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ORIGINAL ARTICLE

Potential of Granular Media Filtration for *Giardia* Cysts: Biophysical Modeling of a Microbial Ecology in Water System

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ABSTRACT
Contamination of drinking water by protozoan cysts can cause disease outbreaks and contribute to background rates of disease. Biophysical processes for removal of cysts from water by granular media filtration is used in drinking water treatment. Giardia cysts were detected in 13% of the 35 filtered water effluents. Polypropylene cartridge filtration removes pathogens primarily by size exclusion (1 µm), and is effective in removing cysts larger than the membrane pore size. Analysis of the physical processes in plant indicated that granular media filtration had an average 2.14 log_{10} removal of Giardia. The average plant effluent turbidity for sites which were parasite positive was ≤ 0.5 NTU in 92.7% of the samples. We conclude that potentially infective Giardia are commonly found in raw surface waters, although cyst viability is frequently low. An action level of one to three cysts per 100 liters in treated drinking water is proposed on the basis of the monitoring data from outbreak situations. Appropriate use of the designation for a water body is a policy decision that can be informed by the use of water quality prediction model of ecosystem . Therefore, we believe that the approach is a perfect model system for studies of microbial risk assessment in water systems.
Keywords: *Giardia, Drinking water, Granular media, Water quality model, Turbidity*

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INTRODUCTION

Surface water is used all over the world as a resource for drinking water. Depending on the quality of the source, there are different methods of purifying the raw water. Such methods consist of physical processes like filtration and sedimentation, biological processes like slow sand filters or activated sludge, chemical processes such as coagulation, flocculation, chlorination, etc. The potential of drinking water to transport microbial pathogens to great numbers of people, causing subsequent illness, is well documented in countries at all levels of economics development. *Giardia intestinalis* (syn. G. lamblia or G. duodenalis) is one of the ten major enteric parasites affecting humans worldwide [1]. The life cycle of *Giardia* is direct, and the infective stage of the parasite, the cyst, is encysted when released into the feces and is immediately infectious [2]. *Giardia intestinalis* is a parasite of public health importance as it can be transmitted through several routes, including water (drinking as well as recreational) and fresh food products [3]. These protozoan parasites are the cause of human morbidity and a significant contributor to mortality. About 200 million people in Asia, Africa, and Latin America have symptomatic infections [4]. A small number of cysts is sufficient for infection. Waterborne outbreaks of giardiasis are a major public health problem in many industrialized nations, including the United Kingdom, Sweden, Canada, and the United States [5,6].
The U.S. enviromental protection agency (U.S.EPA) surface water treatment rule (SWTR) stipulates that surface water treatment plants remove or inactivate levels of *Giardia* cysts by 99.9% between the untreated intake water and the point of the first customer in the distribution system. The U.S.EPA has

developed design guidelines to determine that the proposed treatment will provide the inactivation desired. For example, chemically assisted rapid sand filtration with sedimentation is given a credit of 2.5 $\log_{10}$ inactivation for conventional treatment [7,8]. In conventional water treatment plants the coagulation, flocculation process is followed by filtration. *Giardia* cysts are usually than bacteria ranging in size from 6-15 µm, and can be effectively removed using many common water treatment processes particularly those that include a filtration step. Filtration is the passage of fluids through porous media to remove turbidity (suspended solids, such as clays, silt particles, microbial cells) and flocculated particles. This process depends on the filter medium, concentration and type of solids to be filtered out, and the operation of the filter. Depending on raw water quality, granular filtration can be operated in mode conventional treatment is appropriate for most source waters. Granular filters can be constructed as monomedium (e.g. silica sand), dual media (e.g. anthracite coal and sand) and trimedia (e.g. coal, sand and garnet). Removal of pathogens by granular filtration does not rely on physical processes alone. Low turbidity water, proper chemical coagulation of the water before rapid sand filtration is necessary to achieve a good removal of turbidity and protozoan cysts. The removal of particles by granular filtration is considered to involve two steps: transport of particles from suspension to filter medium, followed by attachment of particles to the medium [9]. The transport step depends on the physical and hydrodynamic properties of the system. Transport mechanisms include diffusion, interception and sedimentation. Factors such as size and density of microbes, size and depth of filter medium, and filtration rate affect transport efficiency. Under optimal conditions, a combination of coagulation, flocculation, sedimentation and granular media filtration can result in better removal of protozoan pathogen with chlorine resistant cysts. Detection of *Giardia* cysts in water sample due their small size and frequently low number is difficult. New genetic probes and monoclonal antibodies are being developed for the detection of pathogens and parasites in water and wastewater. Currently, filtration followed by immunofluorescence detection are regarded as the most effective methods for isolation and enumerating waterborne cysts. The immunofluorescence antibody (IFA) method uses fluorescent labeled antibodies that react with *Giardia* cysts, enhancing and simplifying the visualization of the parasite [10]. By using experimental process control in combination with modeling, the treatment effectiveness can be improved, the optimization and control of process parameters resulting in a better and more stable water quality, and reduction of environmental emissions. The purpose of this study examines the factors related to the occurrence of particles in filtered water and evaluates the data within the context of a risk assessment model.

MATERIALS AND METHODS

Percoll-surcrose, sodium dodecyl sulfate, phosphate-buffered saline (PBS), thiosulfate, fluorescein isothiocyanate (FITC) and Tween 80 used in this study were obtained from Merck Chemical Co. Monoclonal antibody for *Giardia* immunofluorescence detection were obtained from Sigma Chemical Co. Polypropylene filter for surface water samples and window slides were obtained from Delta-Pure Corporation model DW 1-03-9-1 and Cel-Line Associates, Newfield, N.J., respectively. Double distilled water of specific conductance 2–3 µS cm^{-1} was used in all buffers and solutions preparation.

