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USE OF A DYNAMIC SIMULATION MODEL TO UNDERSTAND NITROGEN CYCLING IN THE MIDDLE RIO GRANDE, NM

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ABSTRACT

Water quality often limits the potential uses of scarce water resources in semiarid and arid regions. To best manage water quality one must understand the sources and sinks of both solutes and water to the river system. Nutrient concentration patterns can identify source and sink locations, but cannot always determine biotic processes that affect nutrient concentrations. Modeling tools can provide insight into these large-scale processes. To address questions about large-scale nitrogen removal in the Middle Rio Grande, NM, we created a system dynamics nitrate model using an existing integrated surface water – groundwater model of the region to evaluate our conceptual models of uptake and denitrification as potential nitrate removal mechanisms.

We modeled denitrification in groundwater as a first-order process dependent only on concentration and used a 5% denitrification rate. Uptake was assumed to be proportional to transpiration and was modeled as a percentage of the evapotranspiration calculated within the model multiplied by the nitrate concentration in the water being transpired. We modeled riparian uptake as 90% and agricultural uptake as 50% of the respective evapotranspiration rates. Using these removal rates, our model results suggest that riparian uptake, agricultural uptake and denitrification in groundwater are all needed to produce the observed nitrate concentrations in the groundwater, conveyance channels, and river as well as the seasonal concentration patterns. The model results indicate that a total of 497 metric tons of nitrate-N are removed from the Middle Rio Grande annually. Where river nitrate concentrations are low and there are no large nitrate sources, nitrate behaves nearly conservatively and riparian and agricultural uptake are the most important removal mechanisms. Downstream of a large wastewater nitrate source, denitrification and agricultural uptake were responsible for approximately 90% of the nitrogen removal.

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TABLE OF CONTENTS

TABLE OF FIGURES

TABLE OF TABLES

1. PROJECT DESCRIPTION AND BACKGROUND

1.1 Background

Public mediated resource planning is quickly becoming the norm rather than the exception. Unfortunately, supporting tools are lacking that interactively engage the public in the decision-making process and integrate over the myriad values that influence water policy. Tidwell and others (2007) took steps toward developing a specialized decision framework to meet this need; specifically, a modular and generic resource-planning "toolbox." The original toolbox integrated the disparate systems of hydrology, ecology, climate, demographics, economics, policy and law. Specifically, the toolbox forms a collection of process modules and constitutive relations that the analyst can "swap" in and out to model the physical and social systems unique to their problem.

Development of the toolbox has continued according to the availability of resources. This report documents the development of an additional toolbox module. The model provides a decision support framework for quantifying nutrient dynamics in a semi-arid riverine environment. This work represents the doctorial research of Gretchen Oelsner while attending the University of Arizona. This work was support by Sandia National Laboratories through a Campus Executive Laboratory Directed Research and Development (LDRD) project.

The original tool box report (Tidwell et al. 2007) was organized according to a set of stand-alone papers that were submitted to individual peer-reviewed journals. In this tradition, the second chapter of this report is structured according to a single standalone paper that documents the development of a decision support module for the analysis of nutrient dynamics in the Middle Rio Grande. The first chapter of this report provides the resource-planning toolbox context under which this specific nutrient dynamics module was developed.

1.2 Justification

Persistent conflict between competing interests is becoming more common in water resources management today. Such conflict often results in gridlock (i.e., no decision, continuation of the status-quo) or a protracted and inefficient decision process that is too often resolved through litigation, resulting in only marginal change while imposing exceedingly high transaction costs. In many cases, it simply takes too long and costs too much to make major water resources decisions – and after all that, we often fail to achieve broad consensus in the decisions. Such difficulties arise because of both the complexity of the natural system and the disparity with which water is valued within the community. Finding solutions to these complex problems requires interdisciplinary and multivalued thinking. No single person or institution has a complete knowledge or experience base from which to tackle these problems. For this reason, a process is needed that brings the entire community together so as to expand the collective thinking. Just as the old Jewish proverb states:

"Plans fail for lack of counsel, but with many advisors they succeed."

Beyond the collaborative process there is a need for tools to help structure group thinking and to provide a vehicle for communicating joint understanding within and external to the team. Integration of collaborative processes with interactive decision-support tools provides a venue where group learning can take place, and leads to the identification of mutual gain solutions.

Previous efforts demonstrate the value of applying technically informed collaborative planning and management methods. These methods involve open, collaborative decision-making processes (Connick and Innes 2003; Spash 2001; Claussen 2001; Susskind et al. 2001; Serageldin 1995; Potapchuk 1991) supported by transparent computer models (Tidwell et al. 2004; Costanza and Ruth 1998; van den Belt 1998; Palmer 1993; Johnson 1990; Wallace and Sancar 1988; Jordão et al. 1997). In fact, most water management processes today incorporate some degree of public involvement and collaboration, and most use computer modeling to support analysis and decision making. However, there is still room for improvement in the way traditional planning, public collaboration, and decision-support computer modeling are integrated. Specifically, we suggest an approach that is different in two important ways. First, public collaboration is encouraged in every aspect of the planning process including data gathering and model development so as to establish a foundation of full disclosure and transparency. Giving stakeholders and public representatives more control over technical analysis and the resulting decision-support models builds trust in the process that can help avoid battles of “dueling science.” Second, the cooperatively developed tools provide a portal in which resource managers, decision makers, and stakeholders alike can personally interact with the best available science. In this way, stakeholders can jointly or independently formulate and evaluate new management options, identify tradeoffs, and gain understanding of system behavior. Incorporating rigorous yet accessible analytical tools into collaborative processes empowers stakeholders with greater responsibilities to tackle difficult problems, something often lacking in collaborative decision making today.

Although both a desire for collaboration and the use of computer models are fairly standard in resource management, collaborative modeling methods are relatively new. Various government and nongovernmental organizations have made initial efforts at combining collaboration and modeling but only a few techniques have been developed and relatively few tools have been tailored toward these kinds of techniques. As such, a new generation of computer-aided decision tools are needed that interactively engage the public in the decision-making process and integrate over the myriad values that influence water policy. Specifically, modules of both physical and social processes are needed that can be linked together to capture the unique dynamics of a watershed within the context of a transparent, stakeholder-accessible system dynamics model.

1.3 Objective

Our objective is to develop a resource-planning toolbox to support collaborative watershed management and foster communication between water professionals, decision-makers, and the public. This decision toolbox integrates the broad physical and social dynamics that define the balance between water supply and demand. The toolbox is formulated in a fully generic context allowing application to a wide variety of watersheds across the United States and abroad. This effort is being performed in a series of phases according to available resources. This Sand Report focuses on a single component of the toolbox— quantifying nutrient dynamics in a semi-arid riverine environment. Additional details concerning other toolbox modules can be found in Tidwell et al. 2006.

1.4 Approach

To create a truly multidisciplinary model requires the assemblage of a multidisciplinary team united in their systems thinking. The toolbox team included hydrologists, economists, ecologists, aquatic chemists, watershed scientists, policy analysts, water attorneys, and system modelers. The team was comprised of expertise found both within Sandia and from external sources, including a private consulting firm, the University of New Mexico, New Mexico Institute of Mining and Technology, the University of Arizona, and the University of Chicago.

The team's first task was to develop a "toolbox" conceptual model. This was necessary to organize the overall system into interacting subsystems, while defining the basic processes and constitutive relations which comprise them. Critical subsystems include land surface processes, which encompass precipitation (snow and rain) runoff relations subject to varying vegetation cover and land use practices; surface water hydrology including river routing, tributary inflow and irrigation diversions; groundwater hydrology, subject to river leakage, groundwater pumping and recharge; water quality subject to point and non-point source loading; and environmental health, which tracks the dynamics of riparian and aquatic communities in the basin. In turn, these subsystems are influenced by the temporally variable demands represented by irrigated agriculture, municipal consumption, riparian evapotranspiration, and open-water evaporation subject to applicable legal and political constraints. Finally, alternative water use strategies are integrated as a unique subsystem to allow evaluation of how effectively the alternative can utilize the available water and to quantify its resulting economic and environmental benefit to the region.

It is important to note that these systems are not static in time but rather behave dynamically, expressing the complex interplay of processes that underlie these systems. Additionally, these systems do not operate independently but in complex networks characterized by numerous feedbacks and time delays. That is, the dynamics of one process may depend on the behavior of one or more related processes. Thus, a second aspect of this task, and the most challenging, was to identify the constitutive relations linking disparate processes comprising the water budget. This involved understanding the strength of cause-and-effect relationships, and determining whether the response is immediate or delayed in time.

The next task in the model development effort involved quantifying and structuring the component processes identified in the model conceptualization phase. Furthermore, this disparate set of processes needed to be integrated within a unified framework under a single computational platform. To maintain a tractable solution to this problem, we adopted a system dynamics platform for creating the resource planning toolbox. More detail on system dynamics is given below.

Another important aspect of this work was creating an interface that effectively conveys results to policy-makers and the public. To accomplish this desire, we created an interactive modeling environment comprised of a number of user-friendly interfaces. The interfaces allow the user to easily change external factors influencing water supply (e.g., climate, population growth), policies governing water allocations, and alternative water use strategies. Additional interfaces then convey model output in terms of shifts in water use among different water use sectors, ability to meet legal obligations, changes to in-stream flows, and others.

The final phase of modeling involved calibrating and testing the resource planning toolbox. Specifically, toolbox modules were structured according to the physical and social system governing flows in the Middle Rio Grande Basin, defined here as the stretch of Rio Grande between the Colorado/New Mexico border and Elephant Butte Reservoir. The model was implemented and parameterized according to the best available data. Subsequently, model simulations were calibrated against measured basin data on a monthly basis for a 25-year period of time (1975–1999).

1.5 Model Architecture

Selection of an appropriate architecture for the toolbox model is based on two criteria. First, an environment is needed that provides an “integrated” view of the watershed — one that couples the complex physics governing water supply with the diverse social and environmental issues driving water demand. Second, a modeling environment is needed that can be taken directly to the public for involvement in the decision process and for educational outreach. For these reasons we have adopted an approach based on the principles of system dynamics (Forrester 1990; Sterman 2000). System dynamics provides a unique framework for integrating the disparate physical and social systems important to water resource management, while providing an interactive environment for engaging the public.

System dynamics is a systems-level modeling methodology developed at the Massachusetts Institute of Technology in the 1950s as a tool for business managers to analyze complex issues involving the stocks and flows of goods and services. System dynamics is formulated on the premise that the structure of a system – the network of cause-and-effect relations between system elements – governs system behavior (Sterman 2000). “The systems approach is a discipline for seeing wholes, a discipline for seeing the structures that underlie complex domains. It is a framework for seeing interrelationships rather than things, for seeing patterns of change rather than static snapshots, and for seeing processes rather than objects” (Simonovic and Fahmy 1999).

