UNIVERSITY OF SOUTHAMPTON

FACULTY OF ENGINEERING, SCIENCE & MATHEMATICS

School of Civil Engineering and the Environment

Effect of climatic factors on the design and operation of continental climate waste stabilisation ponds

by

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ABSTRACT

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Doctor of Philosophy

EFFECT OF CLIMATIC FACTORS ON THE DESIGN AND OPERATION OF CONTINENTAL CLIMATE WASTE STABILISATION PONDS

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The thesis consists of a collection of papers appraising the role and design of waste stabilisation ponds (WSPs) in continental climates. It examines some of the underlying principles necessary to understand how these systems work, using laboratory experiments and mathematical modelling techniques. The state of the art of cold and continental climate ponds is reviewed with descriptions of pond types and operating modes, conventions used in process design, special aspects of construction, and a background into the microbiology and pathogen removal.

Design equations developed by the US EPA and normative standards from the former Soviet Union are compared in a simple example using typical wastewater and performance characteristics. Results were similar except at low temperatures, where the Soviet method can give pond depths outside prescribed limits. The construction and operation of extreme climate WSPs in North America, northern Europe and Russia are also compared together with the respective standards. Reasons why WSP systems have not been widely adopted in Russia and the NIS are considered and a case is made for using WSPs in countries in economic transition.

Experimental data is presented on the determination of the light attenuation coefficient k using a near-parallel halogen light source and an array of photodiodes. The relationship between k and suspended solids is non-linear at concentrations above 50 mg l^{-1} and also varies with depth in the range relevant for WSP operation. The results highlight the need to standardise on methods for the measurement and reporting of k values, especially if these are to be applied in pond modelling.

The effect of sediments on water column characteristics was studied in two pilot-scale ponds and showed nitrogen to be a limiting factor controlling growth after pond desludging. This was only the case for a period of some weeks, however, until fresh sludge had accumulated.

A simple model based on first-order kinetics was developed to predict water availability, biochemical oxygen demand (BOD) and faecal coliform removal for intermittent discharge WSPs with different configurations and operating regimes. The results suggest a significant proportion of inflow could be saved for reuse by modifying pond design to suit climatic conditions. Three case studies of how this could be achieved are presented based on ponds in Kazakhstan. The model is used to consider some alternative design and operating protocols, using average and long-term climate records for cities across central Asia. Annual variability in climate parameters has a considerable effect, in particular on the date at which treated wastewater meets standards for discharge or re-use. The model was checked against a small number of datasets from WSPs subject to ice cover for part of the year, using local climate data. Good agreement was shown in predicting effluent BOD, particularly during spring and summer, and sensitivity analysis revealed some key parameters influencing performance. The model offers a powerful tool for simulating the cyclic behaviour seen in continental climate ponds with sufficient accuracy to permit tailoring of the design to local climatic conditions at specific sites. This allowed the production of contour maps covering a large part of central Asia and western China, showing predicted earliest discharge date, required area of maturation pond, salinity, number of months when discharge is not possible, and the ratio between pond inflow and outflow. The overall conclusion was that there is considerable scope for improvement in design methods for continental climate WSPs, although monitoring of full-scale systems is needed to provide verification and calibration data.

CONTENTS

Appendices

STATEMENT OF CONTRIBUTIONS

This thesis includes material that has been produced in collaboration with other authors. The following describes the share taken by the candidate in the work.

Section 2: 'Cold and continental climate ponds'. Heaven, S. and Banks, C. (Chapter in *Pond Treatment Technology*, ed. A Shilton, IWA Publishing 2005). I was mainly responsible for the sections on Process design, Special aspects of construction, Operation of extreme climate ponds. Modification and Trends in design of extreme climate ponds, and Case studies; and for the final collation and editing of the text. Prof Banks was mainly responsible for the Introduction and the sections on Pond microbiology and pathogen removal. Both authors contributed to the section on Future directions.

Section 3: Heaven, S., A. C. Lock, L. N. Pak, and M. K. Rspaev. (2003). Waste stabilisation ponds in extreme continental climates: a comparison of design methods from the USA, Canada, northern Europe and the former Soviet Union. Water Science and Technology 48(2), 25-33. I was the main author, with information on regulations in the former Soviet Union contributed by Dr Pak and Mr Rspaev and advice on implementation of the modelling provided by Dr Lock.

Section 4: Banks, C. J., S. Heaven, and E. A. Zotova. (2005). Some observations on the effects of accumulated benthic sludge on the behaviour of waste stabilisation ponds. Water Science and Technology 51(12), 217-226. The original experimental design was by Prof Banks and myself. Instrumentation of the ponds was by Prof Banks while the monitoring and analysis was mainly carried out by myself and Ms Zotova. Preliminary data analysis and a first draft of the paper were carried out by Heaven. Final interpretation and extensive revisions to the text were by Prof Banks.

Section 5: Heaven, S., C. J. Banks, and E. A. Zotova. (2005). Light attenuation parameters for waste stabilisation ponds. Water Science and Technology 51(12), 143-152. Preliminary design and construction of the equipment was by Prof Banks, with subsequent modifications suggested by Heaven. Experimental runs were carried out by Zotova and Heaven. Data analysis and interpretation were primarily by Heaven, who also wrote the text of the paper.

Section 6: Heaven, S., C. J. Banks, L. N. Pak, and M. K. Rspaev. (2007). Wastewater reuse in central Asia: implications for the design of pond systems. Water Science and Technology 55(1-2), 85-93. I was responsible for the modelling work and for writing the paper. Banks was primarily responsible for the case studies and for planning of the experimental work briefly referred to in this paper. Pak and Rspaev contributed information to the case studies and carried out the monitoring of the experimental ponds.

Section 7: Heaven, S., A. M. Salter and D. Clarke. (To be submitted for publication). Use of a simple model to produce guidelines for waste stabilisation pond performance in continental climate conditions. I was responsible for collection and analysis of existing data, choice of scenarios, modification of the spreadsheet form of the model, running model scenarios in both spreadsheet and visual basic versions, interpretation of the results and drafting of the paper. Dr Salter was responsible for further development of the pond model in Visual Basic format, and implementation of the software to obtain evapotranspiration and climate data. Dr Clarke advised on sources of climate data and their use. Both Dr Salter and Dr Clarke advised on interpretation of the results.

Section 8: Heaven, S., A. M. Salter, and D. Clarke. (2007). Influence of annual climate variability on design and operation of waste stabilisation ponds for continental climates. Water Science and Technology 55(11), 37-46. I developed the original spreadsheet model, carried out the modelling work and analysis and interpretation of the results, and wrote the paper. Dr Salter was responsible for implementation of the model in the form of a Visual Basic program, and Dr Clarke assisted with the production and use of climate and evapotranspiration data. Both Dr Salter and Dr Clarke also advised on interpretation of the results.

Section A1: Banks, C.J., Koloskov, G.B., Lock, A.C., Heaven, S. (2003). A computer simulation of the oxygen balance in a cold climate winter storage WSP during the critical spring warm-up period. *Water Science and Technology*, 48(2), 189-196. Prof Banks was responsible for the original concept, for suggestions on model structure and data sources and for drafting the paper. Koloskov carried out the majority of the modelling and programming, with advice from Lock and Heaven on specific aspects of the modelling.

Section A2: Heaven, S., D. Clarke, A. Salter and L.N. Pak (in preparation - accepted for conference). Matching waste stabilisation pond outputs to irrigation needs in a continental climate region. Dr Clarke is responsible for provision of the information on irrigation practice and crop choices etc and for feedback from the CROPWAT programme into pond design. Dr Salter was responsible for further development of the pond model in Visual Basic format. Dr Pak provided information on wastewater treatment and disposal in Kazakhstan. I was responsible for the spreadsheet version of the model, for running scenarios and analysing results, and drafting the paper.

ACKNOWLEDGEMENTS

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ABBREVIATIONS

BOD	Biochemical Oxygen Demand
BOD_5	5-day Biochemical Oxygen Demand
BOD_u	Ultimate Biochemical Oxygen Demand
C_a	Solubility of oxygen in water for SNiP calculations
C_e	Effluent concentration for US EPA plug flow method
C_{ex}	Oxygen concentration to be maintained in effluent for SNiP calculations
C_i	Influent concentration for US EPA plug flow method
COD	Chemical Oxygen Demand
DO	Dissolved oxygen
EPD	Environmental Protection Department
f	First order rate constant for BOD decay at low temperature
FC	Faecal coliform
F_{lag}	Pond surface area for SNiP calculations
FP	Facultative pond
FS	Faecal streptococci
HRT	Hydraulic retention time
i	Temperature coefficient for Arrhenius relationship
i/o	Inflow/outflow ratio for WSP system
I_o	Subsurface irradiance
I_z	Irradiance at depth z
k	Light attenuation coefficient
k	First order decay coefficient
K_{lag}	Volumetric use coefficient for SNiP calculations
k_P	First order reaction rate constant for US EPA plug flow method
L_{en}	Influent BOD_u concentration for SNiP calculations
L_{ex}	Effluent BOD_u concentration for SNiP calculations
L_{fin}	Residual BOD concentration for SNiP calculations
NIS	New Independent States of the former Soviet Union
P1k	Pond 1 decay coefficient for BOD or similar parameter
P2k	Pond 2 decay coefficient for BOD or similar parameter
PAR	Photosynthetically active radiation
r_a	Atmospheric re-aeration coefficient at unit oxygen deficiency for SNiP calculations
SMP	Storage/maturation pond
SNiP	Construction Norms and Regulations (former Soviet Union)
SS	Suspended solids
TC	Total coliform
t_{lag}	Hydraulic retention time for SNiP calculations
US EPA	United States Environmental Protection Agency
VS	Volatile solids
VSS	Volatile suspended solids
WSP	Waste Stabilisation Pond
θ	Temperature coefficient for Arrhenius relationship

Section 1: INTRODUCTION

A waste stabilisation pond (WSP) can be defined as a method of treatment based on impoundment or retention of wastewater, and relying on a mutualistic relationship between algae and bacteria to bring about the aerobic breakdown of organic pollutants. The main source of the oxygen needed for microbially-mediated degradation is algal photosynthesis, while the algae make use of carbon dioxide from bacterial respiration and nutrients released in the degradation process. This thesis is concerned with factors affecting waste stabilisation pond performance and its prediction, particularly in continental climate conditions. Although a large number of pond systems are in operation in such regions, especially across North America, in comparison with the body of work on tropical and temperate waste WSPs relatively little research has been carried out in this field. In a world of increasing population and uncertainties about the future adequacy of water supplies, any attempt to provide for the demands of agriculture and human settlements will require effective use of all water resources. These demands need to be met with water of adequate quality to protect the environment and public health. The potential contribution of wastewater reuse is small in relation to overall consumption, but may be important where other water resources are scarce. While population densities have historically been low in the arid continental regions, areas such as central Asia and western China are now experiencing considerable economic expansion, and with it a growing demand for basic infrastructure such as wastewater treatment. WSP systems offer an effective low-cost solution, but whilst existing design and operation protocols are robust they are also somewhat conservative. The overall aim of the work is to explore some factors affecting pond design and operation, and suggest how these might lead to effective design protocols tailored more closely to local conditions. In particular the research aims to show that:

- Current design protocols are generally based on steady state operation and/or worst case values (e.g. the temperature of the coldest month of the year), but in many cases transient conditions such as spring warm-up are of more relevance and importance.
- There is a continuum of possible operating modes between the single short discharge period common at high latitudes, and the continuous discharge possible in warm climates, and that current WSP design protocols should be adapted to reflect this.

The thesis focuses on regions subject to cold winters, but the approaches and findings may be applicable to temperate climates or systems that are operated in intermittent discharge mode for other reasons.

The thesis takes the form of a collection of book chapters and papers, the majority of which have been published or accepted for publication. Section 2 consists of a general review of the operation and performance of cold and continental climate WSPs, while Section 3 presents a comparison of design methods used in different continental regions and their implications. From these studies, it appeared that little attention has been given to design for and performance in non-steady state conditions, and a model was therefore developed to simulate the spring warm-up phase: this work does not form part of the main thesis, but is presented in Appendix A1. During model development it became clear that there is a lack of information on several factors of key importance for the simulation of pond behaviour: these include the effect of benthic sediments on nutrient availability, and appropriate parameter values for light attenuation. Section 4 reports the results of some experimental work on benthic feedback into the pond system, including the effects of a period of hot weather leading to thermal inversion of the pond. Section 5 looks at measurement of and values for light attenuation coefficients in WSP systems, from the viewpoint of their impact on modelling pond performance. While this work was in progress, it was also decided to assess the potential for modifying standard pond design and operating protocols to promote water reuse, based on a much simpler mass-balance modelling approach. Section 6 presents the results of this work, including three case studies on evaporation pond systems in Kazakhstan. Section 7 describes the further development of this simple model, and its application across a wide geographical region to predict WSP performance and offer a basis for potential design guidelines. In Section 8 this work is extended to look at the effect of annual variability in climate parameters on aspects of pond design and operation, and to explore some design alternatives. Section 9 presents brief conclusions of the research and identifies areas for future work.

The appendix provides some information in support of the main work, again in the form of published and unpublished papers. Section A1 presents the pond model developed to simulate spring warm-up, while Section A2 gives a brief description of the potential for matching WSP outputs to irrigation needs in a continental climate region, and the implications for pond design and operating protocols.

Section 2: Chapter 16 - Cold and continental climate ponds

S.Heaven and C.J.Banks

16.1 INTRODUCTION

16.1.1 History and development

Investigations into the use of waste stabilisation ponds (WSPs) for wastewater treatment in continental climate conditions were carried out at Lyublinsk fields in Moscow from 1913 onwards (Vinberg *et al.*, 1966), but the results were not put into widespread application. In North America, the introduction of treatment ponds began in the early 1940s. Originally these were simple storage lagoons that held wastewater until self-purification made it fit to discharge into the natural environment. It was soon realised, however, that biological purification processes were occurring in these ponds even in the winter months, and that the processes accelerated through the spring and into the summer period. This realisation and the empirical development of designs to improve treatment efficiency led to a dramatic growth in the use of WSPs in the 1950s-60s. By 1990 approximately 80% of municipally-owned treatment systems in Alberta and Saskatchewan (504 in number) were engineer-designed waste stabilisation pond systems (Prince *et al.*, 1994). In Canada and Alaska there are more than 1000 WSP systems for domestic/industrial wastewater treatment, representing at least 50% of the total treatment capacity in this region and in many cases outperforming conventional plant. The majority (70%) of these systems serve populations of less than 1000. They have also been used for large communities such as Winnipeg, Canada, although this is less common partly because of the large land areas required. Use of WSP systems in extreme climates elsewhere is less widespread, but examples exist in all the main low temperature regions (Pearson *et al.*, 1987; Juanico *et al.*, 2000; Heaven *et al.*, 2003)

16.1.2 Pond types and operating mode

The pond types used in extreme climates are essentially the same as in temperate or tropical areas: anaerobic, facultative and storage/maturation ponds. The main difference is in the discharge mode. Most warm climate ponds operate as continuous discharge systems, where treated effluent discharges into a watercourse at a rate dependent on the inflow. In cold and extreme climates intermittent discharge systems are more common, in which the wastewater is retained for long periods and is only released once or twice a year, usually in spring and/or autumn. The long retention time is based on the fact that in winter the degree of treatment and the capacity of the receiving watercourse are sharply reduced, and ice cover may make discharge impossible. A third category of total containment ponds exists, in areas where evaporation is greater than precipitation: these do not discharge to a water body at all. Suitable conditions for these ponds often arise in continental climates with their warm summers and cold winters, especially in arid regions: but evaporation of the water in this case may represent loss of a valuable resource. Containment or evaporation ponds are relatively rare in North America and northern Europe but are common in the former Soviet Union, especially for industrial enterprises.

A typical extreme climate engineered WSP system consists of all three types of pond: depending on the flow and loading, there may be multiple examples of each type, working in series or in parallel. Figure 16.1 shows the sequence of events in a one-year retention WSP system including anaerobic, facultative and storage/maturation ponds. There is a single batch discharge in the autumn, although continuous discharge through the summer is also possible if the effluent quality is satisfactory. The diagram assumes that at the end of the summer period the soluble BOD in the storage/maturation pond is at a very low level (a reasonable assumption considering the extended storage under oxygen-enriched conditions at warm temperatures). When the temperature falls in autumn, algae in both the facultative and storage/maturation ponds begin to settle, leaving a water body that is free of suspended solids (Prince *et al.*, 1995b). At this point the water in the storage/maturation pond is decanted leaving the algal/bacterial sludge and a minimum volume of bottom waters, to maximise storage capacity. In some cases part of the water in the facultative pond is also decanted, but this is not common.

During the winter wastewater continues to flow through the anaerobic cells, where solids settle out, into the facultative pond, where ice cover exists and water temperatures are around 0-4 $^{\circ}$C. The treated water from the facultative pond is diluted and washed out into the storage/maturation pond.

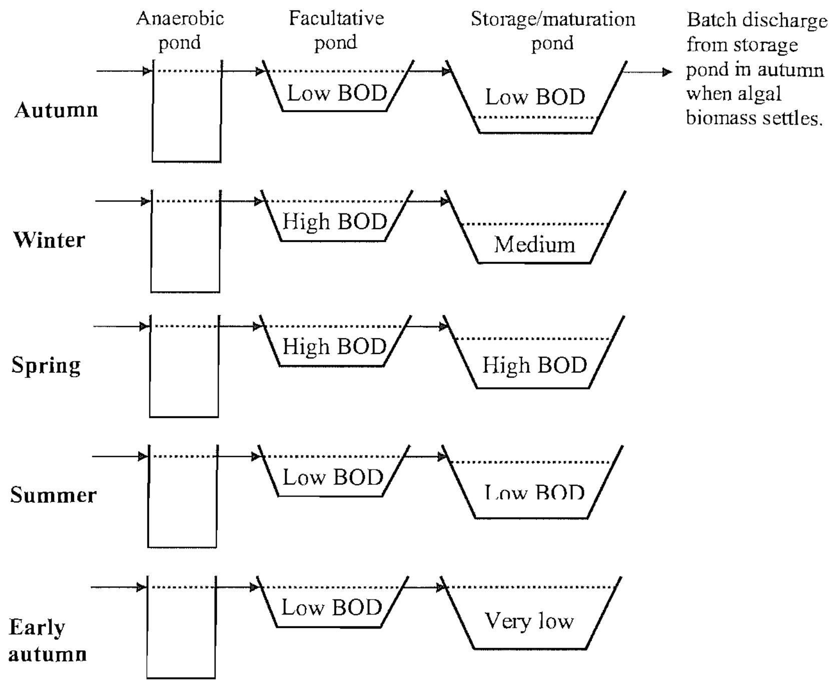


Figure 16.1. Schematic of typical operational pattern for an annually discharged WSP.

By spring the contents of the facultative pond are more or less equal in quality to untreated settled sewage, given the minimal rates of biological decomposition under ice cover. The water in the storage/maturation pond is similar to partially diluted sewage. Hence in spring when the ice melts, the organic load in the system is at its maximum and dissolved oxygen levels are at a minimum. It is at this point that the system can become odorous.

During the spring 'warm up' period the BOD in both the facultative and storage/maturation ponds will start to be reduced as a result of aerobic heterotrophic microbial utilisation of the dissolved organic matter. Oxygen for this will be supplied by the population explosion of algae that grow photosynthetically utilising macronutrients and inorganic carbon sources. During this time the facultative pond will continue to receive settled wastewater whilst the storage/maturation pond will receive a reduced organic load due to its influent having been 'pre-treated' in the facultative pond. In this spring period, both the facultative and storage/maturation ponds will be operating in 'facultative mode', with an excess of soluble organic carbon and nutrients. As spring turns to summer the depth of water in the storage/maturation pond will increase beyond the optimum for a facultative pond. By this time, however, the organic load on it will be very low, and it will start to function as a maturation pond reducing soluble BOD to low levels. Depending upon temperature, initial organic load and other factors both the facultative and storage/maturation ponds should reach a steady state by early summer and continue to operate like conventional WSPs through the summer period. By the end of the summer the soluble BOD in the storage/maturation pond will be very low (e.g. 5 - 15 mg l^{-1}) allowing a high quality autumn discharge, while that in the facultative pond will also be substantially reduced (e.g. to around 30 mg l^{-1}).

Figure 16.2 qualitatively depicts the fate of key parameters in the facultative pond during the 12-month period. These are descriptively divided into three phases: 'accumulation' where BOD load is added but not reduced; 'non steady state' where BOD load is progressively reduced as temperature and microbial numbers (algal and bacterial) increase; and 'steady state' where BOD is consistently low, microbial numbers are in equilibrium, and summer temperatures are more or less constant.

The critical phase is during the 'non steady state' period as the initial organic load is high, the algal oxygen production potential is at its lowest, and the potential for bacterial growth (aerobically or anaerobically) is at its maximum. If the initial load is such that the rate of oxygen depletion is greater than the rate of replenishment then the system will be predominantly anaerobic, leading to lower rates of reaction and potentially to odours. If the initial load and the continuing input is low then the rate of oxygen replenishment may exceed the oxygen demand and the system will become oxygen saturated. In terms of treatment this latter state does no harm, but it does represent a process inefficiency as the system is larger than it needs to be. A well-designed pond system should therefore aim to balance the oxygen input in relation to the oxygen demand exerted by the system: in other words, exactly the same design basis should be adopted for WSPs as for conventional mechanical and forced aeration aerobic biological treatment plant.

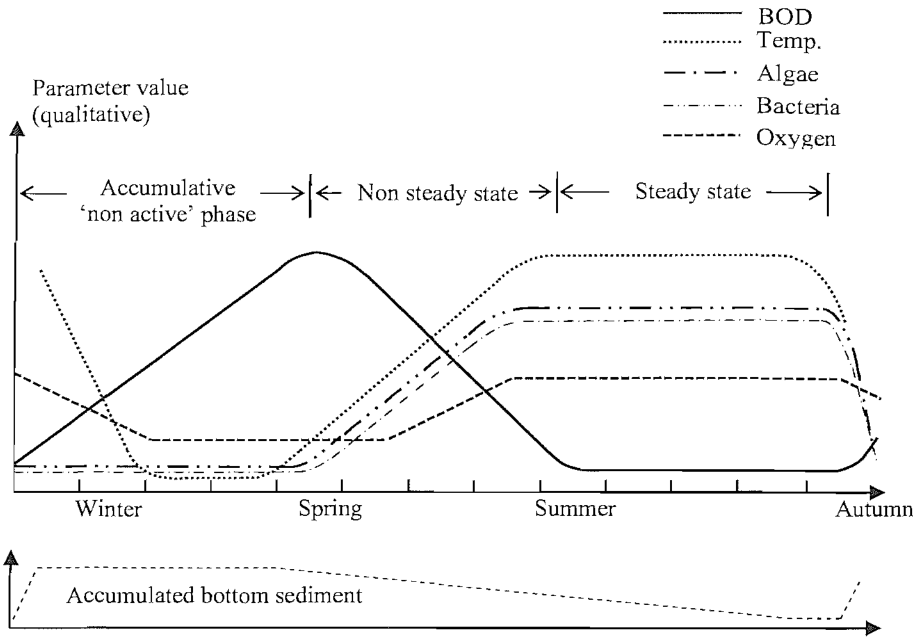


Figure 16.2: Seasonal fluctuations in key parameters in a cold climate facultative pond working in batch mode with a once-yearly discharge

16.2 PROCESS DESIGN

A range of design guides and standards are available for extreme climate WSPs. A key document is the US Environmental Protection Agency's design manual for municipal wastewater stabilization ponds (USEPA, 1983). In Canada, design standards are not consolidated in a single document, but central and provincial authorities have published a wealth of guidance material (Environment-Canada, 1985, 1987; Heinke and Smith, 1988; Heinke *et al.*, 1988; NovaTec, 1996; Saskatchewan, 1996; Newfoundland, 2002). Design and construction in the former Soviet Union is regulated by documents known as the Construction Norms and Regulations (SNiPs). WSP design is covered in SNiP 2.04.03-85 on Water Drainage: External Networks and Structures (SNiP, 1996). Few recent guidelines are available from northern Europe, reflecting the decreasing number of unmodified WSP systems operating in the region (Ødegaard *et al.*, 1987).

16.2.1 Design methods and equations

Various analytical and empirical design methods have been suggested for cold climate facultative ponds. The US EPA manual gives sample calculations at low temperatures based on five design methods: Areal Loading Rate, Gloyna, Marais-Shaw, Plug Flow, and Wehner-Wilhelm (USEPA, 1983). Additional methods have been proposed by Thirumurthi and others (Thirumurthi, 1974; Smith and Finch, 1983; Environment-Canada, 1987). Many of these are not specifically applicable to intermittent discharge ponds or low temperatures, however. Smith and Finch (1983) discuss design and modelling approaches, and conclude that no one method is adequate and simpler methods based on retention time are likely to perform as well as others. Middlebrooks (1987) compared various rational and empirical methods with actual performance data including data from cold or continental climate areas; and concluded that surface loading rates produced the best results, with plug flow the best of the rational methods. Arceivala (1998) offers a rational method of calculating algal oxygen production per hectare (and hence permissible BOD loading) based on the sky clearance factor and visible radiation at different latitudes, with values up to 50^{o} N. A number of attempts have been made to model processes within the pond system, as a basis for rational design (Moreno-Grau *et al.*, 1996; Kayombo *et al.*, 2000; Giraldo and Garzon, 2002; Banks *et al.*, 2003).

In practice, however, the majority of currently-used design methods are based on plug flow with a first order reaction rate, or on a combination of surface loading rate with hydraulic retention time (HRT) or depth. The US EPA design methods for areal loading and plug flow are summarised in Table 16.1. Suggested values for loading rates, pond depths and retention times are available from a wide range of sources. There seems to be a consensus that loading rates of 11-22 kg BOD ha^{-1} day^{-1} are suitable for intermittent discharge ponds in cold regions, and that loading on the first pond should not exceed 40 kg BOD ha^{-1} day^{-1} to prevent odours (Dawson and Grainge, 1969; USEPA, 1983; NovaTec, 1996). Recommended depths vary depending on severity of the climate: typical working depths of 1-1.5 m for facultative and maturation ponds in temperate climates may be increased to 2-3 m allow for ice cover and sludge accumulation (see section 16.3 below).

The plug flow reaction rate constant varies depending on the BOD loading and the temperature. Values and adjustments widely used in North America are shown in Table 16.1. Reaction rate constants and adjustments used in the former Soviet Union differ from those in the west, particularly at low temperatures (Alferova *et al.*, 1973; Samokhin, 1981). SNiP 2.04.03-85 (SNiP , 1996) requires design to be based on plug flow, and states that WSPs can be used for full or tertiary treatment, but there appear to be inconsistencies: in particular, the very limited depths and BOD loadings specified suggest it is mainly intended to apply to high-rate ponds or to tertiary treatment (Heaven *et al.*, 2003).

Table 16.1. Areal Loading Rate and Plug Flow design methods (based on USEPA, 1983)

Areal Loading Rate method for average winter air temperature < 0 °C

BOD_5 loading on whole system limited to 11-22 kg ha^{-1} day^{-1}.

BOD_5 loading on first pond limited to 40 kg ha^{-1} day^{-1}.

HRT = 120-180 days, depending on period of ice cover and discharge conditions. (In practice once-per-year discharge systems are often recommended).

Plug Flow method

$$\frac{C_e}{C_o} = e^{-k_p t} \quad \text{(equation 16.1)}$$

where

C_e = effluent BOD_5, mg l^{-1} and C_o = influent BOD_5, mg l^{-1}

k_p = plug flow 1st-order reaction rate, day^{-1}

t = HRT, days

BOD_5 loading on first pond limited to 40 kg ha^{-1} day^{-1}.

Temperature adjustment $k_{pT} = k_{p20}(1.09)^{T-20}$ (equation 16.2)

where

T = minimum operating water temperature °C.

k_p varies with BOD loading rate as shown:

BOD_5 kg ha^{-1} day^{-1}	22	45	67	90	112
K_{p20} day^{-1}	0.045	0.071	0.083	0.096	0.129

Design of anaerobic ponds is usually based on hydraulic retention time and depth, as in warm climates, although surface or volumetric loading rates are sometimes quoted (Gray, 1999). Dawson and Grainge (1969) suggested depths of around 3 - 7.5 m for short-retention anaerobic ponds in northern latitudes, to conserve heat and allow for sludge accumulation. More recently, Environment Canada recommended depths of 3-5 m and a minimum HRT of 2-5 days (NovaTec, 1996). Abdrashitova *et al.* (2001) suggest the anaerobic pond should be designed to balance the rates of accumulation and digestion of organic solids, in order to reduce the frequency of de-sludging: in future this may lead to a rational basis for the sizing of anaerobic ponds, based on factors such as the temperature and duration of the warmer months.

Design of the storage/maturation pond for an intermittent discharge system is also based on HRT, determined by climatic conditions (e.g. the period of ice cover) and the required frequency of discharge (see section 16.4.2 below). The most common values are 6 and 12 months, sometimes with a safety margin of 2-3 months to allow for conditions such as volume of flow, BOD concentration, or break-up of ice cover in the receiving watercourse. The maximum recommended working depth of storage/maturation ponds is around 2.5 m (Heinke *et al.*, 1991; Prince *et al.*, 1994) This is greater than for facultative ponds, since storage rather than treatment is the main purpose (Prince *et al.*, 1995b), and since for much of the year this type of pond contains treated water with a low BOD.

16.3 SPECIAL ASPECTS OF CONSTRUCTION

16.3.2 Configuration and orientation

Prince *et al.* (1994, 1995a, b) carried out a major review of the effect of system configuration on performance of cold climate ponds, based on data from 190 WSPs in Alberta, Canada. It was concluded that the most robust system consisted of four anaerobic ponds, one facultative pond and one storage/maturation pond with 12 months' storage capacity. This system is often described as 4S, 1T, 1L, using a notation where S refers to sedimentation, T to treatment and L to storage by lagooning, and is widely considered as a standard design in North America. Heinke *et al.* (1991) give design criteria and a flowchart for selection of constructed ponds and systems making use of natural lakes. With respect to the dimensions of individual ponds, there is a sharp division between practice in the west and in the former Soviet Union (Heaven *et al.*, 2003). SNiP 2.04.03-85 requires a length:width ratio of 20:1 or more for naturally aerated ponds. Much smaller ratios are accepted in North America, with 3:1 considered as adequate to promote good plug flow (Environment-Canada, 1985; NovaTec, 1996).

Ponds in extreme climates need additional volume to accommodate ice formation. Design values for ice thickness can be calculated from reference sources (Smith, 1996), or obtained from typical values for the region (Environment-Canada, 1985; SNiP, 1996). Ice depths of over 2 m can occur in ponds in northern Canada; thicknesses in WSPs are usually less than in natural water bodies, however, because of the warm influent (Hanæus, 1991b).

Like WSPs in warm and temperate climates, it is recommended that cold climate ponds are sited downwind of inhabited areas; in cold regions the risk of odour release is highest on thawing, so the direction of the prevailing wind in spring should be considered.

Recommended widths of buffer zones range from 30-300 m for different purposes and in different jurisdictions (NovaTec, 1996; SNiP, 1996).

Winter wind direction and the location of physical barriers such as fencing around the site are also important factors with respect to snow-drifting. A layer of snow improves insulation, reduces ice thickness and may therefore lead to an earlier spring warm-up; but cuts down light, reduces disinfection and prevents photosynthetic aeration, which can occur even under ice and may contribute to odour reduction (Environment-Canada, 1987; Heinke *et al.*, 1991). One investigation found 30% of light penetrating a layer of ice 350 mm thick, whereas only 0.3% passed through 100 mm snow (Environment-Canada, 1985). Different approaches to snow accumulation may be appropriate in different locations. In WSPs located in the far north, for example, the amount of light available in winter is limited and there may be greater benefits from promoting snow cover. In more southerly continental climates, the temperature may briefly rise above freezing in the middle of the day, allowing the surface to thaw and re-freeze and producing very transparent ice. In arid regions such as central Asia, there is relatively little snow, and clear skies and bright sunshine are common in winter, accompanied by periodic strong winds. Under these conditions light penetration can be significant and it may be better to design layout to minimise cover by drifting snow. Much more work is needed, however, to establish the potential influence of layout and operating practice on snow cover and pond performance.

16.3.3 Earthworks and lining

Extensive work has been carried out to establish guidelines for construction in permafrost regions (Environment-Canada, 1985; Smith, 1996). WSPs are particularly sensitive, as storage of large volumes of relatively warm wastewater can cause thawing after construction, leading to catastrophic embankment or liner failures. Damage to liners can also be caused by 'ice rafting', where wind-blown ice rides up the side of the pond, and side slopes of 4:1 are recommended to prevent this (Heinke *et al.*, 1991). Merrill and Stephl (1996) describe a case of liner failure in southern Alaska, to which ice rafting may have contributed. Embankment design may need to provide truck access not only for normal maintenance and de-sludging, but also for discharge of trucked wastes such as septic tank sludges which are often added to WSPs in remote northern areas with scattered populations.

16.3.4 Hydraulics and pipework

The Cold Regions Utilities Monograph (1996) suggests design details for a wide range of pipework and valving systems appropriate to cold climate ponds, including control structures and pond connections. Overflow pipes are particularly problematic, because of their high level: ice may form in or around the pipe and then lift it as the pond level rises. It is therefore good practice to build the pipe level with the berm (NovaTec, 1996). Heinke *et al.* (1991) recommend large submerged inlet pipes to avoid inlet freezing. In cold conditions a submerged inlet may also promote mixing and prevent short-circuiting by formation of a layer of warmer influent above the pond contents.

Ideally pipework and valving arrangements should allow series or parallel operation of ponds, with the option of isolating any section (USEPA, 1983; SNiP, 1996). This is particularly useful in cold climate ponds because of the potential for using empty cells to dewater sludges (see section 16.4.1.below).

In sharply continental climates, the spring warm-up can occur very rapidly. This may impose significant additional hydraulic and/or BOD loads, due to the volume of melt water and the effect of the spring turnover. A common solution in Scandinavia is to install adjustable outlet weirs, so that the water volume remains constant (Hanæus, 1991b). The use of baffles within a pond can help to prevent stratification and reduce short-circuiting, but design of membrane baffle systems is difficult due to ice loading.

Heinke *et al.* (1991) recommend that the storage/maturation pond outlet pipe is 0.5 m above the liner to provide a buffer layer of water that will prevent immediate freezing of the pond bottom after an autumn discharge.

16.4 OPERATION OF EXTREME CLIMATE PONDS

16.4.1 Sludge accumulation and disposal

Rates of sludge accumulation in all pond types are greater in cold climates and under winter conditions, due to the reduced rate of solids digestion at low temperatures. Typically sludge accumulates during the winter period, and may then decrease during the summer (see Figure 16.2). Accumulation rates vary depending on a number of factors, including wastewater temperature, degree of insolation, organic loading, and concentration of inorganic solids in the influent.

Middlebrooks *et al.* (1982) quote accumulation rates for facultative ponds in Canada and Alaska ranging from 0.25 - 0.4 m^3 day^{-1} per 1000 population, with higher values at more northerly sites. Fieldwork on facultative ponds in Utah by Schneiter *et al.* (1983) indicated values of 0.7-0.8 cm $year^{-1}$ and 1.1-1.7 kg total solids m^{-2} $year^{-1}$. For anaerobic ponds, the range of volumetric accumulation rates quoted by Middlebrooks *et al.* (1982) is similar to that for facultative ponds, but there is no clear geographical pattern. A study of the anaerobic ponds in Whitehorse, Yukon, conducted after 7 years of operation found a considerably higher sludge accumulation value of 0.65 m^3 day^{-1} per 1000 population (Allan and Jeffreys, 1987). A summary of results from the 1985 Winnipeg workshop on cold climate WSPs shows a range of 0.17-0.5 m^3 day^{-1} per 1000 population (Environment-Canada, 1987). The above figures give a rather wide range, and for the purposes of estimating sludge accumulation it may be simpler to use a value of 35 g day^{-1} settleable solids per person, and a solids concentration of 5-9% after settlement and consolidation. Assuming a long-term solids degradation rate of 40% (and no wash-out of settled sludge), this gives an accumulation rate of 0.23-0.42 m^3 day^{-1} per 1000 population.

Freezing is well known as an effective means of dewatering sludges. Much of the work carried out has concerned sludges from aerated ponds, chemically-conditioned sludge from precipitation ponds, and septic sludges (e.g. Hanæus, 1991b; Desjardins and Briere, 1996; Hedstrom and Hanæus, 1999). It can be assumed that the properties of ordinary WSP sludges will not differ significantly from these (Schneiter *et al.*, 1984). Extreme climate options for WSP sludges include the use of special sludge freezing beds (Martel, 1993; Hellstrom, 1997). Alternatively, since winter in cold regions is a season of low flows, one option is partially or completely to empty the first pond in autumn and to stockpile the sludge to allow natural freezing and drainage (Ødegaard *et al.*, 1987). A solids content of 50% is reported to have been obtained using this method (Hanæus, 1987; and see section 16.7.2 below).

16.4.2 Pond discharge

Both spring and autumn discharges are common practice in intermittent discharge WSP systems. Spring discharge may have the advantage of coinciding with peak flows and maximum dilution in the receiving watercourse, and of allowing a reduction in storage volume if discharges can be made twice a year. Care must be taken, however, to avoid discharging during a spring turnover when the pollutant load will be high. Autumn discharges, on the other hand, can be made when the algal suspended solids are a minimum and effluent quality is at its peak. In this case, rather than being a problem the discharge can potentially make a useful contribution to maintaining water volume and quality, especially in arid or sharply continental climates where river flows during this period are low. There is a growing consensus, however, that under most conditions an operating regime consisting of a full 12 months of storage and an autumn discharge provides the best results. Extensive surveys carried out by Prince *et al.* (1994, 1995a, b) confirmed that with a single autumn discharge the 4S, 1T, 1L system can out-perform most conventional wastewater treatment plants over a wide range of parameters.

Control of the discharge operation is the most critical factor in performance of extreme climate intermittent-discharge ponds. The US EPA gives detailed guidelines on the requirements for achieving good results (USEPA, 1983). The operator should be provided with discharge instructions and a typical schedule. Pond and receiving water quality must be carefully assessed before and during the discharge. The normal procedure is to isolate the pond to be discharged and to measure a range of parameters including BOD, suspended solids (SS), volatile suspended solids (VSS), pH and dissolved oxygen (DO). Colour, turbidity, and any unusual factors are also noted. Provided that the effluent meets regulatory standards discharge can begin, and can continue as long as the weather is favourable, DO is near or above saturation, and turbidity satisfactory. During the discharge period samples should be taken three times daily in the receiving water near the outlet, and analysed for DO and SS. A typical operating pattern where there are multiple ponds is to draw down the last two ponds to a depth of 0.45-0.60 m. Once one is empty, discharge is interrupted while flow is diverted into the drawn-down pond and the remaining pond is rested before emptying. Similar guidelines have been developed by a number of Canadian provinces (Heinke and Smith, 1988; Saskatchewan, 1996).

16.4.3 Monitoring and maintenance

The majority of monitoring and maintenance tasks for extreme climate WSPs are similar to those for tropical and temperate ponds. Apart from the desludging and discharge operations described above, the main difference is the risk of odour release in spring, at or around the time of ice break-up. Oleszkiewicz and Sparling (1987) looked at factors affecting odour release, and concluded that surface loading rate was the main controllable variable. Experiments with over-winter low-intensity aeration of highly loaded ponds indicated that, while the concentration of sulphides in the aerated ponds was reduced, perceived odour was not significantly affected. A natural odour protection mechanism occurs when, as the pond thaws, a layer of meltwater with very low BOD forms above the ice providing a barrier to release of odours. Possible control mechanisms for investigation therefore include supply of clean oxygen-rich water to form a surface layer, addition of chemicals such as sodium or potassium nitrate, or high-rate aeration, though the latter options are likely to be expensive.

Winter monitoring data should include the thickness of ice and the percentage of the surface area frozen. The Winnipeg workshop on cold climate ponds (Environment-Canada, 1987) recommended monitoring DO levels below the ice: although the results cannot be used immediately for odour control, they add to the database of available information. In general very little information is available on the physical, chemical and microbiological processes occurring in cold climate ponds during the winter period, and adequate monitoring is essential to establish better understanding and control. Other areas requiring monitoring are ground temperatures and embankment faces for signs of icing that may indicate seepage or melting of a permafrost ice lens.

16.5 POND MICROBIOLOGY AND PATHOGEN REMOVAL

16.5.1 Pond biology and microbial activity in cold WSPs

There is little evidence to suggest that microorganisms found in cold climate ponds are significantly different from those in other regions, and the common genera of algae observed during the summer period are those typically encountered in freshwater habitats worldwide. Henry and Prasad (1986) suggested that relative proportions of psychrophilic and mesophilic bacteria may change in the winter and summer months. Detailed studies on the macrobiology of ponds are rare, but any species differences that occur may be expected to reflect the natural distribution of populations of aquatic invertebrates such as fly larvae and worms.

Biologically the most striking feature of an extreme climate pond is the rapid growth and equally rapid death of the population, brought about by transitions in temperature, light intensity and day length. It is clear that formation of ice over the pond surface during winter does not kill all the microorganisms in the pond, and once the ice melts and temperatures increase a rapid increase in numbers of both bacteria and algae follows. This is not only a case of survival at low temperatures: there is also continuing microbial activity within both the ice and the unfrozen water mass. Mackenthun and McNabb (1961) found sub-ice photosynthetic aeration in the Junction City WSP system, with substantial quantities of DO confined to a narrow stratum just below the ice. Concentration increased during daylight hours when the percentage of incident light reached 4-7%. Studies on the ice-covered Siberian Lake Baikal show that convective mixing can occur if the near-ice layer of water warms to the point of maximal density. The effect may be to suspend non-motile phytoplankton in the upper water column, providing cells with enough light for growth during ice-covered periods (Kelley, 1997).

Although there is ample evidence for continuing microbial activity in both ice slush and free water at or near freezing point (e.g. Felip *et al.*, 1995; McKnight *et al.*, 2000; Phillips and Fawley, 2000), rates of reaction are sub-optimal and only a minimal amount of treatment through biological mechanisms occurs during the winter months. During the spring period as water temperatures increase there is typically an explosion of both photoautotrophic and heterotrophic activity within the pond, followed by a lag period which may be as long as 2-3 months before steady state conditions are achieved with cyclic oscillations or succession of secondary feeders (Banks *et al.*, 2002).

16.5.2 Survival of pathogenic micro-organisms

WSPs are quoted as being highly effective in the removal of pathogenic organisms including bacteria, viruses and helminth eggs. The reasons for this are discussed in Chapter 6 and are attributed to a combination of factors such as long retention periods, fluctuations in pH and dissolved oxygen concentration, exposure to UV radiation, and the antagonistic effect produced by certain algal species. In a long retention cold climate WSP these mechanisms are most likely to be effective during the warmer months of the year, when there is no ice cover and microbial population dynamics are similar to those in a conventional WSP. In fact, the total pathogen load discharged is likely to be lower than that from a continuous discharge pond as the likelihood of by-pass through hydraulic short circuiting is eliminated by the mode of operation. In a once-per-year discharge pond the pathogens enter the environment over a relatively short period, however, and with twice-yearly discharge the spring-time pathogen load is likely to be high as some of the destructive mechanisms mentioned above do not function when the pond is ice-covered.