Experimental set-up

Most of the surface water samples were collected from Zayandehrud River in Isfahan. Two thirds of the drinking water in Isfahan (Iran) originate from surface sources and the rest of the water is ensured by ground sources. All of the samples (refers to sites) obtained their raw water from surface waters, and 48 of 70 treated their water by chlorination only, without any filtration. The remaining 22 samples employed some kind of filtration, either in pressurized vessels or direct filtration with coagulant. The sampling equipment was assembled in a metal carrying case, and all components in contact with water were made from either brass, plastic, or stainless steel. Influent water was passed through a pressure reducing valve calibrated to keep back pressure below 19 lb/in^2. Surface water samples were taken by filtering water through an polypropylene filter or a membrane disk (pore size, 1µm) and method 1622 procedures recommended by the U.S.enviromental protection agency [11]. A typical sampling run took place over a 24h periods, with average flow rates of between 1 and 4 liters/min. At least 1,000 liters were filtered, except in some cases of exceptionally high turbidity or high algal loading. All samples were shipped in coolers with blue ice packs by surface transport or by air. Most samples arrived within 48 h of shipping, but some from remote sites did not arrive for 72 h. Samples were picked up immediately whenever they came in and stored at 4°C until analysis (within 2 days on average). Filters were analyzed describ by the method [12], except that filters were entirely hand washed, more of the pellet was stained and examined, and the final pellet (~100 µl) was stained in a 15ml centrifuge tube and dried on a window slide after washing instead of being stained on a membrane [13]. Samples were washed once with phosphate-buffered saline (PBS) between the application of primary antibody and the fluoresce in isothiocyanate (FITC) labelling antibody. After the second incubation, the pellet was washed once with PBS and once with deionized water before spotting on a window slide. Control human source *Giardia* cysts were obtained from the medicine diagnostic laboratory and from experimental infections in during the course of this study. Negative controls were run with each batch. Filters were washed by hand into 1,000 ml of deionized water containing 0.1% sodium dodecyl sulfate and 0.1% Tween 80. All of the wash water was concentrated by centrifugation and clarified by layering over percoll-sucrose, 1.0 M sucrose, or a 50:50 (v/v) mixture of both, and the resulting pellet was split between immunofluorescence analysis (10 to 50%) and infection (50 to 90%). The resulting pellet was split equally between immunofluorescence analysis and infection [14]. Quality assurance tests were performed on sampling units and all materials used in the assay to ensure that they were free of parasite contamination. The recovery efficiency of methodology was evaluated by analyzing a series of control samples made up on the basis of the U.S.EPA protocol for information collection rule laboratory approval. Water treatment systems are generally designed to achieve set reductions of specific organisms such as G*iardia*. The effectiveness of the disinfection is usually communicated using the log removal concept. Log removal is an effective way to quantify the number of pathogens removed during a disinfection process, and thus the risk reduction achieved.

In situ modeling system

Mathematical models are very useful in describing the ecological state of a river system and to predict the change in this state when certain boundary or initial conditions are altered. The models describe the main water quality processes, and typically require the hydrological and constituent inputs (water flows, pollutant loadings). These models include terms for dispersive and advective transport depending on the hydrological and hydrodynamic characteristics of the water body, and terms for the biological, chemical and physical reactions among constituents. The water safety framework is not only applicable to micro-pollutant monitoring of drinking water treatment, it can also be applied to aspects such as turbidity, disinfectant residuals, pressure and particle counts. The modeling package used for the simulation of the water quality is WASP5, developed by the U.S.EPA. In this study the hypothesis is tested on a water quality model developed for the Zayandehrud River. The model EUTRO, a module in the WASP5 simulation package [15] which describes the phytoplankton, nutrient and oxygen dynamics and module DYNHYD calculates the hydrodynamics of a water body, were implemented. The complexity of the model can be easily increased by enabling more state variables, parameters and functions used for the simulation. TOXI, another module in WASP5 for simulating sediment and micro-pollutant transport, is also applied in the framework of this hypothesis using increasing degrees of complexity in describing the sorption kinetics between suspended solids and particles [16,17]. Modeling error decreases with increasing model complexity as the more complex models are able to better simulate reality with more processes included and fewer simplifying assumptions. A mass balance (Eq.1) is used accounting for all material entering and leaving the system by point and non-point loading, advective and dispersive transport and physical, chemical and biological transformations:

$$\frac{\partial C}{\partial t} = -\frac{\partial}{\partial x}(V_x C) - \frac{\partial}{\partial y}(V_y C) - \frac{\partial}{\partial z}(V_z C) + \frac{\partial}{\partial x}\left(D_x \frac{\partial C}{\partial x}\right) + \frac{\partial}{\partial y}\left(D_y \frac{\partial C}{\partial y}\right) + \frac{\partial}{\partial z}\left(D_z \frac{\partial C}{\partial z}\right) + R_B + R_K + R_L \qquad (1)$$

Where C is the substance concentration with $\partial C / \partial t$ representing its change with respect to time t, D_x, D_y and D_z are the longitudinal, lateral and vertical diffusion coefficients, R_B , R_K and R_L are the rates for boundary loading, kinetic transformations and loading from point and nonpoint sources, respectively, and V_x , V_y and V_z are the longitudinal, lateral and vertical advective velocities. The velocities are provided by the DYNHYD hydrodynamic simulations. DYNHYD solves the one dimensional partial differential equation (Eq. 2), which includes the momentum equation for the momentum balance:

$$\frac{\partial V}{\partial t} = -V\frac{\partial V}{\partial x} + A_g + A_f \qquad (2)$$

And the continuity equation (Eq. 3) for the mass balance:

$$\frac{\partial Q}{\partial x} + \frac{1}{W}\frac{\partial H}{\partial t} = 0 \qquad (3)$$

Where A_f frictional acceleration, A_g gravitational acceleration, W flow width, H water surface elevation or head, Q volume discharge, V velocity along the flow longitudinal axis, x distance along the flow longitudinal axis and increasing upstream. Manning's equation (Eq. 4) is the most commonly used equation to analyze open channel flows or the frictional acceleration:

$$A_f = \frac{gN^2}{r^{4/3}}.V.|V| \quad (4)$$

Where N Manning's roughness coefficient, r hydraulic radius (cross sectional area), $|V|$ ensures that friction always opposes the flow direction. The roughness coefficient depends on the characteristics of the stream bottom and is used for calibration. These equations are integrated numerically on a discretized network of the flow course. At each time step the momentum equation is solved using the links giving the required velocities and the continuity equation is solved via the nodes giving the water levels and volumes of each unit of discretization. The discharge Q (Eq. 5) over a weir is based on the Bernoulli equation, which assumes the stream lines of the flow are straight and there are no energy losses:

$$Q = \left(\frac{2}{3}\right)^{\frac{3}{2}}.g^{\frac{1}{2}}.b.h^{\frac{3}{2}} \quad (5)$$

Where b breadth of weir crest, g gravitational acceleration, h height between weir crest and water level upstream of the weir.