In system dynamics a problem is often decomposed into a temporally dynamic, spatially aggregated system. The scale of the domain can range from the inner workings of a human cell to the size of global markets. Systems are modeled as a network of stocks and flows. For example, the change in volume of water stored in a reservoir is a function of the inflows less the outflows. Key to this framework is the feedback between the various stocks and flows comprising the system. In our reservoir example, feedback occurs between evaporative losses and reservoir storage through the volume/surface area relation for the reservoir. Feedback is not always realized immediately but may be delayed in time, representing another critical feature of dynamic systems.

There are a number of commercially available, object-oriented simulation tools that provide a convenient environment for constructing system dynamics models. For purposes of this effort, the resource planning toolbox is developed in Studio Expert 2005, produced by Powersim, Inc. (www.powersim.com). Model construction proceeds in a graphical environment, using objects as building blocks. These objects are defined with specific attributes that represent individual physical or social processes. These objects are networked together so as to mimic the general structure of the system. In this way, these tools provide a structured and intuitive environment for model development.

1.6 Original Toolbox Elements

The result of this work is a generic resource-planning toolbox to support collaborative watershed management and foster communication between water professionals, decision makers, and the public. Within this toolbox resides the overall framework by which an analyst can build a basin-specific model. Comprising the toolbox are subsystem modules and constitutive relations that the analyst can “swap” in and out to capture the physical and social systems important to their problem. This modularity allows the creation of sophisticated, highly integrated models that water professionals can use as well as simplified models aimed at public outreach. This modular toolbox is formulated in system dynamics and subsequently linked to a Geographical Information System (GIS) interface. Toolbox applications are possible at the aggregate watershed scale or the subwatershed scale in which key systems (i.e., land surface, groundwater, surface water) are spatially discretized. Temporal resolution of the toolbox is equally flexible, with options ranging from daily to annual.

Foundational elements of the resource-planning toolbox are briefly described below. Additional details can be found in Tidwell et al. 2006.

Surface Water Process Modules: The surface water package covers the basic elements pertaining to river routing, including reservoir processes, reservoir operations, open water evaporation, irrigation diversions and conveyance processes, crop and riparian evapotranspiration, municipal waste water returns, and surface-groundwater interaction. Modeling is structured according to a reach-based approach with reaches defined by gages or other key inflows/diversions.

Groundwater Process Modules: The groundwater module addresses processes such as groundwater flow, recharge, surface-groundwater interaction, riparian evapotranspiration, groundwater pumping, and spatially varying groundwater head distributions. A spatially distributed, unstructured computational grid approach forms the overarching framework for this modeling.

Land Surface Process Modules: The surface water and groundwater systems receive tributary inflows and recharge, respectively, from precipitation falling on the adjoining watersheds. The land surface model quantifies these precipitation-runoff-recharge processes. The model treats individual watersheds, disaggregating the system into hydrologic response units. Within each unit falling precipitation (as snow or rain) is partitioned into canopy capture, runoff, direct evaporation, infiltration, change in soil moisture, evapotranspiration, interflow, or deep recharge.

Water Quality Modules: The water quality package is used to simulate the transport of conservative solutes in the surface and groundwater systems. Sources, sinks and mixing both in the surface and groundwater systems are addressed. Chloride and bromide concentrations measured in the Rio Grande are used to calibrate and verify the model.

Ecologic Process Modules: A fish ecology module was also developed and tested. The model considers fish fertility and mortality as influenced by river discharge and ammonium concentration. The model also considers ease of fish migration and the impact of stocking from off-stream refugia on the fish population. Application is drawn with the Rio Grande Silvery Minnow.

Economic Process Modules: The economics module explores the relation between water and the economy. There are two economic levels that are included in the model: micro-level components and the overall, regional macro economy. Within the micro theme area, we consider the following sectors: urban residential, agricultural, commercial, industrial, institutional, shoreline use, birding, and non-use values (instream use). Market microeconomic components impact the regional or macro economy through their impact on productivity and employment. Both levels, in turn, impact the demand for water.

Water Marketing Process Modules: The focus of this module is the development of decision-support tools for the exploration and design of water markets. An integrated decision-support framework and market gaming interface was developed to test and quantify participant behavior in a water market environment.

Collaborative Modeling Process: Previous modules have dealt exclusively with the conceptualization, formulation, and testing of physical and social process models. This effort focused on the collaborative modeling process; specifically, the diverse array of experiences and lessons learned concerning collaborative modeling processes. The experiences reported provide insight into how to "do" collaborative modeling.

1.7 Toolbox Application

This toolbox provides a comprehensive resource planning framework that integrates the disparate systems of hydrology, ecology, climate, population, economics, policy, and water law into a coherent decision-support system that is fully generic, allowing application to a wide variety of watersheds spanning a broad range of scales. The intention is that in time the model will be adopted by resource managers at the local, state, and federal levels as their tool of choice for resolving difficult water allocation problems. Finally, development of this resource-planning toolbox represents an important foundational element of the proposed interagency center for Computer Aided Dispute Resolution (CADRe). The Center's mission is to manage water conflict through the application of computer-aided collaborative decision-making methods. The Center will promote the use of decision-support technologies within collaborative stakeholder processes to help stakeholders find common ground and create mutually beneficial water management solutions. The Center will also to serve to develop new methods and technologies to help federal, state and local water managers find innovative and balanced solutions to the nation's most vexing water problems. The toolbox is an important step toward achieving the technology development goals of this center.

2. MODEL FOR EXPLORING NITROGEN CYCLING IN THE MIDDLE RIO GRANDE, NM

2.1. Introduction

Water quality often limits the potential uses of scarce water resources in semiarid and arid regions. To best manage water quality one must understand the sources and sinks of both solutes and water to the river system. In large river systems, water quality is usually examined with nutrient concentration patterns [*Alexander et al.*, 1997; *Hooper et al.*, 2001; *Mueller and Spahr*, 2006], whereas most process based studies are conducted at a small scale [*Johnson et al.*, 1995; *Peterson and Kwak*, 1999; *Richey et al.*, 1990]. Large rivers in semiarid regions often provide water for municipal and agricultural use in addition to adjacent ecosystems [*Feng et al.*, 2000; *Kingsford*, 2000; *Stanley et al.*, 1997; *Williams*, 2001]. Effective management of these rivers requires some method for assessing large-scale nutrient cycling.

Previous studies in the Middle Rio Grande have focused on analyzing the spatial and temporal patterns of nutrient concentrations using monitoring and discharge data [*Anderholm et al.*, 1995; *Passell et al.*, 2004; 2005]. Several studies have looked at how nutrient load patterns change downstream and over time [*Moore and Anderholm*, 2002; *Oelsner et al.*, 2007; *Passell et al.*, 2004; 2005]. Results from these studies suggest that wastewater treatment plants are important point sources of nutrients to the river [*Oelsner et al.*, 2007; *Passell et al.*, 2004] and agricultural diversions can remove nitrogen and improve water quality downstream of wastewater treatment plant point sources [*Oelsner et al.*, 2007]. In addition, Oelsner et al., 2007 showed that there is limited biological modification of dissolved nitrogen and organic carbon in the river within the Middle Rio Grande. However, due to the complex relationships between river water, agricultural conveyance water and groundwater, it is not clear from concentration patterns where and how nitrogen is removed from the system.

The water system in the Middle Rio Grande consists of the river, conveyance channels, agricultural return drains, and a shallow alluvial aquifer which make it difficult to determine locations and methods of nutrient processing. During the irrigation season, water is diverted from the river into conveyance channels and transported to fields for irrigation. Drains collect water seepage from agricultural fields and unused irrigation water for return to the river.

Riverside drains collect agricultural return flows and, because they are lower than the river, they collect water seeping from the river. Many studies have measured rates of nitrogen transformations at the plot scale [*Lowrance*, 1992; *Molles et al.*, 1998; *Mulholland et al.*, 2004; *Schade et al.*, 2005; *Valett et al.*, 2005], but it is not feasible to conduct plot-scale experiments over an entire watershed. Given the complex hydrology in the Middle Rio Grande, modeling is a good approach for exploring the nitrogen dynamics of the Middle Rio Grande.

Widely used nutrient models like SPARROW and Qual2E (Qual2K) can be used on larger rivers, but only examine nutrient dynamics in surface water. Furthermore, neither one of these models have the ability to model groundwater-surface water exchange. There are also many models for nitrogen cycling in soil and on agricultural fields [*Bradbury et al.*, 1993; *de Willigen*, 2005; *Johnsson et al.*, 1987]. Despite the abundance of nutrient cycling models, few consider nutrient cycling on a large scale and integrate surface water and groundwater. Previously, Roach [2007] created an integrated surface water-groundwater model for the Rio Grande using a system dynamics approach that allows for the addition of a solute component. In this paper we use a system dynamics model to examine nitrogen removal in the Middle Rio Grande's riparian and agricultural systems. In particular, this paper addresses three questions about the removal of nitrogen in the agricultural and riparian areas. First, is uptake or denitrification the dominant process responsible for removing nitrogen from the system? Second, how much nitrogen is removed from the system? Third, does most of the nitrate removal take place in the river/riparian system or in the agricultural conveyance system?

2.2. Site Description

The Rio Grande flows from its headwaters in southern Colorado through central New Mexico into Texas where it forms the boundary between the United States and Mexico before ultimately emptying into the Gulf of Mexico. The study area, hereafter referred to as the Middle Rio Grande (MRG), is a 300 km reach of the Rio Grande within central New Mexico that begins at the outflow of Cochiti Dam and ends at Elephant Butte Reservoir (Figure 1). These end points are significant hydrologic boundaries of the system as they are locations where river discharge is regulated and measured. Releases from Cochiti Dam represent the major source of water for the reach and all water diversions are returned to the river upstream of Elephant Butte Reservoir.