The effect of cold on pathogen survival has been extensively researched due to its importance in food preservation. Extension of the results to WSPs must be treated with caution, because of the very different environment: likewise studies in natural surface and ground waters may not be applicable to wastewater. It is clear, however, that bacterial pathogens such as *Salmonella*, *E.Coli* and *Streptococcus* have adaptation mechanisms to withstand cold shock and sub-optimal temperatures, including freezing, which allow not only survival but also growth (Thieringer *et al.*, 1998; Phadtare *et al.*, 1999; Wouters *et al.*, 1999; Horton *et al.*, 2000). There is also evidence that bacteria such as *Salmonella*, *Shigella* and *E.Coli* that can carry a resistance (R) factor as an extra plasmid in their cells may live longer in wastewater treatment systems (Environment-Canada, 1985).

Viruses can survive for much longer periods and at lower temperatures than bacterial pathogens, and are regularly founds in WSP effluents (Environment-Canada, 1985). Viruses can persist and remain infectious at temperatures near freezing for several months: the die-off of Polio 1 and Hepatitis A viruses was found to be lower in wastewater at 10 °C than at temperatures of 20 and 30 °C, and much lower than that of *E.Coli* (Nasser *et al.*, 1993; Nasser and Oman, 1999). It has been suggested that *E.Coli* is not a reliable indicator of the presence and persistence of enteric viruses, particularly with decreasing temperature. Castillo and Trumper (1991) showed that the die off rates of coliphages in WSPs did not reflect seasonal climate changes as much as *E.Coli*, and also found a threefold difference in die-off rates between summer and winter for both faecal coliforms (FC) and *Salmonella*. Work by Torrella *et al.* (2003) confirmed the long-term low-temperature survival potential of bacterial and viral indicator organisms isolated from wastewater, and the importance of wastewater components in enhancing this survival potential.

Data on pathogen survival over the winter period in cold climate WSPs is limited. Vinberg *et al.* (1966) state that the percentage removal of *E.Coli* from ponds in Minsk was 99.99% in summer, 99.9% in autumn and 55% in winter. Prince *et al.* (1994) looked at faecal coliform data from Alberta for different pond types and discharge periods. The results showed FC numbers in a spring discharge are an order of magnitude greater than for effluents discharged in the autumn. Soniassy and Lemon (1986) looked at total coliform (TC), FC and faecal streptococci (FS) numbers in raw sewage and treated effluent for a WSP at Yellowknife, Canada. Total coliform counts reached a peak in March with numbers around 1×10^6 counts per 100 ml, dropping to 20 counts per 100 ml in August, followed by a very rapid rise between September and December. Similar trends were found with both FC and FS, and the ratio between counts of these two groups remained around 4, indicating similar die-off rates throughout the year. Mackenthun and McNabb (1961) reported Junction City WSP coliform counts of 4 per 100 ml in June to 4.9×10^4 per 100 ml in February.

It is clear that from a public health perspective, in the majority of cases discharge of effluent from a cold climate WSP should not be undertaken in spring. The practice is sometimes justified, however, by the argument that the turbidity of the receiving water at this time of the year is too high, due to the thawing of snow and ice, for it to be considered for drinking water.

16.6 MODIFICATIONS AND TRENDS IN DESIGN OF EXTREME CLIMATE PONDS

As with warm climate WSPs, a number of modifications and additions can be applied to extreme climate ponds. The two most common are artificial aeration and chemical precipitation. Aerated ponds perform satisfactorily in cold climate regions, although with reduced efficiency in the cold season: the main difficulties are with power supplies in remote areas and operation of mechanical plant at low temperatures. Surface aerators are usually unsuitable because of icing (Environment-Canada, 1987). In Scandinavia precipitation ponds using alum, ferric salts or lime have almost entirely replaced conventional WSPs, due to the need to protect inland waters from nutrient enrichment (Ødegaard *et al.*, 1987; Hanæus, 1991b). Precipitation ponds are also suitable for small settlements that experience major fluctuations in population during cold periods, such as winter skiing and recreational areas. Hanæus (1991b) suggests that the future for pond systems may lie in systems combining conventional, aerated and precipitation ponds. Browne and Jenssen (2001) report on a system in Norway which uses vertical and horizontal flow constructed wetlands, flowform cascades, an advanced facultative pond and three facultative ponds, and achieves year-round treatment to a high standard including nutrient removal. Macrophyte ponds have been used for some time in central

Asia and new work is in progress (Yunusov, 1983; Sarsenbaev and Atabaeva, 2000), but the severe winters in the region present difficulties with the survival of common temperate-climate macrophyte species.

The idea of introducing additional or special microorganisms has been raised at various periods. It was particularly popular in the former Soviet Union (VNII-SIS, 1987; Zhirkov *et al.*, 1987), but no large-scale applications seem to have been reported. More recently references have appeared to the possibility of adding special enzymes to the ponds at Whitehorse, Canada (TaigaNet, 2002).

16.7 CASE STUDIES

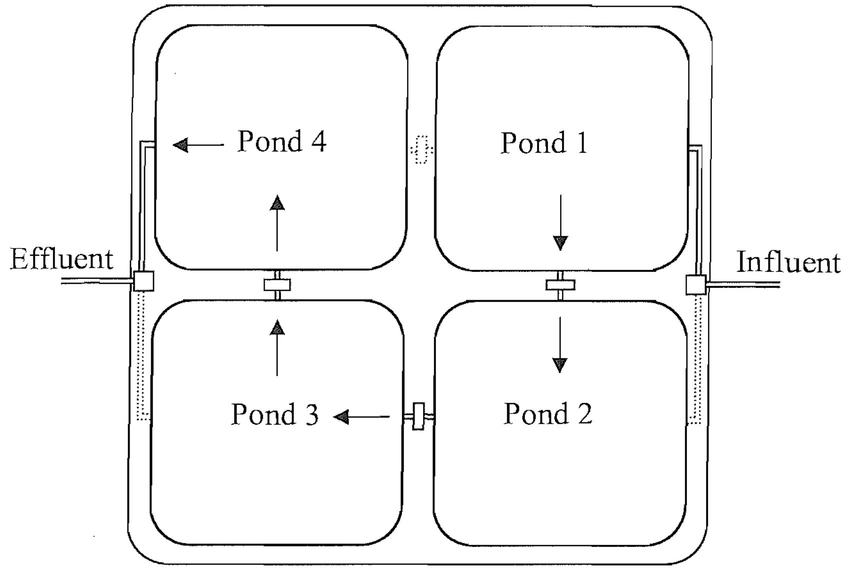
The following section contains two case studies, one of a basic layout successfully replaced by a more modern design, the other of a conventional pond system replaced by chemical precipitation technology.

16.7.1 Whitehorse, Yukon - continuing with WSPs

The city of Whitehorse is located in the Yukon Territory of Canada at latitude 60° 43' N. Mean temperatures range from -18.7 °C in January to 14 °C in July. Per capita wastewater flows are high due to the practice of continuous bleeding from water supply pipes to prevent freezing: this practice is gradually being phased out, but the wastewater remains dilute and cool (Butt and Enns, 2001).

The city's original wastewater treatment plant began operation in 1979. It was designed for a population of 29,700, and at the time of construction served about 10,000 people. The plant consisted of four so-called anaerobic ponds, each with an operating volume of 57,100 m^3 and a depth of 6.1 m intended to promote heat conservation and provide sludge storage capacity. The side slopes were 1:4, with the inlet and outlet 60 m apart and 4 m deep (see Figure 16.3). Design removal values were 60% for BOD_5 and 50% for suspended solids. The ponds were operated in two parallel pairs, giving a retention time of about 20 days. Effluent was discharged continuously into the Yukon River. Discharge conditions based on Yukon Water Board standards allowed an effluent BOD_5 of 45 mg l^{-1}, suspended solids of 60 mg l^{-1} and a faecal coliform (FC) count up to 200,000 per 100 ml. The ponds were monitored over a 2-year period from first filling, to provide data on performance and start-up (Bethell, 1981).

Earth embankments



Pond dimensions 124 x 124 x 6.1 m. Embankment

Figure 16.3: Layout of the original pond system at Whitehorse (based on Alan and Jeffreys, 1987 and Whitley and Thirumurthi, 1992)

By 1985 the ponds were serving a population of 14,800 with a flow of about 12,500 m^3 day^{-1}. Studies in July 1985 revealed short-circuiting and accumulation of sludge (Allan and Jeffreys, 1987). The first two ponds were desludged, and weirs and valving arrangements were modified to change the flow pattern and allow series operation. This led to improvements, but follow-up investigations in 1986 still found hydraulic problems with only 60% utilisation of pond volume. Installation of baffles was recommended, but was not implemented. Meanwhile there were growing concerns about environmental performance and impact, in particular about possible toxicity of the effluent to fish due to ammonia concentrations. In 1989 new standards were introduced requiring a 96-hour LC_{50} bioassay and an FC count of <100,000 per 100 ml.

Research carried out in 1989-90 found an average effluent BOD_5 of 46 mg l^{-1} and suspended solids of 15 mg l^{-1} (Whitley and Thirumurthi, 1992). Removal efficiencies of 52% for BOD5 and 82% for suspended solids compared well with typical figures for this type of plant, in spite of a high system loading of 193 kg BOD_5 ha^{-1} day^{-1}. The ponds were found to be not entirely anaerobic: oxygen was present in pond 4 to the full depth of 6 m from May to July, and pond 3 was partially aerobic in the same period. Measurements of photosynthetically active radiation (PAR) showed penetration to 1-1.5

m in ponds 3 and 4. Chlorophyll concentration in the effluent was zero between December and March and low in October - November and April, but reached a maximum of 190 $\mu g\ l^{-1}$ between May - September. The need to conform to tighter discharge standards and the city's continuing growth led to a series of studies in the early 1990s, aimed at identifying ways to meet projected needs to the year 2012. The review was based on a design population of 33,800 and flows of 19,300 $m^3\ day^{-1}$ (NovaTec, 1992). Options considered included mechanical plant, WSPs, constructed wetlands, infiltration basins, river outfalls and even snowmaking. The recommended solution was a new WSP system consisting of anaerobic, facultative and storage/maturation ponds with once-yearly discharge into the Yukon River.

The new system began operation in 1996, serving a population of 18,464. It consists of two anaerobic ponds 6 m deep, followed by four facultative ponds and a long-detention storage/maturation pond. The facultative ponds range from 125-350 m wide and from 600-1000 m long (see Figure 16.4). The length-to-width ratio of 7:1 was chosen to minimise short-circuiting. The facultative ponds are 2.5 m deep and have a surface area of 42 ha. Wastewater then flows into the storage/maturation pond with a mean depth of about 4 m and a maximum depth of 17 m. The total surface area of the storage/maturation pond is about 177 ha, larger than necessary for 12 months' storage as it occupies a natural depression. Seepage losses are low as the pond lies above an extensive silt deposit. Annual discharge takes place over about 60 days, beginning in August or September and finishing by the end of October. Discharge limits for the new WSP system are specified in the Water Use License issued by the Yukon Territory Water Board (see Table 16.2).

Table 16.2: Discharge limits for the new Whitehorse WSP system

Parameter	Limit
BOD_5, mg l^{-1}	45
Suspended Solids, mg l^{-1}	60
Oils and Grease, mg l^{-1}	5
pH	6-9
Fecal Coliforms MPN 100 ml^{-1}	2000
96-hour static LC_{50} %	100

* based on effluent grab sample

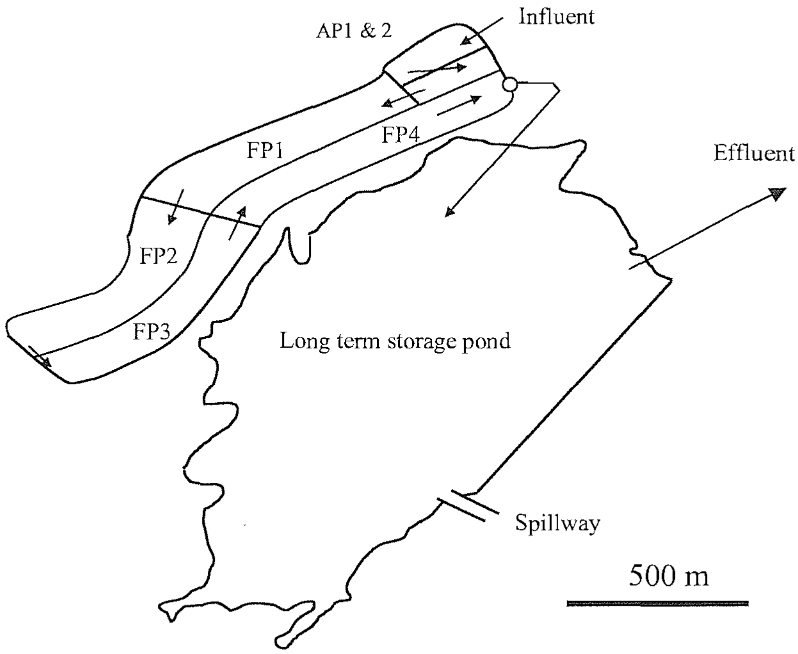


Figure16.4: City of Whitehorse new WSP system (based on Butt and Enns, 2001)

A study of the new pond system in 1999 and 2000 looked at the pollutant removal performance of the WSP and found effluent concentrations of less than 10 mg l^{-1} were consistently achieved for BOD5 and suspended solids, and NH_3-N levels were below 5 mg l^{-1}. The study also found that primary mechanisms for pollutant removal were uptake into algal biomass and volatilisation of NH_3-N stemming from photosynthesis-induced high pH (Butt and Enns, 2001).

The Whitehorse ponds, both old and new, have been the focus of considerable study and extensive public debate. The choice of a new WSP system to replace the old one in preference to other competing technologies is a clear indication of the satisfactory performance and the high degree of local acceptance achieved in this case.

16.7.2 Stugun, Sweden - from WSPs to precipitation ponds

The town of Stugun is situated in the central inland region of Sweden, approximately 450 km northwest of Stockholm. Air temperature ranges from 18 °C in July to -15 °C in January. Stugun has a year-round population of about 1000, which does not vary greatly as it is not in a major tourist region. The main factors affecting wastewater flows are therefore seasonal, particularly snowmelt which occurs in late April.

Wastewater is treated in a series of three ponds that were originally designed as WSPs, but later modified to operate as chemical precipitation ponds. Pond layout and dimensions are shown in Figure 16.5. The total surface area of the system is 0.93 ha and the mean retention time is approximately 30 days (Ødegaard *et al.*, 1987). In the period while they operated as conventional WSPs the ponds received both wastewater, and about 500 m^3 of sludge from individual septic tanks discharged into the first pond in summer (Hanæus, 1987). Influent COD varied from 264 mg l^{-1} in summer to 752 mg l^{-1} in winter, with an annual mean of 451 mg l^{-1}. Pond water temperatures were found to vary from 17 °C in June-August to 1.5 °C in December-March (Hanæus, 1991b).

In the 1960s and 1970s there was increasing concern throughout Scandinavia about the effect of nutrients on inland waters (Ulmgren, 1973; Viitasaari, 1973; Balmer and Vik, 1978). The Stugun wastewater treatment plant discharges into the River Indalsälven, which is used for fishing and recreation. In 1980 a trial was carried out to investigate the possibility of using alum dosing for chemical precipitation, to reduce effluent organic and nutrient concentrations. Results for influent and effluent quality with and without alum dosing are shown in Table 16.3. As a result of the above trials, an alum dosing rate of 150 g m^{-3} was recommended.

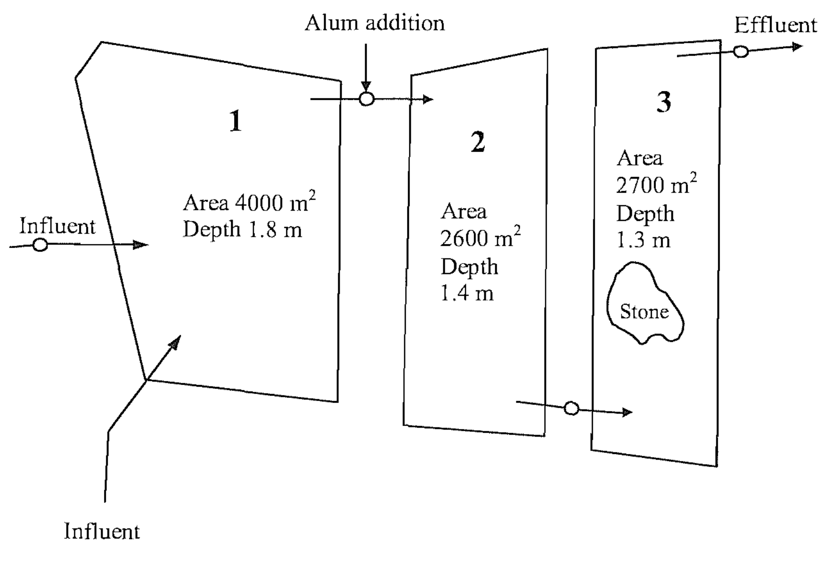


Figure 16.5: Layout of pond system at Stugun (based on Hanaeus, 1991)

Trials were also carried out in summer to investigate the hydraulic performance of the ponds. The wastewater flow in this period averaged 400 m^3 day^{-1}. A preliminary survey showed that approximately 15% of the volume of the first pond was occupied by sludge from biological treatment. Tracer tests were carried with dye released from the inlets and from the corner opposite, and detected at the inlet to pond 2. These indicated a significant degree of short-circuiting: the theoretical detention time was 15 days, but the peak passed in 12-24 hours. A test of the whole system was carried out by adding tracer dye at the inlet and monitoring the outlet: the peak passed in 5-10 days, in comparison with a design detention time of 33 days.

Table 16.3: results of alum dosing trials at Stugun (from Hanæus (1987))

Date	07.01.80 - 27.04.80		28.04.80 - 13.05.80	
Measurement	Mean	Median	Mean	Median
Flow m3/day	260	260	325	295
Alum dose g/m3	0	0	114	100
Influent temperature oC	6.0	6.0	8.5	7.0
Effluent temperature °C	1.5	1.5	6.0	3.0
Influent COD mg/l	866	590	652	280
Effluent COD mg/l	171	170	109	110
COD reduction %	80	71	83	61
Influent total phosphorus mg/l	9.6	9.6	7.1	6.3
Effluent total phosphorus mg/l	6.0	6.5	1.1	1.0
Total phosphorus reduction %	38	32	85	84

Sludge from biological treatment in the first pond was removed by isolating the pond and emptying it (Hanæus, 1987, 1991a). Earth-moving equipment was then used to move the sludge into one quarter of the pond. This area was subsequently separated from the rest of the pond by the construction of an earth dyke. The sludge was left to freeze and dewater during the winter. Through this process a dry solids content of 50% was achieved.

The Stugun ponds now operate as a chemical precipitation system: details of the modifications and subsequent performance can be found in Hanæus (1987). There are approximately 20 plants of this type serving communities of 200-2000 population equivalent in Sweden and Finland, and many more working as one element in a combined system (TemaNord, 1995). Hanæus (1991b) suggests that operation of chemical precipitation ponds could be optimised, for example by having a dose-free period in summer to utilise the biological potential and reduce sludge production. While traditional WSP systems are now rare in Scandinavia, the basic principles may therefore have a continuing role to play.

16.8 FUTURE DIRECTIONS

Despite their widespread use, many uncertainties remain concerning appropriate design procedures for cold and extreme climate ponds. Although empirical methods have developed over the years, reliable relationships still need to be established between temperature, loading rates and pond depths. Because of the variability induced by both geographical and local factors, there is no certainty that generally applicable empirical equations can be obtained. One profitable approach might be to obtain a more fundamental understanding of the kinetics of both algal and bacterial growth at low temperatures and under non steady state conditions, from which kinetic models can be derived to simulate critical phases in the annual cycle within both facultative and storage/maturation ponds. Advances in our understanding of complex systems frequently occur through the interaction of empirical and analytical approaches. A great deal of practical experience has been gathered in recent years, and it may now be time for new developments in the field of modelling and analysis to make their contribution to the process and engineering design of cold climate ponds.

Anaerobic ponds play an important role in the overall pond system, but again there is little information on rates of degradation or stabilisation of WSP sludges at low temperatures. For example it is uncertain whether sludges in a cold-climate pond contribute toward a net loss of organic carbon from the system through methane production, or whether they serve purely to solubilise settleable organics and pass the carbon load through to other parts of the system. It is quite possible that they act as a slow-release carbon sink, helping to balance the load to the aerobic parts of the system. If this is so, more work is needed on optimisation of the design to fulfil this purpose.

Spring time odour is probably one of the biggest problems faced by operators of cold climate pond systems. Several interesting observations suggest that melting of a low-BOD, well-oxygenated layer on top of the ice can produce an effective 'odour buffer'. It is possible that an oxygenated layer created by algal photosynthesis below the ice cover could also act in this way. More research is needed in this area as it could help to resolve the question of whether to design ponds to maximise or minimise snow cover and ice formation. Trapping the odour is one solution but preventing odour formation in the first instance is equally important. Again this requires a better understanding and control of the anaerobic reactions within the accumulating sludge layers of both facultative and anaerobic ponds. Further work is also needed on the possible advantages and disadvantages of supplementing natural aeration with mechanical aeration to meet the demand of the spring time oxygen deficit.

There is clear evidence that over-winter storage of wastewaters can result in a high concentration of pathogens, especially viruses, within the pond system. This raises concern over the potential for reuse of this water for irrigation, especially in arid regions with sharply continental climates where this application appears very promising. Work is needed to determine die-off rates during the non steady state spring-time acclimatisation period, to identify alternative indicator organisms, and to provide guidance on the minimum holding period before continuous summer-time discharge to irrigation systems can be permitted.

In recent years the Scandinavian countries have moved away from biological pond treatment in favour of chemical treatment, but there appears to be scope for both types of system to work in a complementary manner to maintain or increase performance while reducing chemical usage and overall sludge production. Pond systems working in conjunction with wetlands provide further potential for improving final effluent water quality, even in cold climates. The design of hybrid systems, their mode of operation and optimum configuration is a promising area which is receiving growing attention.

WSPs have served communities in extreme climate regions well for more than half a century by providing a low-cost, low-maintenance and reliable means of treating wastewater. The case histories presented above are good examples of the manner in which these systems have developed, not always working adequately at the first attempt! They do work and indeed work extremely well; but to obtain the best results and full benefits, empirical approaches and trial-and-error need to be replaced by a robust design methodology based on an improved fundamental understanding of the system and its biology.

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Section 3: Waste Stabilisation ponds in extreme continental climates: a comparison of design methods from the USA, Canada, northern Europe and the former Soviet Union

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Abstract

The paper presents a brief review of the application of WSPs in extreme climates where ice formation occurs during winter. Design standards and methods are compared and different systems are described. Design equations developed by the US EPA and normative standards from the former Soviet Union are compared in a simple example using typical wastewater and performance characteristics. The results are similar except at low temperatures, where the Soviet method can give pond depths outside the prescribed limits. The paper examines construction and operational aspects of extreme climate WSPs, comparing North American, Northern European and Russian standards. It considers why WSP systems have not been widely adopted in Russia and the NIS, and looks at the advantages these systems may have in countries in economic transition.

Keywords

Waste stabilisation ponds, design methods, former Soviet Union, extreme climates

INTRODUCTION

Waste stabilisation ponds (WSPs) are potentially an appropriate and effective means of wastewater treatment in many parts of the former Soviet Union. The availability of large areas of land is not a problem, while low operating costs and lack of dependence on power supplies, mechanical equipment or imported components are a major advantage in current economic conditions. In the southern republics and Central Asia in particular, WSPs are attractive because of their ability to treat wastewater to standards appropriate for re-use in agriculture, in a region where water resources are scarce. Considerable research was carried out in the Soviet period on the performance of WSPs, and state normative documents and regulations exist. In spite of this, WSPs are not widely used in the former Soviet Union for treatment of either domestic or industrial wastewaters. Possible reasons are the fact that they are often seen as low-technology option suitable only for small settlements or for tertiary wastewater treatment, and a lack of familiarity with recent world practice. This paper compares recommended design and operational practices for WSPs in some of the main cold climate regions: the USA, Canada, northern Europe and the former Soviet Union. Three related areas are considered: process design, structural design and operating procedures. The paper looks at some differences in design standards and approaches, and the implications of these for introduction of WSPs in the New Independent States (NIS).

Design guides and standards: A key document for design of WSPs in North America and elsewhere is the US Environmental Protection Agency's design manual (US EPA, 1983). In Canada, design standards are not consolidated in a single document, but central and provincial authorities have published a wealth of guidance material, and there is naturally a significant degree of exchange with the USA. Northern Europe has few recent guidelines, reflecting the falling number of WSP systems. Design and construction in the former Soviet Union is regulated by documents known as the Construction Norms and Regulations (SNiPs). WSPs are covered in SNiP 2.04.03-85 on Water Drainage: External Networks and Structures.

PROCESS DESIGN

The pond types used in typical WSP systems in extreme continental climates are essentially the same as those in temperate or tropical areas: there are several possible classifications, but a common one is anaerobic, facultative and storage/maturation ponds. The main difference is in the discharge mode. The majority of warm climate ponds operate as continuous discharge systems where treated effluent discharges into a watercourse at a rate dependent on the inflow. In cold and extreme climates intermittent discharge systems are more common, in which the wastewater is retained for long periods and is released once or twice a year, usually in spring and/or autumn. The long retention time is based on the fact that in winter the degree of treatment and the capacity of the receiving watercourse are sharply reduced, and ice cover may make discharge impossible. A third category of total containment ponds exists, usually in regions where evaporation is greater than precipitation: these do not discharge to a water body. Containment ponds are relatively rare in North America but are common in the former Soviet Union, especially for industrial enterprises.

Anaerobic and storage/maturation ponds

Design of anaerobic ponds is generally based on hydraulic retention time (HRT) and depth, as in warm climates. Dawson and Grainge (1969) suggested depths of 3.05 - 7.62 m for short-retention, dominantly anaerobic ponds in northern latitudes, to conserve heat and allow for additional sludge accumulation. Schneiter *et al.* (1983) give values for sludge accumulation in different conditions. More recently, Environment Canada recommended depths of 3-5 m and a minimum HRT of 2-5 days (Environment Canada, 1996). In the former Soviet Union there is no mention of anaerobic ponds in the section of SNiP 2.04.03-85 on WSPs. Vinberg *et al.* (1966) note the need to conserve heat and reduce evaporation and suggest depths of 2.4-3.6 m, but these values refer to work done in the USA. Lukinykh *et al.* (1978) recommend depths of up to 3 m and a HRT of 24 hours. Abdrashitova *et al.* (2001) suggest the anaerobic pond should be designed to balance the rates of accumulation and destruction of organic solids, in order to reduce the frequency of de-sludging: the proper sizing of this unit therefore needs to be resolved and will depend on factors such as the temperature and duration of the warmer months.

Design of the storage/maturation pond for an intermittent discharge system is also based on HRT time, determined by climatic conditions (e.g. the period of ice cover) and the frequency of discharge. The most common values are 12 and 6 months corresponding to once and twice-yearly discharge, sometimes with a safety margin of 2-3 months to allow for conditions in the receiving watercourse. The maximum recommended working depth of storage lagoons is generally around 2.5 m (Heinke *et al.*, 1991; Prince *et al.*, 1994). This is greater than the optimum of 1.5 m for facultative ponds, since treatment is not the main purpose (Prince *et al.*, 1995), and since for much of the year ponds contain treated water with a low biochemical oxygen demand (BOD) (Abdrashitova *et al.*, 2001).

Facultative ponds: design equations and models

The US EPA manual (US EPA, 1983) gives sample calculations for five design methods for facultative ponds: Areal Loading Rate, Gloyna, Marais-Shaw, Plug Flow and Wehner-Wilhelm. The Marais-Shaw Equation was developed for warm climates, and is less suitable at cold temperatures. The remaining methods give reasonable results, but choice of design coefficients can be problematic. The Areal Loading Rate is simplest in this respect, and is often preferred for this reason. Design equations for Areal Loading and Plug Flow methods are shown in Table 1. Pond depth requirements vary in different states. Gloyna's equation uses 1.5 m for areas with significant seasonal temperature variation and 1.5-2 m for severe climates, but 3 m is accepted elsewhere. These values include sludge storage and ice formation, and need to be adjusted for process calculations. Many states also specify a minimum operating depth of 0.6 m. There are no rational or empirical models specifically for the design of intermittent discharge WSPs, but methods for facultative ponds can be adapted by allowing for the extra storage volume required (US EPA, 1975). The US EPA manual comments that conservative loading rates for severe climates may mean retention times are longer than actually needed.

In Canada, Thirumurthi also considered the available analytical and empirical models (Thirumurthi, 1969; Environment Canada, 1987). In addition to those described above, design models potentially suitable for cold temperature application include McGarry and Pescod's empirical method with an allowance for temperature; the Non-Linear Multiple Regression method; Thirumurthi approximate and dispersed plug flow methods; the Dissanayake equation and the Indian empirical method (see Table 1). Again these do not apply directly to intermittent discharge lagoons. Smith and Finch (1983) also discuss design and modelling approaches, but conclude that no one method is adequate and simpler methods based on retention time are likely to perform as well as more complex ones. In practice design in Canada is generally based on a combination of surface loading rate and HRT (Heinke *et al.*, 1991), with each province specifying design standards. Pond depths are similar to those in USA, with a recommended minimum of 1.2 m (Environment Canada, 1996).

In Norway, the Environmental Protection Board suggested a 7-day BOD loading of 3.5 g m^{-2} day^{-1}, and a depth of 1.2-1.8 m (SFT, 1983), while Sweden formerly used a design value of 10 m^2 $person^{-1}$ (Hanæus, 1991). The widespread adoption in Scandinavia of chemical precipitation methods, however, means there have been no recent developments in WSP design theory (TemaNord, 1995).

Table 1 Plug Flow and SNiP 2.04.03-85 design equations

Areal loading rate for average winter air temperature < 0 °C (based on US EPA, 1983)

BOD_5 loading = 11-22 kg ha^{-1} day^{-1} depending on severity of climate. BOD_5 loading on first pond limited to 40 kg ha^{-1} day^{-1}. HRT = 120-180 days, depending on period of ice cover and discharge conditions.

Plug Flow (based on US EPA, 1983)

$$\frac{C_e}{C_o} = e^{-k_p t} \quad (1)$$

C_e = effluent BOD_5, mg l^{-1}; C_o = influent BOD_5, mg l^{-1}; k_p = plug flow 1st-order reaction rate day^{-1}; t = HRT, days. BOD_5 loading on first pond limited to 40 kg ha^{-1} day^{-1}. Temperature adjustment $k_{pT} = k_{p20}(1.09)^{T-20}$ where T = minimum operating water temperature °C. k_p varies with BOD loading rate as shown:

BOD_5 kg ha^{-1} day^{-1}	22	45	67	90	112
K_{p20} day^{-1}	0.045	0.071	0.083	00.096	0.129

SNiP 2.04.03-85

$$t_{lag} = \frac{1}{K_{lag}k}\sum_{1}^{N-1}\ln\frac{L_{en}}{L_{ex}} + \frac{1}{K'_{lag}k'}\ln\frac{L'_{en}-L_{fin}}{L'_{ex}-L_{fin}} \quad (2)$$

t_{lag} = HRT, days; N = no. of pond stages, K_{lag} = volumetric use factor =0.8-0.9 for length: width ratio 20:1, 0.35 for natural reservoirs, interpolated for intermediate values; L_{en} = influent BOD_u, mg l^{-1}; L_{ex} = effluent BOD_u, mg l^{-1}; L_{fin} = residual BOD_u = 2-3 mg l-^{1} in summer, 5 mg l-1in bloom and 1-2 mg l-1 in winter; k = oxygen consumption rate constant = experimentally defined or 0.1 day^{-1} for intermediate ponds and 0.07 day^{-1} for final pond, at 20 °C. Water temperature adjustment $k_T = k_{20}(1.047^{T-20})$ for 5-30 °C and $k_T = k_{20}(1.12(T+1)^{-0.022})^{T-20}$ for 0-5 °C. Symbols marked ' apply to the final pond.

$$F_{lag} = \frac{QC_a(L_{en}-L_{ex})}{K_{lag}(C_a-C_{ex})r_a} \quad (3); \qquad H_{lag} = \frac{K_{lag}(C_a-C_{ex})r_a t_{lag}}{C_a(L_{en}-L_{ex})} \quad \left(= Q\frac{t_{lag}}{F_{lag}}\right) \quad (4)$$

F_{lag} = pond surface area, m^2; Q = wastewater flow, m^3 day^{-1}; C_a = solubility of oxygen in water, mg l^{-1}; C_{ex} = oxygen concentration to be maintained in effluent = 2 mg l^{-1} (Stroyizdat 1981b); r_a = atmospheric aeration at unit oxygen deficiency = 3-4 g m^{-2} day^{-1}; H_{lag} = depth of naturally aerated pond, m.

For WSPs in the former Soviet Union, SNiP 2.04.03-85 specifies that wastewater for full treatment by natural aeration must have an ultimate BOD (BOD_u) of < 200 mg l^{-1}, or < 500 mg l^{-1} for artificially aerated ponds. For tertiary treatment the limits are 25 and 50 mg l^{-1}. If the influent BOD_u is > 500 mg l^{-1} preliminary treatment is required. The design equations, shown in Table 1, are based on BOD removal kinetics but there are a number of differences from the Plug Flow method used in the USA and Canada. The rate constant

k for oxygen consumption at 20 °C is taken as 0.1 day^{-1} for intermediate ponds and 0.07 day^{-1} in the final section. The temperature correction for k differs, particularly below 5 °C where the physical properties of water are different. A factor for pond length-to-width ratio is introduced, and a residual BOD_u is assumed in the final pond. The main difference is in the method of determining the surface area, which is based on atmospheric reaeration. Pond depth is then calculated from the surface area and HRT time. The working depth of the pond is limited to 0.5 m for an initial BOD_u of > 100 mg l^{-1} and 1 m for BOD_u < 100 mg l^{-1}. The same design basis is used for tertiary ponds, with coefficients adjusted accordingly.

The SNiP and US EPA Plug Flow methods were compared, with BOD_u values converted to BOD_5 by a factor of 1.47 (Arceivala, 1999). Figure 1 shows the results for an influent BOD_u [BOD_5] of 200 [136] mg/l, an effluent BOD_u [BOD_5] of 30 [20] mg l^{-1}, and a flow of 1000 m^3 day^{-1} treated in two serial ponds with water temperatures from 0 - 9 °C. SNiP calculations were repeated for K_{lag} = 0.35 and 0.85. For the purpose of comparison surface areas were taken as volume divided by depth, without allowance for side slopes etc. Design effluent BOD_u values for the first SNiP pond in the example shown are based on the US EPA value. The US EPA values for HRT generally lie between the SNiP values, although the upper value diverges sharply below 5 °C (Figure 1a). In the US EPA system, however, the surface area is calculated based on a fixed depth: in the SNiP the depth can theoretically vary, but at low water temperatures the calculated values are much greater than the imposed limits (see Figure 1b). If surface area is recalculated based on required depth it increases significantly, as shown in Figure 1c. Increasing the number of ponds improves depth values for intermediate sections but does not solve the problem. In practice at low temperatures depth limits are a controlling factor, and give more conservative values than the USE PA method. Figure 1d shows comparative k values: it should be remembered that SNiP values are multiplied by K_{lag}.

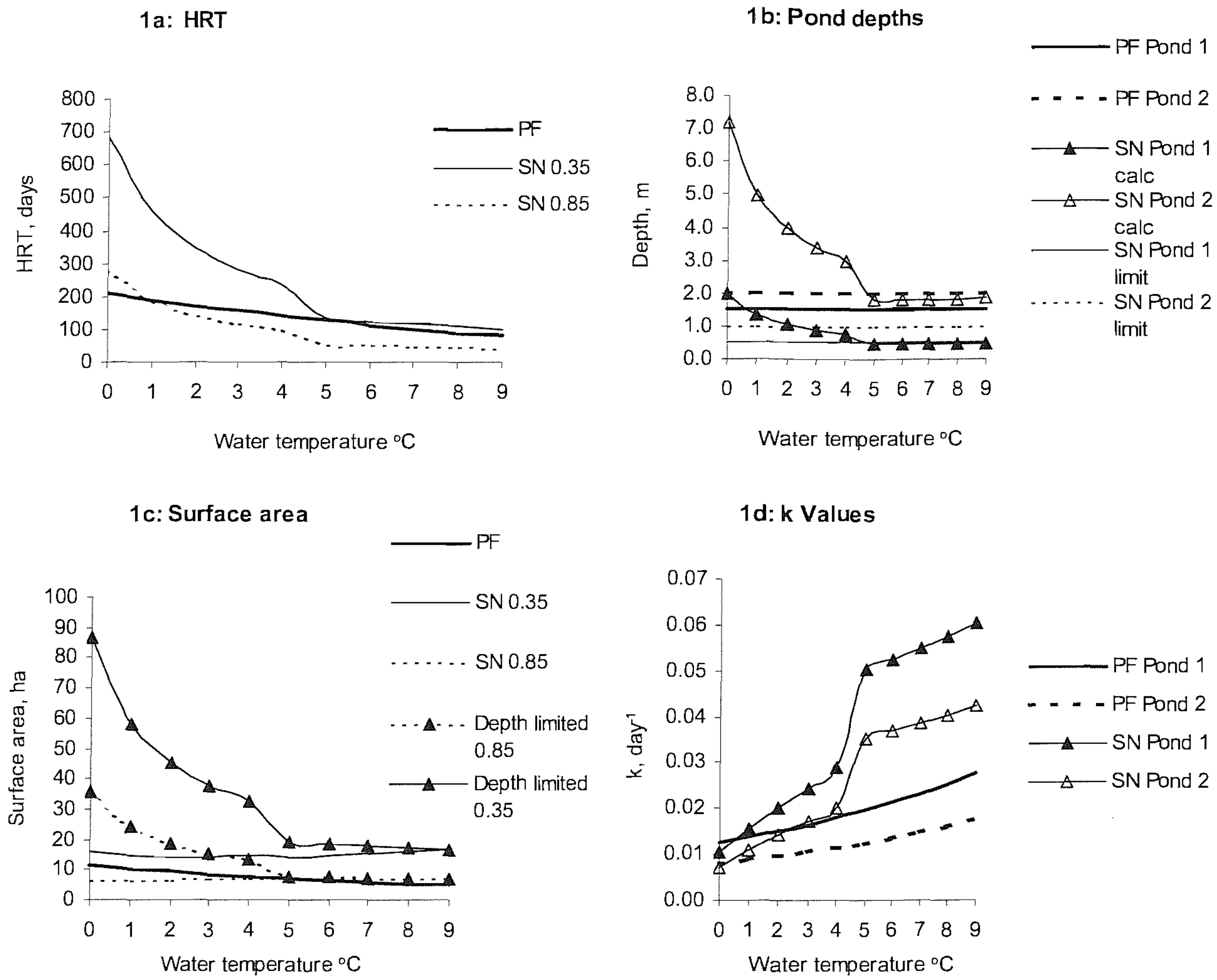

Figure 1 Comparison of designs from US EPA Plug Flow and SNiP 2.04.03-85 methods

Neither the US EPA nor the SNiP plug flow equations are particularly user-friendly for the design of naturally aerated ponds for full treatment of wastewater at low temperatures; but several features of the SNiP, such as the low BOD loadings and restricted depths, indicate that it is mainly intended for design of tertiary treatment and shallow ponds with full-depth light penetration. The SNiP itself states that WSPs can be used for full or tertiary treatment, but standard textbooks and guidelines for interpretation of the SNiP and its predecessors often omit WSPs (Kolobanov *et al.* 1977; VODGEO, 1990). Samokhin (Stroyizdat, 1981b) provides a design nomograph, but Laskov *et al.* (1987) give no guidance on calculation methods, and simply suggest a 60% BOD removal efficiency in each pond. On the other hand several sources state that WSPs are only suitable for warm climates or for use in warm periods of the year (Stroyizdat, 1981a; VNII-SIS, 1987b). Others only mention tertiary treatment, or say WSPs can operate at low temperatures, but recommend shallow ponds that can obtain oxygen from photosynthesis without the risk of stratification (Alferova *et al.*, 1974; Lukinykh *et al.*, 1978). A great deal of work has been done in the former Soviet Union on sophisticated high-rate and contact pond systems (Vinberg *et al.*, 1966; VNII-SIS, 1987a; VNII-SIS, 1987b; Skirdov and Almaneyfi, 1999). Vinberg's work in particular focuses on the potential contribution from photosynthetic oxygen, and is fully aware of all the research carried out in the West on this theme. In spite of this, however, official sources state that photosynthetic aeration should be considered only as a reserve source of oxygen (Stroyizdat, 1981b).

STRUCTURE AND CONFIGURATION

Configuration. SNiP 2.03.04-85 says WSP systems should consist of at least two parallel lines of 3-5 sections each, with the option of isolating any section for cleaning or repair. The length:width ratio for naturally aerated ponds should be 20:1 or more. US EPA (1983) similarly recommends at least three ponds in a treatment line, with pipework to allow parallel or series operation. Large single cells with a central inlet are not preferred, as this may reduce capacity. Much smaller length:width ratios are accepted in North America, however, with 3:1 considered as adequate (Environment Canada, 1985 and 1996). Research on temperate climate ponds (Pearson *et al.*, 1995) also suggests that length:width ratios have little effect on effluent quality. Prince *et al.* (1994, 1995a and b) carried out a major review of the effect of WSP system configuration on performance, based on data from Alberta, Canada, and concluded that the most robust system consisted of four anaerobic ponds, one facultative pond and one storage pond. This system is often described as 4S, 1T, 1L, where S refers to sedimentation, T to treatment and L to lagoon storage, and is now widely adopted for design purposes.

Ice. Ponds in extreme climates need additional volume for ice. According to SNiP 2.04.03-85, if the pond may freeze in winter the depth should be increased by 0.5 m. In northern Sweden ice thickness in lakes can reach 1.2 m but is less in ponds due to the warm influent (Hanæus, 1991). In WSPs in the central mountains, ice reaches 0.6 m deep and melts in a period of 3-4 weeks, imposing a hydraulic and BOD load. A common solution is to install an adjustable outlet weir, so that the water volume remains constant. In northern Canada ice can reach 2 m thick, presenting major problems for mechanical equipment and lining systems. Protection of overflow pipes from ice loading and heave is a special problem due to their high level (Environment Canada, 1987).

