
ELSEVIER
Journal of
Environmental
Management

Journal of Environmental Management 113 (2012) 61–70

Contents lists available at SciVerse ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Modeling riverine pathogen fate and transport in Mexican rural communities and associated public health implications

Agustin Robles-Morua [a], Alex S. Mayer [b,*], Martin T. Auer [b], Enrique R. Vivoni [a,c]

[a] School of Space and Earth Exploration, Arizona State University, Tempe, AZ, USA
[b] Department of Civil and Environmental Engineering, Michigan Technological University, Houghton, MI, USA
[c] School of Sustainable Engineering and the Built Environment, Arizona State University, Tempe, AZ, USA

A R T I C L E I N F O

Article history:
Received 26 June 2011
Received in revised form
10 February 2012
Accepted 23 August 2012
Available online

Keywords:
Wastewater contamination
Pathogen fate and transport

A B S T R A C T

The discharge of untreated or poorly treated wastewater to river systems remains a major problem affecting public and environmental health, particularly in rural communities of less developed countries. One of the primary goals in setting policies for wastewater management is to reduce risks to human health associated with microbial contamination of receiving water. In this study, we apply a surface water quality model to develop an *Escherichia coli* based indicator that reflects the quality of surface water and the potential impact to recreational users in a large, rural river in northwest Mexico (upper Sonora River). The model assesses the relative importance of streamflow variations and the uncertainty in *E. coli* removal coefficient parameters for the predictions of *E. coli* concentrations in the river. Given the sparse information on streamflow, we use a physically-based, distributed hydrologic model to generate tributary contributions to the river. We determined the best estimate and uncertainty of *E. coli* removal rates to explore the impacts of parameter uncertainty on the transport of *E. coli* downstream from two wastewater discharge zones. Our results depict the regions in the river that are in noncompliance with fresh water pathogen norms. The impact of streamflow variability and uncertainty in the removal rates of pathogen indicators was used to derive a range of river distances in noncompliance. The comparison between two sites with different streamflow behaviors was used to illustrate the impacts of streamflow spatiotemporal variability on pathogen indicators. We derive a simple relationship that can be used to assess the relative importance of dilution (ratio of wastewater discharge to river discharge) and pathogen removal (ratio of residence time to reaction time).

1. Introduction

The World Health Organization estimates that roughly 10% of diseases globally are attributable to water quality, sanitation, or hygiene-related issues (Prüss-Űstün et al., 2008). In Latin America, the setting of this work, the significance of water quality degradation and attendant public health problems is seriously underestimated (Byrd et al., 1997; Ingram et al., 1995; McLennan, 2000; Poblete-Davila et al., 2005; Ratner and Rivera, 2004; Robles-Morua et al., 2011). For example, Robles-Morua et al. (2011) found disparity between beliefs about waterborne disease-related risks and the actual risks present in a study region in Mexico, where citizens and local government officials had little awareness of the prevalence and significance of water-borne diseases, believing instead that disease symptoms were an acceptable part of everyday life. This belief structure undermines stakeholder commitment to recommendations relating to disease prevention (McLennan, 2000; Robles-Morua et al., 2011), including the operation and management of wastewater programs. Results from these studies point to the need for broad-based campaigns focusing on prevention and describing the serious nature of waterborne disease, its causes and prevalence.

Hardin (1968) and Rogers (2008), writing some four decades apart, have challenged scientists, engineers and managers to consider the degree to which technical solutions to global water resource issues are defined or tempered by human values. Attention is drawn to the evolving paradigm proposed by Ravetz (1999) shown in Fig. 1, where "system uncertainties" and "decision stakes" are considered to be essential elements of analysis. Accommodation of system uncertainties associated with the transition to post-normal science calls for a replacement of deterministic approaches of the normal science domain with probabilistic techniques. The

* Corresponding author. Tel.: +1 906 487 3372; fax: +1 906 487 2943.
E-mail address: asmayer@mtu.edu (A.S. Mayer).

0301-4797/$ – see front matter © 2012 Elsevier Ltd. All rights reserved.
http://dx.doi.org/10.1016/j.jenvman.2012.08.035

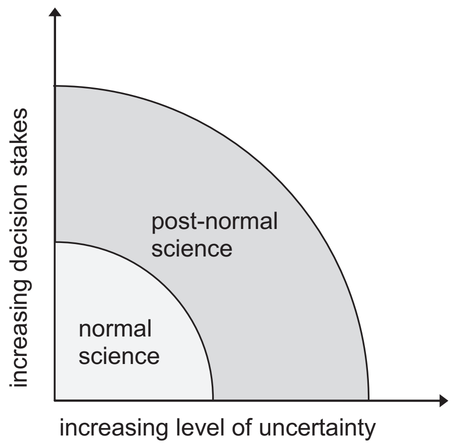

Fig. 1. Contrasting normal and post-normal science in terms of system uncertainty and decision stakes (modified from Rogers, 2008).

tandem increase in decision stakes and uncertainties associated with available information or facts reflects the fact that individual or community stakeholder life styles or livelihoods may be at stake and that issues of social justice and multiple stakeholder values merit consideration (Rogers, 2008). The case considered here, focusing on public health and wastewater management in rural Mexico, offers an exceptional opportunity to address the precepts of post-normal science in one such application.

Recognition of these gaps serves to form the thesis of this research, pointing to the need for tools operating at three levels: empowering the scientific and engineering community to bring to bear contemporary technologies to address public health issues, supporting policy-makers, public health professionals and government officials in effectively communicating the risk of waterborne disease, and offering connectivity between the professional community and lay stakeholders at the local level. Here, the appropriate management focus is reduction in exposure to pathogens associated with untreated or poorly-treated wastewater.

Water quality models have proven to be effective in educating stakeholders and supporting the decision-making processes (Dietz et al., 2004). These tools aid in conceptualization of issues by clarifying the intrinsic complexity of fate and transport processes in natural systems (Chapra, 1997; DePinto et al., 2004) and thus reducing knowledge gaps with respect to waterborne disease-related risk perception. Estimating and conveying the risk associated with exposure to wastewater is further complicated by variability in the physical characteristics of the river system, wastewater and its pathogens, and the uncertainty in model forcing and parameters (Mitch et al., 2010). Streamflows can vary over several orders of magnitude on an inter- and intra-annual basis. In addition, streamflow varies considerably along the length of a river channel as tributary flows enter the channel. The spatiotemporal variability of river flows can have a significant impact on the assimilative capacities of pollutants by influencing both dilution and residence time (e.g. Chen and Ma, 2008; Lence and Takyi, 1992; Lipp et al., 2001; Sincock et al., 2003). Pathogen concentrations are sensitive not only to river flows but also to pathogen removal mechanisms associated with pathogen die-off and settling of particles with attached pathogens. The estimation of pathogen removal rates is a critical component to adequately model the fate and transport of pathogens (Auer and Niehaus, 1993; Canale et al., 1993; Chapra, 1997), but these rates are highly sensitive to environmental conditions and are difficult to measure in situ. Ranges of estimate removal rates vary from less than 0.1 day^{-1} to greater than 10 day^{-1} (e.g. Auer and Niehaus, 1993; Beaudeau et al., 2001; Menon et al., 2003). It is not unusual for estimated pathogen removal rates to have 95% confidence intervals that bracket at least 50% of the best estimates (Menon et al., 2003). Given that pathogen removal is generally modeled as an exponential process, the estimated removal of pathogens can vary by orders of magnitude within these confidence intervals.

In this study, we describe the development of a mathematical model for indicator organisms applied at two sites on the upper Sonora River (USR), an ungauged, rural system in northwest Mexico. Conditions in the USR are representative of those in lesser developed countries throughout the world in three respects. First, community wastewater management along the USR is technologically primitive. Federal standards for monitoring of treatment processes and receiving water quality are not maintained or enforced (Biswas et al., 2005; Robles-Morua et al., 2009; Tortajada, 2003) and thus pollutant loadings and receiving water quality conditions are poorly described. Second, measurements of river flows, critical to model development and application are largely unavailable. This issue is of special significance due to the high degree of spatiotemporal variation in hydrology experienced here, reflecting the impact of the North American monsoon (Gochis et al., 2006; Vivoni et al., 2007). Third, site-specific information on pathogen removal rates, fundamental to modeling applications, is not available.