Water quality pertains to the oxygen balance in a river and can be simulated using varying degrees of complexity. The complexities vary from simple Streeter-Phelps dissolved oxygen (DO) and biological oxygen demand (BOD) description to more complex nutrient limited phytoplankton growth dynamics. This is the simplest complexity and is based on the Streeter and Phelps [18] approach in which the oxygen consumption is characterized in the water using the total maximum biological oxygen demand ($TBOD_{max}$). Oxygen is also consumed in the sediments, which is described in the model by the sediment oxygen demand (SOD). An important source of oxygen into the water body is reaeration via the water surface from the atmosphere. Other complexity is the separation of the $TBOD_{max}$ into its carbonaceous and nitrogenous components, $CBOD_{max}$ and $NBOD_{max}$, respectively. The former is temperature dependent, hence temperature is an input function in the model. The initial rates of increase gives an indication of the deoxygenation rates of each component. The parameter k and the BOD_{max} variables were fit using [19]:

$$BOD = BOD_{max}\left(1 - e^{(-kt)}\right) \quad (6)$$

Where BOD is the oxygen consumed at time t. Complexity is increased at this level by separating the bulk variable $NBOD_{max}$ into its nitrogen components: total organic nitrogen (ON), ammonium (NH_4^+) and nitrate (NO_3). An additional oxygen source and sink are phytoplankton photosynthesis and respiration, respectively. Phytoplankton is, however, not a simulated variable in this complexity level but an input function in the model. At low oxygen concentrations, the denitrification process can be included in the simulation and both carbonaceous deoxygenation and nitrification are oxygen limited. At this complexity, phytoplankton is a simulated variable which can be nutrient limited. Inorganic nitrogen (NH_4^+ , NO_3) and inorganic phosphorus (IP) are the limiting nutrients described using Monod kinetics. A preference factor for NH_4^+ or NO_3 is utilized. Organic phosphorus (OP) is also included in the dynamics and undergoes mineralization and settling. Phytoplankton growth is also light limited and is adjusted with a temperature coefficient. Phytoplankton loss rate is governed by respiration, death, settling and zooplankton grazing. Dissolved oxygen (DO) and BOD dynamics, ecologic photosynthesis and respiration, and nutrient and light limited ecologic growth are combined in a course. Additional processes with corresponding parameters are required to couple the *DO-BOD* and ecologic nutrient cycles.

The transport of suspended solids (SS) requires additional sink and source terms to describe the movement of particles to and from the bottom sediments. Sorption processes must also be included in the reaction term when the transport of heavy metals is simulated. Sorption is the bonding of dissolved chemicals to the particulate solid material in, suspension or in the sediments. The process is described using a partition coefficient K_D which represents the fraction of dissolved and particulate fractions of the heavy metals in relation to the concentration of suspended solids. Many metals have an affinity to sorb either to the fraction of organic carbon f_{OC} of the particulate matter or to bind with the dissolved organic carbon (DOC) fraction to form colloids. DOC remained fairly constant in the flow direction whereas a large variability in f_{OC} was observed. Hence, the dependence of the partition coefficient K_D on f_{OC} was fitted:

$$K_D = f_{OC}.K_{OC} \quad (7)$$

Where K_{OC} is a constant and represents the organic carbon partition coefficient and is calibrated for each heavy metals separately. Although sorption reactions are fast relative to other reactive terms and are assumed to be in equilibrium in which the transfer rates of metals from the dissolved to the solid phase and vice versa are equal. The partition coefficient K_D is a constant and relates the concentrations of the metals phases and the suspended solids as:

$$K_D = \frac{C_{part}}{C_{dis}.SS} \quad (8)$$

Where C_{dis} and C_{part} are the dissolved and particulate fractions of the heavy metals, respectively, and SS is the concentration of suspended solids. The partition coefficient K_D is allowed to increase or decrease in the flow direction at a particular rate. This is particularly the case when large loads of dissolved particles are emitted into a river causing a large increase in the pollutant concentrations in the river.

RESULTS AND DISCUSSION

Analysis for Giardia cysts in filtered water samples

The results of the microscopic analysis of water samples for *Giardia* cysts are summarized in Table 1. Total of 70 finished drinking water samples were examined for *Giardia*. The highest proportion of positive samples was found in (22%) of raw and (20%) treated drinking water. *Giardia* cysts were detected in 12 samples (15%), with a geometric mean (for positive samples) of 3.5 cysts per 100 liters and a range of 0.5 to 45 cysts per 100 liters. Overall, *Giardia* was found in 25 (31.2%) of the finished water supplies. Water samples from 14 of the 5 municipalities sampled contained *Giardia* cysts at least once. *Giardia* cysts were most frequently detected in samples from Zayandehrud River (Fig. 1), but most of the water samples contained fewer than 2 *Giardia* cysts per 100 liters. *Cryptosporidium* oocystes were found much less frequently. Water samples were found to contain cysts at all times of the year but more frequently in late winter early spring and fall.

Table 1. Detection of drinking water and filter effluents analysis from microscopy for *Giardia*

Sample type	Total no.	*Giardia* positive Samples (%)	Mean raw water densities per 100 liter
Total	70	28.5	3.5
Raw drinking water	45	22	2.7
Treated drinking water	15	20	2.3
Dual media	8	12.5	1.5
Mixed media	12	8.3	0.7

Fig. 1. Location of research field.

The finished water had a turbidity of 0.2 to 0.3 NTU (nephelometric turbidity unit), with a free chlorine residual of 0.3 to 0.4 mg/liter, and contained no coliform bacteria. Observation of 35 *Giardia* cysts in drinking water samples showed that 10% of the cysts had a viable type morphology. A viable type morphology does not imply that an organism can exist or infect animals; rather, a cyst that does not have a viable type morphology, i.e., one that has a distorted or shrunken cytoplasm, is probably dead. Five of the six viable type cysts found in tap water samples were from systems that practiced chloramination. Because chloramines react slowly with *Giardia*, these organisms may not demonstrate the same level of destruction as cysts exposed to free chlorine [12]. Results of direct monitoring are reported, which shows the range and mean value of *Giardia* cyst concentration in the outbreak communities compared with those of a control community, which did not report any outbreak of giardiasis during the study period. Cyst concentrations during the outbreaks were 0.5 to 2 orders of magnitude higher than those found after the outbreaks. Analysis of treatment plant configurations showed that granular media e.g. sand filters were more likely to have effluent samples positive for cysts than dual or mixed media filters. More than 20% of the sand filter effluents and 8.3% of the mixed filter effluents was positive for *Giardia*. However, the raw water parasite densities for these filters were generally higher than for parasite negative treatment plants. These results suggest that high raw water parasite densities may overcome filtration and enter finished water supplies. In this study, $\log_{10}$ 1.97-2.02 was obtained for *Giardia* cysts in the two layer filter and $\log_{10}$ 2.27-2.32 was achieved in the three layer filter. This conclusion is supported by the observation that parasite positive treatment plants had an average 2.14 $\log_{10}$ removal of *Giardia*. The removal of protozoan parasites obtained in this study is close to the results of other reports in the literature [12, 20]. The results show that treatment plants can have high removal efficiencies of parasites and still detect organisms in finished drinking water samples. The fact that filtration is not 100% effective places a significant reliance on disinfection, particularly at locations with high source water counts.