Construction and lining. WSPs in the former Soviet Union may be constructed on low-permeability ground, or may be lined. Construction of liners is regulated by SN 551-82 (1982). Engineering use of plastics is still less widespread in the former Soviet Union than elsewhere, and standards for thickness and jointing of materials are less stringent than in the West; but construction details are broadly similar to those shown by the US EPA (1983). Merrill and Stephl (1996) describe a case of liner failure. Ponds in Scandinavia are rarely lined (Hanæus, 1991). To prevent plant growth and erosion, inner slopes should be covered with concrete slabs, asphalt, plastic membrane or stones to 0.5 m above water level and to maximum light penetration depth (usually 1 m) below (Stroyizdat 1981b; VNII-SIS 1987a, Environment Canada 1996). Former Soviet Union guidelines suggest freeboard of 0.5 m for areas up to 0.5 ha, and 0.7 m for more than 0.5 ha (VNII-SIS 1987a). Heinke *et al.* (1991) recommend 1 m freeboard in northern WSPs, however, to allow for frost heave and ice movement. Extensive work has been done in Canada and elsewhere to establish guidelines for construction in permafrost regions (Environment Canada, 1985).

Hydraulics and pipework: The need to position and design inlet and outlet structures to minimise short-circuiting is recognised everywhere. Research carried out in temperate climates by Peña *et al.* (2000) suggests this factor may be more important for anaerobic ponds than has been realised. SNiP 2.04.03-85 requires the transfer pipe between pond sections to be 0.3-0.5 m above the bed, with an upturned end, in order to prevent erosion. Treated water must be discharged 0.15-0.2 times the depth of the pond below the water level or lower surface of ice. Structures that allow draw-off at different levels are recommended, as is heating of pumpwells and pipelines in the north (Stroyizdat, 1981a; VNII-SIS, 1987a). For the USA and Canada, the Cold Regions Utilities Monograph (1996) provides design details for a wide range of cold climate systems including control structures and pipe connections. In Europe and North America, baffling within the pond is suggested to prevent stratification and reduce short-circuiting, but design of baffle systems for cold climates is difficult due to ice loading (Hanæus, 1991).

Location: All countries recommend that WSPs are sited downwind of inhabited areas with respect to the prevailing wind in the warm season. The direction of wastewater flow should be perpendicular to this wind direction to reduce short-circuiting and to enhance mixing, although there is debate about the benefits of stratification. Winter wind direction is also an important factor in snow-drifting: a snow layer improves insulation, reduces ice thickness and accelerates the spring warm-up (Environment Canada 1987; Heinke *et al.*, 1991), but cuts down light, reduces disinfection and prevents photosynthetic aeration (VNII-SIS 1987b; Environment Canada, 1996). Buffer zone widths around WSPs vary from 200 to 300 m in the former Soviet Union depending on the volume treated, and from 30 to 300 m in Canada depending on pond type and property type (SNiP 2.04.03-85, 1996; Environment Canada, 1987 and 1996). In Sweden covered plants are recommended in areas with temperatures of less than -10 ^{o}C for more than 30 days (Ulmgren, 1974).

OPERATION AND MAINTENANCE

In the USA and Canada control of the discharge operation is seen as the most critical factor in performance of intermittent discharge WSP systems (US EPA, 1983). The operator must be provided with discharge guidelines and a recommended schedule. Pond and receiving water quality must be carefully assessed before and during discharge. The normal procedure is to isolate the section to be discharged and to measure a range of parameters including BOD, suspended solids (SS), volatile suspended solids (VSS), pH and dissolved oxygen (DO). Colour, turbidity, and any unusual factors are also noted. The regulatory agency is notified and its approval requested. Once approval is obtained discharge can begin, and can continue as long as the weather is favourable, DO is near or above saturation, and turbidity satisfactory. During discharge samples should be taken three times daily in the receiving water near the outlet, and analysed for DO and SS. A typical operating pattern for multiple ponds is to draw down the last two sections to 0.45-0.60 m. Once one section is empty, discharge is interrupted while the flow is diverted into the drawn-down section and the remaining section is rested before emptying. Both spring and autumn discharge are common. Spring discharge may have the advantage of coinciding with peak flows in the receiving watercourse, and of allowing a reduction in storage volume if two annual discharges are possible. Care must be taken, however, to avoid the spring turnover period. Autumn discharges can be made when algal solids are a minimum and effluent quality at its peak, potentially providing a valuable contribution to river flows in arid or sharply continental climates. There is a growing consensus that 12 months of storage with autumn discharge provides the best results under most operating conditions. Prince *et al.* (1995a and b) confirmed that with a single autumn discharge the 4S, 1T, 1L system can out-perform conventional plants over a wide range of parameters.

In the former Soviet Union the absence of guidelines of this type reflects the lack of familiarity with intermittent discharge operation. On the other hand, it is not uncommon for effluent discharged into total containment ponds to be subject to stringent conditions, even when there is no further release to the environment. These conditions moreover are often applied at the point of discharge into the pond, rather than to the pond contents, making no allowance for treatment processes in the pond.

Extreme climate modifications to WSP operation include the addition of special microorganisms, a popular approach in the former Soviet Union (VNII-SIS, 1987a and b); the use of coagulants and chemical precipitation, now the dominant method in northern Europe (Ødegaard *et al.*, 1987)

According to Soviet norms the staff input required is one qualified wastewater treatment works operator for a WSP system treating up to 400 m^3 day^{-1}, 2 for 400-1400 m^3 day^{-1} and 3 for flows above 1400 m^3 day^{-1} (Stroyizdat, 1981b). In Canada wastewater facilities for more than 1500 persons are also required to have a qualified operator (Environment Canada, 1996).

CONCLUSIONS

Although the theory of extreme climate WSP design is less well established than for other types of wastewater treatment, there is growing consensus on empirically-based parameters. In the former Soviet Union there is a strong theoretical base, but conflicting information on the applications of WSPs and the lack of practical guidance indicate reluctance to translate this into practice. Many of the design criteria are similar, with the exception of those governing pond depth, but operational practices differ and in particular the use of intermittent discharge WSPs is rare in the former Soviet Union. The good results obtained from well-managed WSP systems in North America suggest, however, that this technology is appropriate for similar climatic regions. Exchanges of design models and operational experience between extreme climate regions are likely to lead to improved performance and uptake of WSPs in all areas.

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Section 4: Some observations on the effects of accumulated benthic sludge on the behaviour of waste stabilisation ponds

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Abstract
The effect of accumulated bottom sludge on water column characteristics was studied in two pilot-scale ponds. Parameters measured were ammonia, nitrate, phosphate, COD, suspended solids, dissolved oxygen (DO), temperature and light intensity. The de-sludged pond showed a stronger correlation between DO, light intensity, nutrients and suspended solids with the controlling factor being availability of nitrogen. This was less apparent in the pond with sludge where nutrient levels were higher and more complex mechanisms controlled biomass concentration. Water column characteristics in the two ponds converged rapidly in 7-10 weeks, however, due to accumulation of fresh sludge.

Keywords
Waste stabilisation ponds, sludge, benthic feedback.

INTRODUCTION

One of the prime functions of a facultative waste stabilisation pond (WSP) is to act as a reservoir for the slow accumulation of sediments resulting from death and deposition of the pond flora and fauna and the influent suspended solids. In an ideal design the rate of sediment accumulation would be balanced by that of anaerobic decomposition in the sludge layer, with both soluble organic and inorganic nutrients returning to the water column. This return imposes an additional biochemical oxygen demand on the pond, and affects its performance in terms of nutrient, suspended solids and dissolved oxygen concentrations. Some of the products of anaerobic stabilisation, such as methane and sulphides, may also affect the microbiology of the water column.

A great deal of fundamental research has been carried out to elucidate the mechanisms and quantify the parameters associated with benthic feedback. Much of this has been applied to modelling natural water bodies such as lakes and estuaries. In their review of modelling techniques, Reckhow and Chapra (1999) point out the inappropriateness of simple zero and first-order approaches for systems with a high nutrient loading. Research specifically on WSP systems has been more limited. Bryant and Bauer (1987) reviewed factors affecting feedback of phosphorus and nitrogen, investigated the behaviour of sludge from an aerated stabilisation basin, and produced a model simulating feedback to the water column. Lumbers and Andoh (1987) modelled data from a pond system in the USA and concluded that in the hottest months of the year the benthic feedback was equal in magnitude to the incoming load. Giraldo and Garzon (2002) developed a compartmental model to predict soluble BOD concentrations in facultative ponds. The model was calibrated with data from a pilot-scale system in Colombia and results suggested that solubilisation and return of organic matter from pond sediments to the aerobic layer has a significant influence on effluent BOD. Di Toro et al (1990) carried out extensive work on modelling sediment oxygen demand due to fluxes of methane, nitrogen and ammonia: it is estimated that loss of carbon as methane can account for as much as 30% of influent biochemical oxygen demand.

An experiment was set up to compare water column characteristics from two pilot-scale ponds, one of which had been de-sludged. The work aimed to gather data on the scale of the effect, and its significance in practical terms for WSP operation.

MATERIALS AND METHODS

Small-scale ponds construction and environmental conditions: The experiment was carried out using two ponds each with a volume of 550 litres, a water column depth of 0.6 m and a surface area of 0.9 m^2. These consisted of pre-fabricated semi-translucent polypropylene tanks that were externally insulated with 50 mm of polystyrene foam, preventing any light entering through the tank walls. To protect them from rain and provide a more stable temperature profile they were housed in a greenhouse with a southerly aspect to maximise the incident sunlight in Southampton, UK where the ponds were located. Because of the low light intensities and short daylight hours during certain months each pond also received supplemental illumination from an array of halogen floodlights providing a source wattage rating to each pond of 1600 watts. These gave a surface illumination of the ponds equivalent to 300 W m^{-2}. To prevent excessive localised surface heating of air above the ponds they were ventilated continuously using a 375 mm blade diameter oscillating fan. During the brightest days of summer the combined artificial and natural irradiance could reach 1000 W m^{-2}.

Continuous measurement of light, dissolved oxygen and temperature: Light intensity and temperature were recorded at the surface of each of the ponds and light intensity, temperature and dissolved oxygen concentration were measured at 160, 345 and 545 mm below the surface in each of the ponds. Light intensity was measured using photodiodes (Siemens, type BPW 21) calibrated against a standard photovoltaic cell with an output of 71.4 $\mu V\ W^{-1}\ m^{-2}$. The photodiode output was recorded in mA as this is more stable under temperature change.

Dissolved oxygen (DO) was measured using a galvanic cell type with a zinc anode, silver cathode and teflon membrane (Dryden Aqua, UK). The response of the probes is around 6 $mV\ mg^{-1}\ l^{-1}$ of DO. The accuracy is usually better than +/- 0.2mg/l and they are self-temperature compensating from 0 to 40 deg C. The calibration of the DO probes was checked daily in air by removing the probes from the pond, washing them and suspending them in air for a period of approximately 10-15 minutes. Output was recorded in mV and converted to DO concentration by a single point calibration. Temperature was measured using a type K fine wire exposed junction thermocouple offering a fast response over the temperature range 0-100 °C. Sensor output was internally configured to a direct temperature reading by the data logging equipment software. For temperature, light, and DO readings the probes were continuously sampled using a Data Taker D500 data logger and expansion unit. Under normal operation readings were averaged over a 30 second period and then further averaged to give a stored value for each of the sensors every 10 minutes. During the daily calibration of the DO probes readings were averaged and recorded each minute.

Routine pond feeding and maintenance: The experimental ponds had been running for a period of two years and received a synthetic wastewater that contained (g l^{-1}) semi-skimmed milk, 1.44; freeze-dried blood, 0.057; sterilized bakers' yeast, 0.23; sugar, 0.115; K_2HPO_4, 0.0056. This produced a feed with a COD of approximately 380 mg l^{-1}, BOD 160 mg l^{-1} and suspended solids of 190 mg l^{-1}. Before feeding each day a volume of

25 litres of pond water was siphoned off, in a period of approximately 15 minutes, from a point near the centre of the water body. Each pond was then batch fed 25 litres of the synthetic wastewater over a feeding period of approximately 1 hour. To account for evaporation, the level of the pond was topped up with clean tap water. This method of feeding resulted in a 22-day hydraulic retention time and a surface loading of 45 kg BOD ha^{-1} day^{-1}, while the batch feeding guaranteed a minimum retention period of 23 hours.

Sampling and analysis: During the experimental period samples were taken on alternate days from both ponds. Suspended solids (SS) were measured by filtration through a pre-dried and weighed GFC filter (Whatman, UK). Ammonia, nitrate, phosphate, alkalinity were measured with a Bran & Luebbe Autoanalyser model 3; filtered Chemical Oxygen Demand (COD) was measured by the closed tube digestion method (Standard Methods, 1998). Chlorophyll was determined by filtering a sample through a GFC filter (Whatman, UK) previously dosed with 1 ml of a saturated solution of $MgSO_4$, followed by grinding and treatment with acetone. The resultant colour was measured at 664 and 665 nm using a Cecil Instruments spectrophotometer (3000 series). Absorbance was measured at 664 nm in a colorimeter (Camlab DREL/5). Samples were analysed in duplicate or triplicate. Floating sludge were estimated on a scale of 0-5 where 0 = not present, 1 = isolated small pieces, 2 = < 500 ml, 3 = < 1.5 l, 4 = > 1.5 l and 5 = surface covered. A similar index was used for scum.

Experimental procedure: At the start of the experiment the top water from each of the ponds was pumped out, using a peristaltic pump and taking care not to disturb any of the accumulated bottom sediment. In this way it was possible to remove and mix together the top waters from the two ponds into a temporary holding tank leaving only 45 litres (approximately) of sludge in each of the ponds. The sludge itself was dark green/black in colour with a gelatinous nodular texture and had accumulated to a depth of between 50-75 mm. The sludge was removed from one pond (Pond 2) and left undisturbed in the other (Pond 1). The mixed top water was then returned in equal volumes to each pond, taking care not to disturb the remaining sludge layer in Pond 1. At the beginning of the experiment conditions in both ponds were therefore equal in all respects apart from the presence or absence of a sludge layer. Feeding, monitoring and sampling of the ponds continued as before.

RESULTS AND DISCUSSION

Apart from some short-term transitional differences, in the two years of operation before the reported experiment the water column characteristics of the ponds were similar. Table 1 gives comparative values for some key parameters in the experimental period, the same periods in year 1 and year 2, and for the whole time of operation excluding any periods in which the ponds were treated differently. Figure 1 shows phosphate concentrations in the 16 months preceding the experiment: while one pond sometimes leads, the two clearly respond in similar ways to changing conditions. It was therefore assumed that any differences observed from the start of the experiment onwards could be attributed to the presence or absence of an accumulated benthic sludge layer.

Table 1 Comparative performance of ponds

in mg l^{-1}	SS	COD	NO_3	NH_4	PO_4
Experimental period (July-Nov 2003)					
av P1	106	55	0.10	1.65	1.71
av P2	102	60	0.05	0.13	1.19
R^2	0.34	0.21	0.00	0.27	0.01
Year 1 (July-Nov 2001) - before experiment					
av P1	46	74	0.12	0.35	0.34
av P2	50	55	0.11	0.35	0.32
R^2	0.74	0.61	0.64	0.45	-
Year 2 (July-Nov 2002) - before experiment					
av P1	61	50	0.26	2.63	1.34
av P2	51	53	0.23	2.28	1.33
R^2	0.75	0.12	0.75	0.48	0.60
Whole period (2001 - 2003)					
av P1	67	61	0.19	1.63	1.34
av P2	71	55	0.17	1.61	1.25

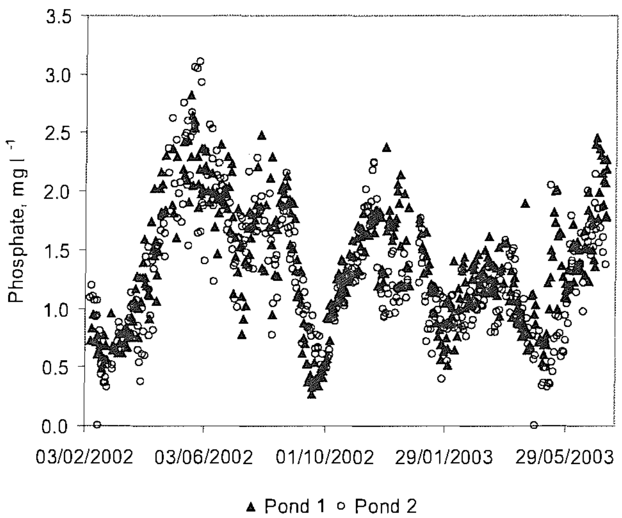

Figure 1 Phosphate concentrations in Ponds 1 and 2 for 16 months before the experiment

Immediately after mixing, the suspended solids concentration of the top water from the two ponds was 160 mg l^{-1}. In the eight days following this fell sharply in both ponds to around 110 mg l^{-1}. The reason for this is unclear, but it is possible that after mixing the algal species balance was out of equilibrium. During this period there was good correlation between values of SS, absorbance, phosphate, and maximum and average DO in the two ponds, as shown in Table 2. COD concentrations were similar throughout, although the correlation was low. Both ponds showed a strong correlation between average light and average DO concentrations (R^2 = 0.95 in Pond 1 and 0.8 in Pond 2). The dominant trend was the fall in SS, but there were slight differences between the ponds: for example SS and chlorophyll were related in Pond 2 (R^2 = 0.77) but not in Pond 1 (R^2 = 0.06). Small quantities of floating sludge appeared in Pond 1 on six of the eight days.

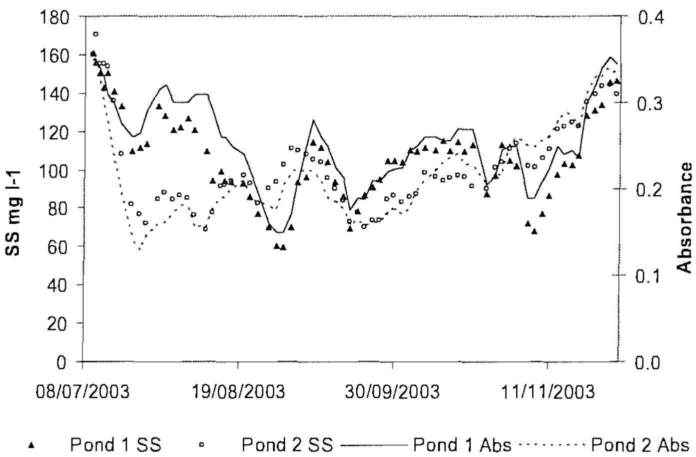

Figure 2 Pond 1 and 2 suspended solids and absorbance during the whole experimental period

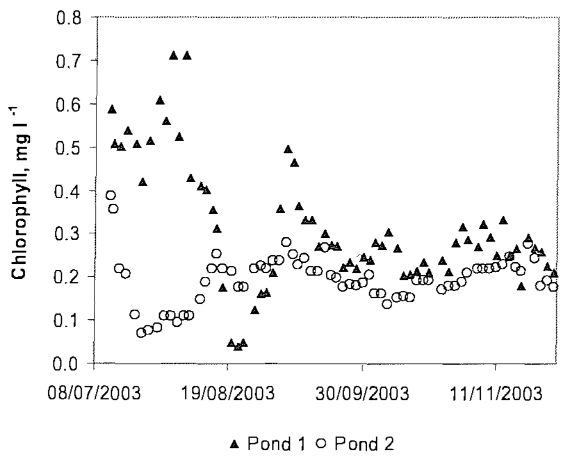

Figure 3 Pond 1 and 2 chlorophyll during the whole experimental period

Table 2 Average parameters and correlation between ponds during phases of experimental period

mg l^{-1}	SS	Chloro	Abs	COD	NO_3	NH_4	PO_4	Alk	Scum[1]	Sludge[1]	DOmax	DOave
Day 1-9												
av P1	148	0.53	0.32	49	0.10	1.27	1.48	123	1.50	1.43	116%	43%
av P2	148	0.29	0.29	55	0.04	0.17	0.74	177	1.00	0.00	205%	97%
R^2	0.74	0.24	0.98	0.06	0.12	0.04	0.85	0.00	0.00	0.00	0.88	0.89
Day 10-35												
av P1	117	0.53	0.30	67	0.14	2.45	2.02	178	2.00	1.36	82%	28%
av P2	80	0.11	0.16	62	0.06	0.18	0.77	216	1.75	0.09	82%	27%
R^2	0.36	0.10	0.27	0.72	0.00	0.34	0.63	0.43	1.00	0.05	0.39	0.19
Day 36-59												
av P1	83	0.21	0.21	76	0.05	5.79	2.62	223	1.75	2.08	89%	31%
av P2	97	0.22	0.20	80	0.10	0.26	1.39	196	1.58	0.25	81%	26%
R^2	0.00	0.71	0.06	0.11	0.04	0.14	0.55	0.04	0.73	0.09	0.37	0.44
Day 60-143												
av P1	104	0.27	0.25	46	0.09	0.19	1.36	280	1.03	0.30	-	-
av P2	102	0.20	0.23	55	0.04	0.06	1.32	291	1.03	0.10	-	-
R^2	0.39	0.22	0.53	0.01	0.12	0.28	0.00	0.74	0.00	0.21	-	-
Whole experiment												
av P1	106	0.32	0.25	55	0.10	1.65	1.74	243	1.34	0.89	-	-
av P2	102	0.20	0.22	60	0.05	0.13	1.19	252	1.19	0.11	-	-
R^2	0.34	0.03	0.21	0.21	0.00	0.27	0.01	0.62	0.67	0.09	-	-

[1] Values for sludge and scum are based on index 0-5; DO is % saturation; all others mg l^{-1}

From day 10 to day 35 suspended solids concentrations appeared relatively stable with an average of 117 mgl^{-1} SS in Pond 1 and 80 mgl^{-1} in Pond 2. During this period the soluble COD, measured on filtered samples taken from the ponds before the daily feed was added, showed low residual values of 65 and 60 mg l^{-1} in Ponds 1 and 2 respectively. While these values cannot be compared directly with those from the same period in previous years, due to changes in pond operating regime, the treatment efficiency is clearly similar to that of the overall period as shown in Table 1.

The major difference between the ponds during the first 35 days was in concentrations of nitrate, ammonia and phosphate which were significantly lower in Pond 2 than Pond 1 (see Table 2). It is clear that during this time the nutrient concentration in Pond 2 was very low, with available soluble nitrogen in the form of nitrate and ammonia at a level likely to be growth-limiting to planktonic algal species. This was not the case in Pond 1 where the average concentration of available soluble nitrogen from day 10-35 was 11 times greater than in Pond 2, and sufficient in itself to explain the difference in suspended solids. The chlorophyll concentration in Pond 1 during this time ranged from 0.4 - 0.7 mg l^{-1} with an average value of 0.5 mg l^{-1}, whilst in Pond 2 it remained between 0.1 - 0.2 mg l^{-1} indicating that at least a proportion of the difference in suspended solids between the two ponds could be attributable to algae. Further evidence that ammonia may be the limiting nutrient in Pond 2 comes from the relatively low correlation between levels of ammonia and phosphate (R^2 = 0.65 and 0.35 for days 1-9 and 10-35 respectively); the corresponding values in Pond 1 are higher (R^2 = 0.93 and 0.55) and may indicate that neither nutrient has been reduced to a limiting value but remains proportional to the amount supplied. Ammonia and phosphate showed negative correlations with SS in Pond 1 in this period (R^2 = 0.69 and 0.62), but little relation in Pond 2 (R^2 = 0.07 and 0.19). In both ponds the concentration of phosphates was positively correlated to water temperature, especially temperature in the bottom layer (R^2 = 0.51 and 0.86 in Pond 1 and 2 respectively).

As the feed to each pond was identical during this period, and as the potential for nutrient uptake was greater in Pond 1 because of the higher biomass density, the difference in soluble nutrient level can only be explained by release from the bottom sediment in Pond 1. Further evidence that the benthic deposits are in a dynamic state of interaction with the water column can be seen by reference to the levels of dissolved oxygen measured over the same period (Figure 4). In considering these it should be remembered that both ponds received the same surface illumination and that Pond 1 has the greater concentration of algal biomass, and hence the greater photosynthetic capacity. Both ponds receive the same external daily BOD load and are therefore subjected to the same external oxygen demand: it would therefore be expected that DO concentrations in Pond 1 would generally be higher than in Pond 2. In practice this was not the case: throughout this period DO levels in Pond 2 are the same as or higher than those in Pond 1. There was also a strong correlation between average DO and SS, chlorophyll and absorbance in Pond 2 (R^2 = 0.81. 0.84, 0.77 respectively), while Pond 1 showed little or no relationship (R^2 = 0.21, 0.00, 0.24). These results indicate that an additional oxygen demand is being exerted by release of soluble organic materials from the sludge layer. The material is effectively degraded, however, as indicated by both the increased oxygen demand and the fact that the residual COD in the two ponds is almost equal. By day 35 values for most parameters in Pond 2 had risen to levels at or near those in Pond 1, suggesting that fresh sludge building up in this period was beginning to make its contribution.

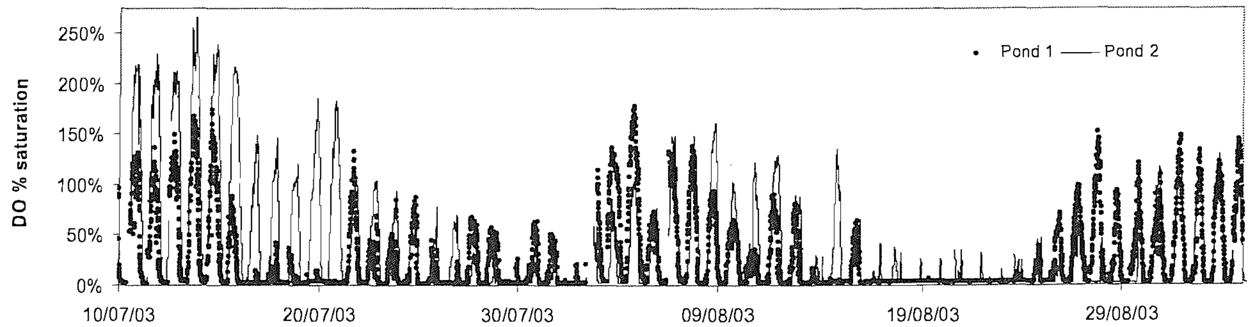

Figure 4 Dissolved oxygen concentrations in Pond 1 and 2 for days 1-71

On day 36 an unexpected event made further direct evaluation of the effect of the benthic sludge impossible. This event was interesting in itself, however, and is reported as it shows the potential impact of the sludge layer on a pond under certain conditions. Water temperature in the ponds has always shown stratification, with diurnal fluctuations that are more pronounced closer to the surface and a relatively steady temperature at the bottom. Typically the surface water temperature rises in the day and then cools in the evening, when it may reach a point where the surface is slightly cooler than the layer below (see Figure 5). This results in a partial turnover, taking dissolved oxygen into the lower layers. During the experiment the weather was very hot, with daytime temperatures in the greenhouse reaching over 40°C. In the early hours of day 36 the surface and middle layer of Pond 1 came to within 0.5 °C of the bottom temperature, resulting in a larger turnover that brought a significant amount of sludge to the surface (index 5). The conditions causing this were unusual for the ponds under study, despite their relatively shallow depth, and were a result of a prolonged series of hot days that raised the average water temperature. On day 38 the bottom temperature actually exceeded that in the upper layers (Figure 6), and a complete turnover occurred in which most of the sludge rose. The rising sludge in Pond 1 had a pronounced effect on a number of parameters:

- Soluble COD peaked at 187 mg l^{-1}, from previous values of about 70 mg l^{-1}.
- Ammonia rose from 2-3 mg l^{-1} to 8-9 mg l^{-1} for 10 days then fell to 0.25 mg l^{-1}.
- Suspended solids fell to around 60 mg l^{-1} from levels above 100 mg l^{-1}.
- Chlorophyll levels averaging 0.5 mg l^{-1} declined very sharply to 0.1 mg l^{-1}.

Figure 5 Pond 1 temperatures around turnover

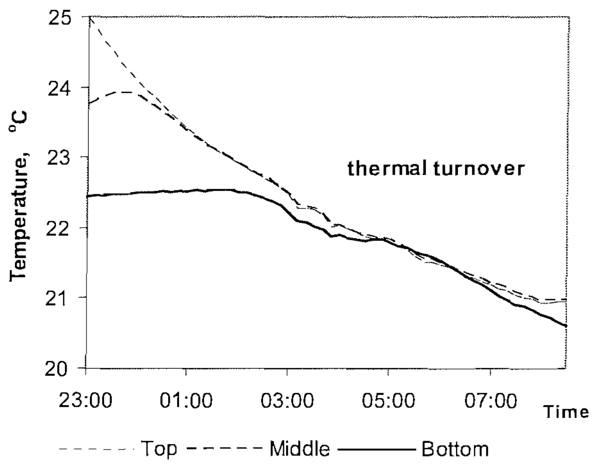

Figure 6 Pond 1 temperatures on turnover

Similar complete turnovers of the water body also took place in Pond 2 but the effects were less marked as the quantity of sludge accumulated since de-sludging on day 1 was small. It was however detectable, with the concentration of ammonia in the water column briefly peaking at 0.86 mg l^{-1} from previous values ranging between 0.1-0.2 mg l^{-1}. Soluble COD also rose briefly to 108 mg l^{-1} from 60-70 mg l^{-1}. It should be noted that when there is an accumulated benthic sludge layer there is always the potential for rising sludge even when the conditions described above do not occur, although the event may be at a much reduced scale. Throughout the reported experimental period of 143 days there was some correlation between the index estimate for rising sludge in Pond 1, and temperature, especially temperature in the bottom ($R^2 = 0.45$). A very small quantity of floating sludge first appeared at the surface of Pond 2 on day 32 and after this there was a weak correlation between temperature and the rising sludge index, reflecting the consistency of the relationship but also the smaller sludge quantity in the benthic layer. Sludge that has risen tends to sink naturally during the day time period and usually has no noticeable adverse effect on pond operation.

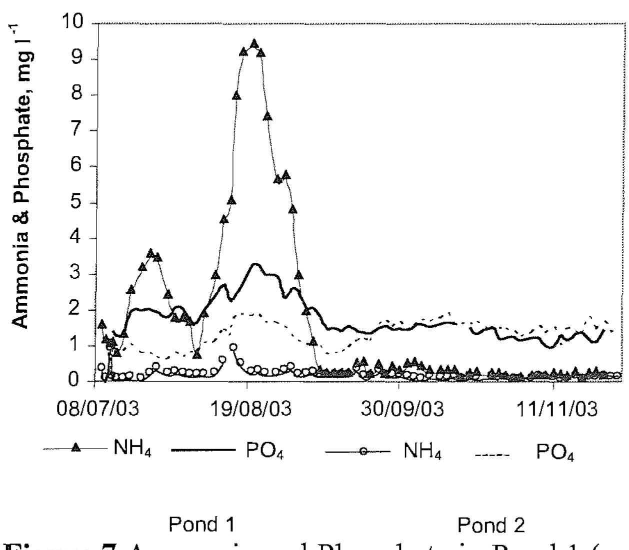

Figure 7 Ammonia and Phosphate in Pond 1 (solid lines and symbols) and 2 (doted lines, open symbols) around turnover

Table 3 R^2 values for prediction of average DO based on other parameters

	Pond 1	Pond 2
Day 1-71		
Light only	0.26	0.29
SS only	0.02	0.73
Chlorophyll only	0.01	0.29
SS and light	0.25	0.78
Chlorophyll and light	0.25	0.43
Day 1-35		
Light only	0.64	0.46
SS only	0.21	0.81
Chlorophyll only	0.00	0.84
SS and light	0.72	0.91
Chlorophyll and light	0.61	0.87

From day 36-59 the behaviour of the ponds reflected the different degrees of impact of pond turnover. As might be expected, while the COD and DO in Pond 2 were still largely determined by the presence of algae in the water column, Pond 1 was much more affected by sludge. Pond 2 showed a reduced correlation between suspended solids and average DO ($R^2 = 0.62$), which disappeared in Pond 1 ($R^2 = 0.04$). COD in Pond 2 was negatively related to DO ($R^2 = 0.71$) and to SS ($R^2 = 0.81$), while Pond 1 showed no relationship ($R^2 < 0.02$). There was a weak negative correlation between phosphates and SS in Pond 2, ($R^2 = 0.54$), which may have been due to algal uptake; but no equivalent in P1. In general the correlations between related internal parameters in Pond 1 were weaker indicating that conditions in this period were more variable. The correlation between sludge and bottom temperature in Pond 1 was R2 = 0.72, and there was zero correlation between SS in the two ponds in this period.

After falling from day 36-53 the concentration of SS in Pond 1 then began to rise, reaching 114 mg l^{-1} on day 61. Chlorophyll concentrations which fell even more rapidly also recovered to 0.5 mg l^{-1} on day 59. These results suggest that the rising sludge initially caused die-off or sedimentation of the algae through shading or toxicity, followed by recovery enhanced by the released nutrients. Chlorophyll concentrations in Pond 1 in this period were related to both phosphate ($R^2 = 0.81$) and ammonia ($R^2 = 0.52$). The relationship between chlorophyll and DO was weak but similar in both ponds ($R^2 = 0.27$ and 0.31). The average DO concentration in P1 was actually fractionally higher than in Pond 2 during this period (see Table 2); DO in Pond 1 fell almost to zero on pond turnover, but rose sharply as the algal population increased stimulated by the release of nutrients.

From day 60 onwards, the behaviour of the two ponds slowly equalised, with similar relationships between parameters in each pond, and individual parameter values moving closer, leading to increased correlation between the ponds (see Table 2). The remaining differences mainly concerned nutrient concentrations, with more nitrate and ammonia in Pond 1 than Pond 2 (Table 2). Bottom temperatures showed some correlation with nitrates in both ponds ($R^2 = 0.69$ in P1, 0.59 in P2), and with ammonia in Pond 1 only ($R^2 = 0.59$). Ammonia in Pond 1 is strongly linked with phosphate throughout, with a correlation coefficient R2 = 0.75 for the whole experimental period of 143 days.

An analysis was undertaken of the relationships between some key operational and discharge parameters (SS, DO, COD, nutrients) and driving factors such as light and temperature. Results for DO concentrations predicted by multiple regression from other parameters are shown in Table 3. In Pond 2 SS is an effective predictor of DO, and still more so when combined with light intensity. It is clear however that even before pond turnover Pond 1 is a more complex system influenced by many parameters: the most significant for DO concentration from Day 1-71 is ammonia followed by light, bottom temperature and phosphates with $R^2 = 0.34$, 0.26, 0.22 and 0.19 respectively.

The experimental period can therefore be split into four distinct phases: days 1-9 showed initial stabilisation following mixing and de-sludging of Pond 2; days 10-35 showed relatively stable operation and allowed direct comparison between the ponds to determine the influence of the sludge layer; days 36-59 showed the response of Pond 1 to a major rising sludge event; and days 60-143 showed conditions in the two ponds slowly equalising. The results show that the two ponds initially responded in a similar manner to ambient conditions, but the pond containing sludge had a significantly higher concentration of suspended solids and chlorophyll, indicating return of nutrients from the benthic sediments. During this time a fresh layer of sediment was building up in the previously de-sludged pond. The recovery of SS, chlorophyll and other parameters in the first 35 days after de-sludging indicates that the activity of the top layer of freshly deposited sludge contributes significantly to the effect on the water column. In general the results suggest that the presence of sludge does not have an inhibitory effect but in fact contributes to the growth of primary producers under normal conditions. Pond turnover had a dramatic short-term effect which gradually declined and the performance of the two ponds drew closer over a period of weeks. Effectively pond turnover fits within the pond's self-adjusting system where the release of nutrients stimulates the growth of algae which then provide oxygen for breakdown of the associated COD.

CONCLUSIONS

Accumulated sludge contributes to the nutrient load in the water column in a way that significantly affects pond behaviour. In particular there is an increase in concentrations of SS and nutrients, which may be of importance if the pond is to be discharged to sensitive natural waters. This behaviour does not appear to be detrimental to overall performance in terms of COD removal or DO concentrations in the pond, however; and given that the nutrient contribution from freshly deposited sludge approached that from a mature sludge layer within 7-10 weeks it seems unlikely that changes in recommended design or de-sludging frequency can be used to regulate it in a practical manner.

ACKNOWLEGEMENTS

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Section 5: Light attenuation parameters for waste stabilisation ponds

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Abstract
Effective modelling of shallow water ecosystems, including waste stabilisation ponds, is strongly dependent on the availability of good estimates of the light attenuation coefficient *k* (m^{-1}). Experimental data is presented on its determination using purpose-built laboratory apparatus with a near-parallel halogen light source and an array of photodiodes allowing measurements of irradiance at different depths. The equipment was used to compare *k* values from 4 different pure cultures, and mixed cultures of algae taken from a pilot-scale WSP. Laboratory values were compared with in situ measurements in the pond. At concentrations above 50 mg l^{-1} the relationship between *k* and suspended solids is non-linear; *k* also varied with depth. This could be modelled by a single equation, suggesting similarity of response in different cultures. At shallow depths and low suspended solids concentrations k values are variable and hard to measure reliably. The results highlight the need to standardise on a method for the measurement and reporting of *k* values if these are to be widely applicable in the development of pond models.

Keywords extinction coefficient, light attenuation, waste stabilisation ponds.

INTRODUCTION

Light plays a vital role in the functioning of a waste stabilisation pond (WSP), providing the energy source for photosynthesis and thus oxygen production. Knowledge of the parameters affecting light intensity or irradiance within the pond is thus crucial to modelling and prediction of WSP behaviour. There is an extensive scientific literature, both theoretical and experimental, on light attenuation in oceans and freshwater bodies, and also in photobioreactors. WSPs fall between these two applications, however, and relevant parameter values can be more difficult to find.

Attenuation of light follows an exponential relationship of the form $I_z = I_0e^{-kz}$ (equation 1) where I_o is the subsurface irradiance, I_z the irradiance at depth z and *k* is a light attenuation coefficient. A huge body of work in the fields of limnology and oceanography concerns values and expressions for *k* (see Kirk, 1994). For many purposes *k* is assumed to be a linear function of one or more components such as suspended solids (SS), dissolved solids or chlorophyll. Numerous expressions have been proposed for conditions similar to those in WSPs, such as eutrophic lakes and estuaries (e.g. Tsirtsis, 1995; Lonin and Tuchkovenko, 2001). Values of *k* in excess of 10 m^{-1} are quoted by Wetzel (2001) for stained and eutrophic lakes. Brawley *et al.* (2003) note that effective modelling of shallow water ecosystems is strongly dependent on the availability of good estimates for *k*

Photobioreactors are designed to operate at concentrations of algal biomass far higher than those usually found in WSPs. In these conditions the assumption of linear dependence of *k* on biomass concentration is known to be invalid. Yun and Park (2001) tried a theoretically-based approach to modelling *k* for *Chlorella vulgaris* in the biomass concentration range 0-2000 mg l^{-1}, using a linear approximation, an equation proposed by Cornet *et al.* (1992), and a hyperbolic model. While Cornet's equation was more satisfactory in the sense of having a physical basis, the hyperbolic model was found to give the best fit. Fernandez *et al.* (1997) also found a hyperbolic model gave the best results for data from *Phaeodactylum tricornutum* at concentrations of up to 3000 mg l^{-1}. At low biomass concentrations the effect of scattering was significant: for the range observed the highest value of *k* was found at 24 mg l^{-1}, while from 40-294 mg l^{-1} the relationship between *k* and biomass concentration appeared linear. Privoznik *et al.* (1978) also noted the relative importance of scattering in cultures of *Chlorella pyrenoidosa* at low concentrations. Ogbunna *et al.* (1995) calculated a specific *k* value per kg m^{-3} of 200 m^2 kg^{-1} for *C. pyrenoidosa*, and applied this to modelling growth curves from 17 - 3000 mg l^{-1}.

A definitive study of light penetration in WSPs, looking at both photosynthetically active radiation (PAR) and monochromatic light, was carried out by Curtis *et al.* (1994). Absorbance played a far more important role than scattering for all ponds in the study, pond-to pond variation was mainly attributable to differences in algal biomass, and variations in attenuation were observed at different wavelengths and depths. Despite this, in practice many models are based on simple empirical linear relationships, and measured absolute values for *k* are hard to find. Bartsch (1961) found *k* values of 6-11 m^{-1} in Dakota WSPs in summer, while Thomann (1987) gives a value of 23 m^{-1}. Mesplé *et al.* (1994) obtained a specific *k* value for total SS minus phytoplankton of 0.05 m^{-1} per mg dry weight l^{-1}, for high rate algal ponds. Juanico *et al.* (2003) used a specific *k* value of 0.29 m^{-1} per mg l^{-1} of carbon, based on literature values, and found that a change in light absorption and self-shading factors produced a two-fold change in average irradiance in spring, but was not significant in summer due to higher concentrations of algae and organic matter. A survey of WSPs in New Zealand found a median euphotic depth of 0.35 m corresponding to a *k* value of 13 m^{-1} (Davies-Colley *et al.*, 1995).

In the current work, *k* values for mixed and pure cultures were measured in purpose-built laboratory apparatus, and also in three pilot-scale pond systems. As photosynthetic sulphate-reducing bacteria can also be present in pond systems in certain conditions, imparting a red-purple colour to the water, a mixed culture rich in these was also grown and tested.

MATERIALS AND METHODS

Light sensors. Light intensity was measured using type BPW 21 photodiodes (RS components, UK), used in similar applications elsewhere for measurement of PAR (Ensminger *et al.*, 2001). Photodiodes for laboratory use were calibrated against a LI-210SA photometric sensor (LiCor, USA), while those used for external measurements were calibrated against a RC/0308 standard photovoltaic cell (PV Systems, UK). Where several photodiodes were to be used in one set of measurements, the output from each was checked against the average output from all under different conditions of illumination. A strong linear relationship (R^2 >0.998) was found, allowing calculation of a normalising factor. Photodiode outputs were continuously sampled using a datalogger (DataTaker D500 and expansion unit). Readings were averaged over a 30-second period and then over longer periods as required. Output was measured in milliamps unless noted.

Waste stabilisation ponds. Measurements were made in three sets of pilot-scale ponds, two located in Almaty, Kazakhstan and one in Southampton, UK. The first set of Almaty ponds A(I) consisted of four circular concrete tanks 2 m in diameter with a water column depth of 1.5 m. The ponds were fed on screened wastewater from Almaty sewage works, with a typical 5-day biochemical oxygen demand (BOD_5) of 200 mg l^{-1}. They were batch fed over a one-hour period each day, to give hydraulic retention times of 7.5, 15, 22.5 and 30 days. An array of photodiodes constructed to read at the surface and at depths of 0.3, 0.6, 0.9 and 1.2 m was moved between the ponds on a 3-day cycle. Readings taken in millivolts were averaged over a 15-minute period. Construction and operation of the second set of Almaty ponds A(II) is described elsewhere (Banks *et al.*, 2002). These ponds were instrumented each with a single photodiode at a depth of 0.25 m, with readings averaged over a 15-minute period. The work in Southampton was carried out on two ponds (SP1 and SP2) each with a surface area of 0.9 m^2 and a water column depth of 0.6 m. The ponds consisted of semi-translucent polypropylene tanks externally insulated with 50 mm of polystyrene foam, preventing any light entering through the tank walls. They were housed in a south-facing greenhouse, and received supplemental lighting from an array of halogen floodlights capable of providing a surface illumination of 300 W m^{-2}. They were batch fed on a synthetic wastewater of the type used for Almaty A(II) ponds, at a hydraulic retention time of 22 days. The ponds were instrumented with photodiodes at depths of 0, 0.15, 0.33 and 0.53 m and in normal operation readings were averaged over 10-minute intervals. For detailed comparison with laboratory measurements an array of eight photodiodes at depths of 0 (two diodes), 0.09, 0.18, 0.27, 0.36, 0.45, and 0.54 m was used, with measurements averaged over 1-minute intervals. In all cases readings were taken immediately after cleaning of the photodiode surface.