This work addresses knowledge gaps relating to public health at two levels. First, we seek to overcome the challenges of characterizing the risk from waterborne disease in remote, less developed regions by integrating hydrologic simulations, a parsimonious program of field measurements, and a mathematical model of indicator organism fate. Second, we seek to quantify that risk, providing public health and regulatory officials with an accessible and scientifically-sound vision of the risks posed locally, including recognition of uncertainties in that vision associated with environmental forcing conditions (e.g. river flows and pollutant loads; Chen and Ma, 2008) and pollutant fate (pathogen removal processes; Ferguson et al., 2003). This work will provide critical technical information and support for stakeholder outreach in the upper Sonora River and offers guidance to those seeking to reduce knowledge gaps in risk perception regarding wastewater discharges and waterborne disease in other less developed regions.

2. Methodology

2.1. Study site

This research was conducted within the upper Sonora River in the state of Sonora, Mexico (Fig. 2), a region experiencing water quality and public health issues similar to those observed across the country. Wastewater collection and treatment systems in this region utilize lagoons or ditches with discharge to the river either directly through channels or indirectly by seepage through berms. Lagoons, favored for their minimal operational demands and maintenance costs, were built in the early 1980's. The lagoons were built without protection against wastewater seepage to groundwater. Based on dates of construction, most of these systems have exceeded their design lifetimes and are currently abandoned. Communities in Mexico are required to monitor wastewater effluent annually (NOM-001-ECOL, 1996). However, access to lagoon sites is difficult, making inspection and maintenance problematic. The local communities are responsible for lagoon operation, maintenance and performance evaluation, but no facility in Sonora has submitted reports indicating compliance with federal

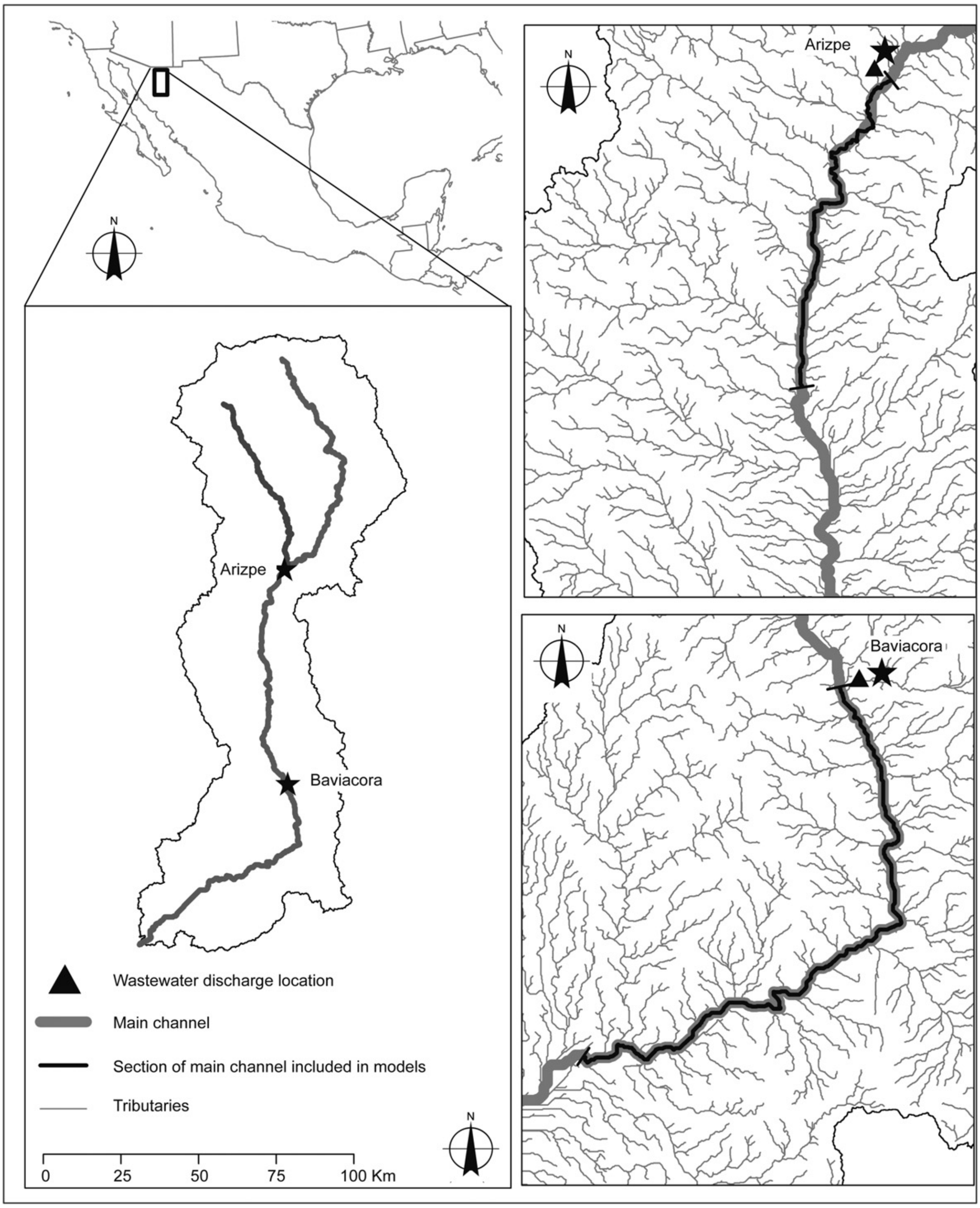

Fig. 2. Study sites at Arizpe and Baviacora along the upper Sonora River, Mexico.

standards for discharge (Robles-Morua et al., 2009; Tortajada, 2003). In addition to the poor state of wastewater treatment in this region, communities located along the main stem of the Sonora River display a high incidence of waterborne disease (Robles-Morua et al., 2011).

The study focuses on two communities located along the USR: Arizpe (population 1623) and Baviacora (population 1885) (Fig. 2) (INEGI, 2005). The basin has an arid to semi-arid climate with average annual rainfall ranging from 327 to 540 mm and a mean annual temperature above 22 °C (INEGI, 1993; Mendez-Barroso et al., 2009). The two locations are representative of similarly-sized, rural communities along the 200-km main stem of the river. The Arizpe collection system discharges into an unlined wastewater lagoon. There is no engineered outlet structure. Effluent seeps through containment berms, entering the river as a diffuse input. Wastewater in Baviacora is directed to an unlined lagoon located about 150 m from the river. An outlet structure provides for the discharge of lagoon effluent directly to the river as a point source flow. The USR, including reaches below lagoon discharges at Arizpe and Baviacora, is valued by local stakeholders for recreational amenities, including bathing.

2.2. Model framework, parameterization and application

Modeling of pathogen fate and transport is performed using U.S. EPA's QUAL2K software (Chapra et al., 2008), a one-dimensional river water quality model. This model was selected because it is able to integrate inputs from direct and diffuse sources to determine impacts on water quality in receiving water bodies, including the fate and transport of pathogens. Model inputs consist of data obtained from field studies that were conducted to determine *Escherichia coli* loads and removal rates. QUAL2K has built-in functions that minimize mass balance and other errors associated with numerical approximations of water quality fate and transport equations. In addition, we used an advanced hydrologic model to estimate streamflow as a function of time and distance along the USR.

2.2.1. Model segmentation

Application of QUAL2K requires that the river system be partitioned into reaches with specified characteristics, e.g. boundary latitude and longitude, reach length, bed slope and channel width. These characteristics are assumed to remain constant for every reach. Reaches are organized in ascending order beginning from the headwaters of the river's main stem. A single headwater reach is used to represent upstream inputs into the main channel of the USR. Reach lengths were established as the distance between tributaries entering the main channel. Flows corresponding to each tributary were established at the beginning of each reach. Fig. 2 shows the modeled sections and the locations of tributaries for each site. A 29-m Digital Elevation Model (DEM) obtained from the Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER DEM, 2008) was used to delineate boundaries for the USR watershed and derive the channel network, location of the tributaries, reach lengths and bed slope.

