995. Pathogen die-off in stored wastewater sludge. W at. Sci. Tech. 31 (5 6): 91 95.

Glass, J. S., and R. T. O Brien. 1980. Enterovirus and coliphage inactivation during activated sludge treatment. W at. Res. 14 (7): 877 82.

Greenway, M., and A. Woolley. 1999. Constructed wetlands in Queensland: Performance efficiency and nutrient bioaccumulation. Ecol. Eng. 12 (1 2): 39 55.

Grimason, A. M., S. Wiandt, B. Baleux, W. N. Thitai, W. J. Bontoux, and H. V. Smith. 1996. Occurrence and removal of Giardia Sp. cysts by Kenyan and French waste stabilisation pond systems. W at. Sci. Tech. 33 (7): 83 89.

Gross, M., and D. Mitchell. 1990. Virus removal by sand filtration of septic tank effluent. J. Environ. Eng. ASCE 116 (July/Aug): 711 20.

Hammer, D. A., and R. L. Knight. 1994. Designing constructed wetlands for nitrogen removal. W at. Sci. Tech. 29: 15 27.

Harvey, R., and S. Garabedian. 1991. Use of colloid filtration theory in modeling movement of bacteria through a contaminated sandy aquifer. Environ. Sci. Tech. 25 (Jan.): 178 85.

Hoglund, C., T.A. Stenstrom, H. Jonsson, and A. Sundin. 1998. Evaluation of faecal contamination and microbial die-off in urine separating sewage systems. W at. Sci. Tech. 38 (6): 17 25.

Horne, A. J. 1995. Nitrogen removal from waste treatment pond or activated sludge plant effluents with free-surface wetlands. W at. Sci. Tech. 31 (12): 341 51.

Hurst, C. J., C. P. Gerba, and I. Cech. 1980. Effects of environmental variables and soil characteristics on virus survival in soil. Appl. Environ. Microb. 40 (6): 1067 79.

Ingersoll, T., and L. A. Baker. 1997. Nitrate removal in wetland microcosms. W at. Res. 32 (3): 677 84.

Jansons, J., L. Edmonds, B. Speight, and M. Bucens. 1989. Survival of viruses in groundwater. W at. Res. 23 (March): 301 06.

Jin, Y. 1997. Sorption of viruses during flow through saturated soil columns. Environ. Sci. Tech. 31 (2): 548 55.

Johnson, W., and B. Logan. 1996. Enhanced transport of bacteria in porous media by sediment-phase and aqueous-phase natural organic matter. Wat. Res. 30 (April): 923 31.

Kadlec, R. H., and R. L. Knight. 1996. Treatment wetlands. Boca Raton, Fla: Lewis Publishers.

Kim, T. D., N. Shiragami, and H. Unno. 1995. Development of a model describing virus removal process in an activated sludge basin. J. Chem. Eng. Jpn. 28 (3): 257 62.

Kim, T. D., and H. Unno. 1996. The roles of microbes in the removal and inactivation of viruses in a biological wastewater treatment system. Wat. Sci. Tech. 33 (10 11): 243 50.

Knight, R. L., R. H. Kadlec, and H. M. Ohlendorf. 1999. The use of treatment wetlands for petroleum industry effluents. Environ. Sci. Tech. 33 (7): 973 80.

Kopchynski, T., P. Fox, B. Alsmadi, and M. Berner. 1996. The effects of soil type and effluent pre-treatment on soil aquifer treatment. Wat. Sci. Tech. 34 (11): 235 42.

Krueger, C. J., L. B. Barber, D. W. Metge, and J. A. Field. 1998. Fate and transport a linear alkylbenzenesulfonate in a sewage-contaminated aquifer: A comparison of natural-gradient pulsed tracer tests. Environ. Sci. Tech. 32 (8): 1134 42.

Labelle, R. L., and C. P. Gerba. 1979. Influence of pH, salinity, and organic matter on the adsorption of enteric viruses to estuarine sediments. Appl. Environ. Microb. 38 (1): 93 101.