Relationship between turbidity and cyst removal

The surface water treatment rule (SWTR) prescribes that utilities must maintain effluent turbidities for conventional filters of ≤ 0.5 NTU in 92.7% of the samples. We found that the average plant effluent turbidity for sites that were parasite positive was 0.70 NTU (Table 2).

For comparison, the average plant effluent turbidity for sites that were parasite negative was 0.33 NTU. The results show that production of low turbidity water did not ensure that the plant effluent would be cyst free. The vast majority of sites that were positive for *Giardia* would have been able to meet the turbidity regulation of the SWTR. Overall, the removal of turbidity within the treatment process was not a statistically significant ($p > 0.05$) predictor of the removal of *Giardia* (Fig. 2).

An even better relationship between turbidity and parasite removal was observed when the data were plotted for an individual site (Fig. 3 and 4).

Particle counts performed close to the date of parasite sampling showed a similar pattern. The $\log_{10}$ removal of total suspended solids (TSS) in the range of 5 to 15μm had a correlation coefficient 0.82 and 0.95 when compared with removal turbidity of *Giardia*, respectively (Fig. 5).

Most of the utilities examined in this study achieved 2 to 2.5 $\log_{10}$ removal of cysts by clarification and filtration as recommended by the SWTR. It is clear that, in many locations, additional disinfection is needed to treat raw water *Giardia* levels. For raw water particle concentrations from 10^2–20×10^3 per ml, the median removal efficiency was 2.04 $\log_{10}$; whereas, when concentrations increased to 10^5–10^6 per ml, the median removal efficiency increased to 3.02 $\log_{10}$. The greater removal efficiency at higher particle concentrations was due primarily to more efficient clarification. This is to be expected because removal of particles by clarification depends significantly on aggregation efficiency.

Fitting the biophysical model for river water quality

Protection of source water can help to minimize microbial risk associated with the water entering a drinking water treatment plant. Potential control of indirect monitoring include eutrophication components, physicochemical characteristics and hydrological events of the water are also important, but may not be easily controlled. It also indirectly affects the efficiency of removal of microbial contaminants by granular filtration . Accordingly, we are using of model framework considered here predicts the transport and biodegradation contaminants applied as particles will collide more frequently and thus be more likely to aggregate in water dynamic conditions. This model is illustrated in Figure 6.

Each water body is divided into segments or volume elements and these segments or volume elements are considered to be in steady state conditions within each simulation time period. Advection or plug flow is assumed during each time period. At the end of each period mixing occurs within each segment or volume element to obtain the concentrations in the segment or volume element at the beginning of the next time step. The indices $i-1$, i and $i+1$ refer river reach segments. The indices t and $t+1$ refer to two successive time periods, respectively. At the beginning of time period t, each segment is completely mixed. During the time interval Δt of period t, the water quality model predicts the concentrations, assuming plug flow in the direction of flow from segment i toward segment $i+1$. The time interval Δt is such that the flow from any segment i does not pass through any following segment $i+1$. Hence, at the end of each time period each segment has some of the original water that was there at the beginning of the period, and its end of period concentrations of constituents, plus some of the immediately upstream segment's water and its end of period concentrations of constituents. These two volumes of water and their respective constituent concentrations are then mixed to achieve a constant concentration within the entire segment. This is done for all segments in each time step. Included in this plug flow and then mixing process are the inputs to the reach from point and non- point sources of constituents. In Figure 6, a mass of waste enters reach i at a rate of V_i^t. The volume in each reach segment is denoted by Q , and the flows from one segment to the next are denoted by W. The drawing on the left represents a portion of a stream or river divided into well mixed segments. During a period t, waste constituents enter segment i from the immediate upstream reach $i-1$ and from the point waste source. Concentration in segment i before mixing at end of time period t:

$$upstream: [C_{i-1}^t W_{i-1,i}^t + V_i^t]\Delta t e^{(-k\Delta t)} \qquad (9)$$

In this illustration, the mass of each of these wastes is assumed to decay during each time period, independent of other wastes in the water. Depending on the types of waste, the decay processes that take place may be more complex than those assumed in this illustration. At the end of each time period these altered wastes are mixed together to create an average concentration for the entire reach segment. Concentration in segment i after mixing at end of time period t:

$$C_i^{t+1} = \left\{(C_{i-1}^t W_{i-1}^t + V_i^t)\,\Delta t + C_i^t(Q_i^t - W_{i,i+1}^t \Delta t)\right\} e^{(-k\Delta t)/Q_i^{t+1}} \quad (10)$$

This illustration applies for each reach segment i and for each time period t. The length, Δx, of each completely mixed segment or volume element depends on the extent of dispersion. Reducing the length of each reach segment or size of each volume element reduces the dispersion within the entire stream or river. Reducing segment lengths, together within creasing flow velocities, also reduces the allowable duration of each time period t. The duration of each simulation time step Δt must be such that flow from any segment or element enters only the adjacent downstream segment or element during that time step. Concentration in segment i before mixing at end of time period t:

$$downstream: C_i^t\left[Q_i^t - W_{i,i-1}^t \Delta t\right] e^{(-k\Delta t)} \quad (11)$$

Since the flows being simulated are not always known, this leads to the selection of very small time steps, especially in water bodies having very little dispersion. While smaller simulation time steps increase the accuracy of the model output, they also increase the computational times. Thus the balance between computational speed and numerical accuracy restricts the model efficiency.

In this model eutrophication is the progressive process of nutrient enrichment of water systems. The increase in nutrients leads to an increase in the productivity of the water system, which may result in an excessive increase in the biomass of algae or other primary producers, such as macrophytes. When it is visible on the surface of the water it is called an algae bloom. Excessive algal biomass could affect the water quality, especially if it causes anaerobic conditions and thus impairs the drinking, recreational and ecological uses. The eutrophication component of the model relates the concentration of nutrients and the algal biomass. Model processes (Fig. 7) of showing the decay of carbonaceous and sediment oxygen demands, reaeration or deaeration of oxygen at the air water interface, ammonification of organic nitrogen in the waste, nitrification (oxidation) of ammonium to nitrate nitrogen and oxidation of organic phosphorus in the sediment or bottom layer to phosphate phosphorus, phytoplankton (biomass) production from nitrate and phosphate consumption, and biomass respiration and death contributing to the organic nitrogen and phosphorus. Parameters include the total deoxygenation rate and the settling velocity of organic matter.