2.2.2. Equation of state

Members of the coliform group of bacteria are widely applied as indicators for assessing public health risks associated with exposure to human and animal wastes, including total coliforms, fecal coliforms and *E. coli*. The presence and abundance of *E. coli* is significantly correlated with contact-associated waterborne diseases (Cheung et al., 1990). U.S. EPA (1986, 2003) and WHO (2003) recommends that concentrations of this indicator not exceed 126 CFU/100 ml in freshwaters. While the Mexican government does not include *E. coli* among its ambient water quality standards, *E. coli* is selected here as the state variable representing a generic pathogen in model simulations in recognition of its broader contemporary application worldwide. The equation of state for the *E. coli* state variable includes terms for loading, advective and diffusive mass transport and removal (loss to death and settling):

$$\frac{\partial X}{\partial t} = W - U\frac{\partial X}{\partial x} + E\frac{\partial^2 X}{\partial x^2} - kX \tag{1}$$

where

X = *E. coli* concentration, CFU·100 mL^{-1}
t = time, d
W = *E. coli* load, CFU·100 mL^{-1} d^{-1}
U = river velocity, m d^{-1}
E = longitudinal dispersion coefficient, m^2 d^{-1}
x = distance, m
k = *E. coli* removal rate coefficient, d^{-1}

2.2.3. E. coli loads

Wastewater discharge was determined directly at Baviacora by measuring wastewater flows, chloride and *E. coli* concentrations directly from the outfall. Wastewater discharges at Baviacora and Arizpe were also estimated through chloride mass balances (Albek, 2003; Jun et al., 2007; Kim et al., 2005, 2002; Marti et al., 2004). The mass balance on chloride in a river receiving a wastewater discharge is given by,

$$Q_{down} \cdot C_{down} = Q_{up} \cdot C_{up} + Q_{ww} \cdot C_{ww} \tag{2}$$

where:

Q = flow, m^3 d^{-1}
C = chloride concentration, g m^3

and the subscripts refer to locations upstream of (*up*), downstream of (*down*) and at the specified wastewater discharge (*ww*).

Equation (2) is solved for Q_{ww}, the wastewater flow, given measured values for all of the other terms. River flows (Q_{up}, Q_{down}) were determined by the velocity-area method. Channel dimensions and river velocity (Marsh-McBirney Model 2000 Flow Mate current meter) were measured at locations upstream and downstream of the lagoon discharges at Arizpe and Baviacora (Fig. 3). Chloride concentrations, analyzed by ion chromatography (Dionex Model DX500, APHA, 2005), were measured at the upstream and downstream locations and for the lagoon outflows (C_{ww}). Effluent chloride concentrations (C_{ww}) for Baviacora were measured directly from the discharge pipe. At Arizpe, where there is no outlet structure, C_{ww} was measured at the lagoon inlet under the assumption that chloride mass would be conserved through the lagoon and during the transport of the lagoon discharge to the river.

The directly measured wastewater discharge at Baviacora was 2130 m^3 d^{-1}. The wastewater discharges determined through chloride mass balances were 1090 m^3 d^{-1} and 1940 m^3 d^{-1} for Arizpe and Baviacora, respectively. The comparison between direct and indirect measurements of wastewater discharge at Baviacora indicates that the chloride mass balance method is reasonably accurate.

E. coli concentrations were measured using the Petrifilm™ plate count method (3M Corporation). At Baviacora, *E. coli* levels were determined for samples collected from an outlet channel connecting the lagoon discharge pipe with the river. For Arizpe, where

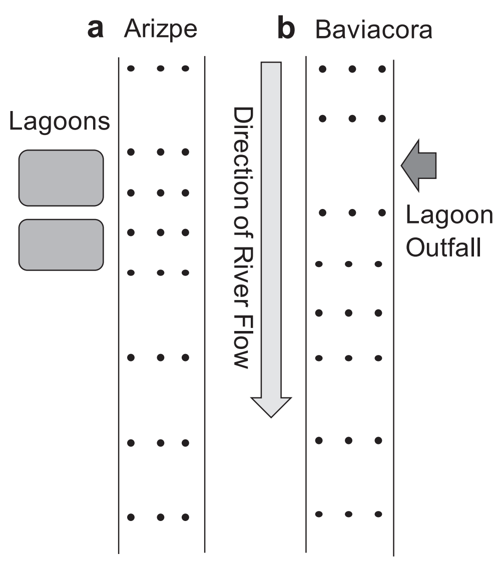

Fig. 3. Sampling sites on the USR at (a) Arizpe and (b) Baviacora.

discharge occurs through berm seepage, *E. coli* concentrations were measured over a 12-station river grid adjoining the lagoon (Fig. 3) and the maximum value obtained was used for calculations.

E. coli concentrations for the lagoon discharges were 71,000 and 212,000 CFU 100 mL^{-1}, for Arizpe and Baviacora. The difference in the *E. coli* concentrations is due primarily to the loss of bacteria via filtration through the lagoon berm in Arizpe. Assuming the flow and *E. coli* concentrations for lagoon discharges to be constant with respect to time, loads for Arizpe and Baviacora were estimated to be 7.7×10^{11} and 4.1×10^{12} CFU d^{-1}, respectively.

2.2.4. Mass transport

In Equation (1), advection is parameterized through the river velocity (U), which is calculated by iteratively solving for river depth, H, using Manning's equation,

$$U = \frac{1}{n} R^{\frac{2}{3}} S^{\frac{1}{2}} \tag{3}$$

where:

n = Manning's roughness coefficient, m$^{-1/3}$ s
R = hydraulic radius = $(WH)/(W + 2H)$, m, assuming a rectangular cross section
W = channel width, m
H = river depth, m
S = bed slope, m m^{-1}

Channel width was measured directly at each study site and assumed constant. Bed slope was derived using the Digital Elevation Model described previously. Estimates of velocity were confirmed by direct measurement at the two study sites. Longitudinal dispersion, E, is calculated in QUAL2K as

$$E = 0.011 \frac{U^2 R^2}{H(gHS)^{0.5}} \tag{4}$$

where:

g = gravitational constant, 9.81 m s^{-2}

There are no flow gauging stations on the portion of the USR studied here. Instead, daily headwater flows and contributions to flow from tributaries to the main channel were calculated through a fully distributed hydrologic modeling simulation (Triangulated Irregular Network (TIN)-Based Real-time Integrated Basin Simulator; tRIBS, Ivanov et al., 2004a,b). This physically-based model is capable of estimating flow in ungauged river basins by relying on parameters that can be related to quantifiable landscape properties. The tRIBS model relies on site-specific information regarding the spatial distribution of landscape variation in topography, soil and vegetation types and depths to the water table and bedrock. We based our simulations on a previous modeling effort in the Sonora River reported by Vivoni et al. (2010) who documented the model performance relative to a spatial set of observations during the North American monsoon (NAM). Forcing conditions for tRIBS (e.g. precipitation and meteorological variables) were derived using a continuous, bias-corrected dataset obtained from the North American Land Data Assimilation System (NLDAS) for a one year period beginning in June 2007. Additional details of these hydrologic simulations are provided by Robles-Morua (2010).