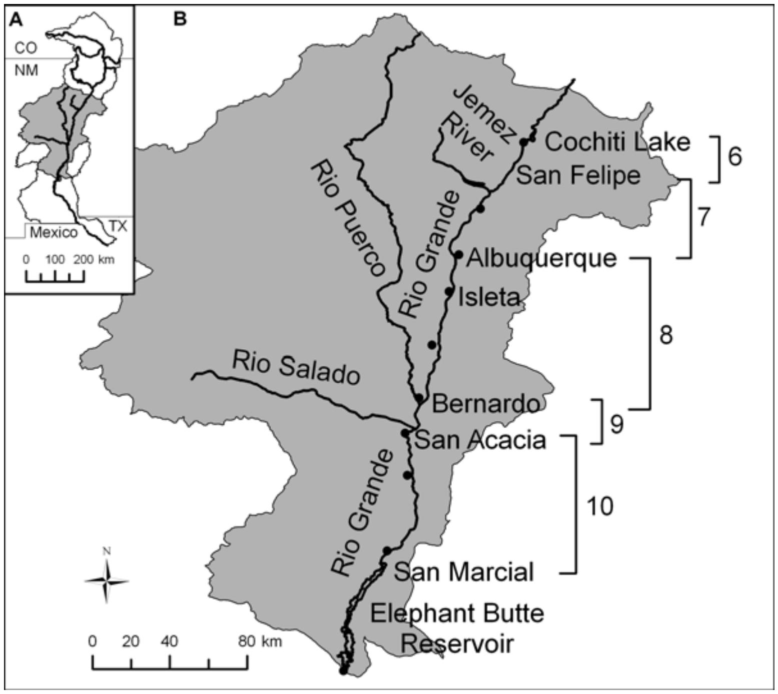


Figure 1: A) Map of Upper Rio Grande watershed shaded in gray. B) Detailed map of Middle Rio Grande study area. The reach extent and number is noted on the right with brackets.

The Middle Rio Grande watershed includes an area of approximately 52,345 km2 with elevations ranging from about 1,300 m at the southern end of the basin to about 3,100 m in the Sandia Mountains outside of Albuquerque and in the Magdalena Mountains west of Socorro. The reach is characterized by an arid to semiarid climate with average annual precipitation at river level ranging from 216 mm in Albuquerque to 241 mm in Socorro [Western Regional Climate Center (WRRC), 2005]. The higher altitudes receive more precipitation; 583 mm at Sandia Crest and 482 mm at South Baldy Peak in the Magdalena Mountains [Langmuir Laboratory, 2006; Western Regional Climate Center (WRRC), 2005]. According to the 2001 National Land Cover Dataset (NLCD), most of the land within the watershed is classified as shrubland (44%) with the remaining classified as forest (29%), grassland/herbaceous (19%), agricultural (3%), urban (2%) and open water, barren and wetlands (1% each) [Multi-Resolution Land Characteristics Consortium (MRLC), 2007]. Basin vegetation is dominated by grass and shrubs including mesquite and creosote at lower elevations transitioning to pinon and juniper at higher elevations [Plummer et al., 2004]. The area in closest proximity to the river, including the alluvial floodplain, the riparian forest (bosque), and the diversion structures and agricultural conveyance channels between Cochiti Dam and San Marcial are maintained by the Middle Rio Grande Conservancy District (MRGCD). Land use within the area managed by the MRGCD is classified as 48% cultivated, 26.4% urban, 15.4% grass and brush and 10.2% is classified as bosque [S.S. Papadopulos and Associates Inc. (SSPA), 2002](1986 numbers). Vegetation within the riparian corridor is dominated by native cottonwood (Populus deltoids ssp. wislizeni) and willow (*Salix* sp.) mixed with invasive Russian olive (Elaeaganus angustifolia) and tamarisk (*Tamarix pentandra* and *Tamarix chinensis*) [*Anderholm*, 1997; *Bartolino and Cole*, 2002].

Water in the Rio Grande is derived mostly from spring snowmelt in the San Juan, Sangre de Cristo, and Jemez mountain ranges in southern Colorado and northern New Mexico at elevations of 3,000 m to 4,000 m [*Hogan et al., In Press; Phillips et al.*, 2003]. Stable isotopes of oxygen (^{18}O) and hydrogen (^{2}H) are relatively light (-14 and -100 $^{0}/_{00}$, respectively) [*Hogan et al., In Press; Phillips et al.*, 2003] and are consistent with winter precipitation for the region [*Adams et al.*, 1995]. Most perennial tributaries discharge to the Rio Grande upstream of Cochiti Lake, and the river receives little surface-water inflow downstream of this reservoir [*Ortiz and Lange*, 1996]. The ephemeral Rio Puerco and Rio Salado and intermittent Jemez River are the three major tributaries within the Middle Rio Grande, however they contribute little water to the river [*Moore and Anderholm*, 2002]. Annual discharge from the Albuquerque wastewater treatment plant to the Rio Grande makes it one of the largest tributaries in the river's upper basin [*Passell et al.*, 2004].

Anthropogenic structures have caused channelization and artificial gradients such that the Middle Rio Grande is a losing reach and water infiltrates from the river into the shallow alluvial aquifer [*Moore and Anderholm*, 2002]. Riverside drains intercept shallow groundwater seeped from the river and return it to the river between Cochiti Dam (471.0 km) and San Acacia (655.3 km) [*Moore and Anderholm*, 2002]. South of San Acacia, the river is a losing reach due to the presence of the Low Flow Conveyance Channel (LFCC) which was built in the severe drought period of the 1950s to deliver water to Elephant Butte Reservoir with as few seepage and evapotranspirative losses as possible [*Anderholm et al.*, 1995; *Moore and Anderholm*, 2002]. The LFCC is the lowest point in the surface drainage system and its presence results in a hydraulic gradient away from the river toward the LFCC. As of the mid-1980s, water is no longer diverted from the river into the LFCC but water from drain returns and river seepage continue to be sources of water to the channel. During the summer months, water from the LFCC is often pumped into the river to maintain sufficient water in the river for the endangered Rio Grande silvery minnow (*Hybognathus amarus*). Several kilometers upstream of Elephant Butte Reservoir, the LFCC and Rio Grande merge into one channel before flowing into the reservoir.

The surface water system within the Middle Rio Grande includes the river and a complex system of agricultural conveyance channels and drains adjacent to the river. A list of Rio Grande USGS gauging stations and the major drains gauged by the MRGCD are given along with the approximate distance of each from the Rio Grande headwaters in Colorado in Table 1. At the upstream boundary of the Middle Rio Grande, discharge is determined by releases from Cochiti Dam (470.0 km) but usually mirrors upstream flows. Further downstream, diversion structures at Cochiti (471.0 km), Angostura (505.0 km), Isleta (570.0 km) and San Acacia (655.3 km) move water from the river into distribution canals to the fields for irrigation between the beginning of March and the end of October. Agricultural return flows are collected in open drains and routed back to the river and discharged either directly into the river north of San Acacia (655.3 km) or into the LFCC south of San Acacia.

Table 1. Description of reaches used in the model.

Reach Number	Location Description	Groundwater Zones[1]
6	Cochiti Dam to San Felipe	AB1, AB2, AB3
7	San Felipe to Albuquerque	AB15, AB16, AB17
8	Albuquerque to Bernardo	AB30, AB31, AB32, AB33
9	Bernardo to San Acacia	AB48
10	San Acacia to San Marcial	SB1, SB2
11	San Marcial to Elephant Butte Reservoir	SB3

[1]AB refers to the Albuquerque Basin and SB refers to the Socorro Basin. See Roach, 2007 for complete details.

2.3. Methods

2.3.1. Introduction to Powersim

Powersim Studio 2005 Service Release 3 (Powersim), is the system-dynamics software program we used to model nitrate in the Rio Grande. System dynamics modeling methodology was developed in the 1950's as a tool for managers, but has since been applied to many different disciplines. It is based on principles of cause and effect, feedback, and delay and allows the user to see patterns of behavior in a system over time.

Powersim uses a graphical modeling language with three different types of variables, which makes the model simple to follow visually. Level symbols represent a stock of water or nitrate that change over time in the simulation. Auxiliary symbols recalculate information that is fed into the system and constants represent information that is not changed during the simulation. A circle shape attached to an arrow represents an equation that governs a flow into or out of a stock at a calculated rate. When variables are connected by the user, Powersim generates equations which integrate flows that cause a level to change.

2.3.2. Water Model Structure

We used a system dynamics hydrology model developed by J. Roach [*Roach*, 2007] as a hydrologic framework and added a nitrate component that considers uptake and denitrification as possible processes for removing nitrate from the river in the riparian and agricultural systems. The hydrology model has three primary components, one each for river water, conveyance water and groundwater. The Middle Rio Grande (between Cochiti Dam and San Marcial) is divided into five reaches based on gauging station locations, and each reach has a river and conveyance level in the model (Table1, Figure 1). The groundwater in the Middle Rio Grande basin was divided into smaller regions, with a total of 51 groundwater zones between Albuquerque and San Acacia and 12 groundwater zones between San Acacia and Elephant Butte Reservoir. However, the number of groundwater zones in direct contact with the river is much smaller; 11 between Cochiti Dam and San Acacia and 2 between San Acacia and San Marcial. For locations of groundwater zones, see Roach [2007].

Flows of water among the river, conveyance and groundwater systems connect these three systems and are the basis of the feedback relationships in the model (Figure 2). Water from the river leaks into the groundwater and is diverted into the conveyance system for irrigation. Water in the conveyance system is transferred to the groundwater by leaking from the conveyance canals and seeping from crop fields. Drains, in turn, collect groundwater and return it to the river. In addition to the internal fluxes among the model components, the water model considers a number of other fluxes to and from the system. Some of the other fluxes include riparian ET, agricultural ET, tributaries, wastewater, and groundwater transfers between shallow aquifer zones. For complete details on the water model structure and calibration, please see [*Roach*, 2007].