Lance, J. C., and C. P. Gerba. 1984a. Effect of ionic composition of suspending solution on virus adsorption by a soil column. Appl. Environ. Microb. 47 (3): 484 88.

Lance, J. C., and C. P. Gerba. 1984b. Virus movement in soil during saturated and unsatured flow. Appl. Environ. Microb. 47 (2): 335 37.

Lance, J. C., C. P. Gerba, and J. L. Melnick. 1976. Virus movement in soil columns flooded with secondary sewage effluent. Appl. Environ. Microb. 32 (4): 520 26.

Manthe, D., and N. Ash. 1993. Nitrogen removal by constructed wetlands for tertiary wastewater treatment. Paper presented at the Water Environment Federation Technical Exhibition and Conference, 3 7 October at Anaheim, California.

Martin, C. D., and K. D. Johnson. 1995. The use of extended aeration and in-series surface flow wetlands for landfill leachate treatment. Wat. Sci. Tech. 32 (3): 119 28.

Mayer, C. L., and C. J. Palmer. 1996. Evaluation of PCR, nested PCR, and fluorescent antibodies for detection of Giardia and Cryptosporidium species in wastewater. Appl. Environ. Microb. 62 (6): 2081 85.

Mayo, A. W. 1995. Modeling coliform mortality in waste stabilization

ponds. J. Environ. Eng. ASCE 121 (Feb.): 140 52.
McCarty, P. L., M. Reinhard, and B. E. Rittman. 1981. Trace organics in groundwater. Environ. Sci. Tech. 15 (1): 40 51.
McDowell-Boyer, L. M., J. R. Hunt, and N. Sitar. 1986. Particle transport through porous media. W at. Resouces Res. 22 (13): 1901 21.
McSparran, P.A. 1998. Optimization of wastewater effluent allocation between recharge and discharge. Master s thesis, Arizona State University.
Melbart, A., and J. Malina. 1974. Inactivation of virus bench-scale oxygenated waste stabilization ponds. Austin: Center for Research in Water Resources, University of Texas.
Mezrioui, N., and B. Baleux. 1994. Resistance patterns of E.Coli strains isolated from domestic sewage before and after treatment in both aerobic lagoon and activated sludge. W at. Res. 28 (Nov.): 2399 406.
Moore, B. E., B. P. Sagik, and J. F. Malina. 1975. Viral association with suspended solids. W at. Res. 9 (2): 197 203.
Moreno, A., J. Ferrer, F. Ruiz-Bevia, D. Prats, B. Vazquez, and D. Zarzo. 1994. LAS monitoring in a lagoon treatment plant. W at. Res. 28 (Oct.): 2183 89.
Pedley, S., and G. Howard. 1997. The public health implications of microbiological contamination of groundwater. Q. J. Eng. Geol. 30 (May) Part 2: 179 88.
Pieper, A., J. Ryan, R. Harvey, G. Amy, R. Illangasekare, and D. Metge. 1997. Transport and recovery of bacteriophage PRD1 in a sand and gravel aquifer: Effect of sewage derived organic matter. Environ. Sci. Tech. 31 (April): 1163 70.
Pike, E., and C. Curds, eds. 1974. Removal of bacteria and pathogenic organisms. In The Microbial Ecology of the Activated Sludge Process. N.p.
Pinney, M. L., P. K. Westerhoff, and L. A. Baker. 2000. Transformations in dissolved organic carbon through constructed wetlands. W at. Res. 34 (6): 1897 911
Polprasert, C., N. R. Khatiwada, and J. Bhurtel. 1998a. Design model for COD removal in constructed wetlands based on biofilm activity. J. Environ. Eng. ASCE 124 (9): 838 43.
Polprasert, C., N. R. Khatiwada, and J. Bhurtel. 1998b. A model for organic matter removal in free water surface constructed wetlands. W at. Sci. Tech. 38 (1): 369 77.
Postel, S. 1997. Last oasis: Facing water scarcity. New York: WW Norton.
Powelson, D., and C. P. Gerba. 1994. Virus removal from sewage effluents during saturated and unsaturated flow through soil