The tRIBS model was used to generate synthetic hourly streamflows in all tributaries of the USR for the interval June 2007 through May 2008 (Fig. 4). Using long term daily historical records of precipitation stations from the Mexican National Water Commission (CONAGUA) we were able to determine that the period selected for our simulations was a normal year compared to the long term annual averages. Hourly streamflow values in the tributaries were summed to daily amounts for use within QUAL2K at the specific locations where tributaries intersected with river reaches. Fig. 2 illustrates the connectivity between tRIBS tributaries and the QUAL2K main channel. Flows are lower at Arizpe than at Baviacora due to its location further upstream. The impact of the NAM is clearly evident in the hydrographs in Fig. 4, with highest flows observed in July–September and again in December. Recreational use of the river is greatest in low flow months (March to mid-July) and it is at these times that users are most vulnerable to risk from waterborne disease. Fig. 5 shows flows downstream from the wastewater discharges as a function of distance downstream from the wastewater discharge for the two sites. The distances on the x-axis correspond to the length of the river that was simulated downstream of the discharge points (33 and 60 km respectively for Arizpe and Baviacora). Greater variability in flow is noted downstream of Arizpe as a result of contributions from larger sub-basin tributaries found below Arizpe.

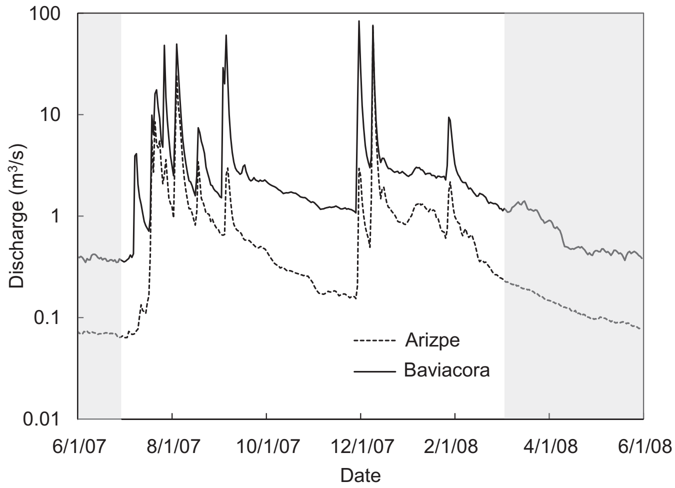

Fig. 4. Simulated daily flow in the USR at Arizpe and Baviacora for the period June 2007 through May 2008. Duration of primary recreational use is indicated by shaded interval.

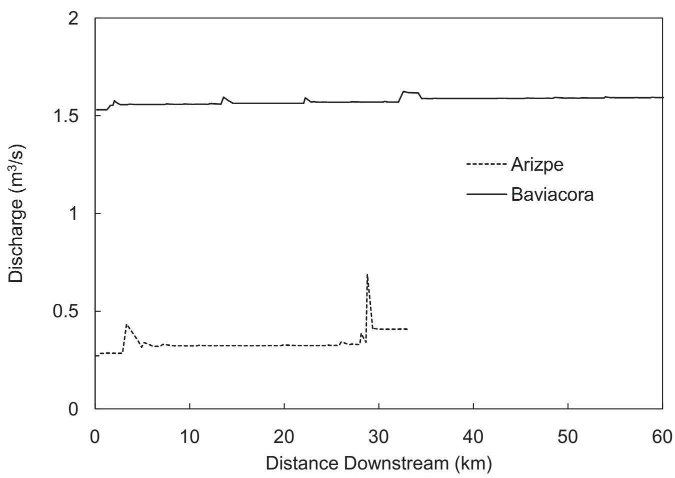

Fig. 5. Simulated, median daily flow in the USR at Arizpe and Baviacora as a function of distance downstream of wastewater discharge.

2.2.5. Kinetics

Traditional treatment of removal kinetics in a model focuses on the first order removal rate coefficient (k in Equation (1)), which reflects losses to mortality and settling (Chapra, 1997). Auer and Niehaus (1993) describe an application of this kinetic framework which was used by Canale et al. (1993) to simulate coliform fate after discharge to a receiving water. Here, we adopt a broadly accepted approach (see Beaudeau et al., 2001) where settling and mortality losses are lumped into a single removal rate coefficient (k). Equation (5) is used to estimate the removal rate coefficient based on field observations of *E. coli* concentration with residence time,

$$\frac{X}{X_0} = \exp(-k\tau) \tag{5}$$

where:

X_0 = *E. coli* concentration at the first sampling station, CFU 100 mL^{-1}
τ = travel time, d

E. coli concentrations were measured directly at the stations identified in Fig. 3 and travel times were quantified from estimates of river velocity.

Paired measurements of river velocity and *E. coli* concentrations were obtained at 10 stations within the Arizpe and Baviacora study sites and an additional site at Aconchi, downstream of Baviacora. River velocity was converted to travel time and the data were fit to the linearized form of Equation (5), yielding an estimate of k (2.15 ± 1.03 d^{-1}, 95% confidence interval; $R^2 = 0.8517$) as the slope of a log-linear regression (Fig. 6). Beaudeau et al. (2001), working with small rivers, reported k values ranging from 0.1 to 17.6 d^{-1} with higher rates being correlated with lower river flows (increased settling loss and more exposure to irradiation). Our result, with its modest level of uncertainty, offers a significant improvement over the level of variability reported by Beaudeau et al. (2001). The k value for this section of the USR compares well with that reported by Auer and Niehaus (1993, 0.5–3.5 d^{-1}) in a review of the literature for temperatures comparable to those of our study, by Menon et al. (2003; 0.19–0.81 d^{-1}) for measurements made in the River Seine, and by Liu et al. (2006; 0.5–1.5 d^{-1}) in a modeling study of Lake Michigan.

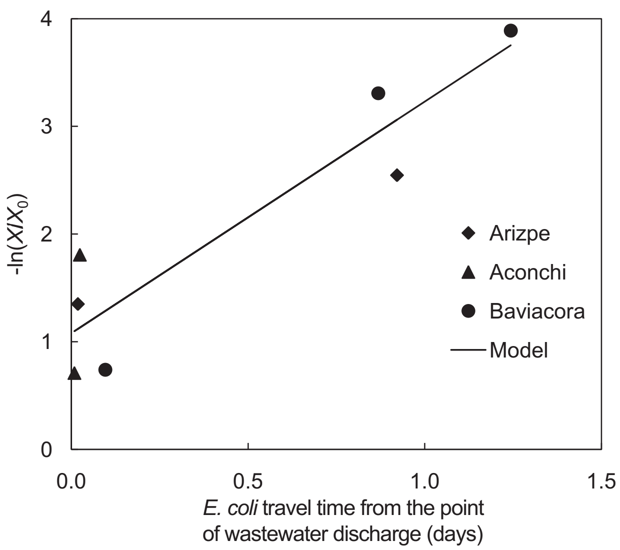

Fig. 6. Determination of the *E. coli* removal rate coefficient based on observed concentrations downstream of Arizpe, Aconchi and Baviacora.

3. Results and discussion

3.1. Model simulations

A suite of model inputs have been developed for the USR at Arizpe and Baviacora quantifying river flow and *E. coli* loads and removal rates. Simulations were performed over river lengths of 33 and 60 km at Arizpe and Baviacora, respectively, with average reach lengths of 0.23 km for the Arizpe simulations and 0.19 km for the Baviacora simulations. To improve the numerical stability of advection–dispersion calculations, each reach was subdivided into five elements. Daily simulations were performed using QUAL2K with the Runge–Kutta integrator run with a time step of 0.04 h. Pathogen source terms and removal rates determined from field studies described in the previous section were incorporated into the model simulations. Calibration and validation of the models were not possible given that water quality data downstream of the wastewater discharges is unavailable. Likewise, given that this is an ungauged river basin, we relied on a physically-based hydrologic model to generate streamflows.

A synthetic parameter, the distance of noncompliance (DNC) was computed as the length of the river downstream of the wastewater discharge with *E. coli* concentrations exceeding the U.S. EPA water quality standard (126 CFU 100 mL^{-1}). This metric was developed to illustrate the impact of spatiotemporal variability in river flow on *E. coli* concentrations and to enhance stakeholder visualization and perception of risk. Simulations utilizing the median flow and ±95% confidence interval for k were conducted to explore the effects of uncertainty in the pathogen removal rate on DNC.