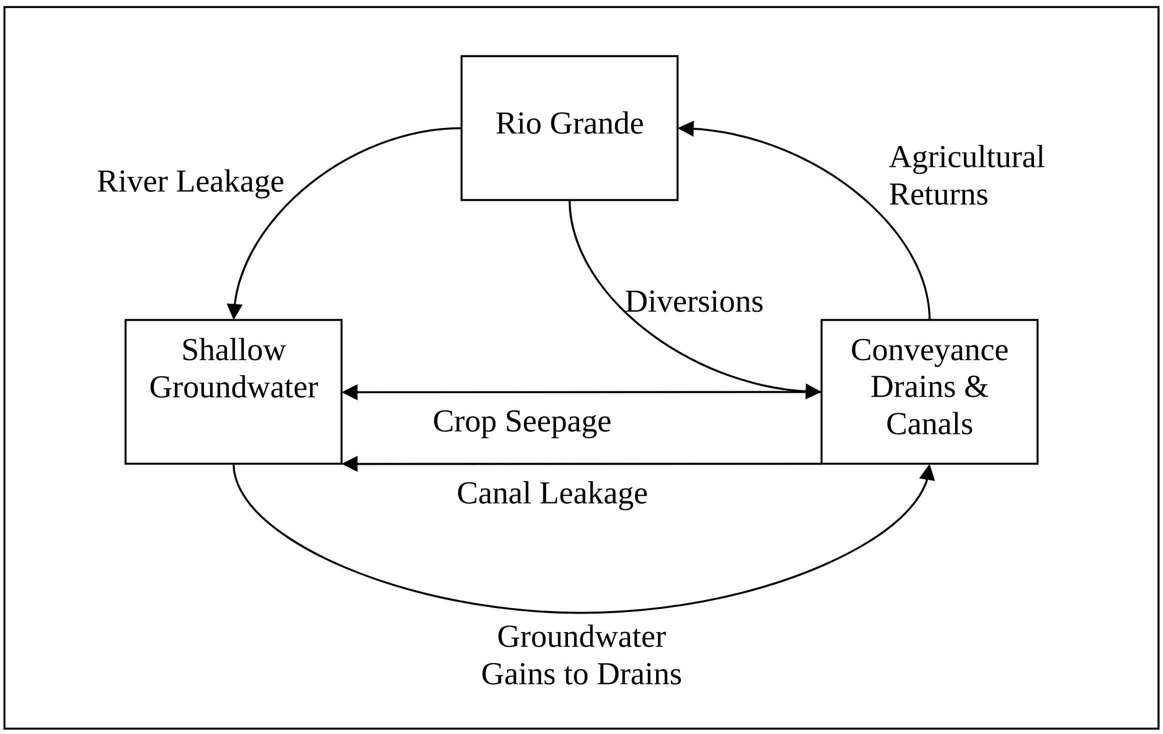


Figure 2. Connections among the Rio Grande, shallow groundwater and conveyance drains and canals stocks in the model

2.3.4. Nitrate Model Structure

Based on previous research in the Middle Rio Grande, we developed a simple conceptual model of the system and possible nitrogen transformations (Figure 3). Our previous work shows that water in the agricultural conveyance channels has lower nitrate concentrations than water diverted from the river into the conveyance canals [*Oelsner et al.*, 2007]. Both immobilization (uptake) and denitrification are known to be important processes for removing nitrogen and reducing nitrate concentrations in the Middle Rio Grande [*Molles et al.*, 1998; *Valett et al.*, 2005]. Therefore, we have chosen to focus on uptake and denitrification as possible removal mechanisms with the understanding that there are many other nitrogen transformations occurring in all parts of the Middle Rio Grande, such as mineralization, nitrification, and volatilization.

Due to the constant recycling of nutrients, our model only considers “net” uptake and denitrification.

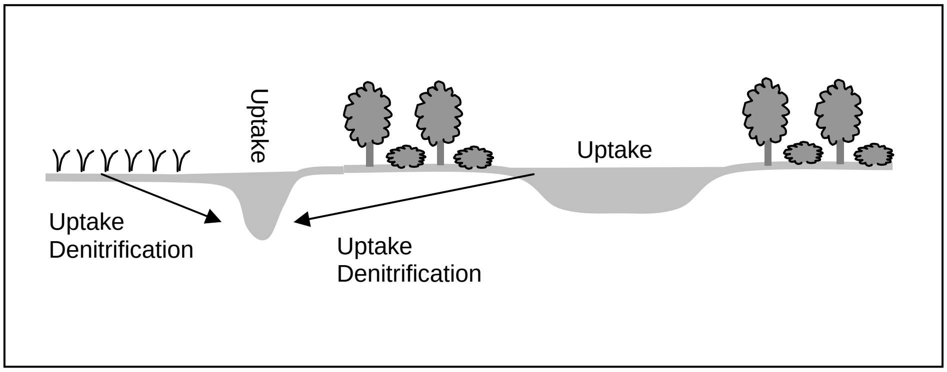


Figure 3. Conceptual model of the river and drain system and the possible mechanisms for nitrogen removal.

Water in the conveyance channels comes from three sources: river diversions, crop seepage, and river leakage. Understanding the hydrologic connections in the system, we hypothesize several locations and mechanisms for nitrate removal. As water seeps from the river into the riverside drains, nitrate could be temporarily removed by riparian vegetation uptake or permanently removed by denitrification (Figure 3). Similarly, nitrate could be removed temporarily via crop uptake or permanently via denitrification as the irrigation water is applied to the fields and seeps from the fields into the drains. Finally, it is possible that nitrate is temporarily removed from the drains via uptake by organisms or vegetation on river and canal banks. Notably, the groundwater in closest proximity to the river and drain system downstream of San Felipe has dissolved oxygen concentrations less than 0.5 mg/l, which would allow for denitrification [*Plummer et al.*, 2004].

The nitrate fluxes (mass/time) are modeled in the same manner as the water fluxes in the water model and generalized equations used in the nitrate model are given in Table 2. Due to differences in the location of diversions, wastewater inputs, and agricultural returns, some of these equations were modified to better fit a particular reach. The river nitrate component includes tributary and wastewater inflows that are calculated by multiplying an average nitrate concentration for each tributary and wastewater inflow by the respective flux of water. Diversions flow from the river nitrate component into the conveyance nitrate component. Nitrate is exchanged between groundwater and surface water according to the exchanges that take place in the water-balance model. When water flows from the conveyance system into the shallow aquifer, the flow is multiplied by the nitrate concentration of the flows going into that reach of the agricultural component. When water flows from the shallow aquifer into the agricultural system, the flow is multiplied by the aquifer nitrate concentration, which is recalculated at each timestep based on the nitrate and water storage. Groundwater/surface-water interaction between the river and shallow aquifer is also represented in the nitrate model. When leakage is positive (leaking from the river into the shallow aquifer) then the flow is multiplied by the concentration of the river after diversions, tributary and wastewater inflow, and evaporation have all taken place in that reach. When the flow is negative (seeping from the shallow aquifer into the river) it is multiplied by the aquifer concentration for that time step. In reaches 6 through 9 (Cochiti to San Acacia) the groundwater model includes a term for “other recharge”, which is a catch-all term for recharge to the shallow aquifer from tributaries, septic tanks, and some mountain front recharge.

Table 2: Generalized equations for the Middle Rio Grande nitrate model.

Variable	**Equation**
Inflow Nitrate	Inflow * Nitrate conc
Tributary Nitrate	Tributary flow * average tributary Nitrate concentration
Wastewater Nitrate	Wastewater inflow * average wastewater Nitrate concentration
Leakage Nitrate	If (leakage>0, leakage Nitrate out of river, leakage Nitrate into river)
Leakage Nitrate out of River	reach loss Nitrate concentration * leakage
Leakage Nitrate into River	aquifer Nitrate concentration * leakage
River Outflow Nitrate Concentration	(inflow Nitrate -ag diversions Nitrate + tributary Nitrate + wastewater Nitrate) / (inflows-ag diversions + tributaries + wastewater)
Ag Diversions Nitrate	ag diversions * reach inflow Nitrate conc
Ag Returns Nitrate	ag outflow Nitrate concentration * ag returns
Ag Outflow Nitrate conc	(diversions Nitrate inflow + Thruflow Nitrate + Nitrate GW-SW interaction rate) / (diversions + Thruflow + GW-SW interaction rate-ag consumption)
Aquifer Nitrate conc	Aquifer Nitrate Storage / Aquifer Volume
Nitrate GW-SW Interaction	If (GW-SW Interaction rate>0, GW-SW Interaction rate * aquifer Nitrate concentration, GW-SW Interaction rate * Reach Inflow Nitrate concentration)
Uptake	ET water loss*Nitrate Concentration*Uptake Rate
Denitrification	Water volume*(Denitrification Rate*Concentration)

Following the water model structure, the river and conveyance system components are designed to not store water. Only the groundwater component is allowed to store water, whereas the level variable in the model for nitrate in river and conveyance reaches remains constant. Therefore, the outflows from both the river and conveyance components delivered to the next downstream reach force a mass balance and are determined as the difference between the other inflows and outflows within each reach.

Uptake is modeled as a proportion of the water removed from the system due to evaporation and transpiration so that the amount of nitrate removed uptake varies seasonally. We assumed that in the riparian areas, nitrate uptake was proportional to transpiration rates. On an annual basis, transpiration in one semiarid riparian woodland was determined to be 90% of evapotranspiration [*Yepez et al.*, 2007]. Therefore, we calculated the nitrate loss due to riparian uptake as 90% of the riparian ET flux multiplied by the groundwater nitrate concentration. Denitrification is modeled as a proportion of nitrate concentration, which is consistent with approaches used elsewhere [*Boyer et al.*, 2006; *Breve et al.*, 1997; *de Willigen*, 2005; *Dettmann*, 2001], and was the most appropriate given our limited data. Denitrification rates are known to vary with temperature, so we used data from the USGS NWIS online database to compared monthly groundwater temperatures. We found that groundwater temperatures remain relatively constant throughout the year between 12.6 °C and 16.2 °C (Table 3) and consequently, we chose to make the denitrification rate temperature independent. These equations assume that there is nothing limiting the rates of denitrification and uptake, such as organic carbon.

Table 3: Average monthly groundwater temperatures calculated using data from USGS water quality samples [U.S. Geological Survey (USGS), 2006].

Month	**Average Temperature (°C)**
January	12.9 ± 2.1
February	12.6 ± 1.7
March	13.9 ± 4.1
April	13.4 ± 2.5
May	13.9 ± 1.9
June	14.7 ± 2.1
July	15.6 ± 2.9
August	16.2 ± 2.6
September	14.6 ± 1.9
October	13.9 ± 2.1
November	14.1 ± 2.3
December	12.7 ± 2.2

Model Inputs

Most concentrations are calculated within the model, but there are several external nitrate sources that require an independent nitrate concentration input. We used monthly averages calculated from the USGS NWIS database to determine nitrate concentration input values for releases from Cochiti Dam, Jemez River, Rio Puerco, and the Tijeras Arroyo (Table 4) [U.S. Geological Survey (USGS), 2006]. We used average values from our synoptic sampling for the wastewater treatment plants. For the Bernalillo and Rio Rancho WWTPs, we used a value of 10 mg/l and for the Albuquerque WWTP we used a value of 8 mg/l. The term for “other” recharge to the groundwater is from septic systems in the groundwater zones considered in the nitrate model. We used a value of 20 mg/l (twice the highest WWTP concentration) as an estimate of nitrate concentrations from leaking septic systems.

Table 4: Nitrate concentrations used as model inputs that vary by month.