Microbial suspensions. Mixed cultures were taken from the Southampton ponds at different seasons; and a culture dominated by purple sulphur bacteria was grown in shallow pond water covering a layer of sulphur-rich pond sediment. Cultures of *Scenedesmus subspicatus* (CCAP 276/20), *Chlorella vulgaris* (CCAP 276/20), *Chlamydomonas reinhardtii* (CCAP 11/32b), and *Microcystis aeruginosa* (CCAP 1450/16) were obtained from the Culture Collection of Algae and Protozoa, Dunstaffnage Marine Laboratory, UK. Cultures were grown on Jaworski's Medium (CCAP JM recipe), modified for *M. aeruginosa* by the addition of 1 ml l^{-1} of trace element solution (Pfennig *et al.*, 1981). Cultures were activated by inoculation into 250 ml flasks containing 100 ml of medium, and incubating for 4-7 days at 20 ^{o}C on an illuminated orbital shaker (Gallenkamp, UK). The contents of each 250 ml flask were then transferred to a 2-litre flask containing 1 litre of medium, and incubated under the same conditions for a further 4-7 days. Two 2-litre flasks were then used to inoculate a 20 litre glass container aerated by a filtered air supply and illuminated by an array of eight 35 W white fluorescent tubes at 18-22 °C for a further 4-7 days.

Light apparatus. Light attenuation measurements were made in a purpose-built column apparatus consisting of a dark grey non-reflective PVC tube 150 mm in diameter and 1.5 m deep, fitted with a horizontal array of six photodiodes located centrally to minimise wall effects. The array could be moved vertically through the column of water and positioned at any depth. Illumination of the water column could be achieved using different light sources, but the source used in the current work was a PAR 36 light with a sealed beam 30 W halogen lamp (General Electric 4515). This allowed near-parallel light to be directed at the water surface. A mathematical correction for divergence is possible (Privoznik *et al.*, 1978) but as each bulb had slightly different characteristics in practice correction was made by measuring the light intensity in air, and deducting the resultant value or slope of the line from values measured in the algal cultures (Fernandez *et al.*, 1997). Sedimentation of algae was prevented by recirculation at a pumping rate in excess of algal settling rate (Stutz-McDonald and Williamson, 1979).

Sampling and analysis. Suspended solids were measured by filtration of an appropriate volume through a pre-dried and weighed GFC filter (Whatman, UK), in accordance with the procedures in Standard Methods for the Examination of Water and Wastewater (1998). Chlorophyll was determined by filtering through a GFC filter (Whatman, UK) previously dosed with 1 ml of a saturated solution of $MgSO_4$. Extraction was by grinding followed by treatment with acetone and centrifugation for 15 minutes at 3000 rpm. The resultant colour was measured at 664 and 665 nm using a Cecil Instruments spectrophotometer (3000 series). Absorbance was measured at 664 nm in a colorimeter (Camlab DREL/5). Samples were analysed in triplicate.

Data handling. At low to medium suspended solids concentrations, where the relationship appeared linear, k values were obtained by plotting $\ln(I_z)$ against depth z, with k as the gradient of a line fitted by the method of least squares. At higher concentrations where the non-linearity of the relationship between $\ln(I_z)$ and z is apparent, local k_z values were calculated for a given depth using the formula $k_z = \ln(I_z/I_o)/z$. For calculation of daily in-pond k values, measurements were rejected if the correlation coefficient for z and $\ln(I_z)$ was $R^2 < 0.98$. Measurements taken in Kazakhstan were also discarded if the irradiance readings indicated passage of clouds during the measurement period. All data processing was carried out using Excel spreadsheet software (Microsoft Excel).

RESULTS AND DISCUSSION

Light column experiments

The results of more than 60 sets of light attenuation measurements in the column apparatus described above carried out with pure and mixed cultures at different dilutions indicated that repeatability and reliability of measurements was good. The difference between values of k obtained from separate runs for the same culture and dilution was usually less than 0.1 m^{-1}. At low to medium concentrations of SS, where the relationship between depth and the natural logarithm of irradiance can be considered as linear, correlation coefficients between z and $\ln(I_z)$ were generally in excess of 0.995 and often of 0.999. The relationship between k values and SS concentrations obtained by dilution of a given culture was also strongly linear at low concentrations, with $R^2 > 0.995$. Gradients for k versus SS were generally in the range of 0.12-0.2 m^{-1} per mg l^{-1} of suspended solids. These values agree well with Bowen (1997), who suggested an average value for algal biomass of 0.17 and a range of 0.06-0.34 m^{-1} mg^{-1} l, based on a value of 19 m^{-1} mg^{-1} l for chlorophyll.

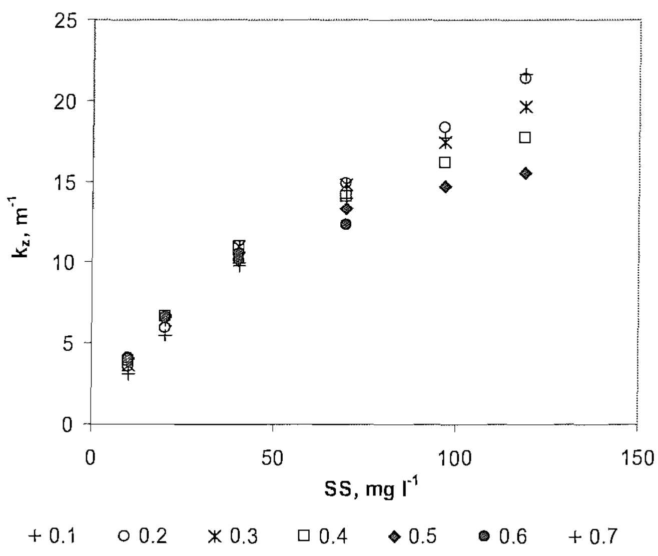

Figure 1 Variation of k_z with SS at different depths (metres) for a culture of *S. subspicatus*

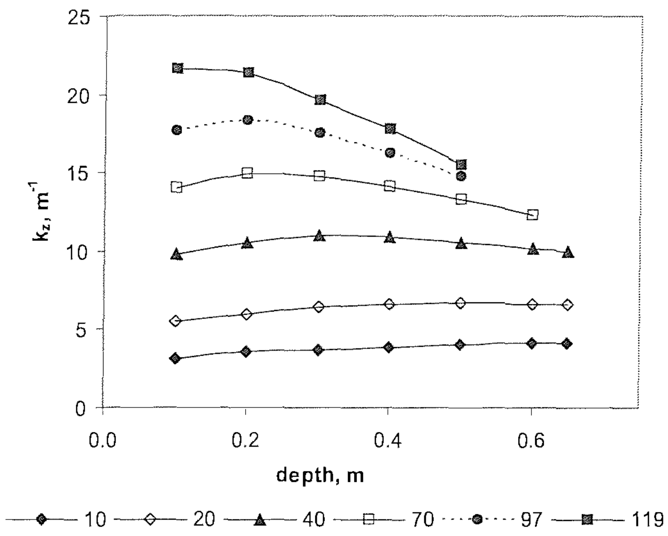

Figure 2 Variation of k_z with depth at different SS (mg l^{-1}) for a culture of *S. subspicatus*.

At higher concentrations the assumption of linearity no longer holds true, initially for the relationship between k and SS and then for depth and $\ln(I_z)$. Once the latter relationship is non-linear, k values can no longer be obtained from the gradient of z versus $\ln(I_z)$ and local values for k_z must be calculated. Figures 1 and 2 show the variation of k_z with suspended solids and with depth for a culture of *S. subspicatus*. At concentrations below 50 mg l^{-1}, values of k_z are similar at all depths and the relationship between k_z and SS is close to linear (Figure 1). Above this concentration, linearity is lost and the overall shape of the curve can be approximated by a hyperbola, in accordance with the findings of other researchers (Fernandez *et al.*, 1997; Yun and Park, 2001). Values for k_z at different depths also show increasing divergence, as described by Kirk (1994). This can be seen more clearly in Figure 2 where for SS concentrations of 10 and 20 mg l^{-1} values of k_z are almost constant. Above this value the line shows increasing curvature, resulting in a steep decline of k value versus depth at the highest measured concentration of 119 mg l^{-1}. This degree of variation supports the decision to investigate actual values of k and k_z on a scale appropriate to typical WSP design.

The results for *S. subspicatus, C. reinhardtii, C. vulgaris* and *M. aeruginosa* showed a similar pattern of variation of k_z with both depth and SS. Figure 3 shows the variation of k_z for *C. vulgaris* plotted as a 3D surface. The response surface for *S. subspicatus* can be modelled by an equation of the form $k_z = 0.4$ x (depth in metres) + 0.0001 x (SS in mg l^{-1})^2 - 0.2 x (SS in mg l^{-1}) (equation 2), giving a correlation of $R^2 = 0.94$ with the experimental data. The same equation applied to results for *C. reinhardtii, C. vulgaris* and

M. aeruginosa gives $R^2 = 0.88$, 0.95 and 0.84 respectively, indicating good similarity. The lower value for *M. aeruginosa* may be due in part to the lower SS concentration in the experiment, and the low chlorophyll content of 0.04 mg l^{-1} or 0.001 mg mg^{-1}. While equation 2 has no physical basis it does indicate that at depths typical for a WSP a linear correction for *k* may be satisfactory, but variations with respect to SS are significant in the range of concentrations likely to be encountered. Figure 4 shows k_z values for a mixed culture dominated by purple sulphur bacteria, with a high suspended solids content. The correlation coefficient for experimental and modelled data with the above expression was $R^2 = 0.94$.

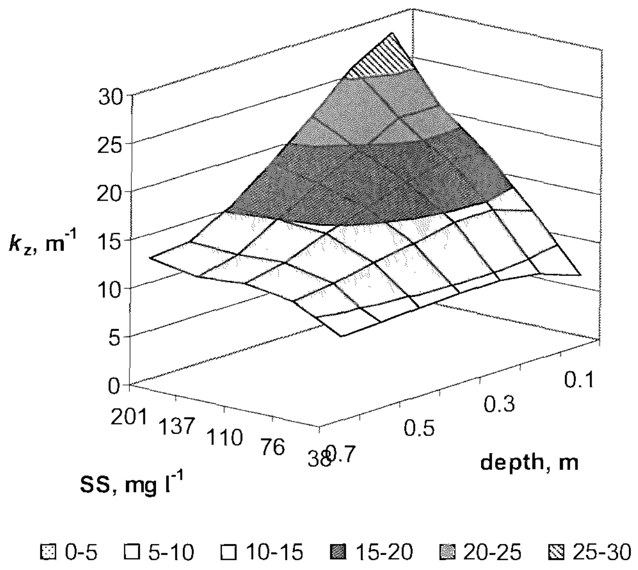

Figure 3 Variation of k_z with depth and SS for *C. vulgaris*

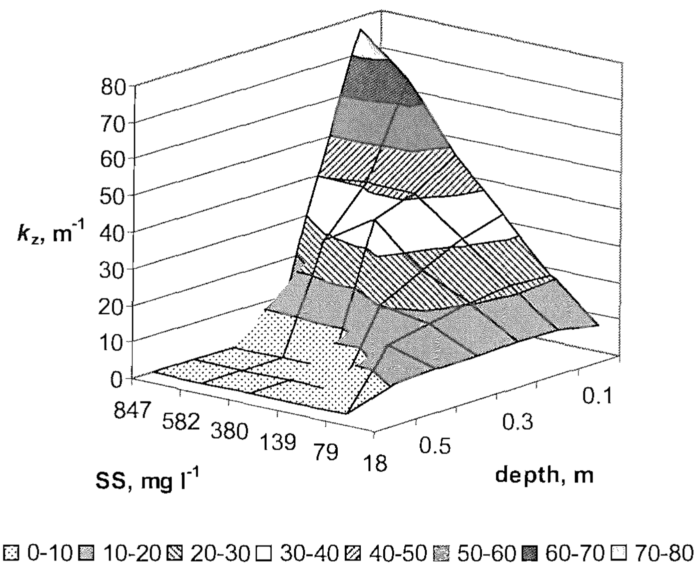

Figure 4 Variation of k_z with depth and SS for mixed culture with purple sulphur bacteria

Values of k and k_z were determined for mixed cultures from SP1 and SP2 in the concentration range 5-58 mg l^{-1} SS. In the region of assumed linearity, k values ranged from 5.2 to 14.7 m^{-1}. The results were compared with values measured in situ in SP1 and SP2 using an array of eight sensors. At SS concentrations above 25 mg l^{-1}, values measured by the two methods showed reasonably good agreement. The main difficulty was in obtaining reliable and reproducible k values from measurements in the pond: accuracy requires bright sunlight, high solar elevation, a cloud-free sky, the absence of local shading, and minimal depth variations, as noted by Curtis *et al.* (1994). Results vary depending on whether the light is bright or diffuse: Kirk (1994) notes that k may decrease with depth in diffuse light. While the photodiode array proved highly effective, optimum measurement conditions are often difficult to achieve in practice, providing an argument in support of laboratory-based methods. Values for k and k_z at SS concentrations below 20 mg l^{-1} were more difficult to measure in either the ponds or the column apparatus, especially at shallow depths (up to 0.2 m). In the column apparatus, this could be due in part to limitations of the equipment: at very low concentrations of suspended solids little absorption takes place, scattering plays a greater role and edge effects from the column walls may become apparent. At shallow depths small errors in depth measurement also have a greater effect. On dilution of a given culture from 25 mg l^{-1} to 10 and then 5 mg l^{-1}, however, the non-linearity of the relationship between k and SS was clearly seen. Further support for the degree of variation in k values at shallow depths can be found from several sources. Curtis *et al.* (1994) noted that rates of attenuation were sometimes lower near the surface. The results in Figure 2 suggest that k values at shallow depth may be lower even at relatively high SS concentrations. In-pond measurements at shallow depth where wall effects are absent also showed a similar pattern of variation. This type of near-surface variation may cause problems for WSP modelling since under normal conditions the majority of photosynthetic activity occurs in the upper 0.2-0.3 metres, and values of k or k_z obtained over a greater depth may therefore not be applicable. The difficulty is compounded at low SS concentrations. This is not generally an issue in operational WSPs with steady-state SS concentrations of 30-100 mg l^{-1}, but may present problems in modelling pond start-up or for non-steady state conditions such as those encountered in extreme climates.

There is clearly a need for standardisation in reporting of *k* values in ponds. In photobioreactors operating at high SS concentrations attenuation constants are frequently reported as specific values in $m^2\ kg^{-1}$ and may be based a particular light path length, giving a value of k_z; while limnologists tend to measure over a long path length and assume linearity of *k* with SS. Algal WSPs fall between these two applications, but in a range where linearity cannot be assumed and it may be necessary to specify depth, SS concentration and possibly even surface irradiance when reporting *k* values.

Pond measurements

While the above discussion indicates some of the difficulties in measuring and calculating attenuation coefficients in WSPs, it was considered useful to have an idea of the range of values that might be expected under varying operating conditions. For this reason measurements were made in pilot-scale ponds in three locations as described above.

Values of *k* for the Southampton ponds ranged from 4.8 to 13.7 m^{-1} throughout a one-year period of observation, while suspended solids ranged from 42-172 mg l^{-1}. As the ponds were kept under semi-controlled conditions of light and temperature, and were subject to different experimental conditions in different periods, no clear seasonal trends were noted. There was a reasonable correlation between *k* values and suspended solids concentrations in both ponds throughout the year ($R^2 = 0.74$) (see Figure 5). The gradients of graphs of *k* in m^{-1} versus suspended solids in mg l^{-1} were 0.066 and 0.056 respectively. This is low in comparison with values for algal cultures and probably reflects the presence of non-algal SS. The relationship between absorbance and *k* in the period for which both parameters were measured was weaker ($R^2 = 0.53$ and 0.65 for SP1 and SP2 respectively), while chlorophyll and *k* were effectively unrelated ($R^2 < 0.05$ for both ponds).

Table 1 R^2 for influence of parameters on irradiance at different depths in SP2

Parameters			
Depth (m)	0.15	0.33	0.53
Surface irradiance (SI)	0.47	0.28	0.14
SS	0.17	0.38	0.37
Chlorophyll	0.00	0.01	0.01
Absorbance	0.16	0.31	0.16
SI and SS	0.57	0.50	0.15
SI and chlorophyll	0.45	0.29	0.10

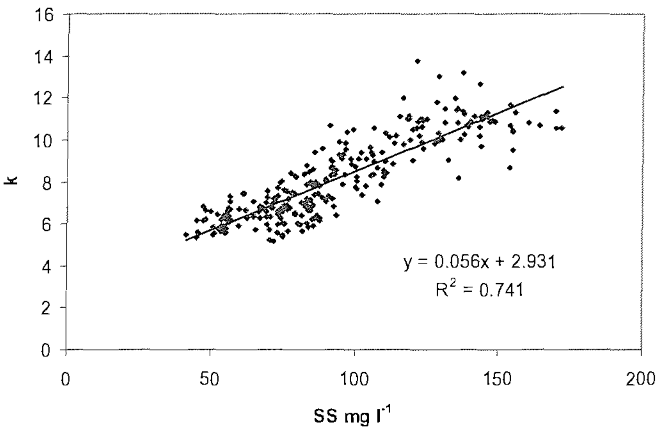

Figure 5 SS and k values for SP2

The absolute value of k on any particular day is of uncertain reliability, due to the method of measurement and in particular to variations in surface irradiance. On average surface irradiance will be less than the clear sky maximum, although as noted above k values with low correlation or low irradiance were discarded. The results, however, provide an interesting view of the range of k values likely to be encountered for a corresponding range of SS and chlorophyll concentrations.

In addition to the calculation of k values, the relationship between irradiance at a given depth and a number of other parameters was investigated by means of regression equations. Results for SP2 are summarised in Table 1. As might be expected, surface irradiance is the most influential single factor. Suspended solids appear to be better predictor of irradiance at 0.33 m than 0.15 m, once again confirming the influence of other factors and the problems of measurement in the upper layers. Surface irradiance and SS taken together are the strongest predictors, accounting for over 50% of variation in the top two layers of SP2. Chlorophyll by itself was a poor indicator of irradiance at depth but improved when considered in conjunction with surface irradiance.

For comparative purposes a similar study was carried out from late March to late May in Almaty, Kazakhstan where seasonal and climatic factors have a strong influence: in winter algal biomass concentrations in the water column fall sharply when the ponds freeze, followed by revival in spring. Values of k based on measurements in ponds A(II)1-3 rose rapidly from around 3-5 m^{-1} immediately after thawing in late March to 13-19 m^{-1} by early May, while suspended solids rose from 3-6 to 60-70 mg l^{-1}. Pond A(II)3 showed the strongest correlation between SS and *k* values for the whole period of observation, with $R^2 = 0.63$ and a gradient of 0.14 m^{-1} mg^{-1} l, showing good agreement with Bowen (1997) and the above results. During this period the pond was not fed, and the suspended solids were therefore mainly of algal origin, arising from nutrients remaining in the pond over winter or released from the bottom sediments in spring. Ponds A(II)1 and 2 showed a weaker correlation between SS and *k* values in the same period. During this time these ponds were fed on a synthetic wastewater containing suspended solids. The correlation between SS and *k* was stronger for Pond A(II)2, which was fed on half-strength wastewater, than for Pond A(II)1 fed on full strength wastewater containing correspondingly more SS ($R^2 = 0.45$ and 0.29 respectively). After peaking in late April, *k* values for these ponds fell until late May, although SS concentrations remained steady or rose slightly. The correlation of *k* with chlorophyll concentration in A(II)1 and 2 appeared promising but data were too few for reliability. In general the results suggested that no single factor determines *k* under non steady state conditions, especially where influent SS may have an effect. The maximum *k* value measured in Ponds A(II)1-3 during this period was 23 m^{-1}.

No detailed analysis of relationships between *k* and other parameters was carried out for the Almaty A(I) ponds due to insufficient data on SS and chlorophyll, but results for measurements of *k* values support those from the A(II) ponds, showing a steady rise from the end of April into May. Subsequent falls appear to have been linked with the appearance of large numbers of grazing organisms. The maximum value of *k* measured during this period was 25 m^{-1}. Values of *k* measured by this method in Almaty in this period are likely to be relatively reliable due to the high solar elevation and long periods of cloudless weather. None of the results from the A(I), A(II) and Southampton ponds showed a good fit with empirical equations devised for other locations (eg. Xu *et al.*, 2002), indicating that these depend on other parameters.

CONCLUSIONS

The light attenuation coefficient *k* was found to be significantly affected by depth and SS concentrations in the range of values typically found in WSPs. For practical purposes it may often be sufficient to consider *k* values as constant, but in certain conditions a more sophisticated approach may be needed taking local variation into account. Examples include modelling of WSP start up, or of the annual spring revival in strongly seasonal climates. Typical values for k in ponds appear to lie in the range of 5-25 m^{-1}. The use of photodiodes to measure local irradiance proved highly successful. In-pond measurement presents many practical difficulties, however, and the column apparatus may offer a reliable means of measurement under standard conditions. It is recommended that a standardised approach is adopted to the measurement and reporting of k values.

ACKNOWLEDGEMENTS

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Section 6: Wastewater reuse in central Asia: implications for the design of pond systems

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Abstract

The paper examines the potential of waste stabilisation ponds to provide water for reuse in extreme continental climates such as those of central Asia, where precipitation is low and summer evaporation rates are high. A simple model is used to predict water availability, BOD and faecal coliform removal for different configurations and operating regimes. The results show a significant proportion of flows could be saved for irrigation or aquifer and river replenishment; if standard designs can be modified to suit these climates, the system is likely to be both more robust and more flexible in terms of types of reuse. The paper concludes with 3 case studies of evaporation pond systems in Kazakhstan, assessing their potential for conversion to full biological treatment systems for water conservation and reuse.

Keywords

continental climate, evaporation, waste stabilisation ponds, wastewater reuse

INTRODUCTION

The Central Asian region has a sharply continental climate characterised by cold winters, hot summers and very low precipitation, making water a precious resource. Despite this, evaporation ponds are widely used as a means of wastewater disposal, for both industrial and domestic effluents. In Kazakhstan alone more than 500 such pond systems are believed to be in operation. In comparison with evaporation ponds, waste stabilisation pond (WSP) systems offer major advantages: they can provide treated water for a variety of uses, including irrigation through summer and high-quality water for top-up of rivers or aquifers in autumn.

Under certain conditions they may also retain water within a catchment, which would otherwise be lost during the winter period. Design guidelines for continental WSPs are less developed than for tropical or temperate areas, however, and in most cases appear to be based on cold climate systems, without taking into account the greater importance of reuse and the significance of high evaporation losses. This paper looks at some aspects of the design and operation of pond systems in Kazakhstan and Central Asia, and considers their implications for water conservation and reuse.

For ponds subject to seasonal ice cover, a widely-adopted design and operating regime is that of intermittent discharge, consisting of treatment plus storage for 6-12 months (US EPA, 1983; Prince *et al.*, 1995). Pond working depths are specified, and the surface loading rate on the first pond is limited. In cold regions this produces a high-quality effluent that can be discharged in a short period, usually in autumn. This approach is robust but may be conservative in warmer continental climates, where the spring warm-up is rapid and treatment capacity in summer months is greater. To determine the impact of some potential design and operational changes on water availability, a simple model was constructed and operated under different scenarios.

MATERIALS AND METHODS

The model simulates a WSP system in central Kazakhstan, consisting of a facultative pond (FP) and one or more storage/maturation ponds (SMP). The wastewater flow rate was taken as 1000 m^3 day^{-1}, with a biochemical oxygen demand (BOD) of 200 mg l^{-1} and a faecal coliform (FC) concentration of 4 x 10^8 l^{-1}.

Model construction. The model calculates mass balances for wastewater volumes, BOD and FC using a one-day time-step. Wastewater volumes are calculated taking into account inflow, outflow, evaporation and precipitation and assuming a lined system with no infiltration. The ponds are assumed to be simple rectangles in plan, with no allowance made for the variation of area with depth and side slope. BOD and FC concentrations are calculated assuming first-order decay kinetics. The decay constant k is assumed to follow an Arrhenius equation of the form $k_T = k_{20}\theta^{(T-20)}$, where k_T and k_{20} are values of k at temperatures of T °C and 20 °C respectively. Parameter values used were $\theta_{BOD} = 1.08$ and $k_{20\ BOD} = 0.25$ (Mara, 1976); and $\theta_{FC} = 1.19$ and $k_{20\ FC} = 2.6$ (Marais, 1974).

The FP is modelled by specifying a BOD surface loading rate and a working depth, thus fixing the surface area, volume and mean hydraulic retention time for a given inflow and influent BOD concentration. Once the surface area is known, daily and total outflows are calculated based on inflow minus evaporation and precipitation. The mass of BOD or FC in the pond is calculated based on the initial value, inputs, decay and discharge, and the daily effluent concentrations are obtained by dividing the total mass of BOD or FC by the pond volume.

The design of the SMP is determined by choosing a maximum working depth and a discharge period. The SMP is assumed to be empty at the end of the discharge period. The procedure is to guess an appropriate surface area for the SMP, from which total and daily values of evaporation and precipitation are calculated. The outflow from the SMP is equal to inflow (corresponding to outflow from the FP minus any direct discharges), minus evaporation and plus precipitation; daily outflows are obtained from the total outflow divided by the discharge period. The maximum volume stored in the SMP is equal to the total outflow, plus evaporation during the discharge period, and minus precipitation in the same period. From the maximum volume and chosen area a depth is calculated. If this depth is greater than the preferred maximum working depth for the SMP, the area must be increased and the calculation repeated. Once a satisfactory result is obtained, daily values are used to calculate pond depth and effluent BOD and FC concentrations. Under some operating regimes a SMP may stand empty for some time, for example if inflow is diverted elsewhere and evaporation exceeds precipitation. In this case total outflow is adjusted by the theoretical contribution from evaporation and precipitation during the period while the pond is empty.

Climate data. Mean monthly climate data were taken for Astana weather station located at 51.2° N, 71.4° E (Hong Kong Observatory, 2005) and are summarised in Table 1. Evaporation was calculated for a reference surface using CROPWAT software (Clarke *et al.*, 1998), and increased by 10% to give values for an open water surface. Daily evaporation and air temperatures were obtained from mean monthly values by polynomial interpolation. Water temperature was assumed to equal air temperature down to 0 °C, and to remain at zero for lower air temperatures. Daily precipitation values were obtained by dividing the mean monthly value by the number of days in the month.

Table 1 Climate data used in standard model (based on Astana weather station)

Parameter	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Year	
Mean temperature °C	-15.8	-15.9	-8.1	4.9	13.1	19.0	21.3	17.7	12.0	2.8	-5.9	-12.6	2.7	(mean)
Precipitation mm	17.4	13.7	14.3	22.0	33.4	34.8	49.5	39.7	24.0	29.6	21.7	17.3	317.4	(sum)
Evaporation* mm	4.4	7.1	18.8	80.2	156.9	196.0	193.0	150.4	101.3	41.3	10.2	3.8	963.3	(sum)
Mean daily sunshine hours	3.3	5.2	6.2	7.9	9.7	11.2	10.8	9.5	7.7	4.4	3.3	3.0	6.9	(mean)

*Based on calculated ET_O for grass surface

Model validation. BOD concentrations predicted by the model were tested using BOD and chemical oxygen demand data from experimental ponds in Almaty, Kazakhstan (Heaven *et al.*, in review;). The model was run with both mean daily air temperatures obtained from monthly values as above, and with actual mean air temperatures.

Modelling scenarios. Options considered are summarised in Table 2 and included:

- A conventional design based on a 2-stage system of a 1 m deep FP and a 2 m deep SMP. The loading rate on the FP is limited to 40 kg BOD ha^{-1} day^{-1} due to seasonal ice cover and the system operates with a single autumn discharge (1-30 September).
- Increasing the loading rate on the FP pond to 100 kg BOD ha^{-1} day^{-1}, thus reducing the surface area and hydraulic retention time
- Increasing the depth of the SMP from 2 to 4 m, to reduce surface area.
- Increasing both FP loading rate and SMP depths.
- As above but replacing a single SMP with two ponds in parallel that store and discharge water in alternate years. Effluent treated to a high standard by the end of the summer is stored over winter without any further addition of incoming wastewater, and is available for irrigation from early spring, thus maximising its economic usefulness.

Table 2 Options considered

Case		Facultative			Storage/maturation		Total
		BOD load	*Depth*	*Area*	*Depth*	*Area*	*Area*
Single storage/maturation pond		kg ha^{-1} day^{-1}	m	ha	m	ha	ha
1a	Standard design	40	1.0	5.00	2.0	11.80	16.80
1b	Standard with discharge July-August	40	1.0	5.00	2.0	11.80	16.80
1c	Increased loading rate FP	100	1.0	2.00	2.0	12.30	14.30
1d	Increased depth SMP	40	1.0	5.00	4.0	6.44	11.44
1e	Increased loading rate FP and depth SMP	100	1.0	2.00	4.0	6.71	8.71
Parallel storage/maturation ponds							
2a	Parallel SMPs	40	1.0	5.00	2.0	12.60	30.20
2b	Increased loading rate FP	100	1.0	2.00	2.0	13.35	28.70
2c	Increased depth SMPs	40	1.0	5.00	4.0	7.17	19.34
2d	Increased loading rate FP and depth SMPs	100	1.0	2.00	4.0	7.59	17.18

RESULTS AND DISCUSSION

Water quantity

Table 3 shows the amount of water potentially available for reuse, based on the results of modelling. The greatest gain comes simply from replacing evaporation ponds with WSPs: in comparison with a theoretical 100% loss, the standard design with a single autumn discharge (1a) allows use of 70% of the original inflow. Experience in Canada and the northern USA shows effluent quality in this period can be extremely high (Prince *et al.*, 1995), making it potentially suitable for aquifer or river replenishment. The water is of greater economic value if it is available in the growing season, which in central Asia can run from April-October depending on crop, latitude and altitude: a more usual period is May-August, but pre-irrigation of the soil in April or May is also common to make up the previous year's moisture deficit. If the water is of a suitable quality for reuse between July-August, it is available for at least the later part of the irrigation period (1b).

Table 3 Water available for reuse under different scenarios

Case		Net loss as % of original inflow			Water available for re-use		
		FP	*SMP*	*Total*	*% of inflow*	*Main discharge*	*Possible uses*
1a	Standard design	9%	21%	30%	70%	September	River/aquifer replenishment
1b	standard but disch July-Aug	9%	21%	30%	70%	July-August	Late irrigation, river/aquifer
1c	Increased loading rate FP	4%	22%	26%	74%	July-August	Late irrigation, river/aquifer
1d	Increased depth SMP	9%	12%	21%	79%	July-August	Late irrigation, river/aquifer
1e	Increased loading FP and depth SMP	4%	12%	16%	84%	July-August	Late irrigation, river/aquifer
2a	Parallel SMPs	9%	52%	61%	39%	April-August	All
2b	Increased loading rate FP	4%	49%	53%	47%	April-August	All
2c	Increased depth SMPs	9%	34%	43%	57%	April-August	All
2d	Increased loading FP and depth SMPs	4%	30%	33%	67%	April-August	All

The volume of water available can be increased by increasing the loading rate on the FP (1c): experimental work on pilot-scale ponds in Kazakhstan suggests this is possible without adversely affecting performance (Heaven *et al.*, in review). A more significant impact is produced by increasing the depth of the SMP. This may be justifiable as its primary function is storage, and light penetration is less important than in a FP, although playing a role in disinfection. An increase in depth implies higher construction costs, although in practice SMPs are sometimes located in natural depressions. If both of these design modifications can be adopted (1e), the amount of available water rises to 84%. In the standard design with two SMPs, availability falls to 39%; but this covers the whole growing season, and all types of reuse. Economic assessment is needed to determine whether the higher capital costs for construction of two ponds are outweighed by the value of the water. If it is possible both to increase the depth of the SMPs and to load the FP more heavily (2b-d), the available volume rises to as much as 67%. The scale of the potential savings for reuse indicates the need for larger studies to confirm whether these or other modifications live up to their promise in practice.

Water quality

BOD. The model fitted the Almaty data moderately well: at high loadings in particular, effluent BOD fell too rapidly in early spring; while summer BOD values tended to be overestimated in FPs and underestimated in SMPs. These discrepancies may be explained by a wide range of factors not included in the model: e.g. effects of nutrient release from pond sediments, existence of a lag phase before the spring algal bloom, higher water temperatures in late summer, and changes in pond population characteristics during the year. More complex models exist, but problems often arise in obtaining parameter values, especially for lower temperature ranges. Better fitting can be achieved by choosing different parameter values for each dataset and season, but this is to overstretch both the model and the available data. While the output is thus indicative rather than exact, the model was considered adequate for its purpose. Figure 1 shows some results from the Almaty ponds. The data also revealed significant year-to-year variability, which may be directly due to temperature variations in a given year (e.g. an early spring or hard winter), or to more complex ecosystem interactions.

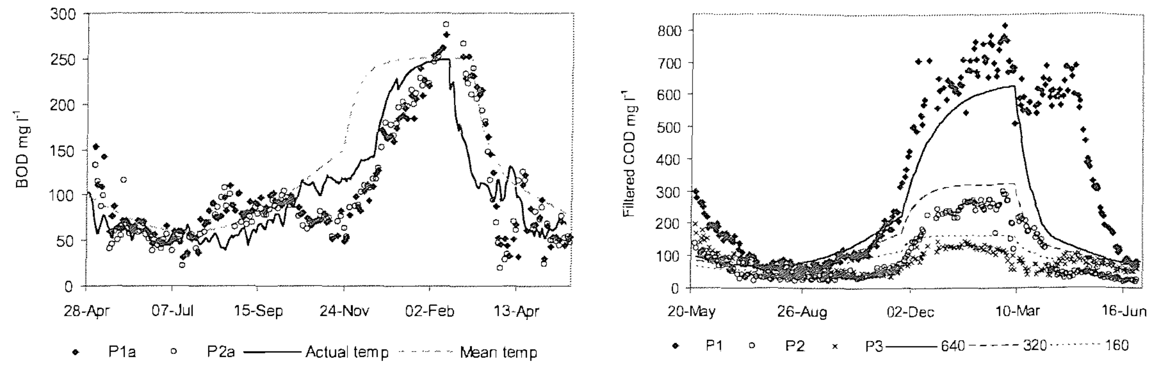

Figure 1 Real and modelled Almaty pond results for a) mean and actual temperature at 250 mg l^{-1} influent BOD and b) mean temperature at 160, 320, 640 mg l^{-1} influent COD

All the single SMP options modelled (1a-e) achieved low BODs (<5 mg l^{-1}) from July onwards. While actual values should be viewed with caution, this matches the observation by Prince *et al.* (1995) that loading rate has little effect on final effluent quality in intermittent discharge ponds. The main gains in water availability would result from increasing SMP depths and FP loading rates: the model indicates this may be possible, but again there is insufficient real data for design guidelines. The cases with two SMPs (2a-d) all had very low BODs throughout the 'rest' period, making water available for all reuse options. Actual results from SMPs may be higher and more variable, but the two-pond system is clearly robust, and may have additional benefits in balancing out the nutrient load from sediments. Figures 2 and 3 present key parameters for cases 1a and 2a, clearly showing the extended period of low BOD in case 2a.

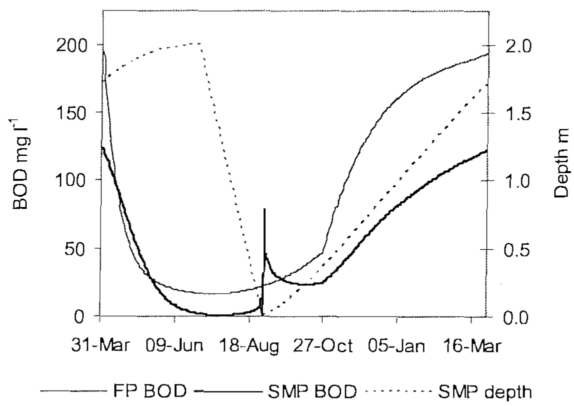

Figure 2 BOD and depth for Case 1a

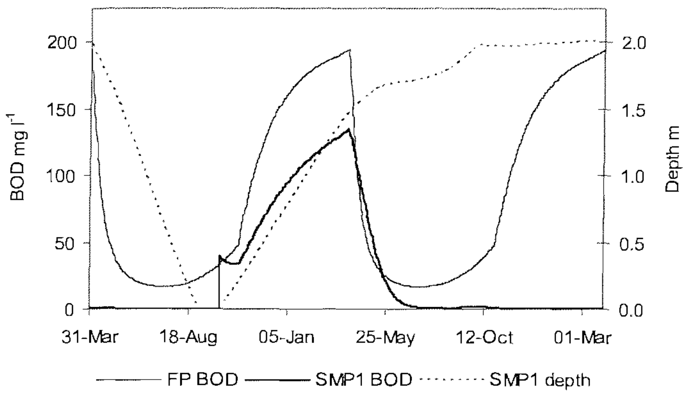

Figure 3 BOD and depth for Case 2a

FC. For reuse in irrigation, microbiological quality parameters are critical. WHO guidelines suggest values of 10^3 FC per 100 ml for unrestricted irrigation and 10^5 FC per 100 ml for restricted irrigation (WHO, in review). Modelling of FC gave results similar in form to those for BOD, but with more rapid winter die-off in the SMP. Reductions in the FP were typically of one order. The single SMP pond achieved levels below 10^5 FC per 100 ml from end April to end August (end September for 1a); and below 10^4 from mid May to mid August for lower FP loadings (1a, 1b and 1d), and early June to late July at higher loadings (1c and 1e). In systems with two SMPs (2a-d), FC concentrations were below 10^3 per 100 ml from November and negligible from May onwards in the 'rest' year.

Results for FC should be viewed with even more caution than those for BOD. Unfortunately no local data were available for model testing, as the Almaty pilot ponds were fed with a synthetic wastewater. Values of k_{FC} and θ_{FC} were taken from Marais (1974), and are said to be valid from 2-21 °C; but they assume aerobic conditions with complete mixing, and were derived from ponds with a 12-day retention period. There is evidence from both laboratory studies and field sampling, however, of extended survival of pathogen indicator organisms in storage ponds at low temperatures (Environment Canada, 1985; Torrella *et al.*, 2003). Once again, more studies are needed to provide an adequate basis for design. Marais (1974) noted that single large ponds for winter storage are a practical solution in cold climates due to the limited pathogen reduction; while above 21 °C there is an apparent reduction in k_{FC}, leading to lower die-off rates. In continental climate areas, which can move relatively rapidly from one temperature range to the other, special guidelines may be needed to ensure both safe and efficient reuse.

CASE STUDIES

Case study 1. The industrial site in this study is located in north-east Kazakhstan, where mean monthly temperatures range from -13 °C in January to +23 °C in July, with an annual mean of 5 °C. Annual precipitation is 302 mm and evaporation 957 mm. Domestic-type wastewater flows of 55 m^3 day^{-1} are generated by the site's administrative block. The wastewater receives primary sedimentation and biological treatment in a package plant, followed by rapid sand filtration and final discharge into a storage/ evaporation pond with a capacity of 20000 m^3. In the Soviet period it was planned to use the treated wastewater for irrigation, but no infrastructure was set up; the land nearby is not very suitable for agriculture and it is now unlikely this will happen. Due to problems with equipment and operator training the package plant does not work well. Performance is assessed on the quality of the discharge into the storage/evaporation pond, which fails to meet a number of the parameters set by the local Environmental Protection Department (EPD). The site operators are coming under pressure to replace the treatment plant, and to line the pond to prevent seepage and potential groundwater contamination. The capacity of the present pond taking into account precipitation, evaporation and infiltration is about 5 years; without infiltration this would be reduced to 18 months. The influent wastewater BOD is extremely low at 60 mg l^{-1}, however, and one alternative would be to modify the pond into a full biological treatment system. If loading on the first FP is limited to 40 kg BOD ha^{-1} day^{-1}, an area of 825 m^2 is required. The existing pond is 2 m deep, giving an HRT of 30 days; if desired this could be reduced to 1 m, as heavy earth-moving equipment and spoil materials are available on site. Using the model and climate details above, a SMP with once-per-year discharge would require an area of approximately 6600 m^2. An FP and SMP could be constructed by subdividing the existing pond: in practice it would be preferable to add more FPs. A small river runs through the site, with high seasonal variations in its flow. The treated water could potentially be discharged to the river in late summer and autumn, when effluent quality is expected to be high, to support increased biodiversity and provide some aquifer recharge.

Case study 2. The study considers a similar plant in central Kazakhstan, where mean monthly temperatures range from -9 °C in January to +28 °C in July. The region is arid, with average annual precipitation less than 150 mm and evaporation around 1000 mm. The potable water supply comes from boreholes located several kilometres from the plant. Wastewater flows generated by on-site accommodation and offices for about 1000 staff are officially reported as 245,000 m^3 $year^{-1}$, but flow measurement suggests a real value of around 45,000 m^3 $year^{-1}$. Wastewater is discharged into unlined holding ponds each 40 m x 60 m and 1.5 m deep: when one pond becomes full another is excavated. The company wishes to upgrade its wastewater management to provide an effective treatment system. Treated wastewater could be used to grow vegetables for on-site consumption, as the site is remote and deliveries are logistically complex. Microbiological quality is therefore critical, and it would be relatively simple to modify the existing layout to a series of ponds with retention times of approximately 30 days. The effect of seasonal variations in HRT due to evaporation is insignificant compared with variations in k_{FC}. The system could generate sufficient water to irrigate approx 2 ha under local conditions. Construction costs for a lined pond system are comparable with those for a package plant, while operation and maintenance should be simpler.