3.2. Insights for public health and regulatory officials

E. coli concentrations, normalized to the water quality standard are shown in Fig. 7 for a median flow day and for the best estimate, lower confidence interval, and upper confidence interval of the removal rate. These results indicate that the downstream length of noncompliance with *E. coli* standards ranges from 9 to 26 km and from 14 to 48 km for Arizpe and Baviacora, respectively, depending on the value of the removal rate. The drops in *E. coli* concentration observed in Fig. 7 (more noticeable in Arizpe) are the result of dilution associated with uncontaminated water entering the main channel from tributaries downstream of the wastewater discharge point. The inflows of water from tributaries entering the main channel can also be observed as peaks in Fig. 5. These results indicate that the wastewater treatment systems at Arizpe and Baviacora are not being operated properly.

If the economic resources and regulatory interest required for treatment upgrades are absent, public health officials must turn to stakeholder education as a means of reducing risk. In particular, we suggest that it is critical to estimate and convey the risk associated with coming into contact with waterborne diseases as a function of variation in streamflows. Soller et al. (2003) used a risk-based modeling approach to determine the level of risk associated to wastewater treatment improvements in terms of discharges per recreation events. Dorner et al. (2006) described how concentrations of pathogens varied with time in relation to streamflows using hydrologic and water quality models. Combining these approaches, we show the model results as probabilistic profiles, to allow assessment of the nature of the risks associated with wastewater discharges (Borsuk et al., 2001). Fig. 8 shows the frequency that the *E. coli* standard is exceeded downstream of the

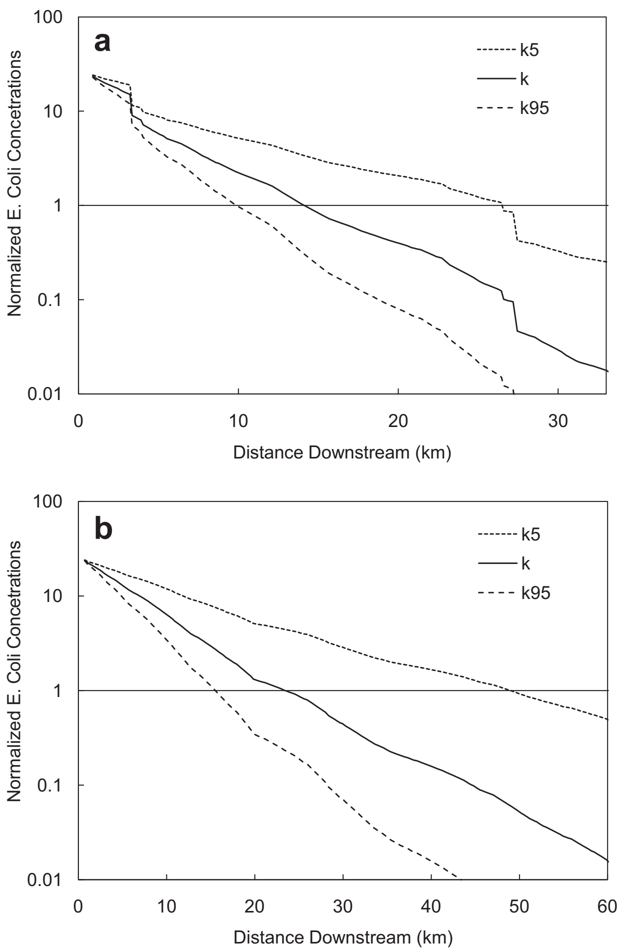

Fig. 7. Profiles of simulated *E. coli* concentration at (a) Arizpe and (b) Baviacora as a function of distance downstream of wastewater discharge for the best estimate, the upper 95% confidence interval, and the lower 95% confidence interval for the *E. coli* removal rate coefficient.

wastewater discharge point for Arizpe and Baviacora. The frequencies are estimated by determining the number of days in a year that a DNC occurs, using QUAL2K output for the best estimate, lower confidence interval, and upper confidence interval of the removal rate.

The results in Fig. 8 indicate the frequency distributions are not symmetric. Approximately 80% of the DNCs occur within 80%–90% of the maximum DNC for both sites and for the best estimate, lower confidence interval, and upper confidence interval of the removal rate. In other words, the shorter distances appear to be significantly less probable. This result may be explained by the exponential relationship between the *E. coli* removal process (see Equation (5)) and the river discharge and the asymmetric nature of the daily discharge distributions. For example, approximately 80% of the daily discharges for Arizpe and Baviacora are below the mean discharges for these sites, as demonstrated in Fig. 4.

These results also clearly show the impact of the uncertainty in the removal rates, where, for example, the location of the 50% frequency varies from 8 km to 13 km to 24 km for Arizpe for the upper confidence interval, best, and lower confidence interval estimates for the removal rate, respectively. The corresponding range for the 50% frequency locations for Baviacora is 14 km–22 km–47 km. The noncompliance distances are greater for Baviacora because of an approximately five-fold higher *E. coli* load in the Baviacora wastewater discharge, as compared to the load in Arizpe.

As a subset of these results, public health and regulatory officials may want to characterize stakeholder risk in terms of the location and timing of standard violations vis-à-vis recreational use. To that end, we present simulation output for the DNC during the recreational season, defined as the March through June interval. The corresponding DNC values during the recreational period are summarized in Table 1 and range from 11 to 13 km at Arizpe and from 15 to 21 km at Baviacora. In order to provide guidance to stakeholders that is both straightforward and risk-averse, we propose limiting recreational contact for a distance downstream of the lagoon discharge equal to the site-specific, period maximum DNC corresponding to the 5% confidence interval, to account for the uncertainty in kinetics: 24.7 km and 46.5 km for Arizpe and Baviacora, respectively.

Fig. 8 indicates these distances along the river channel. Note that in Fig. 8, several small communities are within these distances, in addition to the primary communities of Arizpe and Baviacora. We believe that this approach, founded firmly in the epidemiological evidence supporting public health standards and quantified and visualized through scientific and engineering analysis, will serve well in improving public perception of the risk of waterborne disease in less developed communities such as those of the upper Sonora River.

3.3. Insights for river modelers

It is well established that, at the catchment scale, indicator organism and pathogen loads increase with increasing flow (Dorner et al., 2006; Gonzalez, 1995; Gentry et al., 2006; Grayson et al., 1997; Haydon and Deletic, 2009; Reeves et al., 2004). For example, Lipp et al. (2001) noted in a study of the impacts of seasonal weather variability on microbial contamination in an estuary, that receiving water fecal bacteria levels increased with increasing streamflow. Similarly, Dorner et al. (2006) found that concentrations of *E. coli* were generally underestimated by their model and increased with increasing flow rates. However, consideration of the in-stream impacts of flow variation on mass transport and kinetics is less common. The results in Fig. 9 show that, for wet weather flows (high discharges), the DNC is reduced from that of the average flow condition as increases in streamflow serve to dilute the wastewater inputs. Fig. 9 also shows that the DNC is reduced from the average flow condition under drier conditions (lower discharges) through the effects of extending the residence time (kinetics). The results in Fig. 9 imply that there is a threshold flow below which the DNC is mediated by kinetics and above which DNC is influenced by dilution.

The threshold may be examined quantitatively by use of the Damköhler number, a dimensionless ratio relating the timescales of reaction (here, removal rate)

$$\mathrm{Da} = k\tau \tag{6}$$

and mass transport (here, river flow), where the residence time is defined as

$$\tau = \frac{A_c x}{Q_{up} + Q_{ww}} \tag{7}$$

where:

A_c = river cross-sectional area, m^2
x = river reach length, m.