Month	**Cochiti Dam** NO_3-N mg/l	**Jemez River** NO_3-N mg/l	**Rio Puerco** NO_3-N mg/l	**Tijeras Arroyo** NO_3-N mg/l
January	0.11	0.20	1.11	1.90
February	0.15	0.12	0.87	0.63
March	0.09	0.11	0.71	1.35
April	0.10	0.10	0.49	1.50
May	0.09	0.06	0.68	0.89
June	0.07	0.06	0.43	0.75
July	0.06	0.12	0.71	0.60
August	0.05	0.10	0.81	0.48
September	0.08	0.16	0.78	0.89
October	0.04	0.27	1.00	0.74
November	0.09	0.17	0.93	0.97
December	0.13	0.07	0.52	1.10

2.3.5. Mass Balance

Once the model was constructed, we calculated a mass balance for each reach to ensure conservation of mass when nitrate is treated as a conservative solute. This process also served as an important check on our overall model structure since the river and conveyance systems have a forced mass balance. The change in groundwater nitrate storage was compared to the difference between external inputs to the reach and reach outflows (upstream flows, tributaries, wastewater inputs, interbasin groundwater flow, groundwater pumping, and groundwater recharge to the deeper aquifer, river and conveyance outflows) for each timestep. We were able to achieve mass balance on all reaches from Cochiti to San Marcial.

2.3.6. Model Calibration

Model calibration was difficult due to the limited available nitrate data, particularly for the agricultural conveyance channels. Some historical nitrate data exists for the Rio Grande and the LFCC, but there is no published historical data for the agricultural drains other than values determined by the authors from 2001 – 2005. We addressed this issue by using the average annual groundwater concentration and average monthly LFCC and river concentrations as a test of the rates and processes employed to remove nitrogen.

The average groundwater nitrate concentration within the Middle Rio Grande is 0.18 mg/L NO_3-N when locally high values attributed to proximity of septic systems are excluded [*Plummer et al.*, 2004]. We calculated average monthly concentration values for the river and LFCC to compare with the model results using surface water data from the USGS NWIS online database [*U.S. Geological Survey (USGS)*, 2006] (Figure 4). We chose to use data after 1993 based on work by Passell *et al.*, [2004] which suggested that significant improvements were made to the Albuquerque WWTP that reduced nitrate concentrations in the river around this time. Therefore, while the model runs from 1975 – 2005, we only compared modeled and observed data between 1993 and 1995.

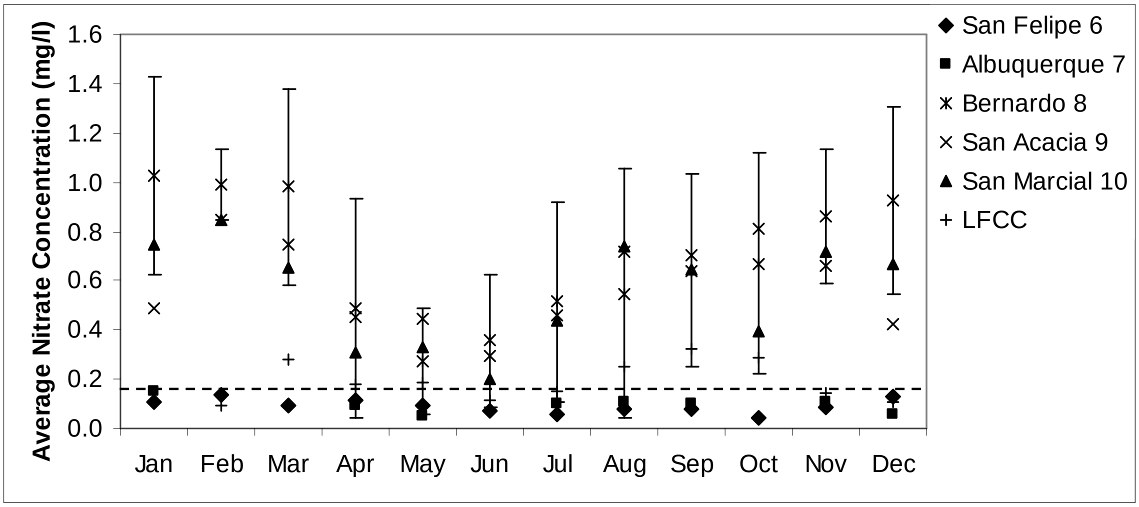


Figure 4. Average monthly nitrate concentrations at each gauging station from 1993-2005. Dashed line represents the average groundwater nitrate concentration (0.18 mg/l).

2.4. Results

2.4.1. Conservative Nitrate Behavior

Without nitrate removal by uptake or denitrification, modeled nitrate concentrations in reaches 6 and 7 are similar to observed values, while nitrate concentrations in reaches 8, 9, and 10 are generally 4 to 6 times higher than observed values (Figure 5). Seasonal variations in observed concentrations are not represented in the modeled concentrations. Modeled river concentrations are highest March through July whereas observed river concentrations are lowest during the same period. Similarly, modeled conveyance channel concentrations are highest during the spring and summer whereas observed concentrations are lowest in summer and winter and highest in March and September.

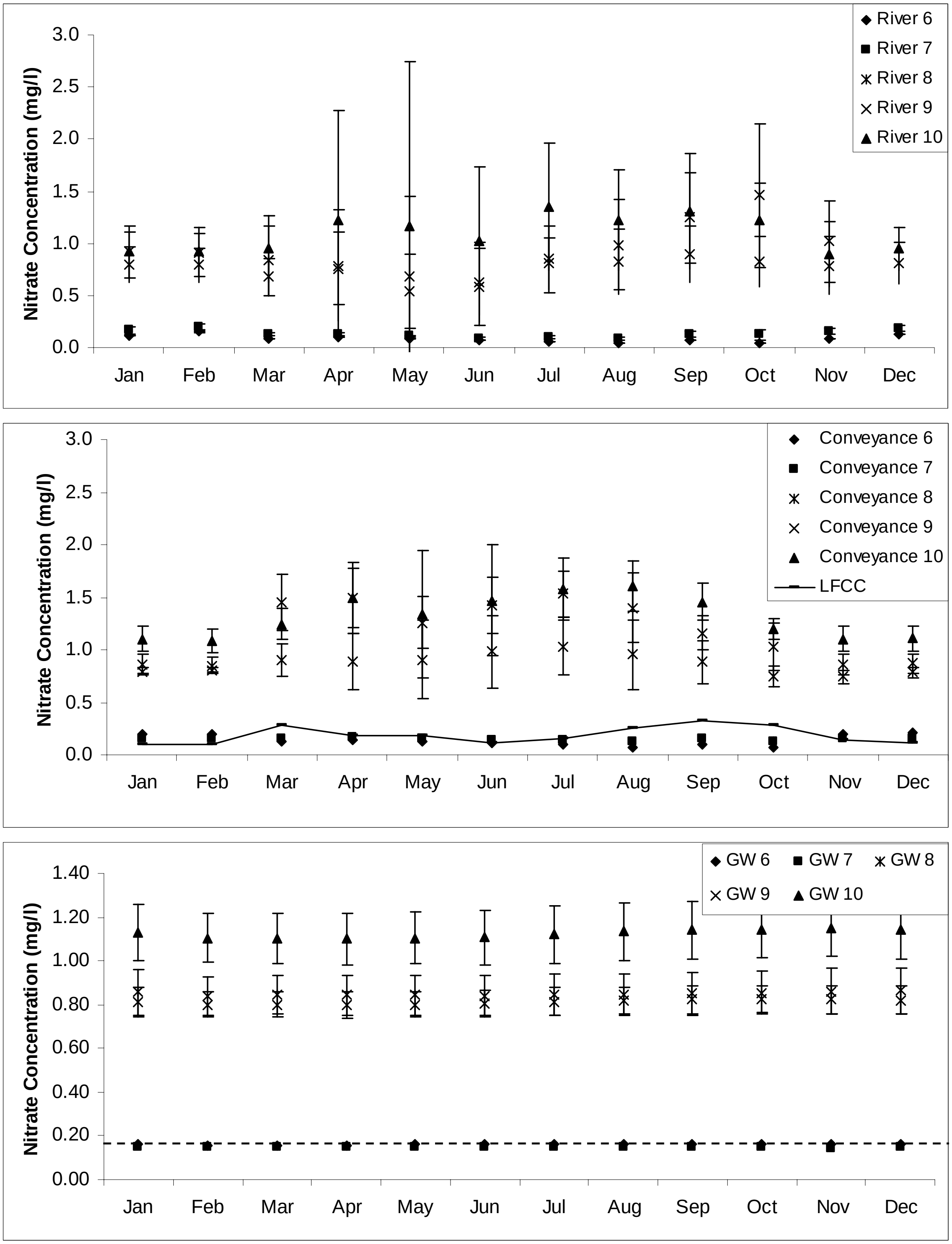
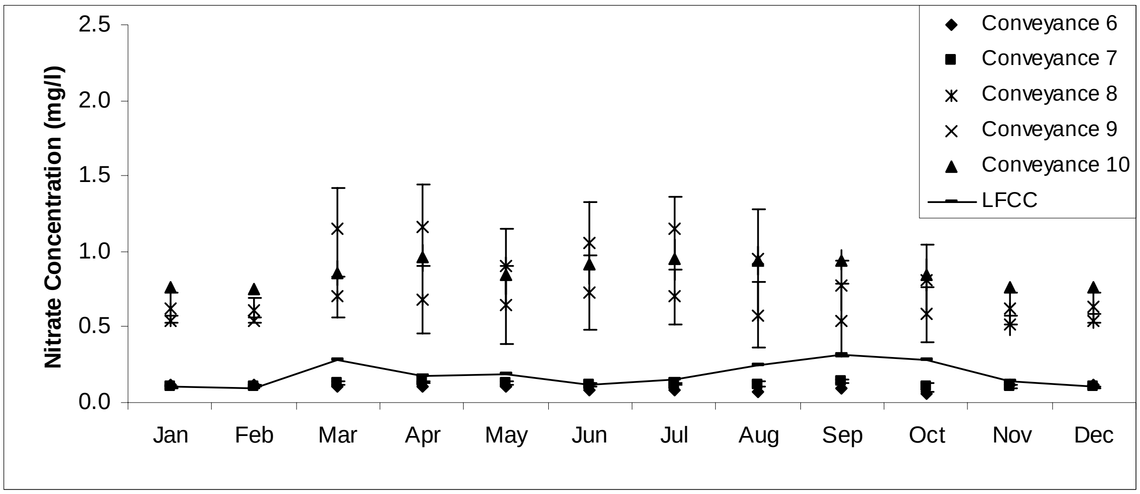


Figure 5. Modeled average monthly nitrate concentrations in the river (top), conveyance channels (middle) and groundwater (bottom) when nitrate is treated conservatively in the model and there is no uptake or denitrification.

2.4.2. Model Calibration with Uptake and Denitrification

Considering both riparian and agricultural uptake, average concentrations in both the conveyance channel and groundwater decrease, but remain higher than observed concentrations (Figure 6). When riparian uptake was calculated using 90% of the riparian ET flux and agricultural uptake was calculated using 50% of the crop ET losses, there was little change to the nitrate concentrations in Reaches 6 and 7 whereas nitrate concentrations in Reaches 8, 9 and 10 decreased between 24 and 41% from the conservative modeled concentrations. Interestingly, nitrate concentrations decreased in the winter months as well as during the summer months.