Model coupling with high level architecture(HLA)

The approach is a biophysical model system for studies of microbial ecology in water systems and that this conceptual model can be used for proposing and testing theories based on ecosystem theories, for the development of new and improved quantitative ecosystem models and is beneficial for future design and management of water treatment systems. The aim is to investigate how uncertainty of parameters and input data propagate through of models and models that interact with one another as they simulate in parallel. More control of information transfer between time steps also allows improved analysis of model system dynamics [21]. Programming in the source code is required to include HLA functionality, which eliminates the need for buffer storage of data for model interaction. Object oriented in the open source project object modeling system [22,23] models are refracted to single processes and only called when required for simulating . The DYNHYD to EUTRO/TOXI coupling using HLA was required for this investigation. The MOCA (Monte Carlo analysis) with the normal distributed parameter settings was repeated and extended with a normal distribution setting of the variation in the flow discharges at the boundaries. The results (Fig. 8) of the MOCA for three eutrophication variables, chlorophyll-a (CHL-a), DO and NH_4^+ based on the uncertainty of the hydrodynamic parameters α(weir discharge coefficient) and n(roughness coefficient).

The variation in the output distributions increased with distance along the flow direction of the river course. The best fit between simulations and observations was obtained for DO. Oxygen reaeration is calculated from the hydrodynamic variables (flow velocity and water depth) and the oxygen deficiency from its saturated concentration in water. Since the water was well saturated with oxygen during the time of sampling, little effect will be observed in the DO concentrations due to varying α and n. The variation observed here is due to the variation in chlorophyll-a concentrations, evident in the high oxygen values (> 9.5 mg O_2/L) within the 85% probability bounds. The effect of hydrodynamic parameters on DO becomes more pronounced as the oxygen deficit in the water becomes larger and the phytoplankton growth diminishes. For most variables, there is an increasing trend in the coefficient of variation (CV) when more parameters are implemented in the Monte Carlo analysis. This is due to the increase in the number of varying parameters in the model which leads to an increased spread in the distributions of the simulated results. An exception is nitrate due to the very high NO_3 concentrations (> 5 mg/l) found in the water. Hence, this substance reacts more to the water transport than to biological factors. Little reaction was found in the C-BOD and ON variables when using only the four most identifiable parameters for the MOCA. The parameters influence C-BOD and ON are only sensitive to these variables and are not very identifiable to the system in its entirety. The variability in O_2 and CHL-a is approximately the same for the MOCA using the four identifiable water quality parameters and the MOCA using the two hydrodynamic parameters. Hence, uncertainty in the hydrodynamic parameters can contribute a significant amount of uncertainty in the water quality modeling. This implies that the parameters characterizing the morphology of the river can contribute almost as much variability in the water quality constituents as the biological factors. This shows the significant impact morphological effects may have on the water quality of a river this size. Figure 9 shows a comparison between the simulated hydrodynamic and water quality river of the ranges in the variable distributions resulting from MOCAs in which only the roughness coefficient n was varied (Simulated: $0.020 < n < 0.028$ $s/m^{1/3}$; Controls iteration: $0.023 < n < 0.032$ $s/m^{1/3}$).

The ranges have been normalised to the mean value. The variation of chlorophyll-a is larger for the river than for the simulated hydrodynamic. Hence, morphological effects pertaining to bed roughness has a larger impact on the river. This is largely due to the larger variation in flow velocity although the discharges remained fairly steady in both cases (Simulated: $57.2 < Q < 59.3$ m^3/s ; Controls iteration: $4.8 < Q < 5.0$ m^3/s). Phytoplankton gives preference to ammonium for their nitrogen source causing the largest effect on this substance in comparison to other nutrients. Coupling EUTRO and TOXI together in the HLA environment allows ease of interactive communication between the two models. Chlorophyll-a concentrations (CHl-a) correlate well with particulate organic carbon (POC) content in the water (Figure 10) .

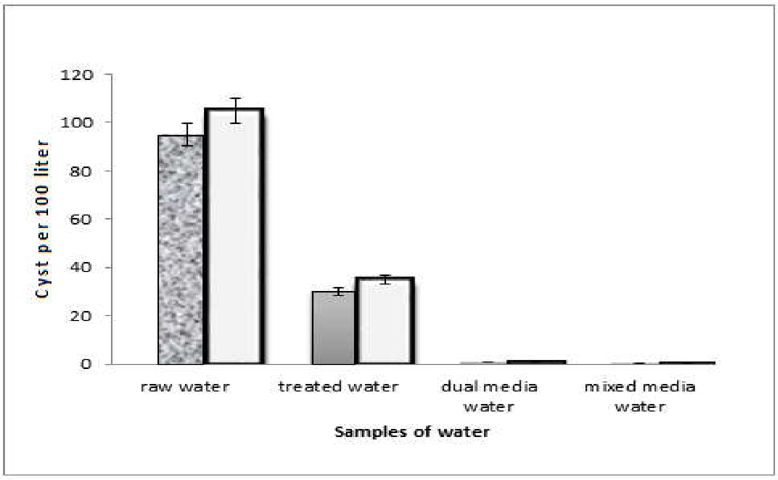

Fig. 2. Mean cyst concentrations in the sample of water tested at Isfahan water treatment plant.

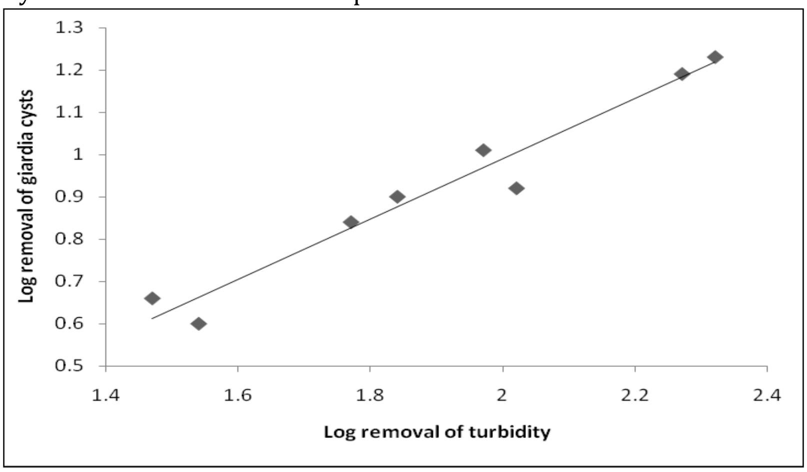

Fig. 3. Relationship between $\log_{10}$ removal of turbidity and $\log_{10}$ removal of *Giardia* cysts. Regression line: $y = 0.7115(x) – 0.4331$; $r = 0.955$, $p< 0.01$