The analysis evolves from a first-order model of *E. coli* concentration as presented in Equation (5) and applying the mass balance

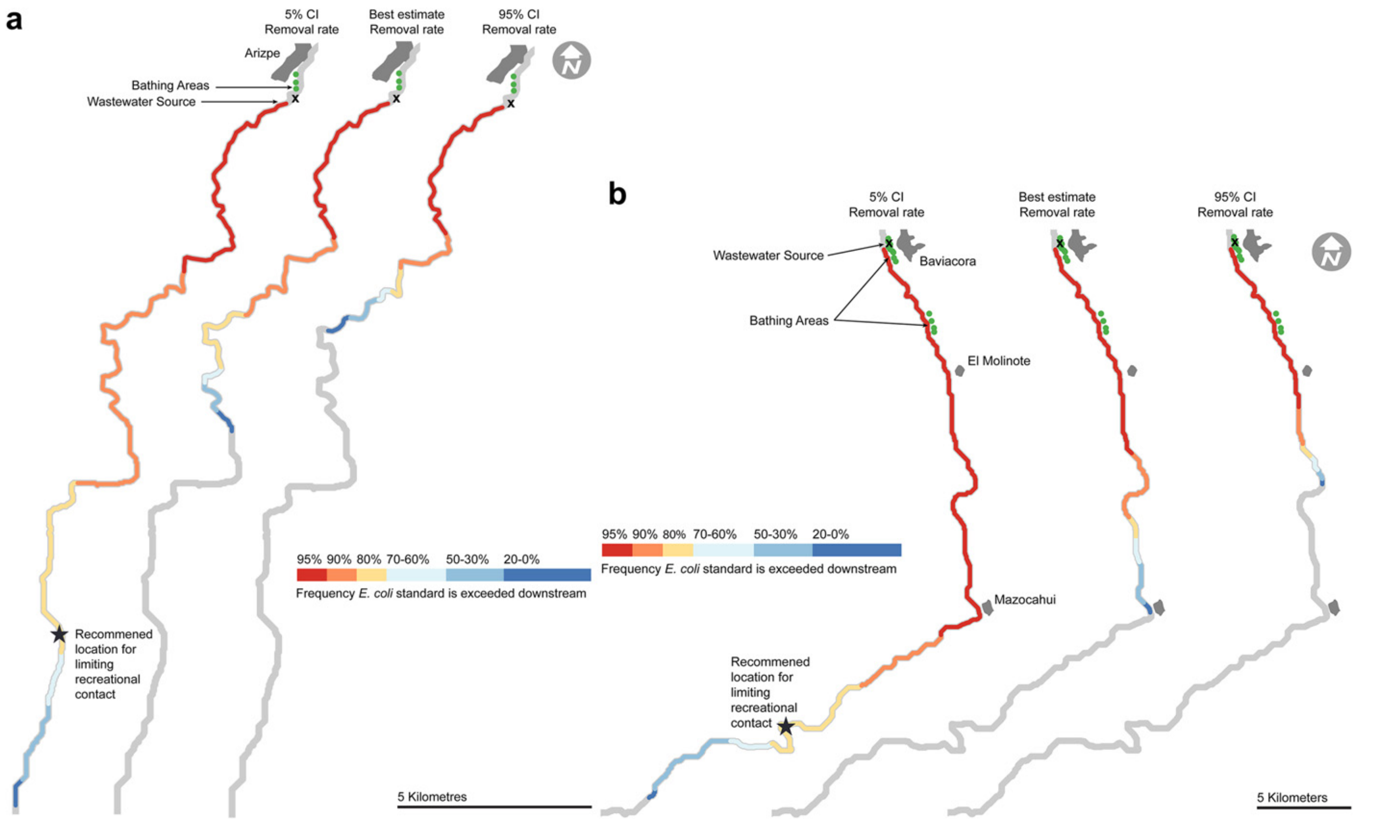

Fig. 8. Frequency of *E. coli* standard being exceeded downstream for (a) Arizpe and (b) Baviacora for the best estimate, the upper 95% confidence interval, and the lower 95% confidence interval for the *E. coli* removal rate coefficient. Starred location indicates downstream extent of recommended river length where recreational contact should be limited.

principles introduced previously in Equation (2). The flow-driven dilution effect on *E. coli* concentration may be determined as

$$X_0 = X_{0,ww}\left(\frac{Q_{ww}}{Q_{up}+Q_{ww}}\right) \tag{8}$$

Substituting Equations (7) and (8) into Equation (5) and taking the derivative $d(X/X_{0,ww})/dQ$ yields

$$\frac{d(X/X_{o,ww})}{dQ} = \left(\frac{kAx}{Q+Q_{ww}}-1\right)\frac{Q_{ww}}{(Q+Q_{ww})^2}\exp\left(\frac{-kA_cx}{Q}\right) \tag{9}$$

We define the dilution fraction as $D = Q_{ww}/Q$. Substituting Da and D into Equation (9), we can solve for the threshold where the slope, $d(X/X_{0,ww})/dQ = 0$:

$$\text{Da} = 1 + \text{D} \tag{10}$$

Thus, at locations where $\text{Da} < (1+\text{D})$, the system is controlled by kinetics and *E. coli* concentrations increase with flow due to reduced residence time – the rising limb of Fig. 9. Where $\text{Da} > (1+\text{D})$, increasing flow serves to dilute *E. coli* concentrations and the system is mass transport controlled, indicated by the falling limb in Fig. 9. The thresholds occur at discharges of approximately 1 $m^3\ s^{-1}$ and 2 $m^3\ s^{-1}$ for Arizpe and Baviacora, respectively, according to the results shown in Fig. 9. These results agree with the theoretical relationship in Equation (10), which suggests that the threshold is linearly related to the distance of noncompliance.

Finally, it is noted in Fig. 9 that there is significant scatter in the predicted DNC for a given discharge. The discharge values in Fig. 9 represent the discharges at the location of the wastewater discharges, and thus do not account for variability in flows downstream of these locations. The scatter occurs as a result of heterogeneity in the spatial distribution of precipitation its effect on streamflows in the main channel and tributaries downstream of the wastewater discharges. In cases where rainfall occurs primarily upstream of the wastewater locations or in a consistent fashion in the tributary watersheds, the DNC-flow relationship is continuous and systematic. However, when rainfall impacts primarily the tributaries downstream of the wastewater discharge locations, the additional dilution reduces *E. coli* levels, resulting in lower values that impart scatter to the relationship. The effect is more significant at Arizpe than at Baviacora due to the fact that the river downstream of Arizpe receives more tributary input. This finding points to the need to consider variability in river flows both temporally and spatially and that river discharges at the wastewater discharge point are not sufficient to predict pathogen transport, since spatial patterns of runoff can vary across the basin over daily time scales.

Table 1
Distance downstream in noncompliance as a function of removal rate coefficients during recreational season.

	Distance downstream in noncompliance (km)					
	Arizpe			Baviacora		
	$\bar{k}$	k_5	k_{95}	$\bar{k}$	k_5	k_{95}
Maximum	12.6	24.7	8.7	21.1	46.5	14.2
Minimum	11.1	22.1	7.3	15.3	27.8	10.1
Average	11.9	22.9	7.9	18.6	40.1	13.0