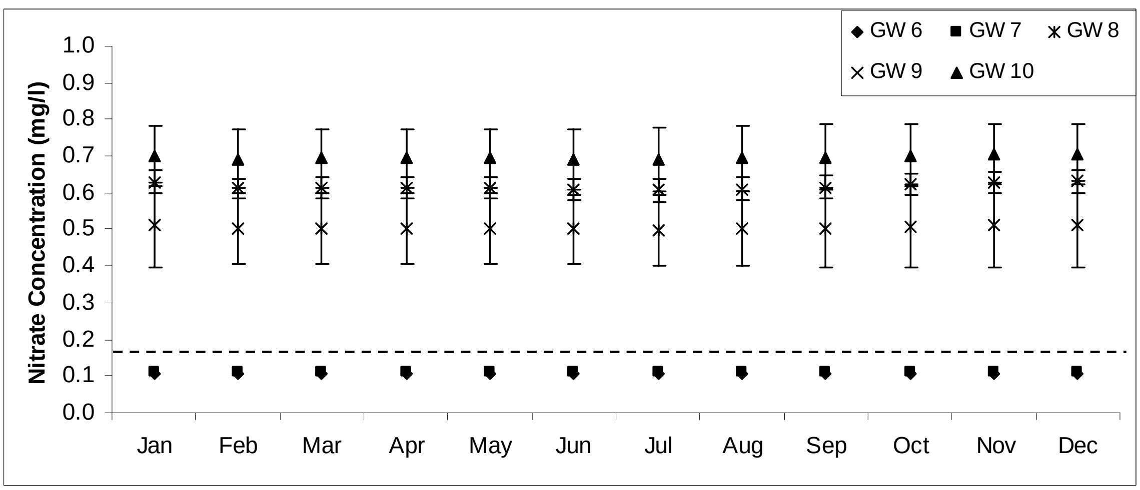


Figure 6. Modeled average monthly nitrate concentrations for each reach with riparian uptake equal to 90% of ET and agricultural uptake equal to 50% of Crop ET in the agricultural conveyance system (top) and groundwater (bottom).

Nitrogen removal by denitrification decreased nitrate concentrations in both the groundwater and conveyance channels (Figure 7). Groundwater concentrations decreased by approximately an order of magnitude using a 5% denitrification rate, and were lower than the average observed groundwater nitrate concentration. Winter conveyance channel nitrate concentrations decreased 75% to 90% and were similar to observed nitrate concentrations. Summer nitrate concentrations decreased 30% to 60% but remained higher than observed concentrations.

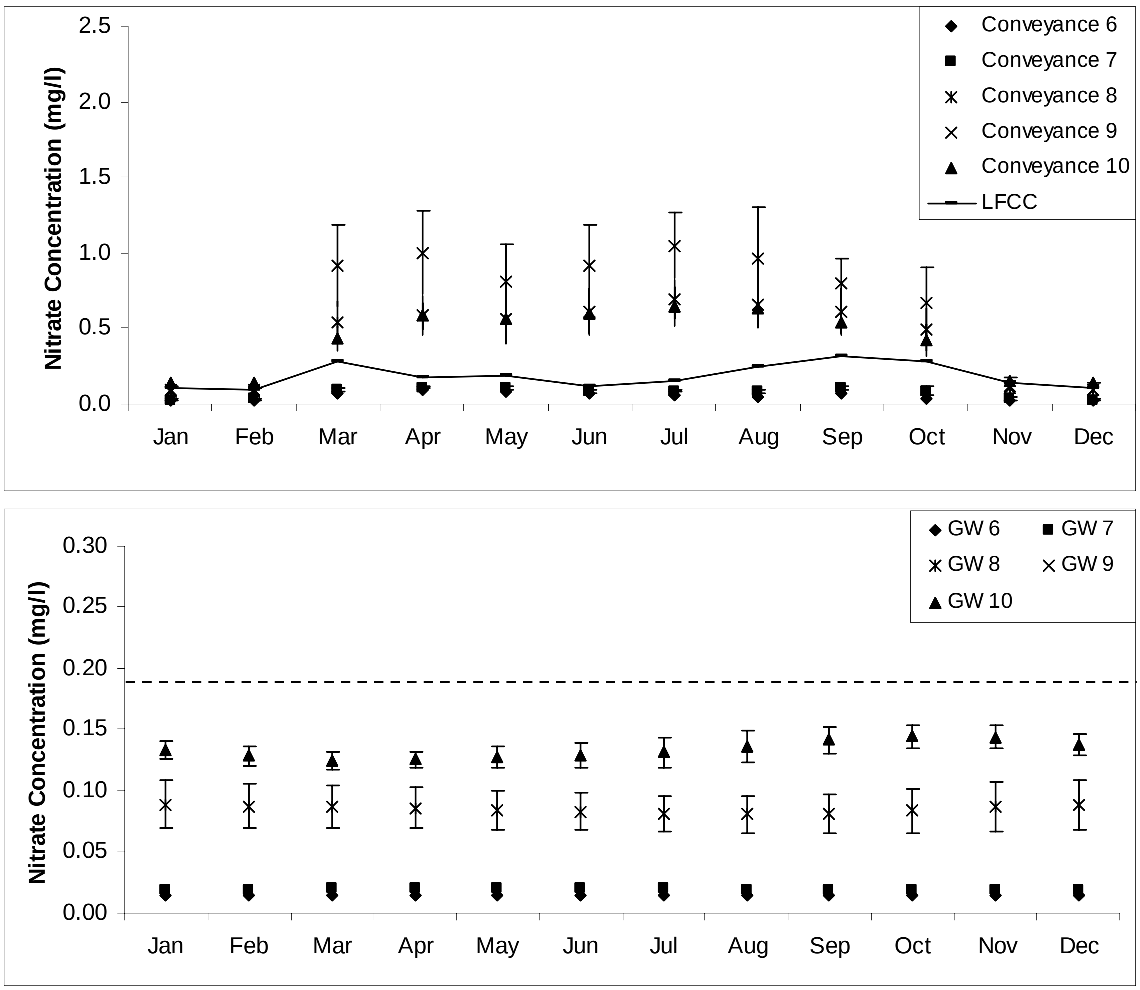


Figure 7. Modeled average monthly nitrate concentrations for each reach with 5% denitrification in the agricultural conveyance system (top) and groundwater (bottom).

When the model used a combination of uptake and denitrification to reduce nitrate concentrations, the modeled concentrations were similar to observed concentrations in the river, conveyance and groundwater system (Figure 8). In addition, the combination of uptake and denitrification produced seasonal variability in the modeled concentrations that agrees with the observed seasonal variability. In the conveyance system, only Reach 9 had average monthly concentrations higher than observed, but the modeled concentrations still captured seasonal variations. Modeled groundwater nitrate concentrations in Reach 10 were average, while the other reaches had lower than average concentrations. Interestingly, the groundwater concentrations were higher when both denitrification and uptake were considered than when only denitrification was included in the model. Modeled and observed nitrate concentrations in the river generally had good agreement, except in October when the modeled concentrations were higher than observed.

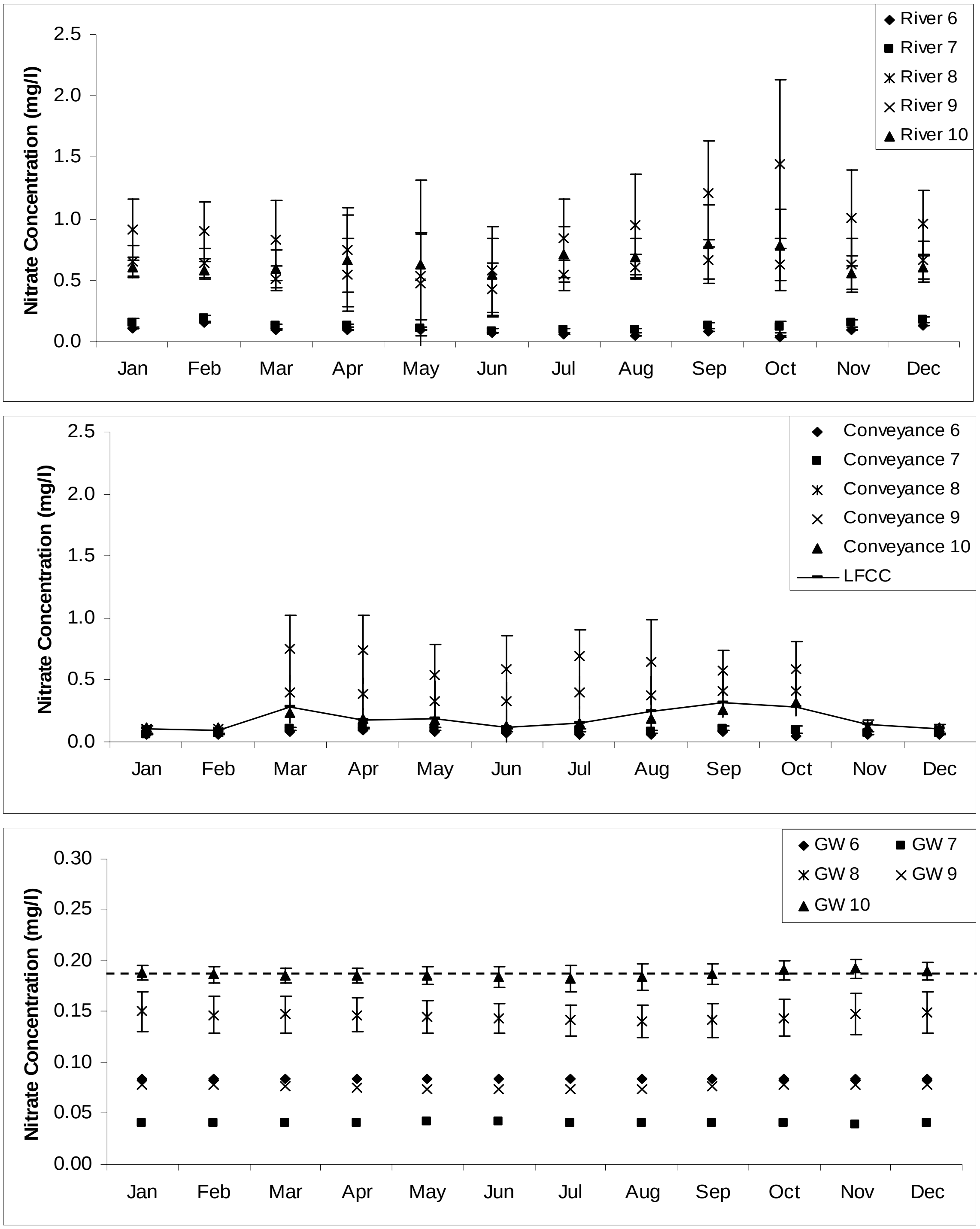


Figure 8. Modeled average monthly nitrate concentrations for each reach with 5% denitrification, riparian uptake equal to 90% of ET and agricultural uptake equal to 50% of Crop ET in the river (top) agricultural conveyance system (middle) and groundwater (bottom).