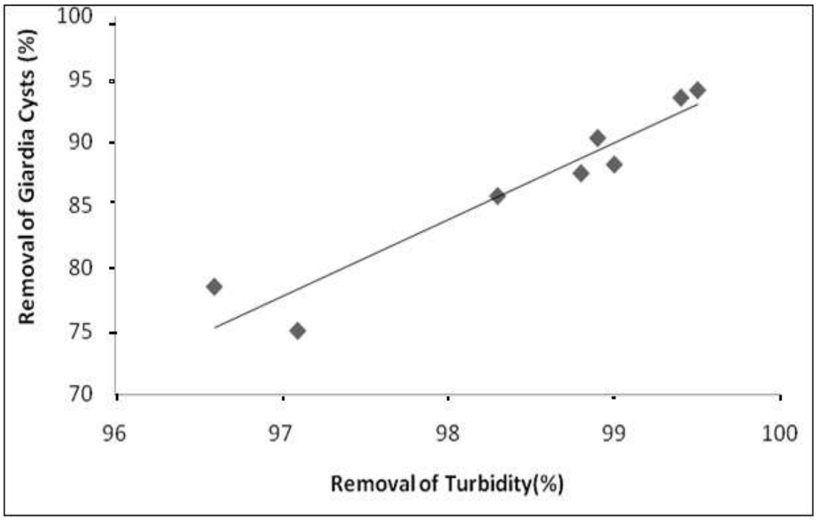

Fig. 4. Relationship between percentage removal of turbidity and percentage removal of *Giardia* cysts. Regression line: y = 6.0848(x) – 512.45; r = 0.9087, p< 0.01.

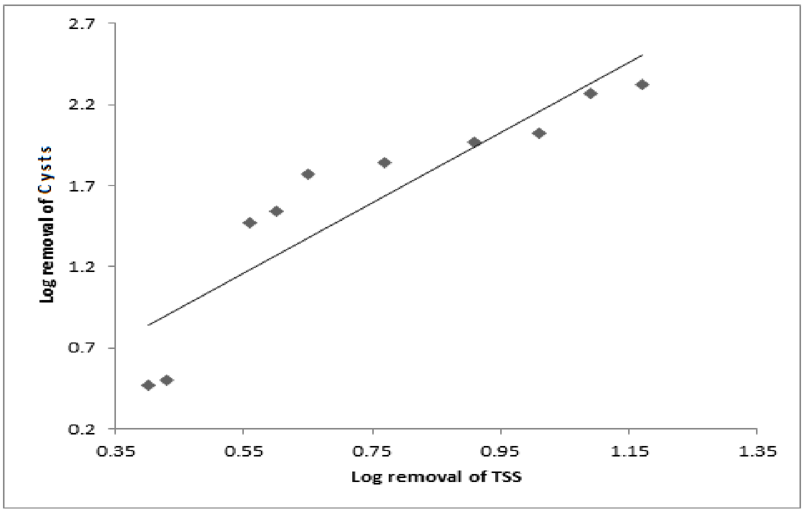

Fig. 5. Relationship between $\log_{10}$ removal of TSS and $\log_{10}$ removal of *Giardia* cysts. Regression line: y = 2.1614(x) – 0.0235; r = 0.8185, p<0.01.

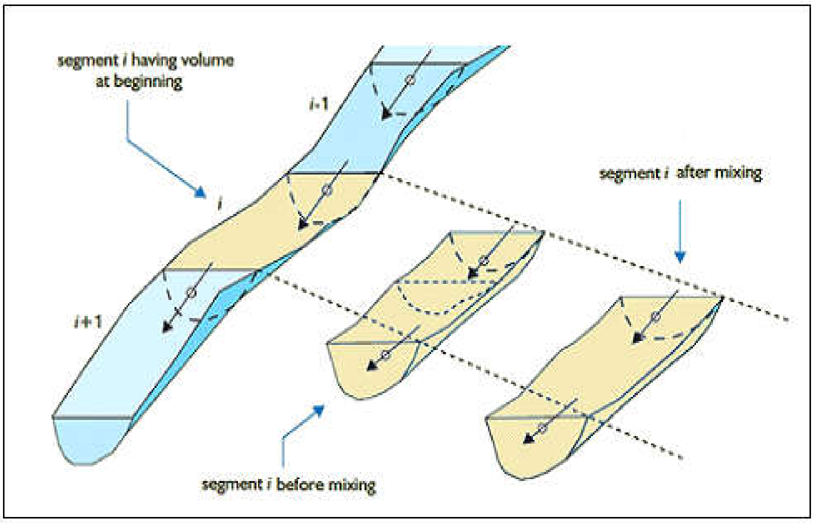

Fig. 6. Water quality modeling approach showing a water system schematized into computational cells.

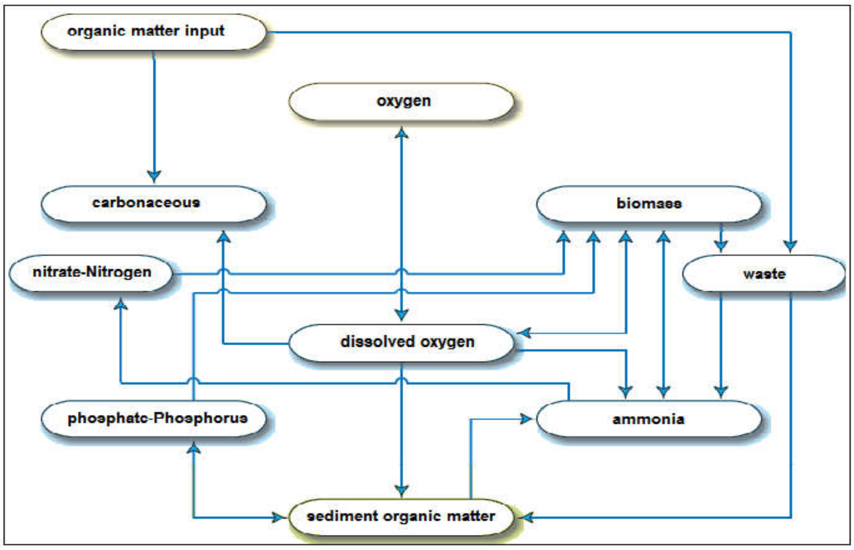

Fig. 7. Model processes of dissolved oxygen, nitrogen and phosphorus cycles, and biomass interactions in a water body.