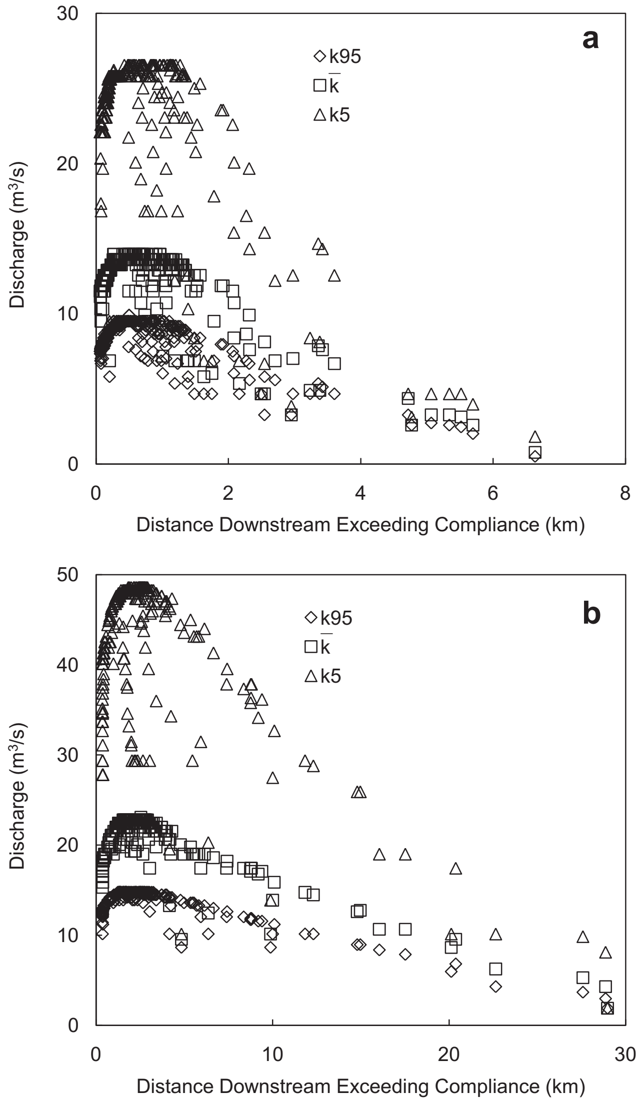

Fig. 9. Distance downstream exceeding *E. coli* standard as a function of river discharge at the location of the wastewater discharge for (a) Arizpe and (b) Baviacora for the best estimate, the upper 95% confidence interval, and the lower 95% confidence interval for the *E. coli* removal rate coefficient.

4. Conclusions

In this work, we have overcome the challenges in characterizing the risk from waterborne disease in a data-sparse, less developed region by integrating hydrologic simulations, a program of field measurements and a model of indicator organism fate. The risk from waterborne disease at two sites in the upper Sonora River, Mexico, was characterized by estimating distances downstream of wastewater discharge points where an indicator organism, *E. coli*, exceeded a well-established standard related to contact with waters contaminated with wastewater. It was shown that these risks are highly sensitive to spatiotemporal variability in river discharges and uncertainty in pathogen removal rates. We provide recommended river distances where contact should be limited during periods where the river discharges are low and recreational use of the river is expected to be high. In addition, by exploring variability in wastewater discharge, river discharges, and pathogen removal rates, we have derived a new, simple relationship that explores the tradeoffs between dilution (river discharge vs. wastewater discharge) and pathogen removal (pathogen removal rates vs. residence time).

This work will provide critical information and support for public health protection in the upper Sonora River and offers guidance to those seeking to reduce knowledge gaps in risk perception regarding wastewater discharges and waterborne disease in other less developed regions. The model developed here could allow stakeholders to explore, in a participatory modeling framework, the impacts of various management strategies, such as improvements to wastewater treatment facilities. Once management strategies are in place, the model also can be used, in coincidence with a monitoring program, to understand the changes in the water quality in the river system and to adapt management strategies in line with stakeholder goals (Borsuk et al., 2001).

While the information generated is relevant for this region, other communities that are also discharging wastewater to the USR were not considered due to difficulty in accessing other sites and because of limited time to travel and analyze samples for estimating *E. coli* loads and removal rates. A drawback of the methodology used in this study is that we only conducted sampling for a single year, therefore interannual variability was not explored here. Although we were able to determine that the simulated year is considered a normal year, the impact of drier years can significantly impact our estimates. It is also critical to quantify other factors that may impair surface water quality (i.e. *E. coli* from cattle and other animals). Overall, however, the methodology suggested in this study is relevant for other regions with sparse or no data and can be used to engage stakeholders and decision makers regarding public and environmental health issues associated with wastewater discharges.

Acknowledgments

We gratefully acknowledge the support of the Mexican federal government of the first author through a CONACYT PhD fellowship. The assistance of Mr. Ryan Biehl while a MS student in the field aspects of this work is acknowledged. Mr. Biehl was supported by an NSF S-STEM fellowship (NSF award DUE-0806569). The assistance of Ms. Cara Shonsey in conducting the QUAL2K simulations is acknowledged as well.

References

Albek, E., 2003. Estimation of point and diffuse contaminant loads to streams by non-parametric regression analysis of monitoring data. Water, Air, and Soil Pollution 147, 229–243.

APHA, AWWA, WEF, 2005. In: Clesceri, L.S., Eaton, A.D., Greenberg, A.E. (Eds.), Standard Methods for the Examination of Water and Wastewater, first ed. American Public Health Association, Washington, D.C.

ASTER DEM, 2008. Provided by Land Processes Distributed Active Archive Center (LP DAAC), Located at the U.S. Geological Survey (USGS) Earth Resources Observation and Science Center (EROS) http://LPDAAC.usgs.gov.

Auer, M.T., Niehaus, S.L., 1993. Modeling fecal coliform bacteria. I. Field and laboratory determination of loss kinetics. Water Research 27 (4), 693–701.

Beaudeau, P., Tousset, N., Bruchon, F., Lefevre, A., Taylor, H.D., 2001. *In situ* measurement and statistical modeling of *Escherichia coli* decay in small rivers. Water Research 35, 3168–3178.

Biswas, A.K., Tortajada, C., Braga, B., Rodriguez, D.J. (Eds.), 2005. Water Quality Management in the Americas. Springer, Berlin, Germany, p. 296.

Borsuk, M., Clemen, R., Maguire, L., Reckhow, K., 2001. Stakeholder values and scientific modeling in the Neuse River watershed. Group Decision and Negotiation 10, 355–373.

Byrd, T.L., VanDerslice, J., Peterson, S.K., 1997. Variation in environmental risk perceptions and information sources among three communities in El Paso. Risk Health Safety and Environment 8, 355–372.

Canale, R.P., Auer, M.T., Owens, E.M., Heidtke, T.M., Effler, S.W., 1993. Modeling fecal coliform bacteria. II. Model development and application. Water Research 27, 703–714.

Chapra, S.C., 1997. Surface Water Quality Modeling. McGraw-Hill, New York, 844 pp.

Chapra, S.C., Pelletier, G.J., Tao, H., 2008. QUAL2K: a Modeling Framework for Simulating River and Stream Water Quality, Version 2.11: Documentation and

Users Manual. Civil and Environmental Engineering Dept., Tufts University, Medford, MA. Available online at: http://www.epa.gov/athens/wwqtsc/html/qual2k.html.

Chen, C., Ma, H., 2008. The uncertainty effects of design flow on water quality management. Environmental Monitoring and Assessment 144 (1–3), 81–91.

Cheung, W.H., Chang, K.C., Hung, R.P., Kleevens, J.W., 1990. Health effects of beach water pollution in Hong Kong. Epidemiological Infections 105, 139–162.

DePinto, J.V., Freedman, P.L., Dilks, D.M., Larson, W.M., 2004. Models quantify the total maximum daily load process. Journal of Environmental Engineering 130 (6), 703–716.

Dietz, T., Tanguay, J., Tuler, S., Webler, T., 2004. Making computer models useful: an exploration of expectations by experts and local officials. Coastal Management 32, 307–318.

Dorner, S.M., Anderson, W.B., Slawson, R.M., Koumen, N., Huck, P.M., 2006. Hydrologic modeling of pathogen fate and transport. Environmental Science and Technology 40, 4746–4753.

Ferguson, C., Husman, A.M.R., Altavilla, N., Deere, D., Ashbolt, N., 2003. Critical Reviews in Environmental Science and Technology 33 (3), 299–361.

Gentry, R.W., McCarty, J., Layton, A., McKay, L.D., Williams, D., Koirala, S.R., Sayler, G., 2006. *Escherichia coli* loading at or near base flow in a mixed-used watershed. Journal of Environmental Quality 35, 2244–2249.

Gochis, D.J., Brito-Castillo, L., Shuttleworth, W.J., 2006. Hydroclimatology of the North American Monsoon region in northwest Mexico. Journal of Hydrology 316, 53–70.

Gonzalez, J.M., 1995. Modeling enteric bacteria survival in aquatic systems. Hydrobiologia 316, 109–116.

Grayson, R.B., Gippel, C.J., Finlayson, B.T., Hart, B.T., 1997. Catchment-wide impacts on water quality: the use of 'snapshot' sampling during stable flow. Journal of Hydrology 199, 121–134.