2.4.3. Nitrogen Removal

Based on our model results, approximately 497 metric tons of nitrate-N is removed from the surface and groundwater systems each year, with the largest amount of removal occurring April through September (Table 5). In Reach 6, 70-100% of the nitrate removed from the reach is via riparian uptake in all months except March when agricultural uptake removes 95% of the nitrate. In Reach 7, both agricultural and riparian uptake are important processes for removing nitrate. In contrast to reaches 6 and 7, denitrification rather than riparian uptake is the uptake is the primary process by which nitrate is removed from the system. In Reaches 8 and 10, approximately 20-40% of the nitrate removal is attributed to agricultural uptake.

Table 5: Total mass of nitrate-N removed from each reach and the percentage of removal by agricultural uptake, riparian uptake, and denitrification.

Reach	Removal	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec
6	Mass (kg)	1.0	1.5	47.4	259.1	677.3	738.0	582.5	393.1	242.7	101.5	56.0	0.7
	%AgUp	0	0	95.0	29.8	14.0	9.9	9.7	10.0	17.5	7.9	0	0
	%RipUp	100	100	5.0	70.2	86.0	90.1	90.3	90.0	82.5	92.1	100	100
	%DNT	0	0	0	0	0	0	0	0	0	0	0	0
7	Mass (kg)	0.7	1.1	243.4	575.4	1031.9	1070.4	897.0	623.1	431.6	155.9	41.6	0.5
	%AgUp	0	0	99.4	68.5	44.1	35.9	39.6	43.6	57.4	51.6	0	0
	%RipUp	100	100	0.6	31.5	55.9	64.1	60.4	56.4	42.6	48.4	100	100
	%DNT	0	0	0	0	0	0	0	0	0	0	0	0
8	Mass (kg)	16,153	15,806	21,434	24,104	24,841	25,897	27,027	25,191	23,507	18,964	15,899	16,116
	%AgUp	0	0	26.1	33.7	34.4	37.3	41.4	38.4	34.4	17.9	0	0
	%RipUp	0.0	0.0	0.0	0.8	2.5	2.8	2.1	1.5	0.8	0.4	0.3	0.0
	%DNT	100.0	100.0	73.9	65.5	63.1	59.9	56.5	60.1	64.7	81.7	99.7	100.0
9	Mass (kg)	1329	1316	1334	1401	1486	1490	1439	1370	1338	1316	1306	1324
	%AgUp	0	0	2.5	3.9	5.0	4.5	4.6	3.8	2.8	0.9	0	0
	%RipUp	0	0	0	4.9	11.1	12.5	9.9	6.0	2.4	0.5	0.2	0
	%DNT	100	100	97.5	91.2	83.9	82.9	85.5	90.1	94.9	98.6	99.8	100
10	Mass (kg)	11,626	11,503	15,965	20,472	18,998	20,191	20,417	18,019	15,511	13,700	11,880	11,748
	%AgUp	0	0	28.6	41.1	34.3	36.6	40.0	34.8	24.5	14.1	0	0
	%RipUp	0	0	0	2.6	5.3	6.7	4.8	2.5	1.2	0.2	0.1	0
	%DNT	100	100	71.4	56.2	60.4	56.7	55.2	62.6	74.3	85.6	99.9	100

2.5. Discussion

2.5.1. Nitrate Removal

According to these modeling results, nitrate removal must be occurring within the system because when nitrate is treated as a conservative solute the modeled concentrations are 4 to 6 times higher than the observed concentrations, except in Reaches 6 and 7. Upstream of Albuquerque in Reaches 6 and 7, our model results suggest that nitrate behaves conservatively. This apparent conservative behavior could be due to the low nitrate concentrations in water released from Cochiti Dam and the lack of a large source of nitrate within each reach. Alternatively, there could be a reduced demand for nitrate by plants and microbes in these regions.

Riparian and agricultural ET reduce nitrate concentrations, but alone are insufficient to match the values and seasonality of observed concentrations. Although our model results indicated conservative behavior in Reaches 6 and 7, we included riparian and agricultural uptake at the same rate as in the other reaches because it seemed unlikely that these processes would not be occurring within the reaches. A somewhat unexpected result was that uptake reduced concentrations in the winter, when there was no ET. By removing nitrate through uptake, nitrate is not concentrated and does not accumulate as much in the groundwater when only water is removed by ET.

Denitrification was very effective at reducing the groundwater nitrate concentrations and winter conveyance channel concentrations, but could not reproduce the summer conveyance channel concentrations. We did not consider denitrification in Reaches 6 and 7 because groundwater and conveyance channel nitrate concentrations in matched or were lower than observed concentrations without any nitrogen removal. Additionally in Reach 6, DO concentrations in groundwater are higher (0.5 – 5 mg/l) than in other areas around the river, suggesting that conditions may be unfavorable for denitrification[*Plummer et al.*, 2004]. A relatively low denitrification rate of 5% reduced groundwater concentrations below average, likely because of high nitrate concentrations without any removal by uptake. During the winter, water is not diverted from the river into the conveyance system so that water in the conveyance system is derived entirely from groundwater. Therefore the low nitrate concentrations produced by denitrification are consistent with the low nitrate concentrations in the conveyance system during the winter.

Combining uptake and denitrification processes results in reasonable concentrations in the river, groundwater and conveyance channels and captures most of the seasonal variability. The groundwater concentrations increase slightly with both processes over denitrification alone because the removal of nitrate through uptake reduces groundwater nitrate concentrations and subsequently the mass removed by the denitrification since denitrification is concentration dependent. Lower than average modeled nitrate concentrations in groundwater may indicate that processes which increase nitrate concentrations such as mineralization and nitrification could be important processes in the groundwater.

2.5.2. Agricultural System vs. Riparian System

According to our model results, denitrification is the primary control on nitrogen removal followed by agricultural uptake in Reaches 8, 9, and 10. Riparian uptake is only an important removal mechanism in Reaches 6 and 7 where denitrification was not required to lower nitrate concentrations. Agricultural uptake likely removes a larger proportion of nitrate than riparian uptake because the concentration in water applied to crops is higher than the concentration in groundwater after denitrification. Reaches 8, 9, and 10 the riparian system is not responsible for removing a large percentage of nitrate and denitrification almost always removed more nitrate than uptake. While our model results suggest that the riparian system is not important for nitrogen removal, it may provide an important source of organic carbon needed for denitrification.

2.5.3. Groundwater Residence Time

The residence time of groundwater in the model has important implications for understanding the nitrate removal mechanisms. Due to the monthly time step of the model and the fact that water is not stored in the river or conveyance systems, groundwater has roughly a one month residence time. Based on stable isotope data collected from the river and conveyance channels in January and August between 2001 and 2006, the actual residence time is much longer than 1 month (Figure 9). The stable isotope values for the LFCC plot between the summer and winter river samples, suggesting that the water in the LFCC is a mixture of summer and winter river water. If the residence time was truly only one month, then the summer LFCC samples would plot very close to the summer river samples and the winter LFCC samples would plot very close to the winter river samples. Therefore, it is possible that the low nitrate concentrations observed during the winter in the conveyance channels is due to uptake that occurred during the previous summer rather than solely denitrification occurring in the winter. The current model structure does not allow us to investigate these possibilities. However, stable isotope (^{15}N and ^{18}O of nitrate) could be used to test the hypothesis presented here that denitrification is the most important nitrate removal mechanism.

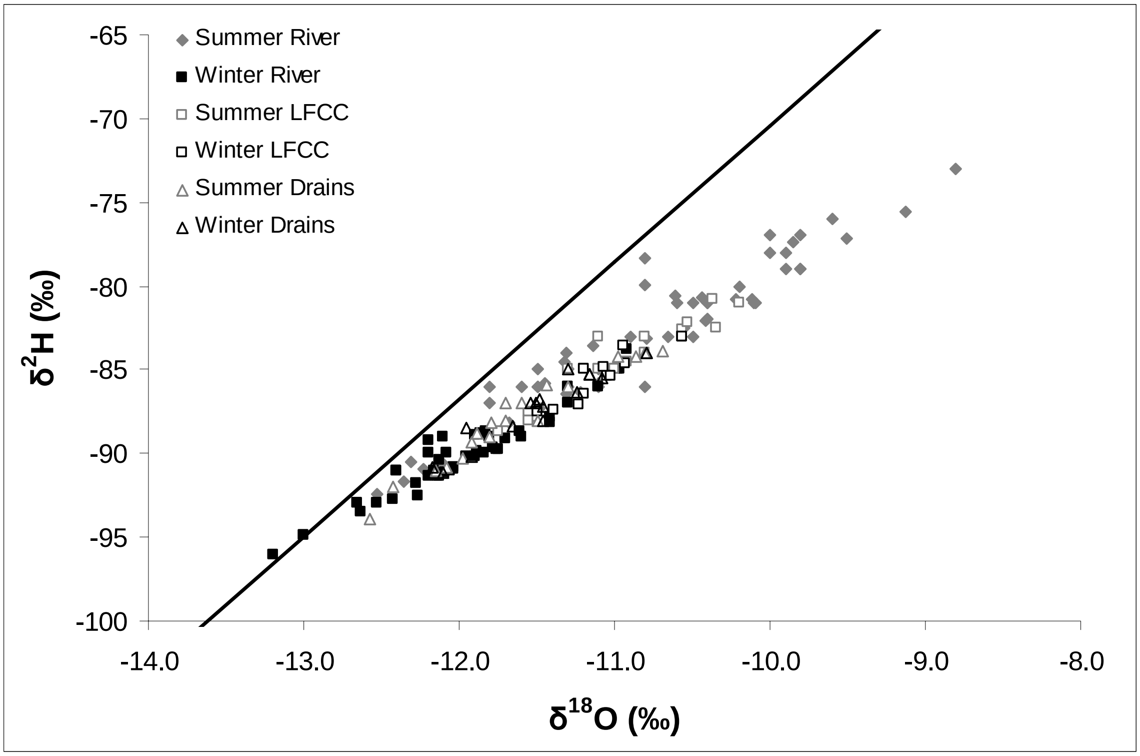


Figure 9. Stable isotope data for river LFCC samples collected between 2001 and 2006. Note that LFCC samples plot between summer and winter river samples.