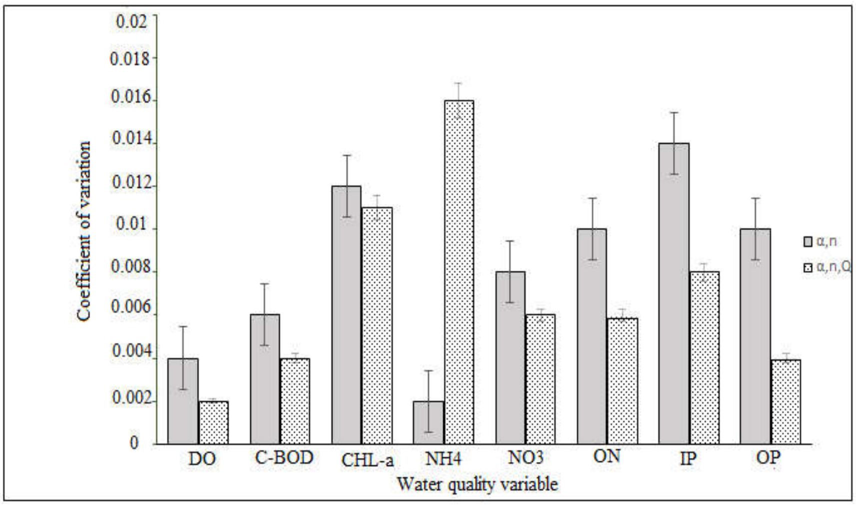

Fig. 8. Coefficient of variation of water quality variables for parameter only (α , n) Monte Carlo analysis and MOCA with parameter and boundary discharge variation (α , n , Q).

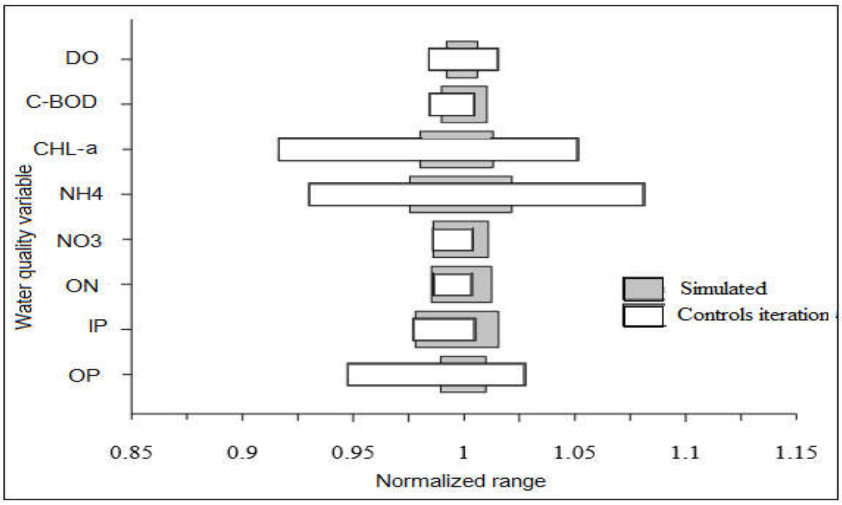

Fig. 9. Normalized ranges of the simulated hydrodynamic and water quality variables for the river.

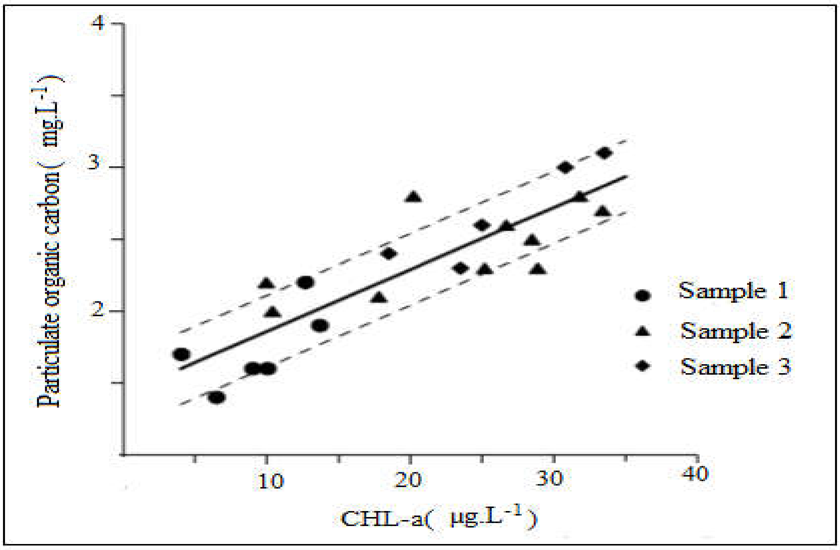

Fig. 10. Correlation between chlorophyll-a (CHl-a) and particulate organic carbon **(**POC). The EUTRO /TOXI coupling using the equation: y= POC=0.034 CHl-a +1.42 ; r^2 = 0.83.

Table 2. Relationship between mean treatment parameters and detection of *Giardia*

Sample type	Turbidity	TSS	pH	Concentration per 100 liter
Raw water	3.28	5.56	8.12	100
Treated water	1.52	2.10	8.07	32.5
Dual media	0.35	0.55	8.01	1
Mixed media	0.23	0.40	7.85	0.5

By dividing POC with the concentration of suspended sediment (SS) simulated in TOXI, the weight fraction of organic carbon in suspended matter (*foc*) is obtained. Therefore, a failure in the removal efficiency of turbidity or particles by granular filtration processes can decrease the inactivation efficiency of disinfection processes. Similarly, clarification affects filter performance. Clarification removes suspended solids, thus reducing the solid loading to the filters and improving filter performance. The results of this work should be identical to the results of the previous [24]. Results gained in steady state computes have special interest. In these cases the DO-BOD and the phytoplankton-nutrient dynamics are only loosely coupled to one another. The latter plays a more significant role in the oxygen balance of the river than the degradation of organic material. Phytoplankton- nutrient dynamics is more important than oxygen-BOD dynamics. TOXI is a model suited for the transport of inorganic substances on the large scale. For small scale applications additional processes must be incorporated which describe the complexity of the transport and transformation dynamic of these substances in the bottom sediments. The roughness coefficient (n) has an effect on the water quality of the river. River channelling has ecological effects: habitats are homogenised and biodiversity decreases. Also retentive capacity of pollutants by the ecosystem is decreased. Channel straightening also removes important refugees for many species since pools of lower current are numerous. Weir system also interrupts ecological continuum simplifying ecosystem structure and reducing ecological diversity. Model predictions and sensitivity analysis indicated that flow velocity (Q) is the major variable determining ecological water quality and suggested that construction of additional dams and water abstraction within the basin would have an adverse effect on water quality. The results show that the integration of ecological models in hydraulic and physicochemical water quality models has an added value for decision support in river management and water policy. The integration of models is a key aspect for the success in environmental decision making. Although the conventional system was effective to remove more than 90% of suspended solid (SS), NH_4^+, chemical oxygen demand (COD), and biochemical oxygen demand (BOD), BOD and COD in effluents still could not meet the discharge standards. The study shows that the aquatic ecosystem played an important role in the wastewater polishment and more than 30% of SS, NH_4^+, COD, and BOD, could be removed further from the effluents of the system, and the treated water could comply with the discharge standards. The main limitation of this approach is the availability of physicochemical, hydraulic and biological data that are collected simultaneously. Therefore, a change in the river monitoring strategy towards collection of data which include simultaneous measurements of these variables is required.