Case study 3. This study concerns a large process plant in western Kazakhstan, where the climate is modified by the nearby Caspian Sea, and temperatures are above 10 °C for 170-180 days per year. Workers living on the site produce domestic wastewater flows of 1200 m^3 day^{-1}, which are discharged without pre-treatment to a system of evaporation ponds. The system consists of two components: a single pond of 25 ha and 1 m depth, which began operation in 1985; and a 5 ha pond divided into four sections of 1.5 m depth, completed in 1999. Water depths in summer are typically no more than a few tens of centimetres. The local EPD regards the ponds as a means of disposal rather than a treatment system: stringent discharge standards are imposed and, since there is no outlet, these are assessed against samples taken from the inlet to the pond system. As a result, a number of parameters routinely exceed permitted concentrations. The system could be redesigned as above, to produce a flow of treated wastewater for on-site irrigation in the summer months; the creation of a defined discharge point might also allow re-negotiation of the discharge consent on a more rational basis. At present water for domestic use is piped 600 km from the Volga river, treated to potable standard and stored: some is used for site irrigation, and reuse would therefore represent a significant saving. In view of the high cost of water, however, and the existence of other year-round uses for it on the site, in this case it may be more effective to provide a conventional mechanical-biological treatment plant. In addition the plant produces 500 m^3 day^{-1} of heavily contaminated water, which is piped to separate treatment plant; and 1700 m^3 day^{-1} of process water and storm drainage, sent to evaporation ponds. The evaporation ponds are undersized and cause periodic flooding: these too could be re-designed, but present a more difficult case for reuse because of potential chemical contamination.

CONCLUSIONS

Simply replacing evaporation ponds with intermittent discharge WSP systems may make a large proportion (70% or more) of wastewater flows potentially available for reuse. Standard designs and operating regimes for these systems are known to work well in cold climates, although the range of reuse applications may be limited by discharge timing. The guidelines may however be conservative for continental climates, where high levels of performance in the warm period may make it possible to reduce surface areas, leading to lower evaporation losses. Results from a simple model give at least qualitative support to these ideas. More complex models exist, but their use is limited by a lack of parameter and coefficient values for lower temperature ranges, and of fundamental understanding of the processes occurring in ponds subject to strong seasonal variation. Currently available information is not reliable enough as a basis for design, but strongly suggests that there may be advantages in developing design and operating protocols specifically for continental climate systems. Year-to-year variability in continental climates may mean that robust systems are needed, such as alternating storage ponds; in this case minimising surface area is vital to reduce losses and maximise the availability of water for reuse. Case studies support the idea that such changes could make an impact in practice.

ACKNOWLEDGEMENTS

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Section 7: Use of a simple model to provide guidelines for waste stabilisation pond performance in continental climate conditions

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Keywords

Continental climate, waste stabilisation ponds, design guidelines

INTRODUCTION

Climate is a key factor in the performance of waste stabilisation ponds (WSPs). In tropical and warm temperate regions pond systems are well known to operate effectively throughout the year, at temperatures capable of supporting the algal growth and oxygen production that stimulate high rates of microbially-mediated BOD removal. In cold and continental climate regions, low winter temperatures combined with reduced light intensity slow down algal growth and reduce the rate of degradation of biochemical oxygen demand (BOD). BOD thus tends to accumulate over the winter period, and pond systems must be operated on an intermittent basis, only discharging when the rate of breakdown exceeds that of accumulation and any excess BOD has been removed. In each case the key reactions are highly temperature-dependent: in long retention systems, this tends to result in seasonal and climate dependency as processes are influenced more by ambient air temperature than the temperature of incoming wastewater. In certain regions other climatic phenomena such as precipitation and evaporation also have a significant effect: Ensink *et al.* (in review) note the impact of climatic conditions on WSP effluent quality and the importance of taking this into consideration for successful design and operation. Climate data have been monitored for long periods worldwide: temperature and precipitation are often available on a daily basis, in some cases for 100 years or more, even in sparsely populated and economically underdeveloped regions. The current work aimed to investigate the potential for using climatic data of this type to indicate design and operating guidelines. As part of this work it was necessary to investigate how climate data are related to pond performance for existing cold and continental climate systems which accumulate BOD in the winter period. Data from intermittent discharge WSP systems are scarce, however, perhaps for the simple reason that they are rarely monitored apart from when the pond is actually discharging. The paper reviews some of the available data and uses the results in a simple model for the prediction of WSP performance, using the central Asian region as a case study.

MODEL CONSTRUCTION AND ASSUMPTIONS

The model is based on that described in Heaven *et al.* (2005), and simulates a WSP system consisting of a facultative pond (FP) and a storage/maturation pond (SMP). The original spreadsheet-based model was extended using a Microsoft Visual Basic program to allow automated analysis with multi-site and multi-year sequences of climate data.

The model calculates mass balances for wastewater volumes, salinity (or any other conserved component) and biochemical oxygen demand (BOD) or any similar degradable component, using a one-day time-step. Wastewater volumes are calculated taking into account inflow, outflow, evaporation and precipitation and assuming no infiltration. The ponds are assumed to be simple rectangles in plan, with no allowance for variation of area with depth and side slope. BOD concentrations are calculated assuming first-order decay kinetics. The decay constant k is assumed to follow an Arrhenius equation of the form $k_T = k_{20} \times i^{(T-20)}$, where i is the Arrhenius constant and k_T and k_{20} are values of k at temperatures of T °C and 20 °C respectively. This relationship is assumed to hold down to a cut-off value of T which can be specified by the user; below this value, the user may specify a second invariant decay constant *f*. Different values can be specified for the FP decay constant *FPk* and the SMP decay constant *SMPk*.

The FP is sized according to the areal loading rate (USEPA, 1983), by specifying a BOD surface loading rate and a working depth, and thus fixing the surface area, volume and mean hydraulic retention time for a given inflow and influent BOD concentration. Once the surface area is known, daily and total outflows are calculated based on inflow minus evaporation and precipitation. The mass of BOD in the pond is calculated based on the initial value, inputs, decay and discharge, and daily effluent concentrations are obtained by dividing the total mass of BOD by the pond volume. The same calculation is carried out for salinity, but without a decay component.

The design of the SMP is defined by choosing a maximum and minimum working depth and a discharge condition. The four possible conditions defining the discharge period are: start and end date specified, start BOD concentration and number of days for discharge specified; end BOD concentration and number of days for discharge specified; and start and end BOD concentration specified. The depth of the SMP is taken to be at its minimum value when the end-of-discharge condition is met. Outflow from the SMP is equal to inflow (corresponding to outflow from the FP), minus evaporation and plus precipitation. As precipitation and evaporation inputs depend on the surface area, any change in area alters the maximum volume to be stored in the SMP. In order to establish the required area for a given maximum and minimum depth, the model is run with an initial estimate of area. If the calculated pond depth at any point in the simulation is greater than the maximum value, the area is incremented and the calculations repeated. This process is iterated until the entire input climate dataset can be analysed without exceeding the maximum depth. Alternatively a fixed area can be specified, in which case the depth is allowed to vary. The daily outflow from the SMP is calculated as the volume at the start of each day divided by the number of days remaining over which it is to be emptied. As evaporation and precipitation vary each day, the amount to be discharged also varies and needs to be recalculated on a daily basis. Daily values are then used to calculate pond depth and effluent BOD concentrations.

The model was used to simulate a 'standard' pond system with a wastewater inflow of 1000 m^3 day^{-1} at a BOD concentration of 200 mg l^{-1}. The standard design was based on a working depth of 1 m for the FP and maximum and minimum depths of 2.5 and 0.5 m for the SMP. The FP surface loading rate was set at 40 kg BOD ha^{-1} day^{-1}.

Climate data. Three types of climate data were used in the modelling: average values for grid points on the earth's surface, average values from specific meteorological stations, and daily values from specific stations. Average grid-point values were taken from the Intergovernmental Panel on Climate Change (IPCC) data sets for minimum and maximum temperatures, cloud cover, wind speed, precipitation and vapour pressure (IPCC, 2006). Data are presented for the majority of the earth's land surface in a 0.5° x 0.5° grid, at monthly intervals, based on average station values over the period 1961 to 1990. These data were used to calculate daily values. For temperature, daily maximum and minimum temperatures are interpolated separately using a sine function applied to the maximum and minimum monthly values. The daily average temperature is taken as the mean of the calculated daily maximum and minimum. Daily values for wind speed, rainfall and vapour pressure are calculated by interpolation between monthly values, assuming the monthly value to be that on the middle day of the month. Daily values are calculated by linear interpolation between successive monthly values. Monthly figures for sunshine hours are calculated from the cloud cover and sunrise/sunset period based on time of year and solar declination according to latitude. Daily values are then calculated based on interpolated values between the months, as for wind speed, rainfall and vapour pressure. Relative humidity is based in temperature and vapour pressures. Daily saturated vapour pressures (maximum and minimum) are calculated based on the daily maximum and minimum temperature. Daily relative humidity values are then obtained by dividing the vapour pressure by the saturated vapour pressure for maximum and minimum separately. Using these values and interpolated wind speed, temperature and latitude, daily figures for evapotranspiration from an open water surface (ETw) are derived using the Penman-Monteith equation (Allen *et al.*, 1998).

Average monthly climate data for specific stations, based on data from the World Meteorological Organisation (WMO) 1961-1990 Global Climate Normals as compiled by the US National Climatic Data Center (NCDC), were taken from the Hong Kong Observatory website (HKO, 2006). Daily records of temperature and precipitation for specific sites in the US were taken from the US NCDC (NCDC, 2006). Long series records of temperature and precipitation for cities across the central Asian region were taken from the archive of the All-Russia Research Institute of Hydrometeorological Information - World Data Centre (RIHMI-WDC, 2006). These datasets consist of daily records, in some cases starting from the 1880s, but with some years missing or only partially complete. As the model requires data for complete years, any years with missing periods of more than five continuous days were eliminated. Where data were missing for five days or less, temperature values were interpolated from adjacent days. Missing precipitation values were assumed to be zero. Water temperature was assumed to equal a 5-day lagged moving average of daily air temperature down to 0 °C and to remain at zero for lower air temperatures.

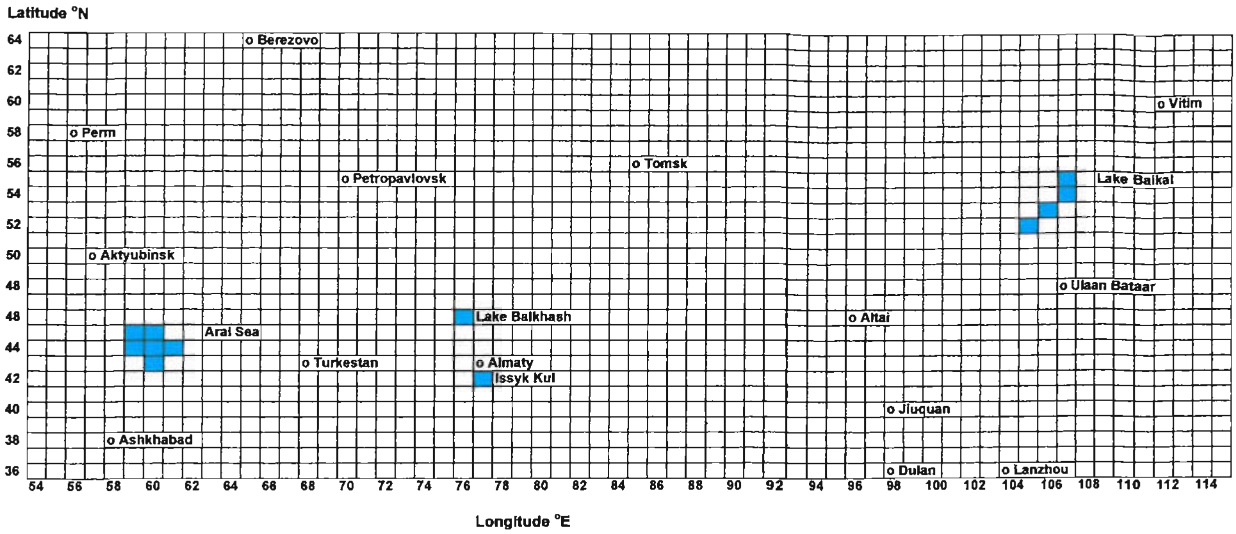
RESULTS AND DISCUSSION

Climatic data. Climate data for use in pond modelling at one-degree grid spacings from 36°N 54°E to 64°N 116°E were generated as described above. Figure 1a shows the area as a rectangular grid with locations of large water bodies and some cities, while Figure 1b shows a general image of the region taken from Google Earth (http://earth.google.com). The generated values were not tested extensively against other data sets, since effectively there is no source of independent data: the IPCC database is generated from station climatological normals. Spot checks were carried out for a number of locations, however, by comparing monthly average climate data from WMO stations with generated data at the nearest grid point. Results for 14 stations across the region are given in Table 1 with some examples in Figure 2. In general the model input data corresponds well to the nearby WMO station data, although a slight temperature lag can be seen in spring. The agreement is less good for sites at high altitude, particularly for precipitation, although this may attributed to strong local variations in rainfall in mountainous terrain.

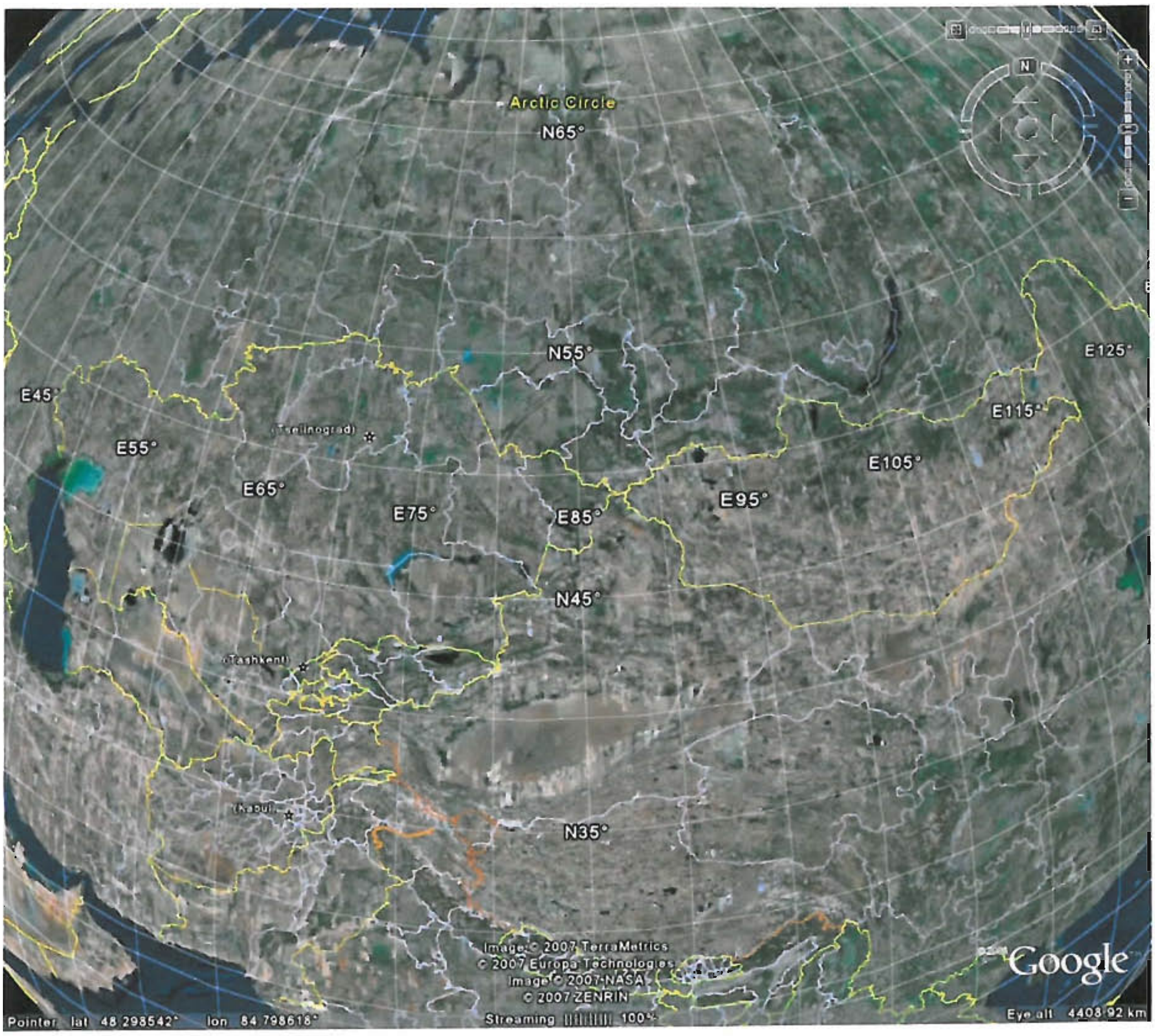
Table 1 Correlation between WMO station and nearest gridpoint values for selected sites

Station	Country	N	E	altitude m	Mean monthly Temp R^2	Mean monthly Precip R^2
Lanzhou	China	36.1	103.9	1517	0.977	0.975
Dulan	China	36.3	98.1	3191	0.987	0.913
Ashkhabad	Turkmenistan	38.0	58.4	208	0.998	0.986
Jiuquan	China	39.8	98.5	1477	0.987	0.986
Almaty	Kazakhstan	43.2	76.9	851	0.994	0.927
Turkestan	Kazakhstan	43.3	68.2	207	0.990	0.986
Altai	Mongolia	46.4	96.3	2181	0.996	0.992
Ulaan Bataar	Mongolia	47.9	106.9	1306	0.985	0.985
Aktyubinsk	Kazakhstan	50.3	57.2	219	0.989	0.901
Petropavlovsk	Kazakhstan	54.8	69.2	142	0.993	0.972
Tomsk	Russia	56.5	84.9	139	0.998	0.967
Perm	Russia	58.0	56.2	170	0.995	0.973
Vitim	Russia	59.5	112.6	190	0.995	0.957
Berezovo	Russia	63.9	65.1	32	0.992	0.991

Figure 3 shows the mean January and July temperatures, annual precipitation and theoretical open water surface evaporation in the trial area.

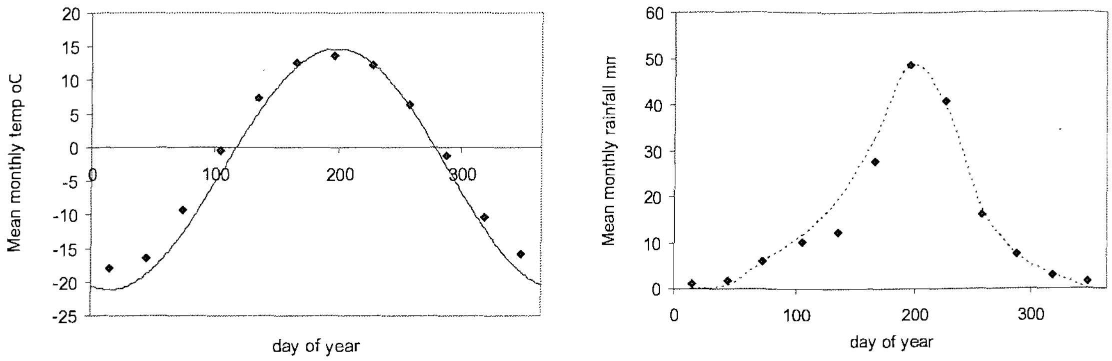


a) Study area represented on a rectangular grid, with selected cities and major water bodies

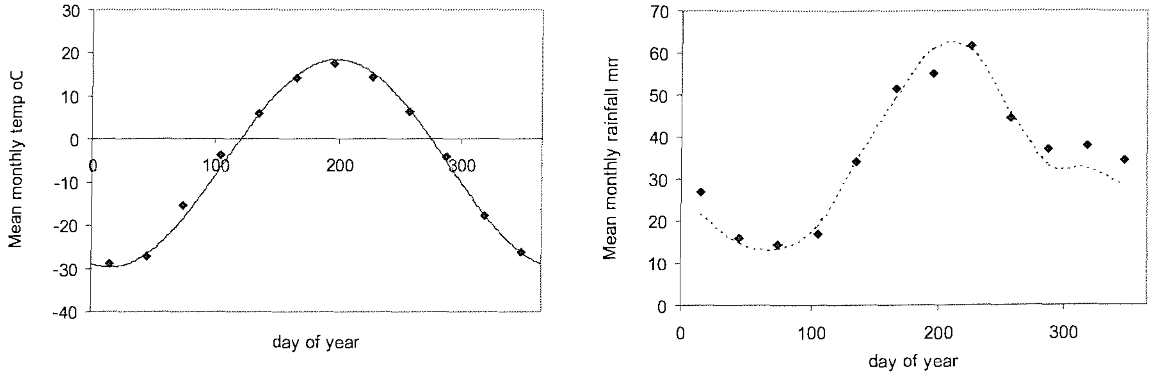


b) General view of study region (taken from Google Earth (http://earth.google.com), April 2007

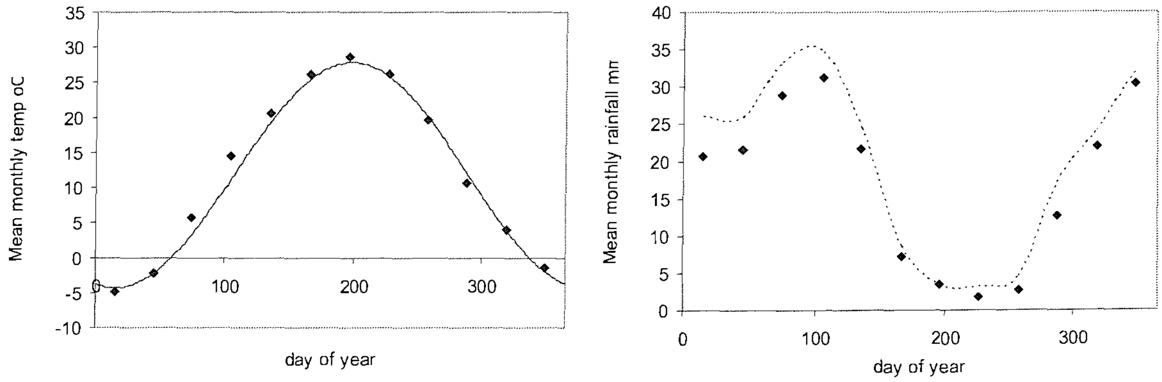
Figure 1 Location of study area



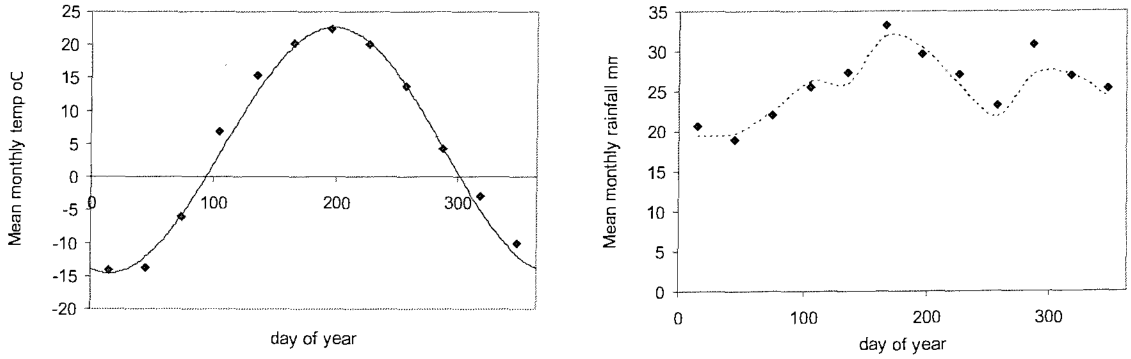
Altai, China (36.1°N 103.9°E 1517 m)



Vitim, Russia (59.5°N 112.6°E 190 m)

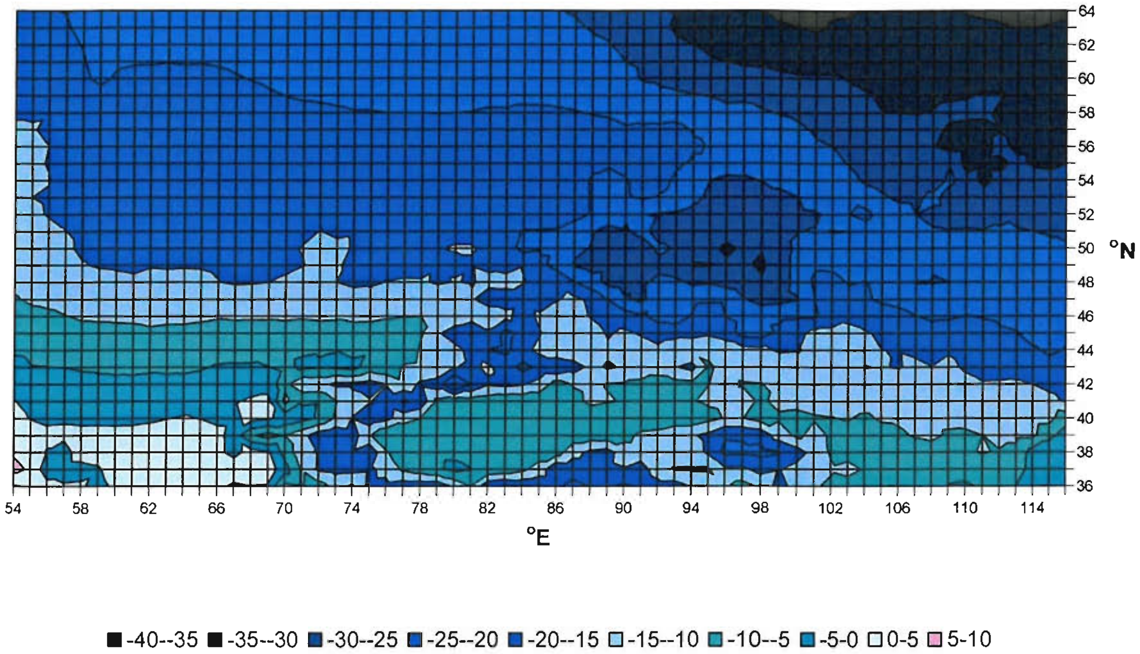


Turkestan, Kazakhstan (43.3°N 68.2°E 207 m)

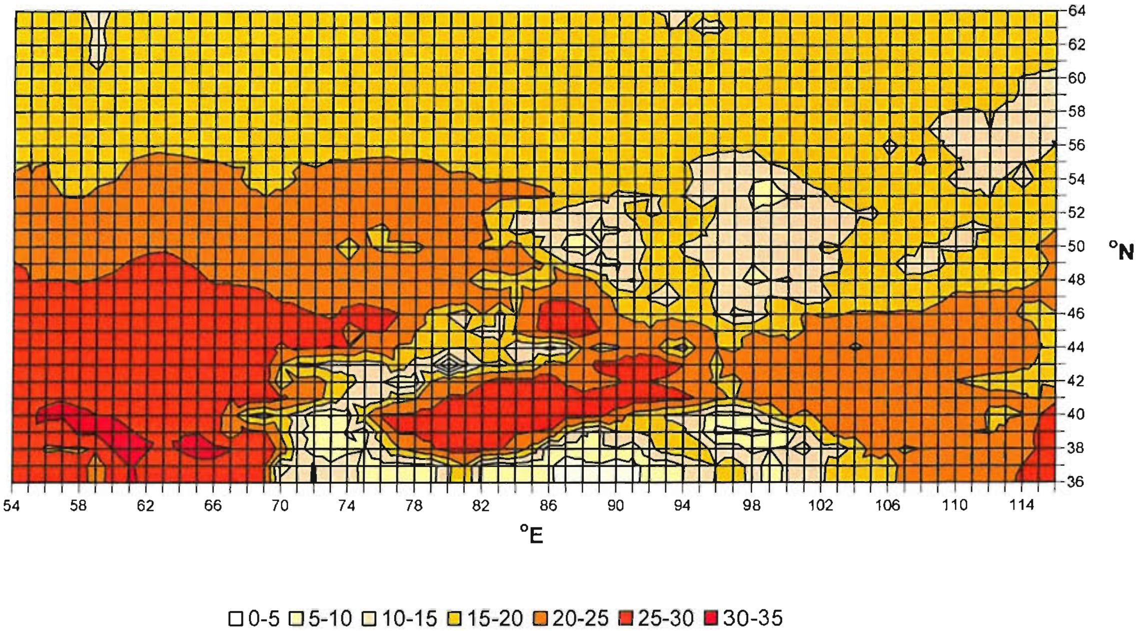


Aktyubinsk, Kazakhstan (50.3°N 57.2°E 219 m)

Figure 2 WMO station data and modelling input for selected stations

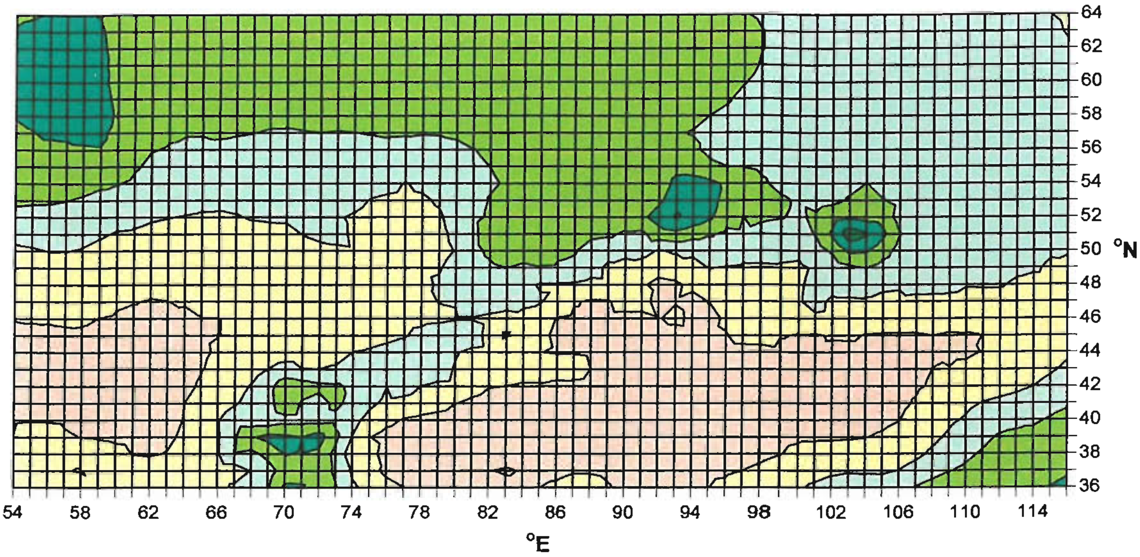


a) Average January temperature, °C



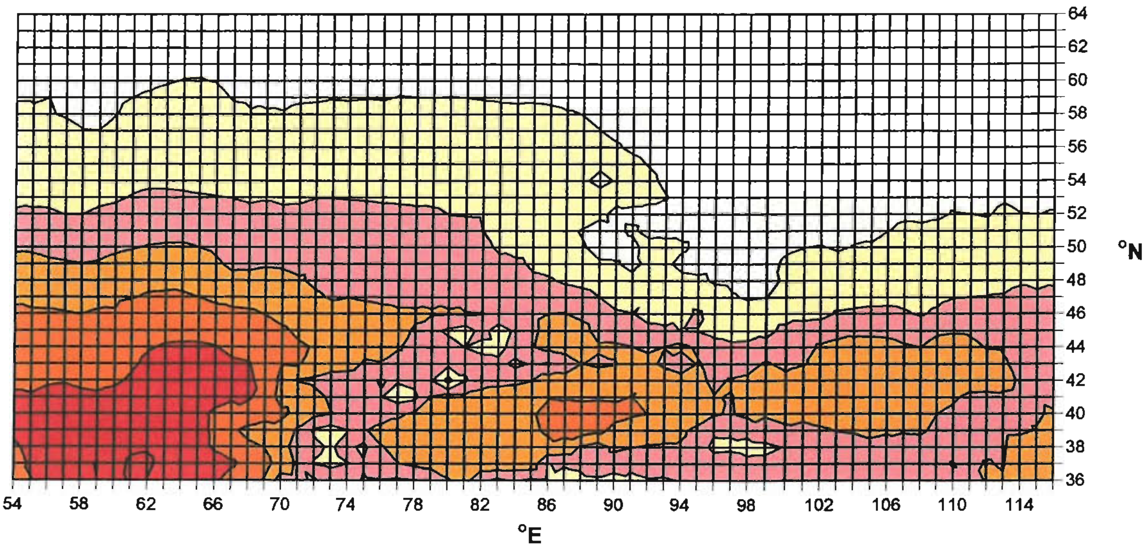
b) Average July temperature, °C

Figure 3 Temperature, precipitation and ETw for study area



□0-150 □150-300 □300-450 □450-600 □600-750 □750-900

c) Average annual precipitation, mm year $^{-1}$



□400-600 □600-800 □800-1000 □1000-1200 □1200-1400 □1400-1600

d) Calculated average annual ETw, mm year^{-1}

Figure 3 (continued) Temperature, precipitation and ETw for study area

Model calibration and verification

Calibration data. The data for model calibration and verification came from a range of locations and types of treatment plant, and was gathered over a 50-year period, making some assumptions necessary. The most extensive survey carried out to date is that reported in Middlebrooks *et al.* (1982) and in the US EPA design manual (USEPA 1983). This involved monitoring of ponds at Kilmichael, Mississippi; Eudora, Kansas; Corinne, Utah; and Peterborough, New Hampshire. The first two sites were apparently not subject to freezing, with the lowest reported temperatures in the study period being respectively 9 ^{o}C (December) and 3 ^{o}C (February), and were therefore not used in the present work. The Corinne system consists of seven cells in series, with a working depth of 1.2 m and a surface area of 1.49 ha for the first cell. The influent varied considerably through the study period, with BOD_5 ranging from 40-140 mg l^{-1} and hydraulic retention times (HRT) from 17.6 to 63.7 days, giving loading rates from 19 - 60 kg BOD_5 ha^{-1} day^{-1} on the first cell and 7.3 - 23.2 kg BOD_5 ha^{-1} day^{-1} on the whole system. The Peterborough plant has three cells in series with a working depth of 1.2 m and areas of 3.4, 2.3 and 2.6 ha. Loading rates on the first cell varied from 31.1 - 49.4 kg BOD_5 ha^{-1} day^{-1}. Peterborough is 90 km from the Atlantic coast and receives an average of 1130 mm precipitation, making it untypical of a continental climate. As monthly figures for flow and loading rates varied at these sites, intermediate values were calculated by linear interpolation.

Neel *et al.* (1961) carried out a detailed study of an experimental WSP system in Fayette, Missouri, USA. The system consisted of five parallel facultative ponds each with a working depth of 0.75 m and a surface area of approximately 0.3 ha, operated at loading rates of 22.8, 45.4, 68.1, 90.9, and 113.4 kg BOD_5 ha^{-1} day^{-1}. These emptied into a sixth pond with a working depth of 0.91 m and a surface area of 6.1 ha, which also received some raw sewage and was therefore not completely typical of a maturation pond. Fayette is close to the southern limit of the area in which ponds freeze in winter, with an average January temperature of around -3.5 ^{o}C: in the study period, however, ice cover was noted on ponds 1-5 more or less continuously from mid December until late February. It is assumed that BOD values quoted are for unfiltered samples, in which case late summer increases may represent algal BOD.

Mackenthun and McNabb (1961) studied three WSP systems in Wisconsin. Of these, the Junction City unit showed consistently low effluent BOD_5 concentrations in the first part of the study period, with no strong seasonal variation. In the second half of the study there was a major increase in load when milk wastes were added to the plant. Insufficient detail is given to determine weekly or monthly loading rates, but the available data suggest these may have been highly variable. This plant was therefore not used in calibration or verification. The WSP at New Auburn consisted of a single pond of 1.1 ha and 1.5 m depth, which was not discharged during the study period. It had a theoretical HRT of 106 days and a loading rate of 31.1 kg BOD_5 ha^{-1} day^{-1}. The plant at Spooner was also a single pond with a surface area of 8.9 ha, depth 1.5 m, HRT 125 days and a loading rate of 36.9 kg BOD_5 ha^{-1} day^{-1}. No information is given on the discharge mode, which was therefore assumed to be continuous, but the data provided suggests that the pond may have experienced some variation in influent concentrations and volumes during the study.

Oleszkiewicz and Sparling (1987) carried out studies on a number of WSP and aerated lagoon systems in Manitoba, Canada, the most relevant and detailed of which took place in Portage La Prairie. The date of the study is not given, but based on weather records it appears likely to be 1983-4. The system comprised two ponds in series. The ponds were sampled every 8-10 days from November to April, making this a valuable record of winter performance. Both ponds show very high values for effluent BOD_5 on a single date in late February, of 520 and 390 mg l^{-1} respectively, greater than the quoted average influent concentration of 381 mg l^{-1}. This could be due to several factors including variable influent BOD_5, concentration of pollutants in the water remaining unfrozen under a thickening ice layer, or analytical error. These two points were therefore omitted for the purposes of model calibration and verification. There is no information on the average inflow or hydraulic retention time, so these were calculated using surface loading rates and pond depths given in the paper, and surface areas estimated from Google Earth (http://earth.google.com) and an aerial photograph provided by one of the authors. Precise dates for pond discharge are not given, and it was assumed that discharge occurred from 15 May to 29 November in accordance with the overall general guidelines.

Soniassey and Lemon (1986) evaluated the performance of a WSP at Yellowknife, North West Territory, Canada. The system was constructed out of a series of three small lakes, with a total surface area of 75 ha, a maximum depth of 5 m and a volume of approx 2245000 m^3. Treated wastewater was discharged from July - October, leaving a residual 2 m depth: no information is given on residual volume. The length of the main lake is approximately 10 times greater than its width, and this with its depth make it particularly unlikely that complete mixing will occur. The lagoon effluent was sampled at distance corresponding to 30%, 60% and 90% of its volume and at the outfall; for the purposes of model verification values were used from the 90% sampling point and the outfall.

Parameter selection. The model was used to simulate the ponds at Corinne, Fayette (cells 3 and 4), Peterborough (Cell 1), New Auburn, Spooner, Portage La Prairie and Yellowknife, for the conditions shown in Table 2.

Table 2 Site parameters and input data for model calibration and verification

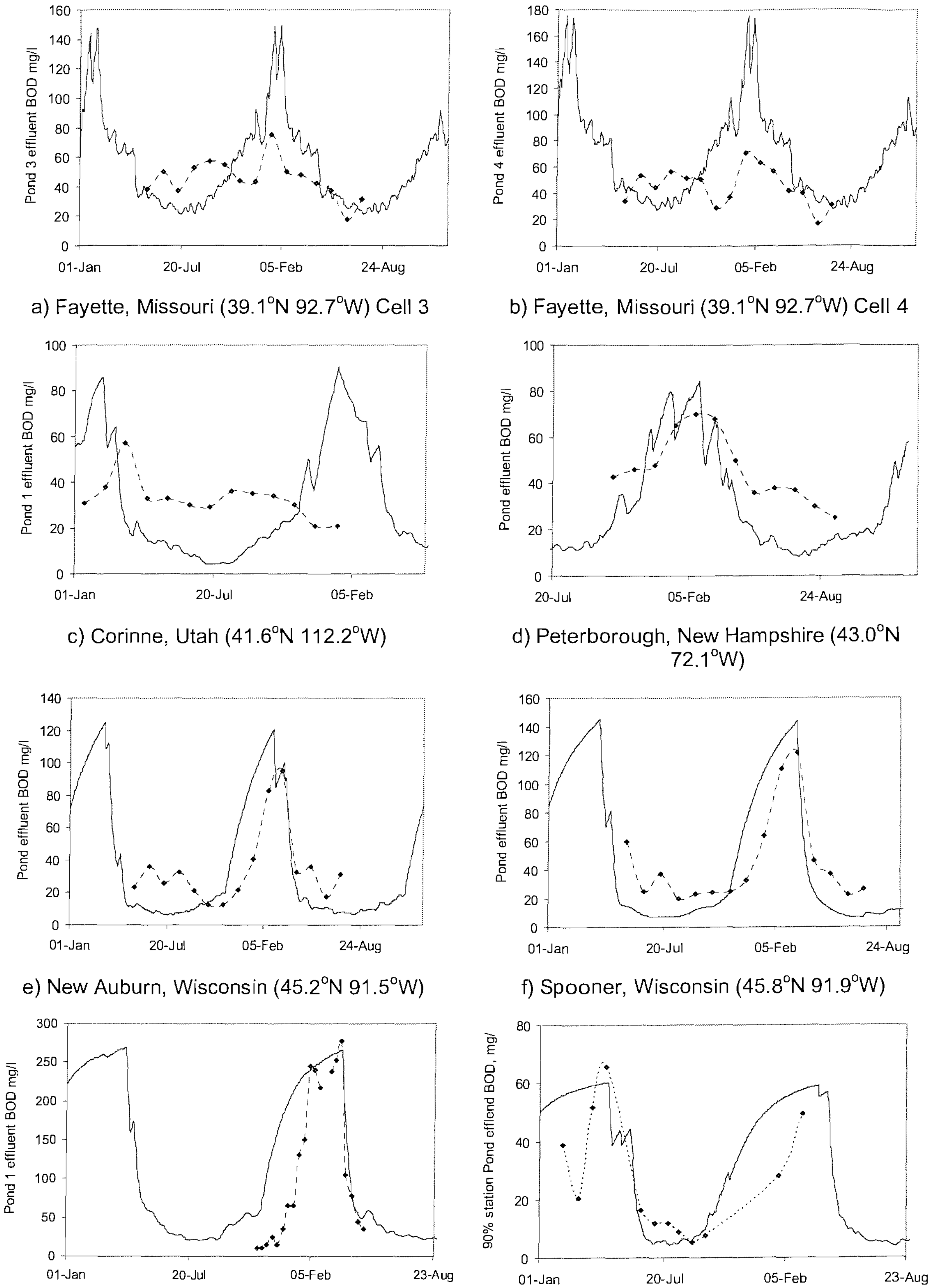
City	N	W	altitude	area	depth	inflow	BOD_5	Discharge
			m	ha	m	m^3 day^{-1}	mg l^{-1}	
Fayette	39.14	92.67	187	0.3	0.76	79	274	continuous
Corinne	41.54	112.11	1283	1.5	1.22	690	67	continuous
Peterborough	42.91	71.93	211	3.4	1.22	1011	128	continuous
New Auburn	45.21	91.55	325	1.1	1.52	106	321	none
Spooner	45.82	91.91	319	8.9	1.52	1090	302	continuous
Portage La Prairie	49.95	98.27	260	2.4	1.50	617	381	15 May - 29 Nov
Yellowknife	62.45	114.57	233	75.0	2.99	7850	98	1 July - 20 Oct

Daily air temperature and precipitation records were obtained for each site during the period in which the WSP data were recorded, while ETw was estimated as described above using average climate parameter values. Calibration was carried out assuming initial values of $FPk = 0.3$ day^{-1} and $i = 1.05$, as suggested by Mara (1976). Using these values, a suitable value for f was determined by correlation of real and model output values for peak winter BOD concentrations. The model was then run with values of FPk and i ranging from 0.2-0.5 and 1.04-1.09 respectively, and the output was checked for the effect on the peak winter concentration at values of f from 0.005-0.1. This approach was adopted as it was found that, while a very slight increase in the overall correlation could be obtained by taking $f = 0$, this gave unrealistically high BOD values in winter making the model less useful in predicting these. Correlation coefficients for real and model output were calculated for the whole data set and for spring only (defined as the period during which BOD concentrations fell sharply), and for these two data sets excluding Corinne. Correlation coefficients ranged from $R^2 = 0.605$ to 0.680 (0.851 to 0.967 without Corrine) for all data, to $R^2 = 0.783$ to 0.838 (0.924 to 09.85 without Corinne) for spring only. The correlation decreased slightly with increasing values of i, although Mara (1976) suggested that in general higher values of i might be needed where $T < 15$ °C. R^2 tended to increase with FPk although there is little difference above $FPk = 0.3$ for values of i =

1.04-1.06. On the basis of these results and in view of the limited quantity and quality of data available, values of $FPk = 0.3$, $i = 1.05$ and $f = 0.007$ with a cut-off value of T = 0 °C were adopted for modelling as being reasonably conservative and in accordance with Mara (1976). Figure 4 shows real data and model output using these parameters.