2.6. Summary

Within the Middle Rio Grande, a combination of riparian uptake, agricultural uptake and denitrification are required to modify nitrate concentrations to match observed concentrations and seasonal patterns. Between Cochiti Dam and Albuquerque, only riparian and agricultural uptake are required. Downstream of the major wastewater inputs at Albuquerque, denitrification in addition to riparian and agricultural plant uptake is required to achieve observed concentrations. We modeled riparian uptake as 90% of riparian ET, assuming that 90% of ET is transpiration. Crop uptake was modeled as 50% of crop ET because of larger evaporation rates associated with irrigated agriculture and the large amount of alfalfa in the region which does not require fertilization. Denitrification was modeled as 5% removal of groundwater nitrate. Based on this model calibration, a total of 497 metric tons nitrate-N is removed from the Middle Rio Grande annually. In the upper region of the study area, both riparian uptake and agricultural uptake were equally important as removal mechanisms. In the reaches downstream of Albuquerque, denitrification was responsible for 55-100% of the nitrate removal. The second most important removal mechanism for nitrate was agricultural uptake. Interestingly, uptake by riparian vegetation was responsible for less than 13% of the nitrate removal suggesting that the majority of the nitrogen removal takes place in the agricultural system rather than the stream/riparian system. Alternatively, the riparian system may provide an important source of carbon to facilitate denitrification in the groundwater under the riparian area.

REFERENCES

Adams, A. I., F. Goff, and D. Counce (1995), Chemical and Isotopic Variations of Precipitation in the Los Alamos Region, New Mexico: Los Alamos National Laboratory, Los Alamos, 1-35 pp, Los Alamos, NM.

Alexander, R. B., J. R. Slack, A. S. Fitzgerald, and T. L. Schertz (1997), Data from selected U.S. Geological Survey national stream water quality monitoring networks, *Water Resources Research*, *34*(9), 2401-2405.

Anderholm, S. (1997), Water-Quality Assessment of the Rio Grande Valley, Colorado, New Mexico and Texas: shallow ground-water quality and land use in the Albuquerque Area, central New Mexico, 1993, U.S. Geological Survey, Water Resources Investigations Report, WRI 97-4067, 73 pp.

Anderholm, S. K., M. J. Radell, and S. F. Richey (1995), Water-quality assessment of the Rio Grande Valley study unit, Colorado, New Mexico and Texas; analysis of selected nutrient, suspended-sediment, and pesticide data, U.S. Geological Survey Water Resources Investigations Report, WRI 94-4061, 203 pp.

Bartolino, J. R., and J. C. Cole (2002), Ground-Water Resources of the Middle Rio Grande Basin, New Mexico, U.S. Geological Survey Circular 1222, 132 pp.

Boyer, E. W., R. B. Alexander, W. J. Parton, C. Li, K. Butterbach-Bahl, S. D. Donner, R. W. Skaggs, and S. J. D. Grosso (2006), Modeling Denitrification in Terrestrial and Aquatic Ecosystems at Regional Scales, *Ecological Applications*, *16*(6), 2123-2142.

Bradbury, N. J., A. P. Whitmore, P. B. S. Hart, and D. S. Jenkinson (1993), Modelling the fate of nitrogen in crop and soil in the years following application of [15]N-labelled fertilizer to winter wheat, *Journal of Agricultural Science 121*(3), 363-379.

Breve, M. A., R. W. Skaggs, J. E. Parsons, and J. W. Gilliam (1997), Drainmod-N, a Nitrogen Model for Artificially Drained Soils, *Transactions - American Society of Agricultural Engineers*, *40*(4), 1067-1076.

de Willigen, P. (2005), Nitrogen turnover in the soil-crop system; comparison of fourteen simulation models, *Nutrient Cycling in Agroecosystems*, *27*(2-3), 141-149.

Dettmann, E. D. (2001), Effect of Water Residence Time on Annual Export and Denitrification of Nitrogen in Estuaries: A Model Analysis, *Estuaries*, *24*(4), 481-490.

Feng, Q., G. D. Cheng, and M. K. Masao (2000), Trends of water resource development and utilization in arid north-west China, *Environmental Geology*, *39*(8), 831-838.

Hogan, J. F., F. M. Phillips, S. K. Mills, J. Hendrickx, J. Ruiz, J. T. Chesley, and Y. Asmerom (*In Press*), Geologic origins of salinization in a semiarid river: the role of sedimentary brines, *Geology*.

Hooper, R. P., B. T. Aulenbach, and V. J. Kelly (2001), The National Stream Quality Accounting Network: a flux-based approach to monitoring the water quality of large rivers, *Hydrological Processes*, *15*(7), 1089 - 1106.

Johnson, B. L., W. B. Richardson, and T. J. Naimo (1995), Past, present, and future concepts in large river ecology, *Bioscience*, *45*(3), 134-141.

Johnsson, H., L. Bergstrom, J. P.-E., and K. Paustian (1987), Simulated nitrogen dynamics and losses in a layered agricultural soil, *Agriculture, ecosystems & environment 18*(4), 333-356.

Kingsford, R. T. (2000), Protecting rivers in arid regions or pumping them dry?, *Hydrobiologia*, *427*(1-3), 1-11.

Langmuir Laboratory (2006), http://www.ee.nmt.edu/%7Elangmuir/ August 2006.

Lowrance, R. (1992), Groundwater Nitrate and Denitrification in a Coastal Plain Riparian Forest *Journal of Environmental Quality 21*(3), 401-405.

Molles, M. C., C. S. Crawford, L. M. Ellis, H. M. Valett, and C. N. Dahm (1998), Managed flooding for riparian ecosystem restoration - Managed flooding reorganizes riparian forest ecosystems along the middle Rio Grande in New Mexico, *Bioscience*, *48*(9), 749-756.

Moore, S. J., and S. K. Anderholm (2002), Spatial and Temporal Variations in Streamflow, Dissolved Solids, Nutrients, and Suspended Sediment in the Rio Grande Valley Study Unit, Colorado, New Mexico and Texas, 1993–95, U.S. Geological Survey, Water Resources Investigations Report, WRI 02-4224, 52 pp.

Mueller, D. K., and N. E. Spahr (2006), Nutrients in streams and rivers across the Nation -1992-2001: U.S. Geological Survey Scientific Investigations Report 2006-5107, 44 pp.

Mulholland, P. J., H. M. Valett, J. R. Webster, S. A. Thomas, L. W. Cooper, S. K. Hamilton, and B. J. Peterson (2004), Stream denitrification and total nitrate uptake rates measured using a field 15N tracer addition approach, *Limnology and Oceanography*, *49*(3), 809-820.

Multi-Resolution Land Characteristics Consortium (MRLC) (2007), National Land Cover Database (NLCD 2001), http://www.mrlc.gov/mrlc2k_nlcd.asp, 2007.

Oelsner, G. P., P. D. Brooks, and J. F. Hogan (2007), Nitrogen sources and sinks within the Middle Rio Grande, NM, *Journal of the American Water Resources Association*, *43*(4), 14pp., doi: 10.1111 j.1752-1688.2007.00071.x.

Ortiz, D., and K. M. Lange (1996), Water Resources Data, New Mexico, Water Year 1995., U.S. Geological Survey, Water Data Report, NM-95-1, 628 pp.

Passell, H. D., C. N. Dahm, and E. J. Bedrick (2004), Hydrological and geochemical trends and patterns in the Upper Rio Grande, 1975 to 1999, *Journal of the American Water Resources Association*, *40*(1), 111-127.

Passell, H. D., C. N. Dahm, and E. J. Bedrick (2005), Nutrient and organic carbon trends and patterns in the upper Rio Grande, 1975-1999, *Science of the Total Environment*, *345*(1-3), 239-260.

Peterson, J. T., and T. J. Kwak (1999), Modeling the effects of land use and climate change on riverine smallmouth bass, *Ecological Applications*, *9*(4), 1391-1404.

Phillips, F. M., J. F. Hogan, S. K. Mills, and J. M. H. Hendrickx (2003), Environmental tracers applied to quantifying causes of salinity in arid-region rivers: preliminary results from the Rio Grande, southwestern USA, in *Water Resources Perspectives: Evaluation, Management and Policy*, edited by A. S. Alsharhan and W. W. Wood, pp. 327-334, Elsevier, Amsterdam, The Netherlands.

Plummer, L. N., L. M. Bexfield, S. K. Anderholm, W. E. Sanford, and E. Busenberg (2004), Geochemical characterization of ground-water flow in the Santa Fe group aquifer system, Middle Rio Grande Basin, New Mexico, U.S. Geological Survey Water Resources Investigations Report, WRI 03-4131, 395 pp.

Richey, J. E., J. I. Hedges, A. H. Devol, P. D. Quay, R. Victoria, L. Martinelli, and B. R. Forsberg (1990), Biogeochemistry of carbon in the Amazon River, *Limnology and Oceanography*, *35*(2), 352-371.

Roach, J. D. (2007), Integrated surface water groundwater modeling in the Upper Rio Grande in support of scenario analysis, Ph.D. thesis, 341 pp, University of Arizona, Tucson, AZ.

S.S. Papadopulos and Associates Inc. (SSPA) (2002), Evaluation of the Middle Rio Grande Conservancy District Irrigation System and Measurement Program, Prepared for: New Mexico Interstate Stream Commission, 596 pp, Boulder, Colorado.

Schade, J. D., J. R. Welter, E. Marti, and N. B. Grimm (2005), Hydrologic exchange and N uptake by riparian vegetation in an arid-land stream, *Journal of the North American Benthological Society*, *24*(1), 19-28.

Stanley, E. H., S. G. Fisher, and N. B. Grimm (1997), Ecosystem expansion and contraction in streams, *Bioscience*, *47*(7), 427-435.

U.S. Geological Survey (USGS) (2006) USGS Water Quality Data, http://waterdata.usgs.gov/nwis/qw. September 2006.

Valett, H. M., M. A. Baker, J. A. Morrice, M. C. Crawford, M. C. Molles Jr., C. N. Dahm, D. L. Moyer, J. R. Thibault, and L. M. Ellis (2005), Biogeochemical and metabolic responses to the flood pulse in a semiarid floodplain, *Ecology*, *86*(1), 220-234.

Western Regional Climate Center (WRRC) (2005) Western U.S. Historical Summaries (individual stations). http://www.wrcc.dri.edu/CLIMATEDATA.html March 2005.

Williams, W. D. (2001), Anthropogenic salinisation of inland waters, *Hydrobiologia*, *466*(1-3), 329-337.

Yepez, E. A., R. L. Scott, W. L. Cable, and D. G. Williams (2007), Intraseasonal Variation in Water and Carbon Dioxide Flux Components in a Semiarid Riparian Woodland, *Ecosystems*, *10*.

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