CONCLUSION

The quality of the water in nature also impacts the condition of ecosystems that all living organisms depend on water resources management involves the monitoring and water quality. The substance transport modeling from land surfaces in which water and substances are transported and the mass balance at any position is dependent on the transformations at that position and the flux from that position. It is essential to use new technologies in water treatment plants to reach high quality potable water. Their purpose is to help guide water treatment operators and regulators in applying appropriate treatment technologies. The current study demonstrates that *Giardia* can be frequently isolated from filtered drinking water. Compliance with criteria outlined by the SWTR does not ensure that filtered water will be free of waterborne parasites. Treatment plants with high levels of cysts in raw water supplies were more likely to have the organisms detected in finished drinking water. The conclusions of the present work, provides of the mechanisms and influencing factors of pathogen removal by granular filtration and highlights computational issues, are related with the effects on the removal of particles and environmental microbes.

CONFLICT OF INTERESTS

The authors declare that there is no conflict of interests regarding the publication of this paper.

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REFERENCES

1. Sulaiman, I.M. and Cama, V. (2006) The biology of *Giardia* parasites. Y.R. Ortega (Eds.), Springer Inc., United States, pp 15-32.
2. Huang, D.B. and White, A.C. (2006) An updated review on *Cryptosporidium* and *Giardia*. Gastroenterol. Clin. North. Am., 35(2): 291–314.
3. Nichols, G.L. (2000) Food-borne protozoa. Br. Med. Bull., 56: 209–235.
4. Yason, J.A. and Rivera, W.L. (2007) Genotyping of *Giardia duodenalis* isolates among residents of slum area in Manila, Philippines. Parasitol. Res., 101(3): 681–687.
5. Jungstrom, I.L. and Castor, B. (1992) Immune response to *Giardia lamblia* in a waterborne outbreak of *Giardiasis* in Sweden. J. Med. Microbiol., 36(5): 347–352.
6. Moore, A.C., Herwaldt, B.L., Craun, G.F., Calderon, R.L., Highsmith, A.K. and Juranek, D.D. (1993) Surveillance for waterborne disease outbreaks. United States, 1991–1992, MMWR CDC Surveill. Summ., 42(5) : 1–22.
7. Dawe, P. (2011) Study on pathogen inactivation in drinking water systems in Newfoundland and Labrador (Project No. 103042.00), CBCL Limited, Canada B3J 2R7.
8. U.S. Environmental Protection Agency (2011) Water treatment rules; what do they mean to you? (EPA 816-R-11-009). Office of Water (4606M), U.S. Government Printing Office, Washington, D.C.
9. Lechevallier, M.W. and Au, K.K. (2004) Water treatment and pathogen control; process efficiency in achieving safe drinking water. First Ed., IWA publishing, London.
10. Mara, D. and Horan, N. (2003) Handbook of water and wastewater microbiology. Academic Press, California.
11. U. S. Environmental Protection Agency (1997) *Cryptosporidium* and *Giardia* in water by (filtration/IMS/FA), method 1622. Office of research and development, U.S. Government Printing Office, Washington, D.C.
12. Lechevallier, M.W., Norton, W.D. and Lee, R.G. (1991) Occurrence of *Giardia* and *Cryptosporidium* spp. in surface water supplies. Appl. Environ. Microbiol., 57(9): 2610–2616.
13. Wallis, P.M., Erlandsen, S.L., Isaac-Renton, J.L., Olson, M.E., Robertson, W.J. and Vankeulen, H. (1996) Prevalence of *Giardia* cysts and *Cryptosporidium* oocysts and characterization of *Giardia* spp. Isolated from drinking water in Canada. Appl. Environ. Microbiol., 62(8): 2789–2797
14. U. S. Environmental Protection Agency (1995) ICR protozoan method for detecting *Giardia* cysts and *Cryptosporidium* oocysts in water by a fluorescent antibody procedure (EPA814-B-95-003), Office of groundwater and drinking water. U.S. Government Printing Office, Washington, D.C.
15. Ambrose, R.B., Wool, T.A. and Martin, J.L. (1993) The water quality simulation program, WASP5: model theory, user's manual, and programmer's guide. U.S. Environmental Protection Agency, Athens, GA.
16. Lindenschmidt, K.E. (2006) The effect of complexity on parameter sensitivity and model uncertainty in river water quality modeling. Ecolo. Model., 190(1-2): 72-86.

17. Lindenschmidt, K.E., Wodrich, R. and Hesse, C. (2006) The effects of scaling and model complexity in simulating the transport of inorganic micro-pollutants in a low land river reach. Water Qual. Res. J. Canada, 41(1): 24-36.
18. Chapra, S. (1997) Surface water quality modeling. McGraw Hill.
19. Thomann, R.V. and Mueller, J.A. (1987) Principles of surface water quality modeling and control. Harper & Row, Publishers, New York.
20. Walker-Coleman, L., York, D.W. and Menendez, P. (2002) Protozoan pathogen monitoring results for Florida's reuse systems. Proceedings of symposium XVII, Water reuse association, Orlando, FL. September 8-11.
21. Lindenschmidt, K.E., Hesser, F. and Rode, M. (2005) Integrating water quality models in the high level architecture environment. Adv. Geosci., 4 :51-56.
22. David, O. (1997) A kernel approach for interactive oriented model construction in Java, Concurrency:Practice and Experience, 9(11) : 1319–1326.
23. Maamar, Z. (2003) Design of a simulation environment based on software agents and the high level Architecture. Inform. Software Technol., 45(3):137-148.
24. Bongartz, K. , Steele, T. D. , Baborowski, M. and Lindenschmidt, K.E. (2007) Monitoring, assessment and modelling using water quality data in the Saale River Basin, Germany. Environ. Monito. and Assess., 135: 227-240.