Selection of a suitable value for *SMPk* was more problematic, as none of the datasets found contained suitable information for calibration. The WSP systems at Corinne, Peterborough and Portage La Prairie all included at least two cells, but in each case the second or subsequent ponds were similar in size to the first and therefore did not provide the very long retention times characteristic of a maturation/storage pond as recommended by Prince et al (1995a or b). The Yellowknife pond system has a theoretical retention time of 286 days but its configuration is untypical and very few data points are available; while cell 6 at Fayette had a HRT of around 132 days, but received some raw sewage and showed almost no seasonal variation in effluent quality. The former Soviet Union standard for wastewater treatment suggests using $SMPk = 0.07$ day^{-1} at 20 °C for a final pond, on the basis that readily degradable components will have been oxidised and only recalcitrant materials will remain (SNiP, 1996). Mara (2005) suggests a value of 0.1 day^{-1} for secondary facultative ponds. Figure 5 shows the results for the final ponds at Portage La Prairie (HRT approx 44 days) and Yellowknife with $SMPk = 0.07$ day^{-1} and 0.1 day^{-1}. In the absence of further data a conservative value of $SMPk = 0.08$ day^{-1} at 20 °C was adopted for modelling.

Model calibration assessment and discussion. From Figure 4 a number of weaknesses in the model can be seen. For sites in the south where the mean daily air temperature in winter is not consistently below zero, the 5-day moving average used for water gives insufficient smoothing, leading to short-term fluctuations in BOD concentration in the model output. A 10-day average gives a smoother result, but must either look forward to temperatures a few days ahead, or imposes a greater time lag. Other smoothing techniques are possible. In the current application, however, the model is mainly used with daily temperatures generated from mean monthly data. This provides a smooth data set, and therefore no changes were made.



a) Fayette, Missouri (39.1°N 92.7°W) Cell 3

b) Fayette, Missouri (39.1°N 92.7°W) Cell 4

c) Corinne, Utah (41.6°N 112.2°W)

d) Peterborough, New Hampshire (43.0°N 72.1°W)

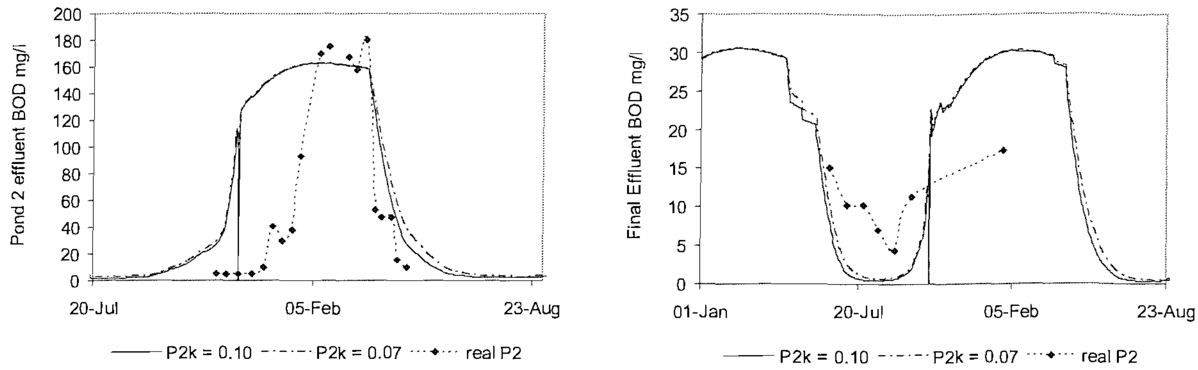
e) New Auburn, Wisconsin (45.2°N 91.5°W)

f) Spooner, Wisconsin (45.8°N 91.9°W)

g) Portage La Prairie, Canada (50.0°N 98.3°W)

h) Yellowknife, Canada (62.5°N 114.5°W)

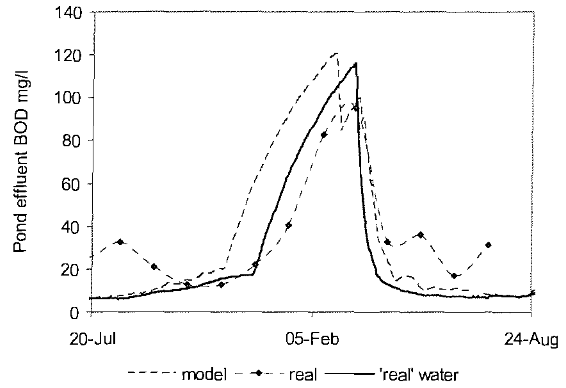
Figure 4 Real and simulated FP effluent BOD values for calibration sites



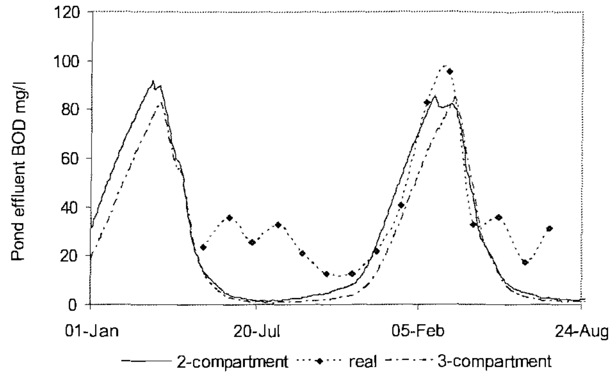
a) Portage La Prairie, Canada (50.0°N 98.3°W) b) Yellowknife, Canada (62.5°N 114.5°W)

Figure 5 Real and simulated SMP effluent BOD values for two sites

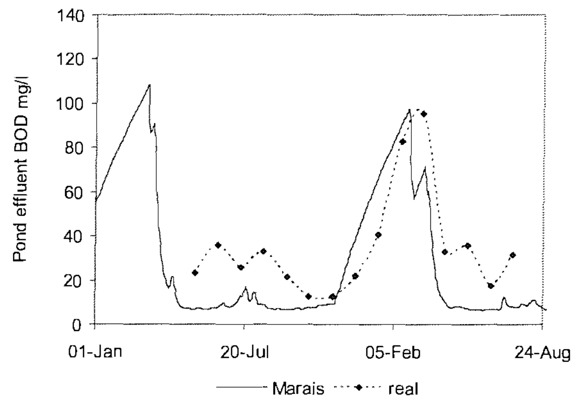
A more serious problem is the difference between real and predicted BOD values in autumn. In each case the model predicts that BOD concentrations will begin to rise earlier than actually occurs: in other words, pond performance at this time of year is better than the model suggests. This may be accounted for by two main factors. One is actual water temperature: at the end of summer the water and surrounding ground retain stored heat and may remain slightly above air temperature for several weeks. This phenomenon has been widely noted in lakes and rivers (e.g. Webb and Nobilis, 1994; Livingstone and Dokulil, 2001; Buyukalaca *et al.*, 2003), while for shallow water bodies like WSPs the resulting temperature lag may be more pronounced in autumn than in spring. The sinusoidal approximation used to generate temperatures also introduces a slight spring lag, as noted above. Some very good models exist for predicting lake water temperature and ice cover (e.g Kettle *et al.*, 1984; Gu and Stefan, 1990; Ottosson and Abrahamsson, 1998). It was decided not to attempt to incorporate these, however, firstly because many of them have much greater data requirements than the current version of the model, e.g. for daily or hourly wind speed, solar radiation and relative humidity. This information is not always available, and given the simplicity of the base pond model represents a degree of over-sophistication. In addition, for some of the modelled sites actual water temperature data are available, but these do not necessarily show a noticeable difference from the 5-day averaged air temperature in autumn. Where there is a difference, this is not always sufficient to account for the temperature lag. At New Auburn, for example, monthly values of actual water temperature were recorded and remained between 1-2 °C from mid-November to mid-January, above the model cut-off value of 0 °C. The model was run using linear interpolation between these values, which removed approximately half of the apparent lag between modelled and real BOD concentrations (Figure 6a). At other locations however, including Spooner, the second site in Wisconsin, this effect was less pronounced. The model's use of a cut-off temperature is a simplification that may not accurately describe the behaviour of individual ponds in terms of actual water temperature or BOD degradation, but provides a basis for a general application when more specific data are not available.



a) New Auburn, 'real' water temperature



b) New Auburn compartmentalised



c) New Auburn, Marais model

Figure 6 Model modifications to eliminate autumn lag

The second factor accounting for the difference between modelled and actual BOD values in autumn is likely to be the degree of mixing. The model assumes complete mixing, and arguments for doing so are presented by Marais (1970). In practice, however, this may not occur, and in this case at the end of summer the pond contains a large volume of treated water which needs to be displaced before the effect of the reduced treatment capability in autumn becomes evident. This effect can be reproduced by introducing separate completely mixed compartments into a model: Figure 6b shows model output for New Auburn with two and three compartments using daily air temperature data, FPk = 0.3 day^{-1} for the first compartment and 0.08 day^{-1} for subsequent compartments. Compartmentalisation reduces the apparent lag and may give a better fit for the winter peak, but leads to very low BOD values in summer. It also requires some justification in terms of local features that might determine mixing behaviour (e.g. length/width ratio and configuration, inflow velocity and inflow/volume ratio, wind exposure), since otherwise there is no difference between modelling a single large pond with simulated compartments and a series of small ponds, as at Corinne or Peterborough. In the interest of keeping the model general and as the current work mainly concerns spring discharge dates, it was therefore decided not to adopt the above modifications.

Low summer BOD values are a consequence of the very simple base model used for BOD decay, which does not include components for BOD contributed by benthic sediments or from the growth of algal biomass. In his classic paper on the dynamics of pond behaviour, Marais (1970) proposed a model in which a fraction of the incoming BOD is considered as settling out, and a fraction of this is subsequently solubilised and returned to the pond liquid, at a rate dependent on temperature; a further term is included for the effect of algal BOD. A version of the Marais model including the benthic contribution but without the algal correction was also tested. This gave good results at some sites: results for New Auburn are shown in Figure 6c. The Marais model is rather sensitive to the choice of coefficient defining the settleable fraction of influent BOD, however, which has a major effect on the peak winter BOD concentration. Marais suggested a coefficient value between 0.4-0.6, while values for the modelled sites varied from 0.4 to 0.8 or more for Portage La Prairie. This may reflect conditions at individual sites: for example the Portage La Prairie system was reported to receive some of its wastewater from a potato processing plant, which may have a high proportion of soluble BOD. Although the Marais model is a more elegant representation of actual processes, it was not possible to find a single coefficient value that worked well for all of the modelled sites. As the aim of this work was to suggest a model that could be generally applied, the approach of using a single f value was therefore preferred.

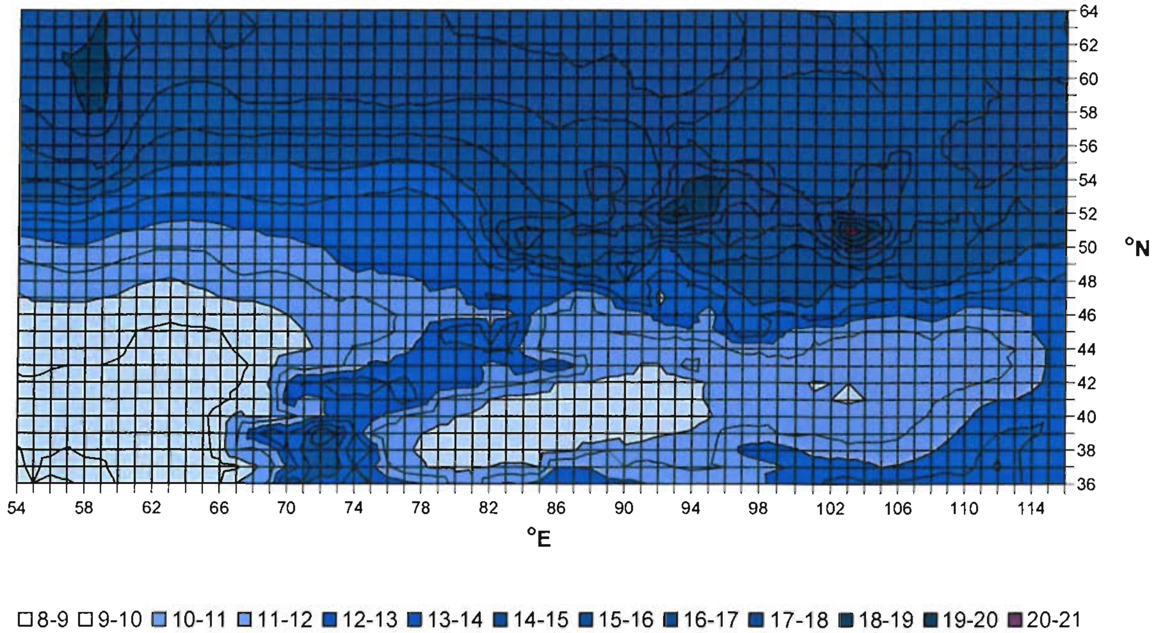
Modelling

The model was run with the standard pond configuration to simulate two design options. The first was based on maximising the length of the discharge period, by discharging from the first day when the BOD concentration in the effluent from the storage/maturation pond reaches a selected value to the last day before it rises above this value (i.e. option with start and end BOD concentration specified). For a two-pond system where both ponds are in continuous operation, this minimises pond size while maximising the potential for water reuse. The second scenario was based on discharging for a 21-day period at the last date before the effluent BOD concentration rises above a specified value (i.e. end concentration and duration specified); this option simulates the classic north American operating protocol of a single autumn discharge. For the maximum discharge period the threshold BOD concentrations were set at 15 mg l^{-1}, corresponding approximately to a 20 mg l^{-1} discharge standard; while for the 21-day discharge option the threshold was chosen as 5 mg l^{-1} to simulate a period when the effluent quality is at its highest (Prince *et al.*, 1995 a and b).

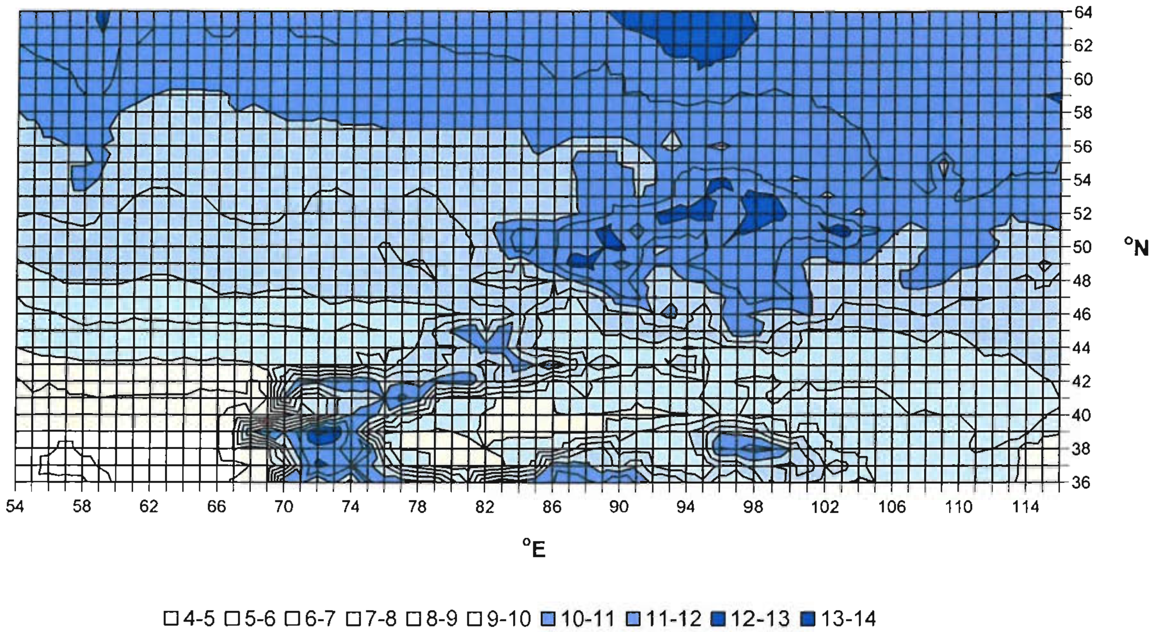
Simulation of ETw at points in large water bodies produces artificially high values, and this affects several of the modelled output values. Points affected were Lake Balkhash in Kazakhstan, Issyk Kul in Kyrgyzstan, Lake Baikal in Russia, and the Aral Sea (see Figure 1). To provide a continuous contour map, output data for these points were replaced by values generated by linear interpolation from neighbouring grid points. Grid point 42°N 80°E lies on top of a mountain at altitude 5840 m, and while the model results may be valid they make graphical presentation difficult, so this point was also replaced by linear interpolation.

Regional output. Figure 7 shows the required area of the storage/maturation pond for the two options. In the north of the area studied this may be up to 4.5 times larger for the 21-day discharge than for the long-discharge option, reflecting the greater precipitation and lower evaporation. Where an SMP is constructed out of a natural lake or depression, area may not be a critical consideration. For a lined and engineered pond, however, there are significant cost implications. Lined systems are already required in the former Soviet Union, and small communities of the type that could be served by pond systems in central Asia are likely to need an engineered solution.

The ratio between the inflow and outflow from the pond system is shown in Figure 8. In the north-west quadrant, corresponding to Russia and northern Kazakhstan, there is relatively little difference between the options: the maximum ratio for the 21-day option

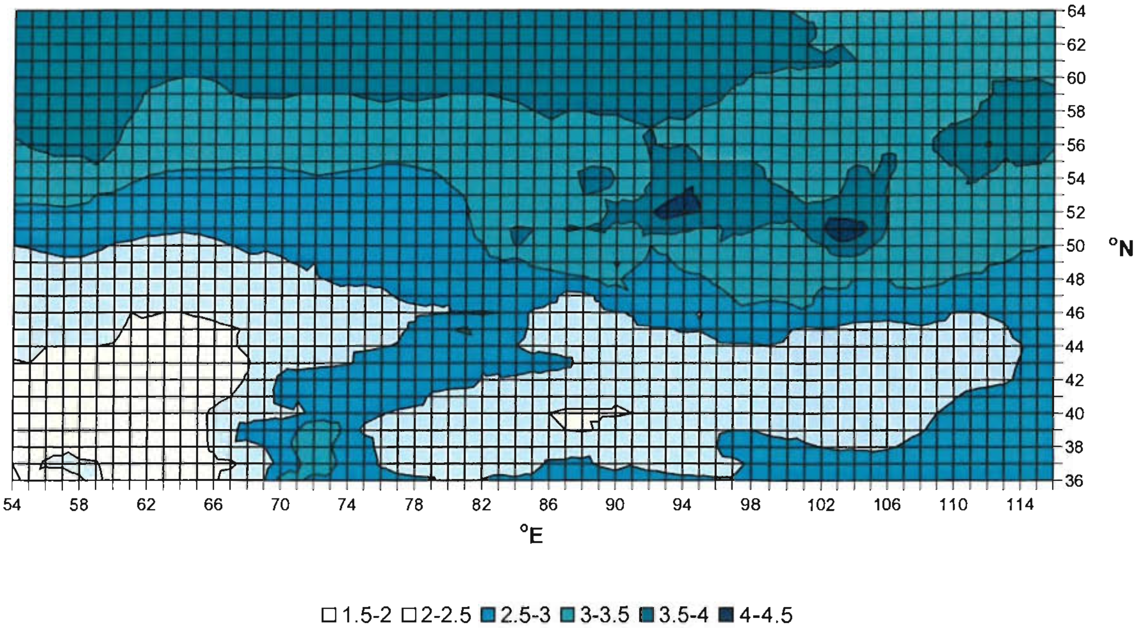


a) SMP area for 21-day discharge (hectares)



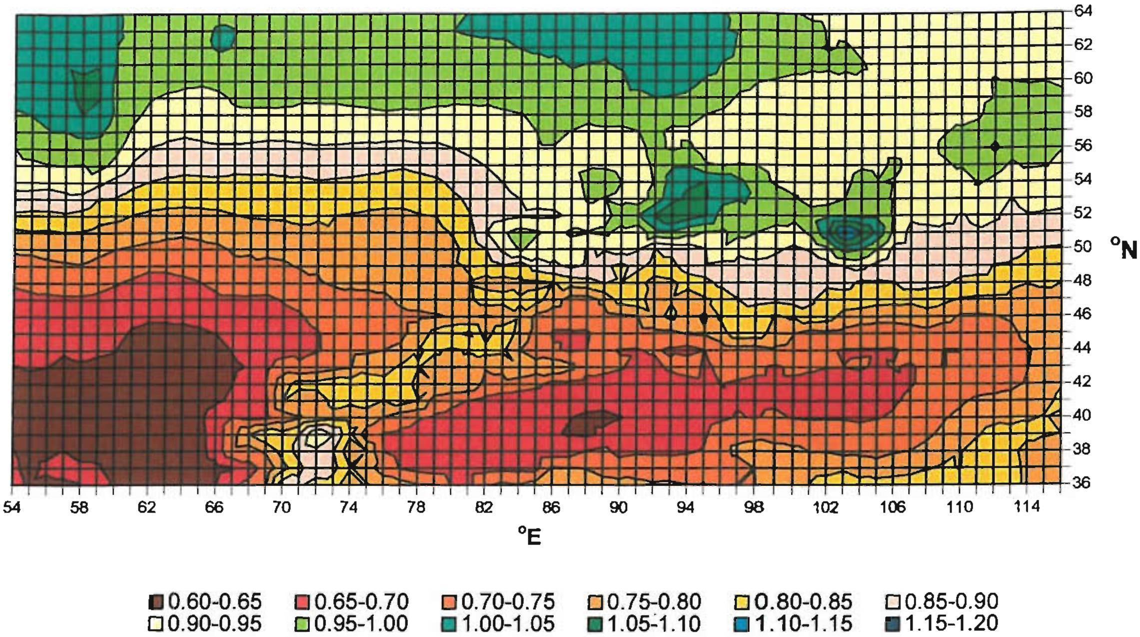
b) SMP area for long discharge (hectares)

Figure 7 Area of SMP required for 21-day and long duration designs

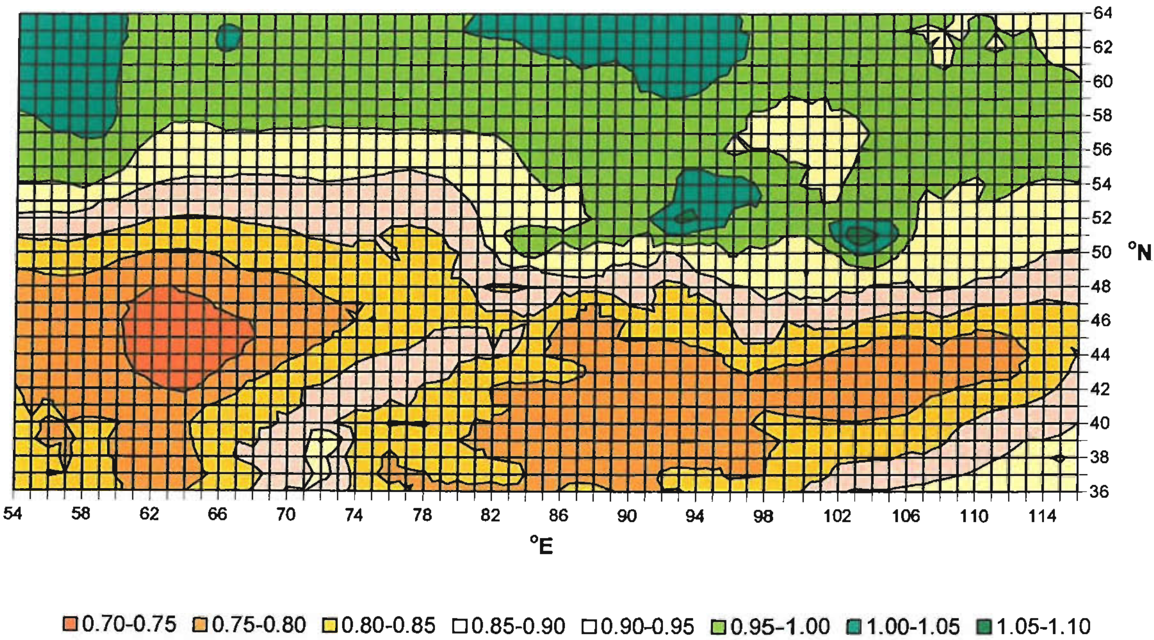


c) Ratio of SMP areas for 21-day and long discharge

Figure 7 continued Area of SMP required for 21-day and long duration designs



a) Inflow/outflow ratio for 21-day discharge



b) Inflow/outflow ratio for long discharge

Figure 8 Inflow/outflow ratio for 21-day and long discharge designs is 1.17 compared to 1.10 for the long-discharge, with the contours running 1-2 degrees further north. In the north-east differences of 5-10% appear between the two options; but communities in Siberia tend to be situated on major waterways, and with increasing latitude there is a diminishing need for water use in agriculture or elsewhere. The biggest differences occur in central Asia and areas of China, however, where there are both population centres and a scarcity of water resources leading to pressures for wastewater reuse. Significant population influxes occurring in western China are also likely to create a need for simple, robust wastewater infrastructure, and this part of the region has a tradition of re-use in aquaculture and agriculture. The larger SMP area required for the 21-day discharge option leads to greater evaporation, representing a loss of resource.

Figure 9 shows the discharge start date for the long-discharge option. In much of the south-west quadrant, up to latitude 40°N, the results suggest that ponds are able to operate in continuous discharge mode, as indicated by the earliest discharge date of 1 January. Outside the mountainous regions in the south, the latest start dates are typically in mid-June, while the overall pattern clearly follows the onset of spring in the region. Figure 10 shows the number of months per year in which a pond operating under this regime is unable to discharge. At 9-11 months the two operating modes are similar or identical, with a single short discharge period. The majority of the north-eastern quadrant, above 46°N is unable to discharge for more than half the year, whereas in the north west this restriction only occurs above latitude 60 - 62°N. An interesting comparison can be made with north American guidelines: in Canada the typical recommendation for ponds subject to freezing is to provide 12 months' storage capacity while regulations in the northern US state commonly specify 6 or 12 months. The results for the study region are clearly supportive of such approaches: intermittent discharge pond systems should not fail if constructed in accordance with guidelines of this type. The picture is at once more detailed and more complex than allowed for by a broad brush approach, however, as indicated by the intricate contours in Figure 10. Modelling based on climatic data provides a potential tool for tailoring designs much more closely to local conditions, with both cost and environmental benefits.

Figure 11 shows the salinity of the treated wastewater at the start and end of the discharge period for a long-discharge system, expressed as an index relative to an incoming value of 1. Problem areas can be clearly seen, in central Asia and on the Tibetan plateau. When

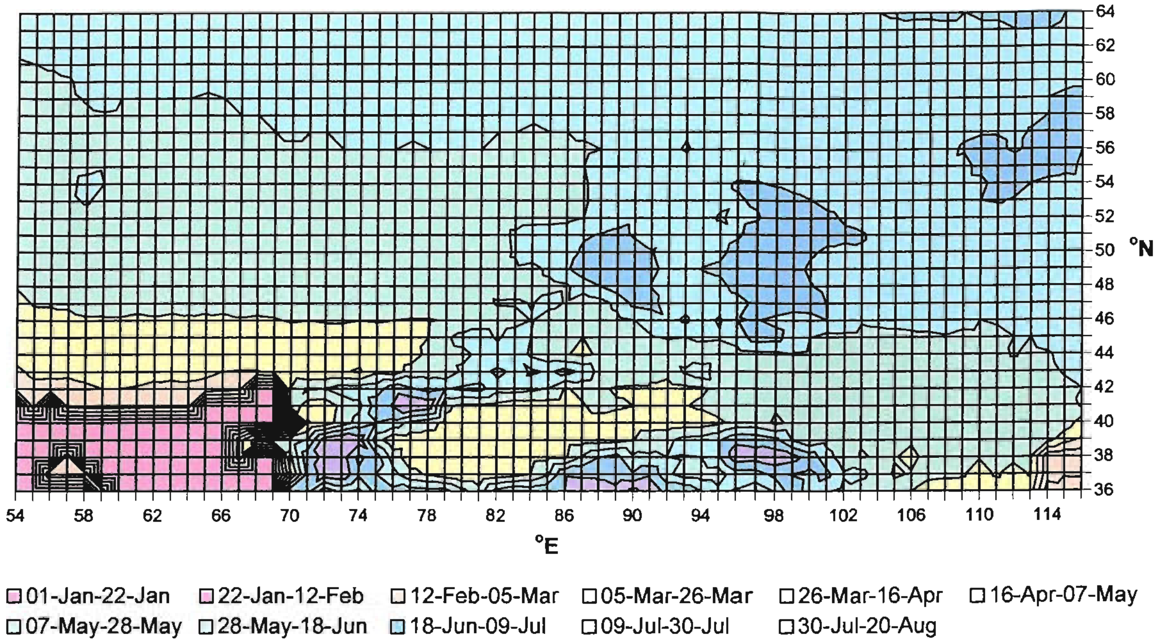


Figure 9 Discharge start date for long-discharge option

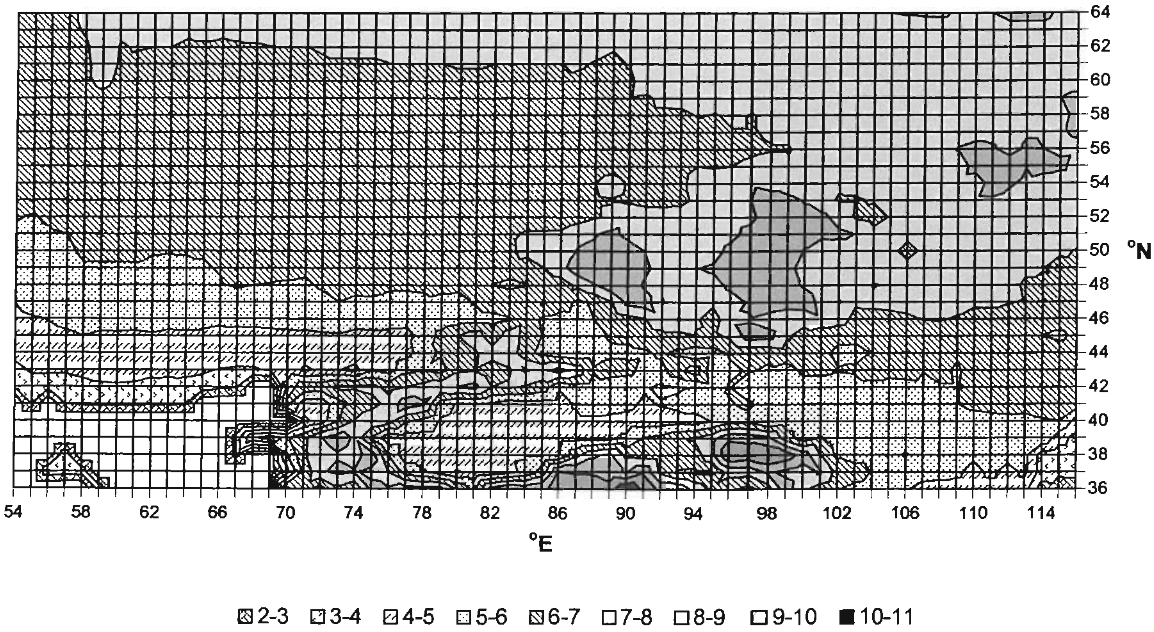
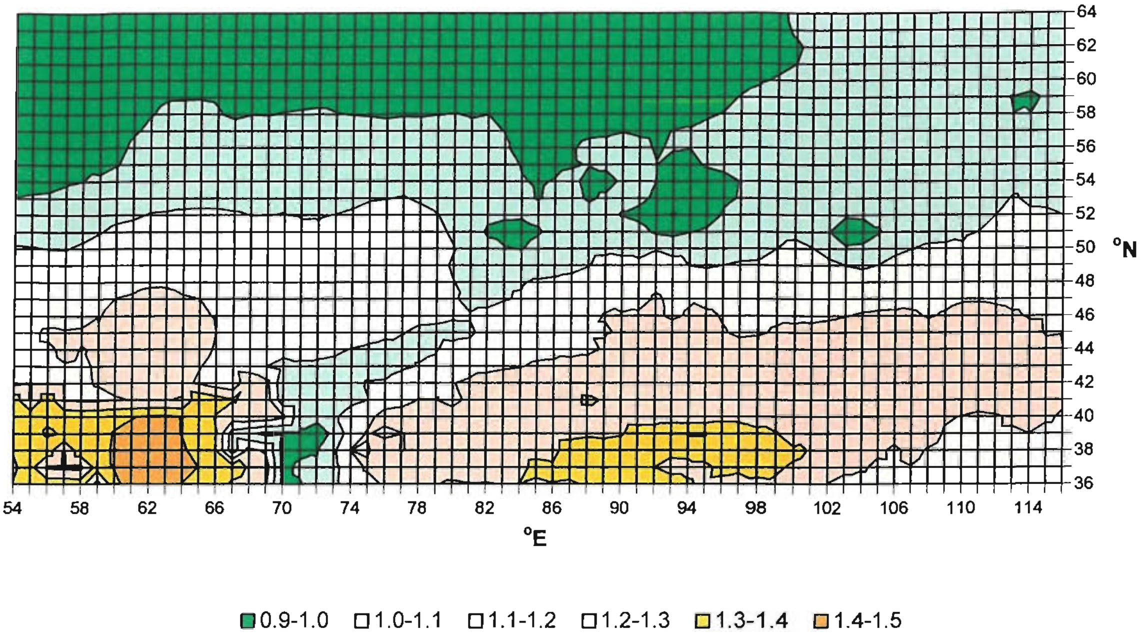
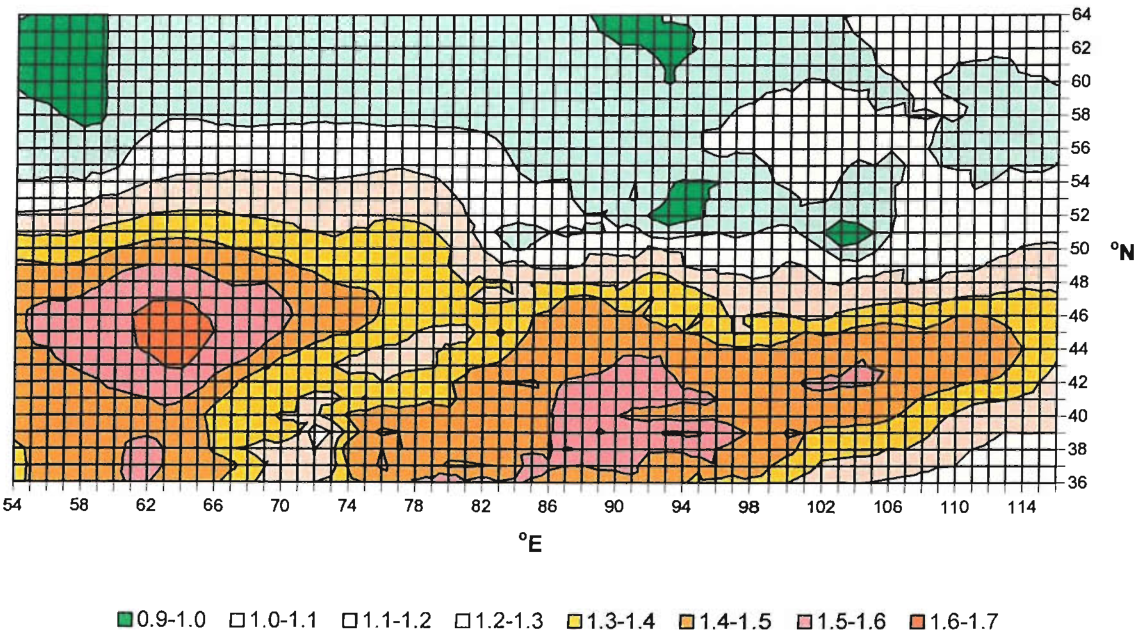


Figure 10 Number of months per year in which pond cannot discharge under long-discharge option



a) Salinity index at start of discharge period



b) Salinity index at end of discharge period

Figure 11 Salinity index at start and end of discharge period for long-discharge option (relative to incoming salinity =1) taken with the inflow/outflow ratio (Figure 8) these contour maps provide a potential means of identifying areas which are unsuitable for WSP systems from the viewpoint of water conservation.

Model drivers: Table 3 gives the correlation between some key input and output parameters for the model. For the long-discharge option, as might be expected there are very strong relationships between days less than 0 °C and the number of months in which discharge is not allowed ($R^2 = 0.96$), and with the start and end of discharge dates. There also is a 90% correlation between SMP area and mean annual temperature, which may be useful for predictive or guidance purposes. For the 21-day discharge, the balance between precipitation and evaporation (P - ETw) shows the strongest relationship with SMP area and inflow/outflow ratio, and accounts for 85-6% of the variation in salinity at the on and off dates. Because of the close linkage between parameters such as latitude, mean annual temperature, precipitation etc no attempt was made to establish a multiple regression equation for output values.

Table 3 Correlation between input parameters and output values for two scenarios

a) long discharge							
R^2	*SMP area*	*Date on*	*Salinity on*	*Date off*	*Salinity off*	*i/o ratio*	*Closure*
Latitude °N	0.5962	0.2221	0.7439	0.3284	0.6652	0.6671	0.2714
Longitude °E	0.0732	0.1607	0.0013	0.2236	0.0295	0.0264	0.1924
Mean annual temp °C	0.9059	0.7122	0.3530	0.9252	0.4310	0.4484	0.8290
Days < 0 °C	0.8524	0.8929	0.3354	0.9335	0.3240	0.3327	0.9636
Mean Jan temp °C	0.8529	0.6251	0.4088	0.8306	0.4500	0.4602	0.7341
Mean Jul temp °C	0.7032	0.6249	0.1740	0.7827	0.2735	0.2955	0.7165
Precipitation mm/year	0.4308	0.0940	0.7640	0.1402	0.8347	0.8354	0.1150
ETw mm/year	0.8936	0.6066	0.6268	0.7523	0.7304	0.7298	0.6947
P - ETw mm/year	0.8025	0.4225	0.7747	0.5389	0.8658	0.8674	0.4874
Slope	ha/unit	day/unit	unit/unit	day/unit	unit/unit	unit/unit	month/unit
Latitude °N	0.1884	2.0029	-0.0124	-1.2469	-0.0174	0.0081	0.1088
Longitude °E	0.0304	0.7838	0.0002	-0.4738	-0.0017	0.0007	0.0421
Mean annual temp °C	-0.3091	-4.7741	0.0114	2.7895	0.0186	-0.0088	-0.2531
Days < 0 °C	0.0366	0.6527	-0.0014	-0.3425	-0.0020	0.0009	0.0333
Mean Jan temp °C	-0.2231	-3.3264	0.0091	1.9640	0.0142	-0.0066	-0.1771
Mean Jul temp °C	-0.3444	-5.6563	0.0101	3.2529	0.0188	-0.0091	-0.2976
Precipitation mm/year	0.0082	0.0668	-0.0006	-0.0418	-0.0010	0.0005	0.0036
ETw mm/year	-0.0069	-0.0985	0.0003	0.0561	0.0005	-0.0003	-0.0052
P - ETw mm/year	0.0044	0.0561	-0.0003	-0.0326	-0.0004	0.0002	0.0030

Note: Closure = number of months for which discharge is not allowed

Table 3 continued

b) 21-day discharge						
R^2	*SMP area*	*Date on*	*Salinity on*	*Date off*	*Salinity off*	*i/o ratio*
Latitude °N	0.7000	0.3043	0.5678	0.0060	0.5983	0.7208
Longitude °E	0.0442	0.2198	0.0644	0.0007	0.0477	0.0337
Mean annual temp	0.6409	0.9178	0.1717	0.0079	0.6264	0.6096
Days < 0 °C	0.5529	0.9427	0.6249	0.0139	0.5715	0.5213
Mean Jan temp °C	0.6269	0.8209	0.6170	0.0008	0.5886	0.6051
Mean Jul temp °C	0.4646	0.7799	0.5428	0.0308	0.4896	0.4291
Precipitation mm/ye	0.7646	0.1174	0.6387	0.0043	0.6709	0.7837
ETw mm/year	0.8765	0.7233	0.8829	0.0006	0.8501	0.8567
P - ETw mm/year	0.9465	0.5024	0.8702	0.0001	0.8604	0.9411
Slope	ha/unit	day/unit	unit/unit	day/unit	unit/unit	unit/unit
Latitude °N	0.2785	-1.3079	-0.0279	18.9947	-0.0260	0.0125
Longitude °E	0.0322	-0.5115	-0.0043	-2.9391	-0.0034	0.0012
Mean annual temp	-0.3548	3.0237	0.0404	29.1047	0.0354	-0.0152
Days < 0 °C	0.0402	-0.3742	-0.0048	-4.7071	-0.0041	0.0017
Mean Jan temp °C	-0.2610	2.1269	0.0288	7.0878	0.0256	-0.0113
Mean Jul temp °C	-0.3821	3.5255	0.0460	72.6631	0.0396	-0.0162
Precipitation mm/ye	0.0149	-0.0417	-0.0015	0.8243	-0.0014	0.0007
ETw mm/year	-0.0092	0.0599	0.0010	0.1757	0.0009	-0.0004
P - ETw mm/year	0.0066	-0.0341	-0.0007	0.0505	-0.0006	0.0003

Sensitivity analysis: To determine the sensitivity of the results to variations in modelling coefficients, the model was run in the long-discharge operating mode with a series of values covering the expected range in *f*, *i*, *FPk* and *SMPk*. Table 4 shows the coefficient values used and Table 5 gives the results of the sensitivity analysis.

Table 4 Coefficient values used in sensitivity analysis

	Base value	*Range*	*Basis*
f	0.07	0.05 - 0.09 (0.01)	model calibration
i	1.05	1.05 - 1.09 (0.01)	Mara, 1976
FPk	0.3	0.20 - 0.40 (0.05)	Mara, 1976
SMPk	0.08	0.06 - 0.10 (0.01)	SNiP, 1996; Mara, 2005

Figures in brackets give increment size (5 increments giving a total of 20 combinations).