Hardin, G., 1968. The tragedy of the commons. Science 162, 1243–1248.

Haydon, S., Deletic, A., 2009. Model output uncertainty of a coupled pathogen indicator-hydrologic catchment model due to input data uncertainty. Environmental Modeling & Software 24, 322–328.

INEGI (Mexican National Institute of Statistics, Geography and Informatics), 1993. Estudio Hidrológico del Estado de Sonora. México. Aguascalientes, Aguascaliantes, 88 pp.

INEGI (Mexican National Institute of Statistics, Geography and Informatics), 2005. II Censo de Poblacion y Vivienda. Aguascalientes, Aguascaliantes, http://www.inegi.org.mx/est/contenidos/Proyectos/ccpv/cpv2005/default.aspx.

Ingram, H., Laney, N.L., Gillian, D.M., 1995. Divided Waters: Bridging the U.S.-Mexico Border. University of Arizona Press, Tucson, AZ, 262 pp.

Ivanov, V.Y., Vivoni, E.R., Brass, R.L., Entekhabi, D., 2004a. Catchment hydrologic response with a fully distributed triangulated irregular network model. Water Resources Research 40 (W11102), 23. http://dx.doi.org/10.1029/2004WR003218.

Ivanov, V.Y., Vivoni, E.R., Brass, R.L., Entekhabi, D., 2004b. Preserving high-resolution surface and rainfall data in operational-scale basin hydrology: a fully distributed physically-based approach. Journal of Hydrology 298, 80–111.

Jun, K.S., Kang, J.W., Lee, K.S., 2007. Simultaneous simulation of model parameters and diffuse pollution sources for river water quality modeling. Water Science & Technology 56, 155–162.

Kim, G.H., Choi, E., Lee, D., 2005. Diffuse and point pollution impacts on the pathogen indicator organism level in the Geum River, Korea. Science of the Total Environment 350, 94–105.

Kim, K., Lee, S.J., Oh, C., Hwang, G., Kim, J., Yeo, S., Kim, Y., Park, S., 2002. Inorganic chemicals in an effluent-dominated stream as indicators for chemical reactions and streamflows. Journal of Hydrology 264, 147–156.

Lence, B.J., Takyi, A.K., 1992. Data requirements for seasonal discharge programs: an application of a regionalized sensitivity analysis. Water Resources Research 28 (7), 1781–1789.

Lipp, E.K., Kurz, R., Vincent, R., Rodriguez-palacios, C., Farrah, S.R., Rose, J.B., 2001. The effects of seasonal variability and weather on microbial fecal pollution and enteric pathogens in a subtropical estuary. Estuaries 24 (2), 266–276.

Liu, L., Phanikumar, M., Molloy, S.L., Shively, D.A., Nevers, M.B., Schwab, D.J., Rose, J.B., 2006. Modeling the transport and inactivation of *E. coli* and Enterococci in the near-shore region of Lake Michigan. Environmental Science and Technology 40, 5022–5028.

Marti, E., Aumatell, J., Gode, L., Poch, M., Sabater, F., 2004. Nutrient retention efficiency in streams receiving inputs from wastewater treatment plants. Journal of Environmental Quality 33, 285–293.

McLennan, J.D., 2000. Prevention of diarrhea in a poor district of Santo Domingo, Dominican Republic: practices, knowledge, and barriers. Journal of Health, Population and Nutrition 18, 15–22.

Mendez-Barroso, L.A., Vivoni, E.R., Watts, C.J., Rodriguez, J.C., 2009. Seasonal and interannual relation between precipitation, surface soil moisture and vegetation dynamics in the North American monsoon region. Journal of Hydrology 377, 59–70.

Menon, P., Billen, G., Servais, P., 2003. Mortality rates of autochthonous and fecal bacteria in natural aquatic ecosystems. Water Research 37, 4151–4158.

Mitch, A.A., Gasner, K.C., Mitch, W.A., 2010. Fecal coliform accumulation within a river subject to seasonally-disinfected wastewater discharges. Water Research 44, 4776–4782.

NOM-001-ECOL, 1996. Norma Oficial Mexicana (NOM) que establece los límites máximos permisibles de contaminantes en las descargas de aguas residuales en aguas y bienes nacionales. Available at: http://www.semarnat.gob.mx/leyesynormas/ (accessed 31.07.10.).

Poblete-Davila, S., Bennett, V., Nieves-Rico, M. (Eds.), 2005. Opposing Currents: the Politics of Water and Gender in Latin America. University of Pittsburgh Press, Pittsburgh, PA, p. 264.

Prüss-Űstün, A., Bos, R., Gore, F., Bartramm, J., 2008. Safer Water, Better Health: Costs, Benefits and Sustainability of Interventions to Protect and Promote Health. World Health Organization, Geneva, Switzerland.

Ratner, B., Rivera, A., 2004. Reasserting community: the social challenge of wastewater management in Panajachel, Guatemala. Human Organization 63, 47–57.

Ravetz, J.R., 1999. What is post-normal science? Future 31, 647–654.

Reeves, R., Grant, S.B., Mrse, R.D., Copil Oancea, C.M., Sanders, B.F., Boehm, A.B., 2004. Scaling and management of fecal indicator bacteria in runoff from a coastal urban watershed in Southern California. Environment Science & Technology 38, 2637–2648.

Robles-Morua, A., Mayer, A.S., Durfee, M.H., 2009. Community partnered projects: a case study of a collaborative effort to improve sanitation in a marginalized community in northwest Mexico. Environment, Development and Sustainability 11, 197–213.

Robles-Morua, A., 2010. Integrated Water and Sanitation Risk Assessment and Modeling in the Upper Sonora River Basin (Northwest Mexico). Ph.D. thesis, Michigan Tehnological University, Houghton, MI. 222 pp.

Robles-Morua, A., Halvorsen, K.E., Mayer, A.S., 2011. Waterborne disease-related risk perceptions in the Sonora River basin, Mexico. Risk Analysis 31 (5), 866–878.

Rogers, K.H., 2008. Limnology and the post-normal imperative: an African perspective. Verhandlungen des Internationalen Verein Limnologie 30 (2), 171–185.

Sincock, A.M., Wheater, H.S., Whitehead, P.G., 2003. Calibration and sensitivity analysis of a river water quality model under unsteady flow conditions. Journal of Hydrology 277, 214–229.

Soller, J.A., Olivieri, A.W., Crook, J., Cooper, R.C., Tchobanoglous, G., Parkin, R.T., Spear, R.C., Eisenberg, J.N., 2003. Risk-based approach to evaluate the public health benefit of additional wastewater treatment. Environmental Science & Technology 37, 1882–1891.

Tortajada, C., 2003. Policy failures prevent water quality progress in Mexico. International Journal of Water and Wastewater 18, 25–26.

U.S. Environmental Protection Agency (USEPA), 1986. Bacteriological ambient water quality criteria availability. Federal Register 51 (45), 8012.

U.S. Environmental Protection Agency (USEPA), 2003. Bacterial Water Quality Standards for Recreational Waters (Freshwater and Marine Waters). Status Report. Office of Water, EPA-823-R-03-008, Washington, DC.

Vivoni, E.R., Gutiérrez-Jurado, H.A., Aragón, C.A., Méndez-Barroso, L.A., Rinehart, A.J., Wyckoff, R.L., Rodríguez, J.C., Watts, C.J., Bolten, J.D., Lakshmi, V., Jackson, T.J., 2007. Variation of hydrometeorological conditions along a topographic transect in northwestern Mexico during the North American monsoon. Journal of Climate 20 (9), 1792–1809.

Vivoni, E.R., Rodriguez, J.C., Watts, C.J., 2010. On the spatiotemporal variability of soil moisture and evapotranspiration in a mountainous basin within the North American monsoon region. Water Resources Research 46 (W02509). http://dx.doi.org/10.1029/2009WR008240.

World Health Organization (WHO), 2003. Guidelines for Safe Recreational-water Environments: Coastal and Freshwaters. Geneva, Switzerland, 219 pp.