Table 5 Results of sensitivity analysis for variation in modelling coefficients

		SMP area	**Date on**	**Date on***	**Salinity on**	**Date off**	**Date off ***	**Salinity off**	**i/o ratio**	**Closure**
		ha/unit	day/unit	day/unit	unit/unit	day/unit	day/unit	unit/unit	%/unit	months/unit
Slope	*f*	-58.0	-1530.3	-1247.5	-2.1	281.0	221.9	-0.9	0.56	-59.6
	i	19.2	463.2	322.9	0.5	-185.4	-147.1	0.6	-0.25	18.3
	FPk	-1.7	-9.6	-6.6	0.0	29.0	22.4	-0.1	0.02	-1.3
	SMPk	-10.8	-364.2	-299.3	-0.5	14.1	10.7	-0.1	0.13	-12.4
		ha	day	day*	unit/unit	day	day*	unit/unit	unit/unit	month
Variation	*f*	-0.2	-6.1	-5.0	0.0	1.1	0.9	0.0	0.00	-0.2
across	*i*	0.8	18.5	12.9	0.0	-7.4	-5.9	0.0	-0.01	0.7
range	*FPk*	-0.3	-1.9	-1.3	0.0	5.8	4.5	0.0	0.00	-0.3
tested	*SMPk*	-0.4	-14.6	-12.0	0.0	0.6	0.4	0.0	0.01	-0.5

Dates marked * exclude values for pond systems able to discharge all year round

Table 5 shows that the SMP area is not greatly affected by any of the model coefficients across the range tested: for comparison, the variation with latitude across the area studied is 5.3 ha. The date at which discharge can begin is affected by the value of *i* and of *SMPk*, although the effect is reduced slightly when sites capable of year-round discharge are removed. Discharge end date is affected by *i* and also *FPk*. The duration of closure is principally affected by the value of *i*, but also by *FPk* and *SMPk*. The change in modelling coefficients had little effect on the correlation between input parameters and output values as shown in Table 3. In general the model seems reasonably stable, but given the limited information available to support choice of coefficient values, especially for *SMPk,* the results highlight a need for research in this area.

The sensitivity analysis also helped to identify some limits on practical pond design. With values of $i > 1.06$, problems were encountered at some high altitude locations and on the Tibetan Plateau. Between 36°N 87-91°E with $i = 1.09$, the storage/maturation pond effluent never reaches a BOD concentration below 15 mg l^{-1}. The same result occurs at 38°N 96 and 98°E. At adjacent points the pond nominally reaches this criterion, but discharge is only possible over a period of 4-6 days. In practice these areas are extremely dry and cold, with few or no inhabitants; there is unlikely to be a great need for WSP designs meeting this or any other standard. The example again highlights, however, the need for a better knowledge and understanding of applicable design coefficients and of the factors that affect them.

Single sites and long series data. In addition to offering a potential basis for regional guidelines, the modelling approach provides a powerful design tool for exploring performance and options at individual sites. As an example, Figure 12 shows outputs for FP BOD and SMP BOD, salinity index and depth, for each latitude along longitude 64E, with the discharge condition specified as a 90-day period once the SMP effluent BOD has reached 10 mg l^{-1} (i.e. start BOD concentration and discharge duration). This mode of specifying the discharge condition is particularly useful, as it simulates a typical situation where treated effluent is discharged to a water course or an irrigation system once it has reached a specified standard. It is also comparatively robust, since as noted above the agreement between simulated and real results is strongest in spring, while the end date has relatively little effect on conditions early in the year (Heaven et al., 2006).

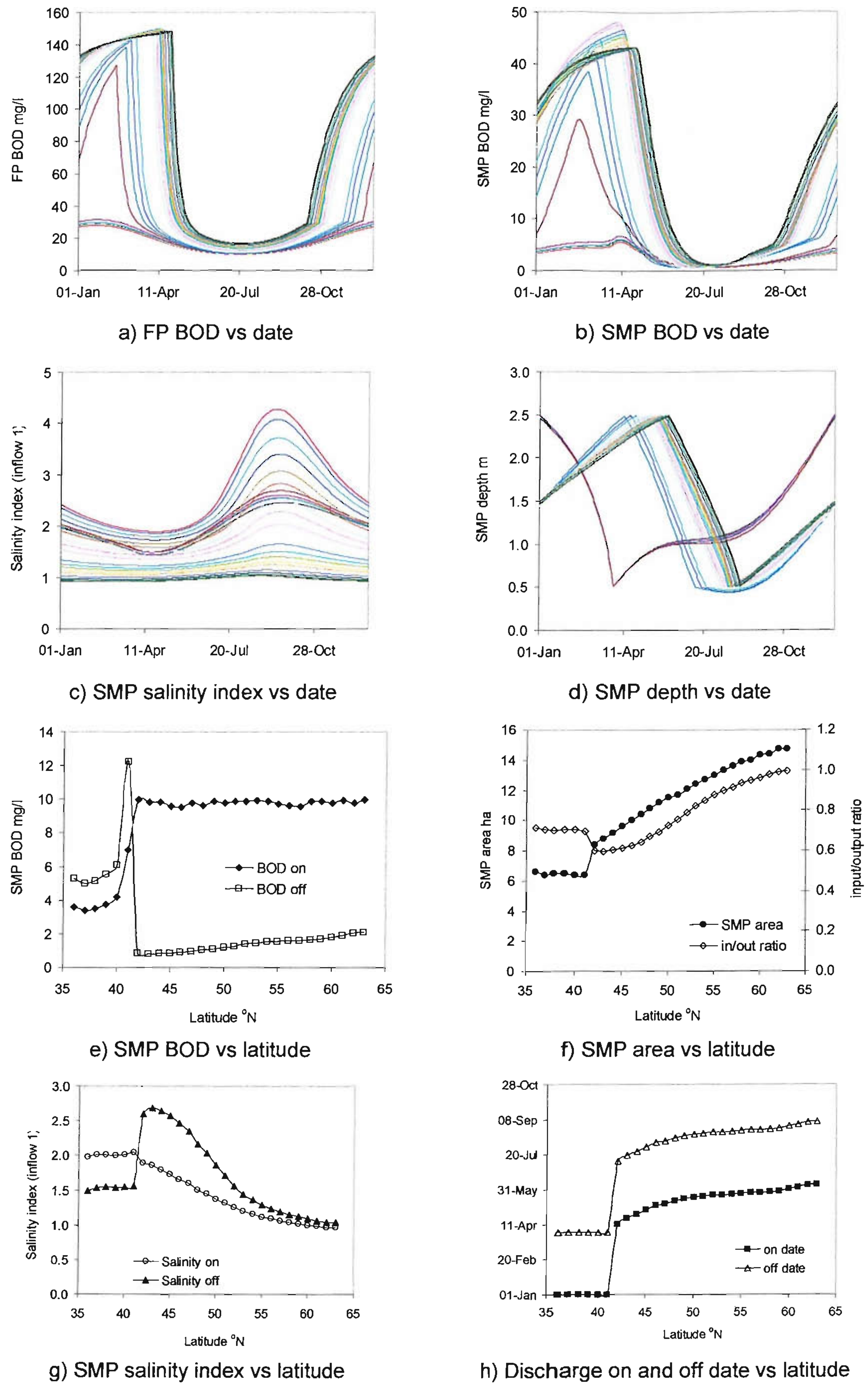


Figure 12 Output for 90-day discharge when SMP BOD = 10 mg l^{-1} at 64E 36-64N

The series with FP BOD < 40 mg l^{-1} all year round (Figure 12a and b) corresponds to latitudes 36-40, where the system is capable of discharging from the beginning of the year; the reduction in SMP depth for this group starting from 1 January can be seen in Figure 12d. At latitude 50°N and above, discharge starts from late May onwards and the characteristic sharp fall in BOD at the onset of spring can be clearly seen in both the FP and SMP. SMP area and discharge dates increase smoothly with latitude, while there is a slight rise in BOD at the off date and a slight fall in salinity (Figure 12e-h). Latitude 41 °N is a transition point: the SMP BOD is < 10 mg l^{-1} in January but is still rising: if the pond is emptied from this point it will in fact fail to meet the discharge condition by the end of the period (Figure 12b and e). For latitudes 42-48°N, the effluent BOD reaches the discharge standard between mid April - mid May. 90 days later the discharge ends, but SMP depth does not start to rise again until late summer due to high evaporation rates: in fact the depth falls slightly as precipitation and pond inflow are less than evaporation (Figure 12d). All the water entering the pond system in this period is effectively lost, and during the discharge period itself there is a sharp increase in salinity (Figure 12c and g). An alternative design with a reduced area for the storage/maturation pond is needed, and one way to achieve this might be to subdivide it and allow one section to dry out. In fact 40-45°N 64°E corresponds to the Kzyl Kum desert in central Asia, and while the area has some population centres they are mainly concerned with oil and mineral extraction, making alternative wastewater treatment systems affordable: but the example illustrates the potential of the model as a site-specific design tool. Further work on the effect of possible alternative designs is described in Heaven *et al.* (2006 and in preparation).

The data used in looking at pond behaviour on a regional basis are derived from average values of climatic parameters, but at certain sites long series data for daily temperature and precipitation are available. There is insufficient information to calculate daily values of ETw for these sites, but average daily ETw values for the nearest gridpoint can be used, as there is relatively little variation in this parameter on a year-to-year basis (Hidalgo *et al.*, 2005). The model was run at selected sites first using long series data to establish the average SMP area required at that location, then again with this area set as a fixed parameter to determine the maximum annual depth. The resulting outputs for first discharge date, total discharge duration, SMP area and SMP maximum depth appeared to be normally distributed. The multi-year depth data were used to estimate the maximum depth likely to occur with a 50-year return period: results are given in Table 6, and Figure

13 shows a probability plot of the depth data for Astana, Kazakhstan. Output of this type could be used to determine appropriate construction depths for the individual sites, or to suggest design guidelines on a regional basis. For the latter purpose more sites should be included in the analysis, but as an example, from Table 6 it appears that 0.25 m or 10% added to the assumed working depth might provide adequate capacity to deal with the 1 in 50 year event for sites north of latitude 50.

Table 6 SMP mean area for multi-year data, with estimated 50-year maximum depth

Site	N	E	Alt	Country	Period		Full data	ETw	Precip	SMP area	50-year depth
			m		from	to	years	mm	mm	ha	m
Khorog	37.3	71.3	2080	Tajikistan	1899	1994	93	1252	254	6.05	2.84
Ashkhabad	38.0	58.3	208	Turkmenistan	1938	1995	52	1490	230	4.63	2.69
Bishkek	42.8	74.5	760	Kyrghyzstan	1936	1991	56	1163	413	6.07	2.82
Turkestan	43.3	68.2	207	Kazakhstan	1886	1995	94	1556	186	5.27	2.99
Almaty	43.2	76.9	851	Kazakhstan	1921	1994	72	976	654	7.30	2.82
Atyrau	47.1	51.9	23	Kazakhstan	1881	1995	94	1337	159	6.39	2.87
Aktyubinsk	50.3	57.2	219	Kazakhstan	1905	1995	78	1009	269	7.95	2.73
Semipalatinsk	50.4	80.2	196	Kazakhstan	1902	1995	84	918	277	7.38	2.74
Astana	51.2	71.4	350	Kazakhstan	1882	1995	100	962	289	8.33	2.73
Kustenai	53.2	63.7	156	Kazakhstan	1903	1995	85	938	325	9.07	2.70
Ulan Ude	51.8	107.6	515	Russia	1887	1995	100	721	248	8.72	2.62
Petropavlovsk	54.8	69.2	142	Kazakhstan	1901	1993	78	756	346	8.68	2.72
Krasnoyarsk	56.0	92.5	276	Russia	1915	1995	78	616	431	8.92	2.72
Perm	58.0	56.0	170	Russia	1883	1995	111	602	619	8.12	2.76

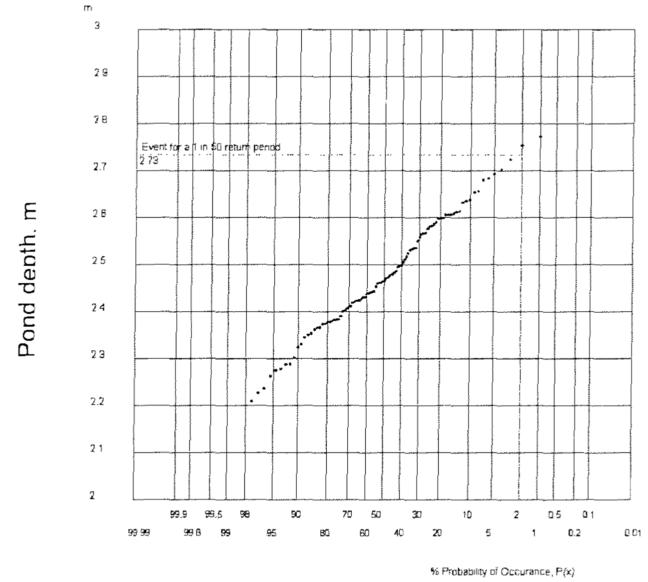


Figure 13 Normal probability plot for maximum SMP depth calculated using 100 years of data for Astana, Kazakhstan

Microbiological quality. Some of the sites used in calibration of the model parameters also had a limited amount of data on effluent microbiological quality (e.g. Neel *et al.*, 1961; Middlebrooks et al., 1982; Soniassey and Lemon, 1986), but no detailed analysis of this was carried out. Problems with data and coefficients values for modelling coliform and other pathogen removal in long storage ponds at low temperatures are briefly discussed in Heaven et al. (2005). In lieu of calibrated values, the model was run using the first-order rate constant coefficient $k_T = 2.6(1.19)^{(T-20)}$ as proposed by Marais (1974), with an influent Faecal Coliform (FC) concentration of 5×10^8 FC l^{-1} and discharge values of 10^5 and 10^6 FC l^{-1} (Mara, 2005). The result at the 10^6 FC l^{-1} level was a considerable increase in required SMP areas compared to the long discharge option, with ponds in the south-west quadrant able to discharge for only part of the year and a few sites on the Tibetan Plateau unable to meet the condition (Figure 14). At 10^5 FC l^{-1} more locations were unable to reach compliance. This is as expected, since the key to pathogen removal is provision of a series of maturation ponds. Microbiological quality is critical for use in irrigation, however, and the results again indicate the need to modify the standard design of a single large storage/maturation pond in transitional areas between continuous discharge and cold climate ponds.

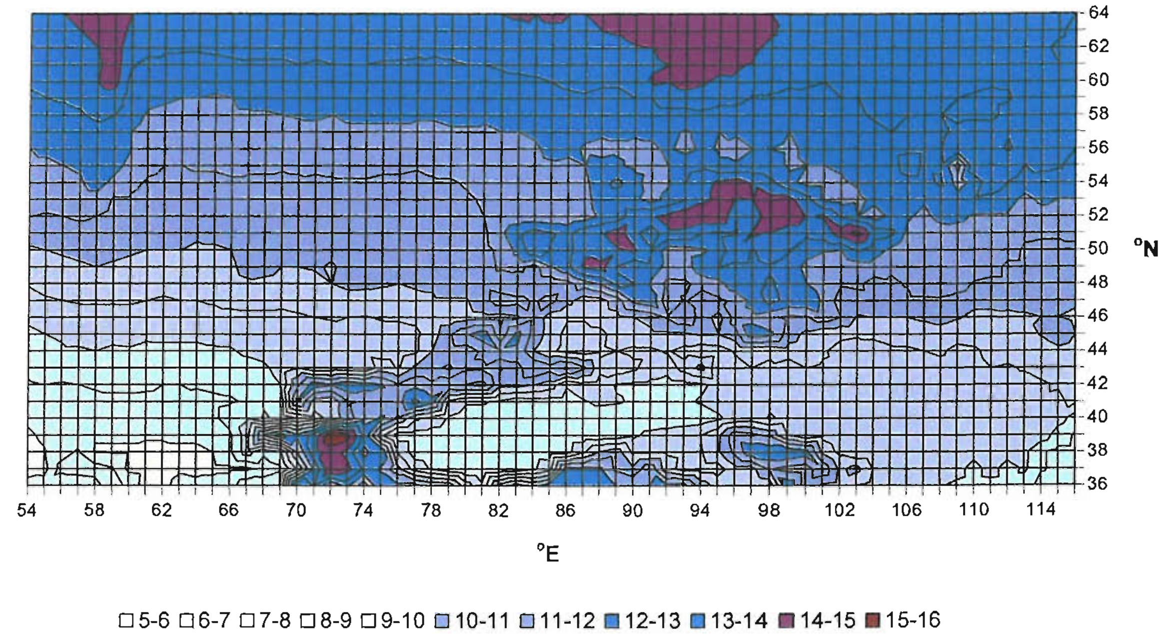


Figure 14 SMP area required for longest possible discharge with 10^6 FC l^{-1} discharge standard

CONCLUSIONS

The base model to predict WSP performance, using mass balance and first-order decay equations for BOD, is extremely simple. It could be upgraded in a number of ways and to different levels of sophistication. For this purpose, data from real pond systems is needed for calibration and verification; within the current work it has only been possible to do this with reference to seven sites. While the real and simulated results show good agreement in spring, it appears that pond performance in autumn is better than that suggested by the model. In practice this is not a major issue, as spring is the critical period for effluent quality reaching a standard suitable for discharge, and under most operating regimes conditions in autumn have relatively little effect on the following year. Even in its current form, the model offers a powerful tool for simulating the cyclic behaviour seen in ponds subject to continental climatic conditions, with a degree of accuracy that permits tailoring of the design to local conditions at specific sites. The results of the work illustrate the importance and value of modelling based on climate parameters, particularly where these are likely to result in dramatic seasonal shifts.

To show clearly the impact of climatic factors the results have been presented in graphical form as regionally-based contour maps. These cover a wide area across central Asia and central China from latitude 36 to 64°N and longitude 54 to 166°E, a land mass of approximately 12 million km^2 ranging from 100 - 5800 m in altitude. These maps are potentially useful in their own right as basis for selecting WSP design and operation parameters. The results broadly confirm that discharge periods typical of those used in North American guidelines could safely be applied; but also suggest that a more detailed approach is worthwhile to optimise the operating regime. The discharge period has a significant effect on the required size of the storage/maturation pond, the inflow/outflow ratio, and on salinity during the discharge: all factors with a considerable impact on construction costs and on the availability and suitability of water for reuse. Long storage periods are appropriate in the north, where water conservation is not critical and the discharge duration is restricted by the dates at which the wastewater can meet notional discharge standards. In areas of the south, all-year round operation may be possible. In between, a graduated approach is needed to match local climatic conditions and maximise the benefits associated with water availability. As the majority of the region's population lives in these areas, and most agricultural production takes place there, this is especially important. The model also provides a means of identifying areas where extremely high evaporation losses or poor treatment performance make pond systems a non-ideal solution for wastewater treatment. These are mainly in the driest parts of the study region or at high altitudes, for example on the Tibetan plateau, where population densities are low and there are few human settlements.

The work described looks at the application of a simple 2-pond system across a wide geographical area, but the results suggest that it may be useful to consider other configurations and operating modes, especially for locations that require an intermediate design between the 'one short discharge per year' mode and all year round operation. To date little work has been carried out on optimising pond configurations and factors such as the surface area to depth ratio in long retention systems where BOD accumulates over part of the year. Attention to this could provide further economies in pond size, land area requirements, potential water availability and enhanced quality.

The current work focused mainly on BOD as a simple example of a non-conserved parameter, but the same approach can readily be adapted to modelling other parameters such as nutrient levels or pathogen indicator organisms. There is a lack of data to support the choice of coefficient values for design and modelling: this needs to be remedied by monitoring the operation of real systems, and by research aimed at providing a better knowledge of applicable parameter values and an improved understanding of the factors that affect them.

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Section 8: Influence of annual climate variability on design and operation of waste stabilisation ponds for continental climates

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Abstract

WSPs are widely used in North America, and offer huge potential for other continental climate regions. The standard design and operating protocol is robust even at high latitudes, but may be conservative elsewhere. A simple model based on first-order kinetics for biochemical oxygen demand (BOD) is used to consider some alternative design and operating protocols, using long-term daily climate records for cities across continental central asia. Options include changing the discharge period; retaining treated water in the pond over the winter; and changing the facultative pond loading. Annual variability in climate parameters has a major effect, in particular on the date at which treated wastewater meets appropriate standards for discharge or re-use: the earlier the discharge, the greater the variability in effluent quality. Skilful management of these systems may therefore be required to maximise their performance. While current models require development, it is clear modelling could provide tools and guidelines that would allow the design of continental climate WSP to be tailored to specific regional and local climate conditions.

Keywords

continental climate, waste stabilisation ponds, wastewater reuse

INTRODUCTION

Waste stabilisation ponds (WSPs) are widely used in Canada and the northern states of the USA, and offer enormous potential for other continental climate regions. The recommended standard design in North America is the intermittent discharge system, based on treatment combined with storage for 12 months, followed by release over a short period in autumn (Prince *et al.*, 1995). This approach has proven successful even at high latitudes, but may be conservative when applied in more southerly areas. The operating mode also assumes that discharge will be to a large watercourse, at a time when effluent quality is high and sufficient flow is available for dilution. In the extreme continental climates of southern central Asia, however, the sharp improvement in effluent quality noted at the onset of autumn in colder climates may not occur. In addition, these regions are typically arid or semi-arid: in some areas perennial water courses suitable for receiving a discharge may not exist, and the overall scarcity of water resources makes re-use of much greater importance. Climates of this type stretch from China and Mongolia across central Asia and southern Russia to the Caspian and beyond. In the more extreme locations, populations have historically been small: but factors such as economic development in western China mean that increasing numbers now live in these areas, creating a growing need for effective technology of this type.

Heaven *et al.* (2005) suggested that alternative WSP design and operating protocols should be developed for these regions, to make more effective use of the treatment capacity present in the warm summer period. Since the transition from accumulation of load in winter to rapid breakdown in spring and summer is driven by climatic factors, however, annual variation in these is likely to be critical to any modified design or operating protocol. The paper looks at some effects of annual variability in climate parameters across continental Asia, using a simple model to allow prediction of the effects on pond performance.

MATERIALS AND METHODS

Model construction and assumptions. The model is based on that described in Heaven et al (2005), and simulates a WSP system consisting of a facultative pond (FP) and a storage/maturation pond (SMP). The original spreadsheet-based model was extended using a Microsoft Visual Basic program to allow automated analysis with climate data for multi-year sequences. The model calculates mass balances for wastewater volumes and biochemical oxygen demand (BOD), or any similar degradable component, using a one-day time-step. Wastewater volumes are calculated taking into account inflow, outflow, evaporation and precipitation and assuming no infiltration. The ponds are assumed to be simple rectangles in plan, with no allowance for variation of area with depth and side slope. BOD concentrations are calculated assuming first-order decay kinetics. The decay constant k is assumed to follow an Arrhenius equation of the form $k_T = k_{20}\theta^{(T-20)}$, where k_T and k_{20} are values of k at temperatures of T ^oC and 20 ^oC respectively.

The FP is sized according to the areal loading rate (US EPA, 1983), by specifying a BOD surface loading rate and a working depth, and thus fixing the surface area, volume and mean hydraulic retention time for a given inflow and influent BOD concentration. Once the surface area is known, daily and total outflows are calculated based on inflow minus evaporation and precipitation. The mass of BOD in the pond is calculated based on the initial value, inputs, decay and discharge, and daily effluent concentrations are obtained by dividing the total mass of BOD by the pond volume.

The design of the SMP is defined by choosing a maximum and minimum depth and a discharge period, at the end of which the depth is assumed to be at its minimum. The outflow from the SMP is equal to inflow (corresponding to outflow from the FP minus any direct discharges), minus evaporation and plus precipitation. As precipitation and evaporation inputs depend on the surface area, any change in area alters the maximum volume to be stored in the SMP. In order to establish the required area for a given maximum and minimum depth, the model is run with an initial estimate of area. If the calculated pond depth at any point in the simulation is greater than the maximum value, the area is incremented and the calculations repeated. This process is iterated until the entire dataset can be analysed without exceeding the maximum depth. Outflow from the SMP is calculated based on discharge over a fixed period that starts and finishes on the specified dates each year. The daily outflow is calculated as the volume contained in the SMP at the start of each day divided by the number of days remaining over which it is to be emptied. As evaporation and precipitation vary each day, the amount to be discharged also varies and needs to be recalculated on a daily basis. Daily values are then used to calculate pond depth and effluent BOD concentrations.

The validity of specific output values for BOD is uncertain, for reasons discussed briefly below and in Heaven *et al.* (2005); but the model results are adequate to indicate key factors and trends.

Climate data. Records of temperature and precipitation for twelve cities across the central Asian region were taken from the archive of the All-Russia Research Institute of Hydrometeorological Information - World Data Centre (RIHMI-WDC, 2006). The datasets consist of daily records, starting in some cases from the 1880s, but with some years missing or only partially complete. As the model requires data from complete years, any years with missing periods of more than 5 continuous days were eliminated. Where data were missing for 5 days or less, temperature values were interpolated from adjacent days. Missing precipitation values were assumed to be zero. For evaporation, Penman-based estimates for reference crop evapotranspiration were obtained from the International Water Management Institute climate database (IWMI, 2006). These monthly values were converted by polynomial interpolation to daily potential evapotranspiration (ET_{wat}) for open water less than 2 m deep, using a factor of 1.05 (Allen et al, 1998). Water temperature was assumed to equal mean daily air temperature down to 0 °C and to remain at zero for lower air temperatures, with a 5-day time lag. Details of weather stations and climate data used are given in Table 1.

Table 1 Weather stations and climate data used in modelling

Site	Latitude	Longitude	Altitude	Country	Period		Full data	Days <0 °C	ET_{wat}	Precipitation
	°N	°E	m		from	to	years		mm	mm
Khorog	37.3	71.3	2080	Tajikistan	1899	1994	93	97	1252	254
Ashkhabad	38.0	58.3	208	Turkmenistan	1938	1995	52	22	1490	230
Yerevan	40.1	44.5	907	Armenia	1886	1991	89	68	1134	295
Bishkek	42.8	74.5	760	Kyrghyzstan	1936	1991	56	75	1163	413
Turkestan	43.3	68.2	207	Kazakhstan	1886	1995	94	70	1556	186
Atyrau	47.1	51.9	23	Kazakhstan	1881	1995	94	116	1337	159
Aktyubinsk	50.3	57.2	219	Kazakhstan	1905	1995	78	150	1009	269
Astana	51.2	71.4	350	Kazakhstan	1882	1995	100	167	962	289
Ulan Ude	51.8	107.6	515	Russia	1887	1995	100	178	721	248
Petropavlovsk	54.8	69.2	142	Kazakhstan	1901	1993	78	169	756	346
Krasnoyarsk	56.0	92.5	276	Russia	1915	1995	78	168	616	431
Yakutsk	62.0	129.7	101	Russia	1889	1995	98	210	556	210

Modelling parameters and scenarios. Wastewater inflow rates were taken as 1000 m^3 day^{-1}, with a BOD of 200 mg l^{-1}. BOD decay constant values were θ_{BOD} = 1.08 and $k_{20\,BOD}$ = 0.25 and 0.08 for the FP and SMP respectively (Mara, 1976). For modelling purposes, wastewater was considered nominally acceptable for discharge when the 95-percentile BOD concentration reached 20 mg l^{-1}. The standard design was based on working depths of 1 m for the FP and 2 m for the SMP; a FP surface loading rate of 40 kg BOD ha^{-1} day^{-1}; and a single autumn discharge. Other cases considered included discharge from the SMP with different durations, start dates, and volumes of over-winter storage; and alternative FP loading rates.

RESULTS AND DISCUSSION

Sites and climate parameters

The sites were chosen to provide a range of latitudes and conditions, and for the quality of their data records, rather than any specific need for or association with WSP systems. In Ashkhabad mean winter temperatures are sufficiently high that in most years ponds are unlikely to freeze: but the dataset shows an average of 22 and a maximum of 61 days each year below zero, including continuous periods of over a month. Khorog is at the southern limit of the group, but experiences freezing temperatures due to its altitude. Yakutsk is too far north for potential re-use of treated wastewater in irrigation: in practice large-scale agriculture ceases around 55 °N, but the site was included to provide an example of high latitude parameters. Figure 1 shows examples of climate data for selected sites: all are typified by large variation in summer and winter temperatures, with especially high variability in spring and autumn. Sites in the middle and northern latitudes (Atyrau, Astana, Petropavlovsk) are characterised by a skewed temperature distribution in winter: there are many low values but a long period in which maximum mean daily temperatures seldom exceed zero, followed by a sudden sharp increase. All of the sites are quite dry (Table 1), but there are also differences in the distribution of precipitation through the year, which have considerable significance for potential re-use in agriculture or river recharge. Figure 1 gives examples of three typical modes for average daily precipitation and ET_{wat} for the period covered by each dataset. In the south summers are very dry, and precipitation occurs in the winter (Khorog, Ashkabad, Turkestan) or bi-modally in spring and autumn peaks (Erevan, Bishkek). In the north (Astana, Krasnoyarsk, Petropavlovsk, Ulan Ude, Yakutsk) precipitation occurs as rain in summer; while in mid-latitudes (Atyrau, Aktyubinsk) rainfall is more evenly distributed through the year.

Modelling results

The distribution of calculated effluent concentrations on a given date was found to be log normal, while the date on which the calculated concentration reached a given value each year was approximately normally distributed.

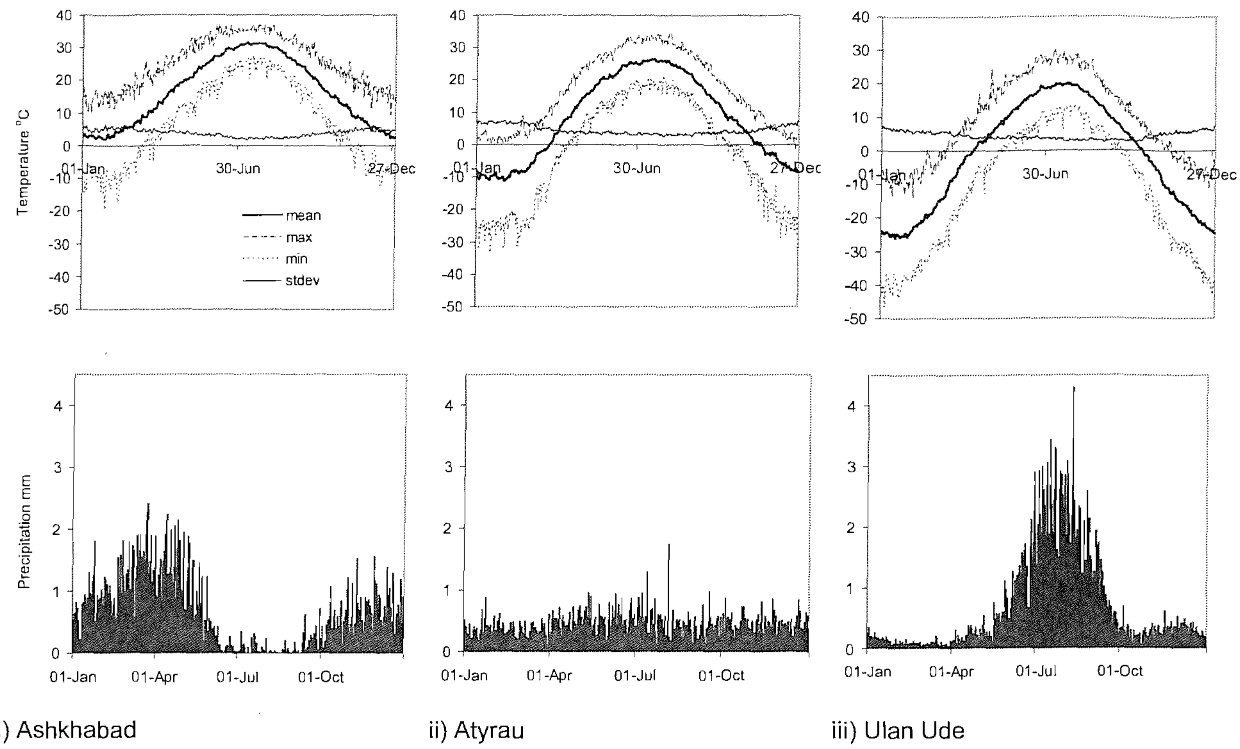
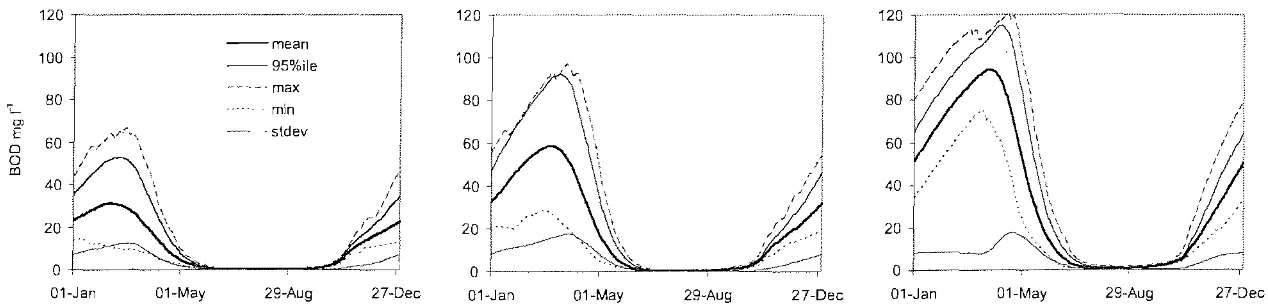


Figure 1 Mean, max and min daily temperatures and mean daily rainfall for selected sites

Standard design. Table 2 shows key output parameters for a standard design at each site, assuming discharge from 1-30 October and maximum and minimum SMP depths of 2.5 m and 0.5 m. Examples of SMP effluent BOD concentrations for selected sites are shown in Figure 2. Overall the results indicate that the classic north American design is robust in terms of the likelihood of achieving low concentrations by the discharge period, and annual variations in climate are unlikely to have much impact on water quality in late summer. There is little or no carry-over in performance from year to year, as the sequence is broken by the long summer retention period in which effluent concentrations reach a steady-state value; in northern areas, a similar effect may also arise from the period of minimal treatment in winter. This implies that the fact that continental climates are subject not only to extreme variations but also to sequences of wet or dry and warm or cold years is not likely to be critical to WSP design. The fact that the nominally acceptable quality is generally achieved much earlier than the actual discharge date suggests, however, that the systems are over-designed, leading to unnecessarily large ponds and high evaporative losses due to the long storage of treated wastewater. For the standard design at the chosen sites, approximately 73% of variation in the day of discharge is accounted for by latitude, rising to 85% if Khorog (altitude 2080 m) is omitted; latitude also accounts for about 56% of variation in required pond size.

Table 2 Model output for standard design with discharge from 1-30 October

Site	Days					Concentration					Area	% for use
	Start				*End*	*Start*				*End*	*FP+SMP*	
	Mean	95%ile	Day	Range	95%ile	Mean	95%ile	Day	Range	95%ile	ha	
Khorog	22-Apr	09-May	128	17	21-Nov	24-Apr	06-May	125	12	03-Dec	15.9	57%
Ashkhabad	-	-	1	-	-	-	-	1	-	-	14.6	60%
Yerevan	25-Mar	22-Apr	111	28	19-Dec	01-Apr	21-Apr	110	20	19-Dec	16.3	55%
Bishkek	17-Mar	09-Apr	98	23	13-Jan	22-Mar	10-Apr	99	19	06-Dec	17.2	53%
Turkestan	20-Mar	15-Apr	104	26	12-Nov	26-Mar	12-Apr	101	17	03-Dec	13.5	63%
Atyrau	26-Apr	16-May	135	20	29-Dec	30-Apr	13-May	132	13	20-Nov	14.5	60%
Aktyubinsk	14-May	26-May	145	12	06-Nov	16-May	26-May	145	10	12-Nov	16.7	54%
Astana	23-May	04-Jun	154	12	09-Nov	25-May	04-Jun	154	10	11-Nov	18.0	51%
Ulan Ude	05-Jun	11-Jun	161	6	04-Nov	06-Jun	12-Jun	162	6	05-Nov	18.7	49%
Petropavlovsk	27-May	07-Jun	157	11	12-Nov	26-May	07-Jun	157	12	11-Nov	19.7	46%
Krasnoyarsk	26-May	06-Jun	156	11	09-Nov	27-May	06-Jun	156	10	13-Nov	20.8	43%
Yakutsk	18-Jun	23-Jun	173	5	01-Nov	19-Jun	26-Jun	176	7	30-Oct	18.9	48%



a) Turkestan b) Atyrau c) Astana

Figure 2 Model output for SMP effluent BOD concentrations with standard design at selected sites

Longer discharge period. Table 3 shows results for model runs with discharge from the 95%ile value for the first day with effluent BOD concentration less than 20 mg l^{-1}, up to 30 October. Discharging from an earlier date over a longer period has little effect on the average performance or on the earliest and latest date on which the quality is acceptable: one reason is that the influent from the FP is also of good quality in this period. Advantages of an earlier discharge are a reduced pond area, leading to lower evaporation losses and increased availability for potential reuse.

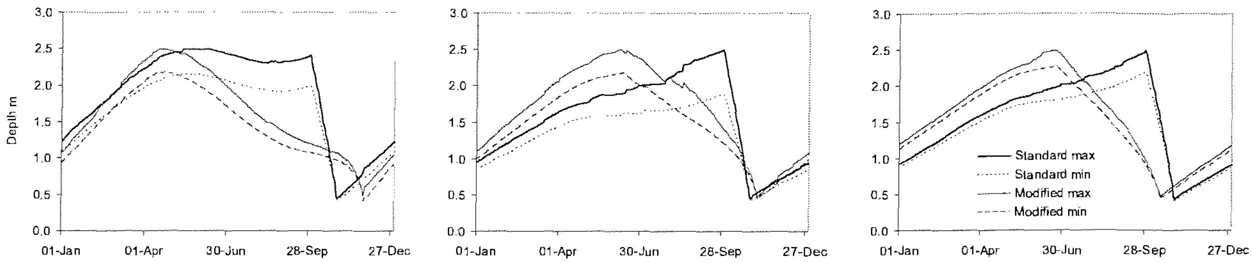
The reduction in pond size seen in Table 3 is mainly due to removal of the need to provide storage for water discharged during the summer period, but may also be due to climatic factors. Figure 3 shows model output for pond depths for Turkestan, Petropavlovsk and Yakutsk. At sites with high evaporation, with a standard design the maximum depth that determines pond size occurs early in the year (e.g. mid-May in Turkestan) and levels fall thereafter. Earlier discharge decreases the depth range at the time of emptying, and thus increases volume utilisation. Further north, the maximum depth occurs immediately before emptying. In Astana, for example, depth is determined by a small number of years with relatively high rainfall in the late spring season: these account for an additional 175 mm of depth which, because of the limit on maximum working depth, contribute approximately 7% to the pond area. Similarly in Petropavlovsk and Ulan Ude the majority of both precipitation and variability in it occurs in summer, so an extended discharge period has a major effect on the required depth (Figure 3). In Yakutsk variability in depth is relatively small, but even so precipitation events affect the maximum value: the greatest depth is determined by an unusually wet June in 1984, not itself the wettest year.

Table 3 Model output for discharge from first acceptable day to 30 October

Site	Discharge			Area	Depth range*	% for use
	Start	*Last**		*FP+SMP*		
	95%ile	95%ile	no. of days	ha	m	
Khorog	09-May	02-Dec	207	13.2	0.58	64%
Ashkhabad	22-Mar	16-Dec	269	11.0	0.43	70%
Yerevan	27-Apr	12-Dec	229	12.0	0.38	67%
Bishkek	16-Apr	28-Nov	226	12.8	0.41	65%
Turkestan	16-Apr	27-Nov	225	12.1	0.33	67%
Atyrau	14-May	20-Nov	190	12.9	0.26	65%
Aktyubinsk	27-May	10-Nov	167	14.6	0.31	60%
Astana	05-Jun	08-Nov	156	15.0	0.36	59%
Ulan Ude	13-Jun	30-Oct	139	14.8	0.18	60%
Petropavlovsk	09-Jun	07-Nov	151	15.2	0.34	58%
Krasnoyarsk	08-Jun	09-Nov	154	15.5	0.30	58%
Yakutsk	25-Jun	16-Oct	113	16.7	0.23	54%

* Last = last day with effluent BOD <20 mg l^{-1}. Depth range = range in max-min values of annual maximum depth

Other modifications. Various different strategies can be adopted to influence the volume of water available and the date at which it reaches the nominal standard for discharge or re-use. One option is to change the pond depth: if the working depth is maintained while the maximum and minimum are increased, this effectively retains a volume of treated water within the pond at the end of summer to provide buffering and dilution for the incoming wastewater. Table 4 illustrates the effect of increasing the maximum and minimum depth, for a working depth of 2 m, using Turkestan and Astana as examples. The result is to bring the earliest discharge date slightly forward, and the final date back. In practice there may be little use for water in November-December in these regions, if air temperatures are below freezing. Table 5 shows the effect of choosing a fixed end-date for the discharge, reflecting different potential options for reuse or disposal, on the earliest start date, the pond area and the re-use potential, for two depth ranges. Once again the earliest start date is brought forward, by a relatively small margin: a maximum of 11 days for Astana and 14 for Turkestan, for a change in end date of 4-5 months. The earliest start date shown for Turkestan corresponds to the time at which water might be of use, for example, in spring pre-irrigation. In this case, the trade-off for an earlier start date is a reduced proportion of water potentially available for reuse.



a) Turkestan b) Petropavlovsk c) Yakutsk

Figure 3 Maximum and minimum SMP depths at selected sites for standard design and modified design with maximum discharge period (as in Table 3)

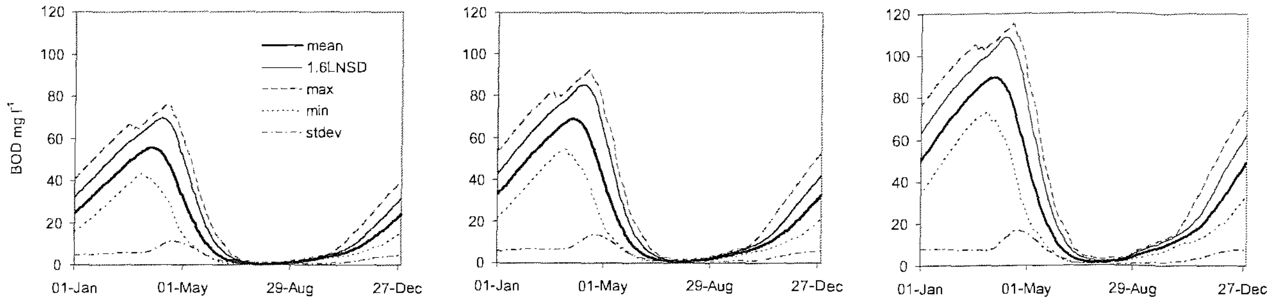
Table 4 Model output for different maximum depths with working depth 2 m

SMP max depth	Turkestan				Astana			
m	*Start (95%ile)*	*End (95%ile)*	*Area (ha)*	*% for use*	*Start (95%ile)*	*End (95%ile)*	*Area (ha)*	*% for use*
2.50	16-Apr	27-Nov	7.1	55%	05-Jun	08-Nov	10.6	71%
3.00	16-Apr	10-Dec	6.4	57%	02-Jun	18-Nov	9.7	73%
3.50	13-Apr	19-Dec	5.9	59%	30-May	27-Nov	9.1	74%
4.00	11-Apr	27-Dec	5.5	61%	26-May	19-Nov	8.6	75%

Table 5 Model output for chosen discharge end-date with working depth 2 m

SMP max depth	Turkestan				Astana			
m	*Start (95%ile)*	*End (95%ile)*	*Area (ha)*	*% for use*	*Start (95%ile)*	*End (95%ile)*	*Area (ha)*	*% for use*
2.50	16-Apr	27-Nov	7.1	55%	05-Jun	08-Nov	10.0	72%
	14-Apr	30-Oct	8.5	49%	04-Jun	30-Oct	10.5	71%
	12-Apr	30-Sep	9.4	46%	01-Jun	30-Sep	12.0	69%
	11-Apr	30-Aug	9.7	45%	30-May	30-Aug	13.0	67%
	11-Apr	31-Jul	9.4	46%	30-May	31-Jul	14.3	64%
4.00	11-Apr	27-Dec	5.5	61%	-	-	-	-
	07-Apr	30-Nov	6.8	56%	26-May	19-Nov	9.2	74%
	03-Apr	30-Oct	8.1	51%	24-May	30-Oct	10.2	72%
	30-Mar	30-Sep	8.8	48%	21-May	30-Sep	11.8	69%
	28-Mar	30-Aug	9.1	47%	18-May	30-Aug	12.8	67%
	28-Mar	31-Jul	9.0	47%	17-May	31-Jul	14.2	65%

The above examples assume a FP pond depth of 1 m and a BOD loading rate of 40 kg ha^{-1} day^{-1}. Table 6 and Figure 4 show the effect of changing the FP area and loading rate while keeping a depth of 1 m, using Astana as an example. There is a significant effect on the earliest date for discharge, once again at the expense of water availability as evaporation losses rise with increasing area. For FP areas of 6.7 and 10 ha, the system is moving towards ponds of equal sizes, but with intermittent discharge: a sort of hybrid between the classic cold and temperate climate designs.



a) 20 kg BOD ha^{-1} day^{-1} b) 30 kg BOD ha^{-1} day^{-1} c) 60 kg BOD ha^{-1} day^{-1}

Figure 4 Astana SMP effluent BOD for discharge to 30 August at different FP loading rates (2.5 m)

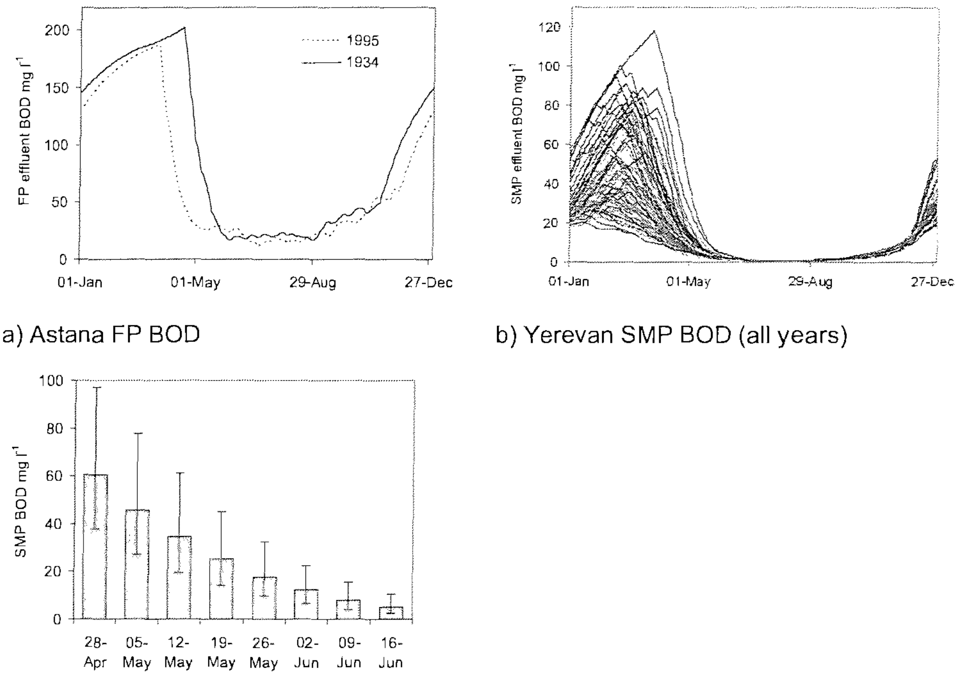
Table 6 Model output for Astana with discharge to 30 August at different FP loading rates

FP loading rate	FP area	SMP max depth 2.5 m			SMP max depth 4 m		
kg BOD ha^{-1} day^{-1}	ha	*Start (95%ile)*	*Area (ha)*	*% for use*	*Start (95%ile)*	*Area (ha)*	*% for use*
20	3.3	02-Jun	16.3	70%	22-May	16.1	70%
30	5.0	30-May	18.0	67%	18-May	17.8	67%
40	6.7	28-May	19.6	64%	16-May	19.5	64%
60	10.0	23-May	22.8	58%	11-May	22.8	58%

Inter-annual variation

The above examples indicate how it may be possible significantly to influence discharge dates and/or volumes of water available. From the results shown and from further analysis (not reported here), however, it is clear that variability between years in operating and performance parameters is a key issue. Some examples are given above for pond depth; more examples for effluent quality are presented in Figure 5. Figure 5a shows FP effluent BOD for the years 1934 and 1995 for Astana (standard design): while values in late summer are similar, there is a one-month difference in the date at which the wastewater first reaches a steady-state condition. Figure 5b shows SMP effluent BOD for discharge from 27 April - 30 October for the whole dataset for Erevan (Table 3): the wide range is clearly seen, as is the exceptionally cold winter of 1933.

The problem of annual variability is particularly acute if it is desired to bring forward the date of first discharge. Not only does the value of the mean effluent concentration rise steeply for earlier dates, but the variability also increases, with a sharp rise in standard deviation during the spring period. This is a direct consequence of annual variation in climate parameters, and can be clearly seen for the cases in Figures 2, 4 and 5b. Figure 5c gives a further example, showing BOD in Astana SMP on the first day of discharge, for discharge from a range of dates to 30 October. These examples indicate the variability of these systems, and suggest that careful management may be needed to ensure acceptable discharge quality. If it is essential to guarantee that water of suitable quality will be available early in the year, it may be necessary to adopt other strategies, such as the use of alternating parallel SMPs to provide separate storage for treated wastewater.



a) Astana FP BOD

b) Yerevan SMP BOD (all years)

c) Astana SMP BOD

Figure 5 Examples of annual variability in output parameters

Model limitations and development

The limitations of the modelling approach are discussed briefly in Heaven et al (2005). Sensitivity analysis and statistical parameters are dealt with in another paper (Salter *et al.*, in preparation), and are not considered here. In summary, the model oversimplifies pond behaviour, and tends to give SMP effluent concentrations that are too low in summer. Choice of parameters is based on mid-range values from Mara (1976), with very limited validation on an experimental scale in Almaty, Kazakhstan (43.2^{o} N, 76.9^{o} E). Further questions concern validity across a wide geographical range, in particular for places like Ashkhabad and Bishkek where ponds may not freeze every year. In the current application, the method of determining SMP area from the maximum depth according to historic data is unsatisfactory, as it may be influenced by extreme values: a more sophisticated approach would consider the distribution of depths. With high variability, 99%ile values may be more appropriate than the 95%ile conventionally used in wastewater treatment. Despite these points, it is clear that modelling is potentially a powerful design tool for improved performance and that further research providing relevant parameter and validation data would be of great value.

CONCLUSIONS

The results of the modelling work confirm that the standard North American design with 12 months storage is robust. There is little or no carry-over in performance from year to year, due to the long period of treatment in summer which allows steady-state conditions to develop. This means the fact that continental climates can experience sequences of wet or dry and hot or cold years is unlikely to be critical for design. The standard design and operating protocol may be conservative in many locations, however, and there is potential for modification to reduce the overall size of the pond system and increase the volume of treated water available for potential reuse. Possible options include changing the discharge period; changing maximum and minimum depths to retain treated water in the pond over the winter period; and altering the facultative pond loading. In most cases these involve a trade-off between discharge date and water availability. If design and operating protocols are to be modified, however, annual variability in climate parameters will have a significant effect, in particular on the date at which the treated wastewater is likely to meet appropriate standards for discharge or re-use: the earlier the discharge, the greater the variability in effluent quality. To eliminate the effect of year-on-year variations, it may be necessary to consider alternatives such as separate storage of treated water over the winter period. Skilful management may be needed if the performance of these systems is to be maximised. While current models require development, it is clear modelling could provide tools and guidelines that would allow the design of continental climate WSPs to be more closely tailored to regional and local conditions.

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Section 9: CONCLUSIONS

WSP systems have considerable unexploited potential for continental climate regions. With increasing economic affluence, communities often choose to adopt more complex and sophisticated wastewater treatment methods. Simpler low-cost options can perform extremely well, however, and in some cases may offer advantages in terms of effluent quality, water conservation and removal of certain types of pathogenic organisms. The current work does not consider nutrient removal and looks only briefly at microbiological quality, although some of the methods used could be adapted for these parameters. It seems possible, however, that if cold climate WSPs received the level of design attention that has been devoted to optimising tropical systems, they too might show a marked degree of performance improvement and a proliferation or diversification in design types for different purposes.

WSP technology appears to have particular applications in central Asia and western China, due to the presence of existing infrastructure in form of evaporation ponds and wastewater storage reservoirs. In some case these could be readily adapted to form full treatment systems. From the work carried out, it appears that WSP designs could be tailored to suit local conditions without losing their essential robustness. Existing design and operation protocols provide a safe system but are conservative, and this is likely to be at the expense of water resources in arid regions. Unfortunately there is a lack of field data for use in exploring alternative design options, as these systems are only rarely monitored outside the discharge period. It appears however that even simple models can provide useful tools for predicting pond behaviour, and there is considerable potential for the development of better models. To validate these, further monitoring of full-scale systems is needed to provide robust data covering a wide geographical area.

The research described in this thesis has made the following specific contributions:

The light attenuation coefficient k was found to be significantly affected by depth and suspended solids concentrations within the range of values typically found in WSPs. For practical purposes it will often be sufficient to consider k values as constant, but under certain conditions a more sophisticated approach may be needed taking local variation into account. Examples where this may be the case include modelling of WSP start-up, or of the annual spring revival in strongly seasonal climates. Typical values for k in ponds appear to lie in the range of 5-25 m^{-1}. In-pond measurement of k values presents many practical difficulties, however, and the column apparatus tested in the work may offer a reliable alternative means of measurement under standard conditions. The use of photodiodes to measure local irradiance proved to be highly successful. It is recommended that a standardised approach is adopted to the measurement and reporting of k values.

It was shown that accumulated benthic sludges contribute to the nutrient load in the water column in a way that significantly affects pond behaviour. In particular there is an increase in concentrations of suspended solids and nutrients, which may be of importance if the pond is to be discharged to sensitive natural waters. This behaviour does not appear to be detrimental to overall performance in terms of COD removal or DO concentrations in the pond, however; and given that the nutrient contribution from freshly deposited sludge approached that from a mature sludge layer within 7-10 weeks it seems unlikely that changes in recommended design or de-sludging frequency can be used to regulate this in a practical manner.

The work has identified some of the limitations of existing design procedures and protocols from cold and continental climate regions: in the former Soviet Union regulations on pond depth effectively control the design at low temperatures, while typical north American guidelines are robust but potentially conservative if applied in all areas where ponds are subject to freezing. The research has demonstrated the importance of climatic factors in determining pond behaviour. In particular it has shown the potential of simple models, using basic widely-available data on parameters such as temperature and precipitation, to predict pond performance in a manner that will allow tailoring of the design to local conditions, in order to reduce construction and operating costs while helping to meet potential demand for wastewater reuse. The output from such modelling can also be presented in the form of contour maps which can be used to provide simple regionally-based design guidelines.

Plans for further work include

- use of more sophisticated models for predicting water temperature profiles and ice cover on the basis of readily available climatic data, to see whether these lead to better prediction of WSP performance in the autumn period.

- application of the current modelling approach in other climate regions including central and northern Europe, North America and north-east China to assess the appropriateness of regional guidelines and the potential for use of alternative design and operating protocols.

Section A1: A computer simulation of the oxygen balance in a cold climate winter storage WSP during the critical spring warm-up period

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Abstract

The paper considers factors that determine the oxygen balance in extreme climate waste stabilisation ponds during the critical spring warm-up period. At this time BOD load on the pond is a maximum, due to accumulation of wastewater under the ice during the winter. The paper describes the operation of a typical cold climate WSP and the events leading to a balanced steady state system as spring develops into summer. A mathematical model to simulate conditions within a batch fed experimental pond over the transient period is described. To model temperature changes in the water body experimental data were fitted to a generalised equation based on diurnal fluctuations in air temperature. The results are plotted in a normalised form and show the diurnal fluctuation and time lapse as the depth of the pond increases. Maximum daily water temperature lags behind maximum light intensity. Bacterial growth is simulated by a Monod kinetic model in which growth rate depends on initial substrate concentration; temperature compensation is applied using a temperature activity coefficient. Oxygen utilisation is calculated from substrate removal. Algal growth rate is more complicated as it is affected by temperature and light availability. Algal oxygen production potential is considered in terms of its primary metabolite yield, which is then used in a Monod equation to estimate the growth rate. The model uses a mass balance approach to determine dissolved oxygen concentration in the pond. The model is still in a simple form but shows reasonable agreement, in terms of events and time lapses, to measured parameters in experimental ponds recovering from ice cover.

INTRODUCTION

Waste stabilisation ponds (WSPs) have been used in extreme climate conditions for the treatment of wastewater since the early 1940s. Originally they were nothing more than primitive lagoons that held wastewater in storage until natural self-purification processes made it fit to discharge into the natural environment. Since then the design of WSP systems has become more sophisticated. In Canada and Alaska there are in excess of 1000 systems for domestic/industrial wastewater treatment representing least 50% of total wastewater treatment capacity in the region. The use of WSP storage and treatment systems in extreme climates elsewhere has been less common, although research into this method of treatment was carried out and design standards derived for in the countries of the former Soviet Union (Heaven *et al.*, 2002).

Description of an extreme climate WSP

The most successful extreme climate WSPs are those that use 3 types of ponds and only discharge once every year (Prince *et al.*, 1995). The anaerobic ponds(s) act as first stage treatment and may comprise a number of separate cells operating in series to give a total retention time of 3-10 days. The second stage consists of a facultative pond(s) working in conjunction with the third stage maturation/storage pond(s): both these serve to reduce the biochemical oxygen demand (BOD) of the wastewater by the process of biological oxidation. As in conventional WSPs, the system in cold climates relies on the symbiotic relation between bacteria and algae to remove BOD and supply the oxygen necessary for this.

Figure 1 below shows qualitatively the operation of a one-year retention WSP system. There is only one batch discharge of effluent in the autumn, although continuous discharge through the summer months is possible if the effluent quality is satisfactory. The diagram assumes that at the end of the summer period the soluble BOD in the storage/maturation pond is at a very low level (a reasonable assumption considering the extended storage under oxygen-enriched conditions at warm temperatures). When the temperature cools in autumn, algae in both the facultative and maturation ponds begin to settle leaving a body of water that is free of suspended solids; it is generally considered that this response is induced by a fall in water temperature (Prince et al., 1995). At this point the water in the maturation pond is decanted leaving the algal/bacterial sludge and a minimum of bottom waters, thus maximising storage capacity. In some cases part of the water in the facultative pond is also decanted, but this is not common. During the winter wastewater continues to flow through the anaerobic cells (where solids settle) into the facultative pond where ice cover exists and water temperatures beneath this equilibrate at around 4°C.

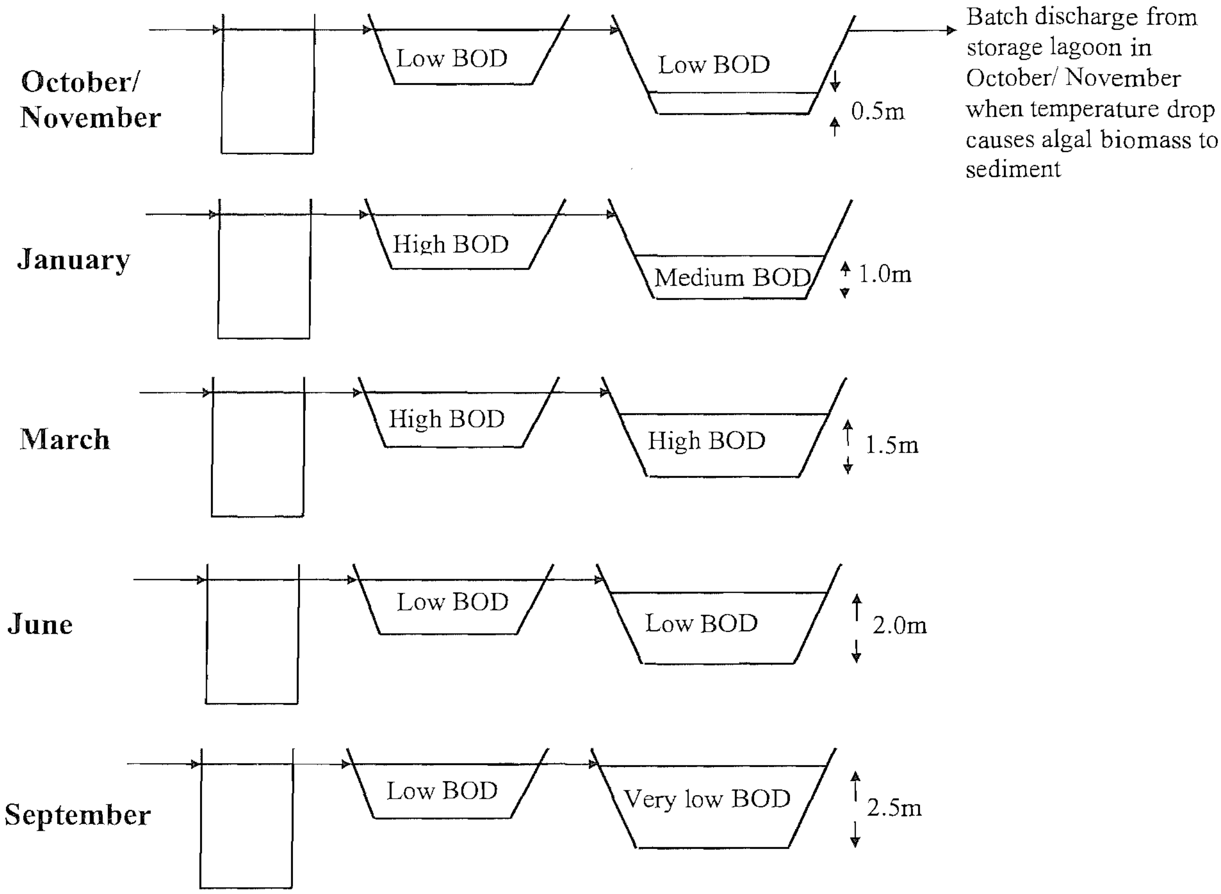

Figure 1 Schematic diagram showing nutrient status in facultative and storage ponds, and water level in the storage pond at different times of the year

The 'clean' water from the facultative pond is diluted out into the storage pond resulting in a situation in spring where the contents of the facultative pond are more or less equal in quality to untreated settled sewage (assuming minimal rates of biological decomposition under ice cover). Those of the maturation pond are similar to partially diluted sewage. Hence in spring, when the ice melts, the organic load in the system is at its maximum and dissolved oxygen levels are probably at their minimum. It is at this point (if any) that the system is likely to be odoriferous.

During the spring 'warm up' period the BOD in both the facultative and maturation ponds will start to be reduced as a result of aerobic heterotrophic microbial utilisation of the dissolved organic matter. Oxygen for this will be supplied by the population explosion of algae that grow photosynthetically at the expense of macronutrients and inorganic carbon sources. During this time the facultative pond will continue to receive settled wastewater whilst the maturation pond will receive a reduced organic load due to its influent having been 'pre-treated' through the facultative pond. In this spring period, both the facultative and maturation ponds will be operating in 'facultative mode' with an excess of soluble organic carbon and nutrients. As spring turns to summer the depth of water in the maturation pond will increase above the optimum for a facultative pond. By this time, however, the organic load on it will be very low, and it will start to function as a maturation pond reducing soluble BOD to low levels. Depending upon temperature, initial organic load and other factors both the facultative and maturation ponds should reach a steady state by June and continue to operate throughout the summer period in a conventional manner. By the end of the summer the soluble BOD in the maturation pond will be very low (possibly between 5-15 mg l^{-1}) resulting in a high quality autumn discharge.

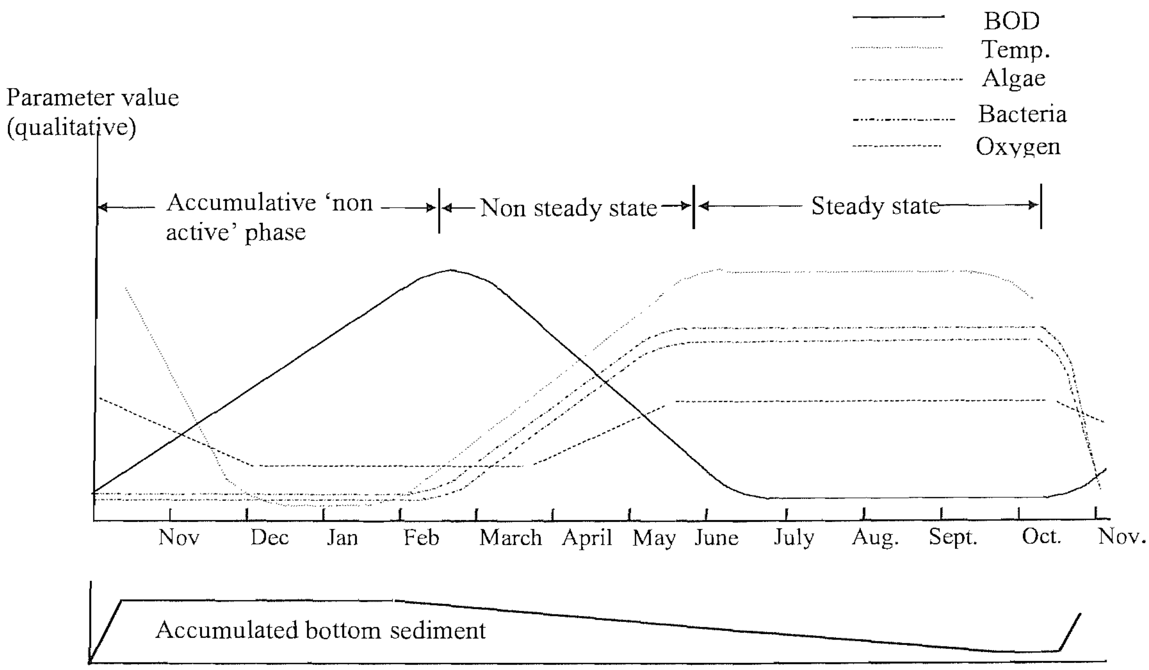

Figure 2 Qualitative graphical description of events in a cold climate facultative pond

Figure 2 qualitatively depicts the fate of key parameters in the facultative pond during the 12-month period. These are descriptively divided into three phases: 'accumulation' where BOD load is added but not destroyed; 'non steady state' where BOD load is progressively destroyed as temperature and microbial numbers (algal and bacterial) increase; and 'steady state' where BOD is consistently low, microbial numbers are in equilibrium, and summer temperatures are more or less constant.

The critical phase appears to be during the 'non steady state' period as the initial organic load is high, the oxygen production potential (algal numbers) is at its lowest, and the potential for bacterial growth (both aerobically or anaerobically) is at its maximum. If the initial load is such that the rate of oxygen depletion is greater than the rate of replenishment then the system will be predominantly anaerobic during this phase leading to odours and lower rates of reaction. If the initial load, and continuing input, is low then the rate of oxygen replenishment may exceed the oxygen demand and the system will become oxygen saturated. In terms of treatment this latter state of affairs will do no harm but it does represent a process inefficiency as the system is larger than it needs to be.

MODELLING

The current work is an attempt to model the potential oxygen balance in a facultative pond during the non steady state spring warm-up period. In effect the microbial systems in a cold climate pond are inactivated during the winter period where water temperature drops to freezing in the surface water and close to freezing throughout the rest of the water body. Any photosynthetic activity will also cease as light penetration beneath the ice is minimal due to a snow covering. Relatively little is known about the survival of the microorganisms in the pond over this period, and hence their effectiveness as an inoculum for fresh growth once there is free water and light penetration.

It is assumed that the survival of macro-invertebrates, such as fly larvae, that inhabit the water column is very restricted and that these will invade the system only when conditions are favourable in the late spring and early summer. Due to the relatively slow growth rates of these secondary feeders it is also assumed that their contribution to the removal process during spring is negligible and that the system is dominated by phototrophic algae and heterotrophic bacteria, algae and protozoa. The system in early spring is thus microbial and the model considers only the growth, substrate utilisation, and photosynthetic activities of microbes. These activities are influenced by temperature, substrate availability, and light, which are all in a dynamic state during the non steady state condition as depicted in Figure 2.

Development of a temperature model: Temperature in the water column is influenced by air temperature and direct irradiance of the surface water. Experimental data from many sources for shallow lakes and ponds, where a thermocline does not develop, shows a gradual reduction in temperature with depth. Our experimental observations show that daily air temperature fluctuations are mirrored by changes in water temperature, but with a slight lag and an increasing damping effect with increasing depth. The daily air and water temperature data has a characteristic sinusoidal form (Figure 3), which is applicable to any daily temperature range and can be approximated for specific locations by the following equation:

$$f(t) = a_1 \sin\left(\frac{2\pi}{T_1}(t - t_{10})\right) + a_2 \sin\left(\frac{2\pi}{T_2}(t - t_{20})\right) + b \qquad (1)$$

Values can be derived for a_1, t_{10}, a_2, t_{20}, and b using initial values of 24 for T1 and 12 for T2. Examples for Almaty, Kazakhstan are given in Figures 3 and 4 with normalised values. These profiles are generally applicable unless extreme changes occur over a short period. Variation in air temperature over longer periods, for example a year, can also be fitted to the same equation using a different set of parameter values. Knowing daily average air temperatures, it is therefore possible to simulate the average temperature behaviour at different depths in the water column as in Figure 5.

Biological growth and oxygen balance model. The model is based on Buhr and Miller (1983) and considers the growth of both bacteria and algae. It uses the Monod equation to relate the specific growth rate of a microbial population and the limiting substrate concentration (Monod, 1949).

$$\mu_B = \frac{\mu_{\max} S}{K_S + S} \qquad (2)$$

For bacterial growth there are two important components: substrate and oxygen, so the growth rate μ is expressed as the product of two Monod equations:

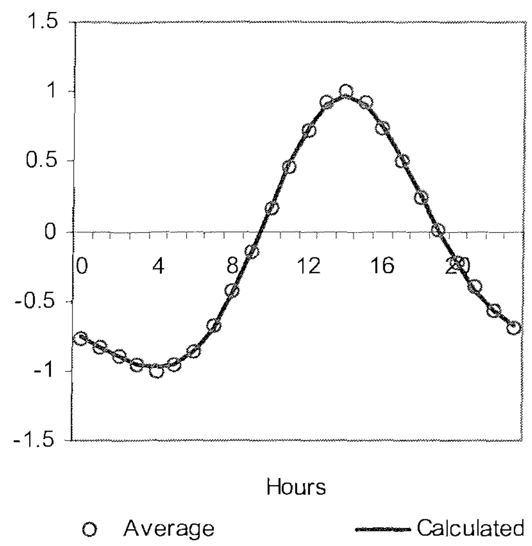

Figure 3 Almaty hourly air temperature (normalised)

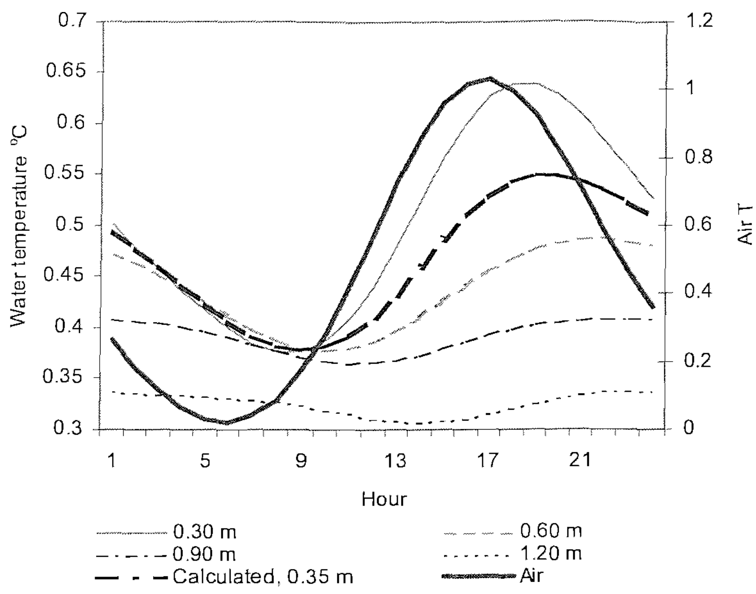

Figure 4 Average hourly water temperature (normalised to [0..1] range). Almaty Summer 1999 data

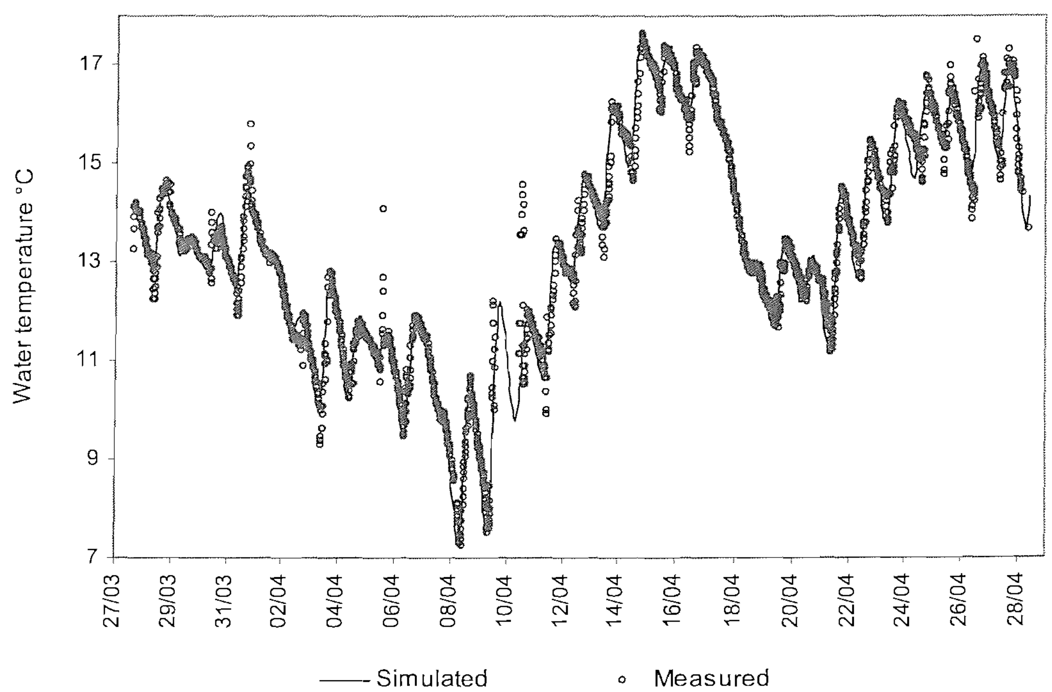

Figure 5 Comparison of measured and simulated data for Almaty experimental pond

$$\mu_B = \mu_{\max B} \cdot \frac{S}{K_s + S} \cdot \frac{O_2}{K_{O_2} + O_2} \tag{3}$$

where μ_B = bacterial specific growth rate in days^{-1}; μ_{max} = maximum bacterial specific growth rate in days^{-1}; S = substrate concentration in mg l^{-1}; K_s = half saturation substrate concentration in mg BOD l^{-1}; K_{O_2} = half saturation oxygen concentration in mg l^{-1}. The rate of change of substrate concentration depends on substrate inflow, outflow and bacterial consumption and is given by

$$\frac{dS}{dt} = \frac{F}{V}(S_o - S_e) - (\mu_B \frac{X}{Y}) \tag{4}$$

where S = substrate concentration in mg l^{-1}; F = flow to reactor in l day^{-1}; V = reactor volume in l; S_o = influent or initial substrate concentration in mg l^{-1}; S_e = effluent or final substrate concentration in mg l^{-1}; X = bacterial biomass density in mg l^{-1}; Y = yield coefficient for bacteria in mg mg^{-1}. The rate of change of bacterial biomass density is dependent on inflow to and outflow from the reactor, and growth and die-off within the reactor, and is given by

$$\frac{dX}{dt} = \frac{F}{V}(X_o - X_e) + \mu_B X - k_{DB} X \tag{5}$$

where k_{DB} = endogenous decay coefficient for bacteria in day^{-1}, X_o = influent bacterial biomass concentration in mg l^{-1}; X_e = effluent bacterial biomass concentration in mg l^{-1}.

For algal growth, the 'substrates' are CO_2 and light. The Monod equation is used for CO_2 (Kayombo *et al.*, 2000), while the effect of light is modelled by multiplying μ_{max} by two functions. The function L(t) allows for the variation in light intensity during the day, and can be modelled by theoretical or experimental data. Light intensity is also reduced by the presence of suspended solids in the water, in accordance with an exponential relationship. The amount of light scattered and absorbed depends on the nature of the suspended solids (Kirk, 1994; Curtis *et al.*, 1994). The algal growth rate is therefore given by

$$\mu_A = \mu_{\max A} \frac{CO_2}{k_C + CO_2} . e^{-F_D (S_D.S + B_D.X + A_D.A)} . L(t) \tag{6}$$

where μ_A = algal specific growth rate in days^{-1}; μ_{max} = maximum algal specific growth rate, days^{-1}; CO_2 = CO_2 concentration in mg l^{-1}; K_c = CO_2 half saturation concentration in mg l^{-1}; F_D = a factor representing scattering and absorption in l mg^{-1}; S_D = substrate density factor; B_D = bacterial density factor; A_D = algal density factor (representing the relative influence of each component).

The rate of oxygen production is a function of the production of new algal and bacterial biomass

$$\frac{dO_2}{dt} = \frac{F}{V}(O_{2_o} - O_{2_e}) + Y_{OA}\mu_A A - Y_{OB}\mu_B X \qquad (7)$$

where O_{2o} = influent oxygen concentration in mg l^{-1}; O_{2e} = effluent oxygen concentration in mg l-1;Y_{OA} = oxygen production yield coefficient for algae in mg^{-1}; Y_{OB} = oxygen consumption yield coefficient for bacteria in mg^{-1}. The rate of CO_2 production is related to the rate of oxygen consumption and is similarly a function of the rate of production of new biomass

$$\frac{dCO_2}{dt} = \frac{F}{V}(CO_{2_O} - CO_{2_e}) + Y_{cB}\mu_B X - Y_{CA}\mu_A A \qquad (8)$$

where CO_{2o} = influent CO_2 concentration in mg l^{-1}; CO_{2e} = effluent CO_2 concentration in mg l-1;Y_{CB} = CO_2 production yield coefficient for bacteria in mg^{-1}; Y_{CA} = CO_2 consumption yield coefficient for algae in mg^{-1}.

The rates of growth of bacteria and algae are highly temperature-dependent. The model takes this into account by adjusting values of μ_{max} according to the following equation:

$$\mu_{max_{(t)}} = \mu_{max_{20}} \theta^{(t-20)} \qquad (9)$$

The user can choose typical values for μ_{max} and θ from the literature, or can input experimental data. The model requires input values for the coefficients defined above. The user enters the start date and time, the duration of the simulation and the time step size. The model can operate in both batch feeding and continuous flow modes. In batch mode it checks at the start of each time step to see if feeding is scheduled. For each time step the average water temperature is updated and the model checks the current value of all components. It calculates current values of μ for algae and bacteria with equation 9. It then calculates the change in substrate concentration, bacterial biomass, algal biomass, oxygen and CO_2 consumption and production using equations 3-8. The change in each value is compared with the current value to check that a negative total has not been generated. If a negative total occurs, it is reset to zero. For example, if the initial oxygen concentration is 2 mg l^{-1} and the calculated oxygen consumption in the given time period is 4 mg l^{-1}, the demand cannot be satisfied so it is assumed that all the available oxygen is consumed and a correspondingly smaller amount of substrate is utilised. The model then recalculates the amount of substrate and the yield of cells.

RESULTS AND DISCUSSION

The model was run to simulate the batch operation of a small-scale experimental WSP in Almaty during the spring warm-up period. The experimental data for this pond and the mode of operation are reported in Pak *et al.* (2002), although some assumptions are made in the modelling as to the initial substrate concentration in the pond. Experimental data suggested that this was quite high due to remobilisation of soluble material from the bottom sediments, despite the pond not receiving wastewater since ice formation in late November of the previous year. The model parameters were thus set to give an initial substrate concentration of 500 mg l^{-1} and a daily feed concentration of 300 mg l^{-1} at a 30-day retention time. Values for the constants used in the model were those suggested by Buhr and Miller (1983) with the exception of the value for μ_{max} and the temperature coefficient θ which were taken from the work of Bartosh *et al.* (2002); there was little difference, however, between these and the earlier work. The temperature simulation in the model used the approach described earlier in this work and the temperature profile as shown in Figure 5. Values for the initial numbers of bacteria and algae are difficult to ascertain experimentally because the viability of organisms remaining after ice formation is unknown. Values of 10 mg l^{-1} were used for both bacteria and algae, and this appeared to give a reasonable approximation to observed values.

It is clear that temperature has a substantial effect on degradation with a low rate occurring in the first period up until 14 April and then an acceleration in rate as the temperature increased above 15°C eventually rising to a daily mean of 25°C. Oxygen concentrations were limited until almost the middle of May and substrate concentration levels did not reach a steady state until about the same time. The model equilibrium time was earlier than was observed in practice, where the pond was found to reach steady state in the middle of June. The other difference lies in the final steady state concentrations of algal numbers, where the model predicts higher values than were found in reality. This may account for the more rapid BOD removal in the model. Adjustment of model constants could reduce these levels to give a better fit. This will be done in later sensitivity studies, but has not been attempted here: the values of the constants used are those typically found in reference books and reported by other researchers.

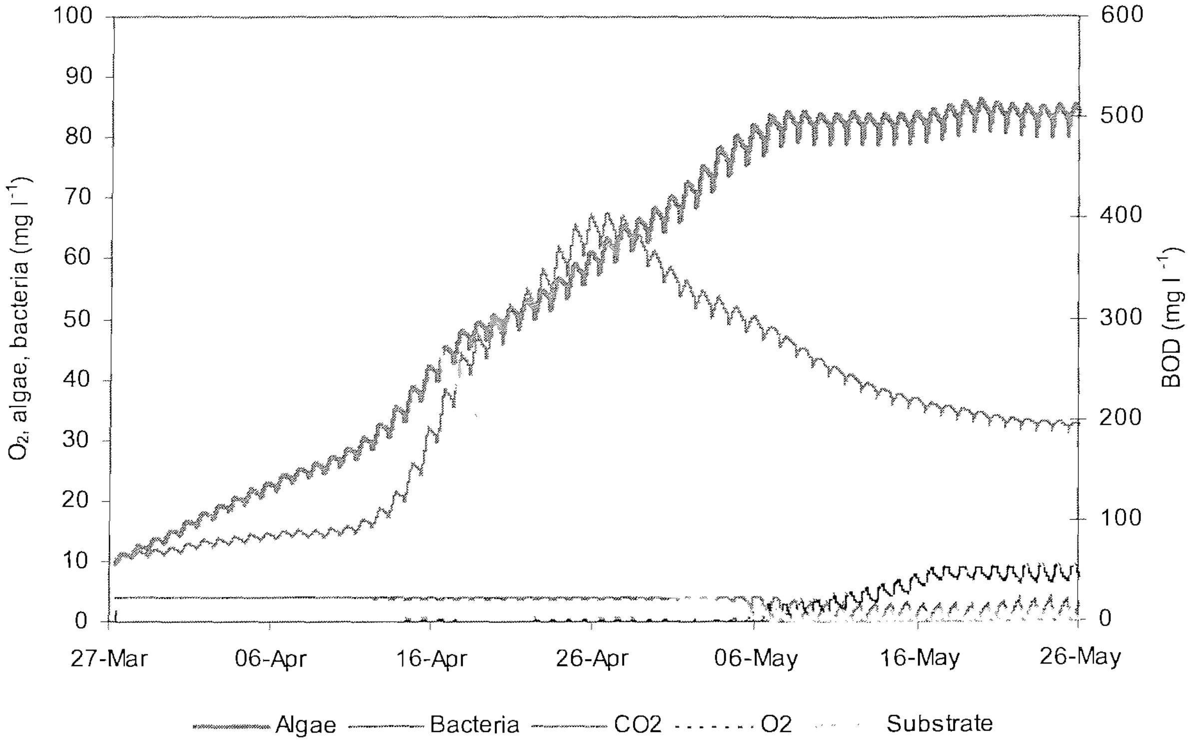

Figure 6 Results of a typical model simulation run for experimental small-scale WSP in Almaty

Table 1 Model parameters used in simulation

Parameter	InFlow	Pond	Unit
Volume	26	750	litres
Algae	0.00001	10	mg l^{-1}
Bacteria	0.00001	10	mg l^{-1}
Substrate	200	500	mg l^{-1}
CO_2	2	4.67	mg l^{-1}
Oxygen	2	1	mg l^{-1}
N	7	7	mg l^{-1}

Constants	Values	Units
$\mu_{max A}$	1.13	days^{-1}
$\mu_{max B}$	4.95	days^{-1}
Y	0.4	(mg bacterial cells).(mg BOD consumed)$^{-1}$
Y_{CA}	2.1824	(mg CO_2 consumed).(mg algae)$^{-1}$
Y_{OA}	1.5872	(mg O_2 produced).(mg algae)$^{-1}$
Y_{CB}	3.432	(mg CO_2 produced).(mg bacteria)$^{-1}$
Y_{OB}	2.496	(mg O_2 consumed). (mg bacteria)$^{-1}$
K_S	150	mg BOD l^{-1}
K_{O2}	0.128	mg l^{-1}
K_C	0.044	mg l^{-1}
K_{DA}	0.05	1 days^{-1}
K_{DB}	0.1	1 days^{-1}

CONCLUSIONS

The fact that there is reasonable agreement without any 'fitting' of values to experimental data derived from the ponds indicates that the model is inherently robust.

The model is still in its development stage and at present operates as a single cell in which the concentrations of substrate and organisms are uniform throughout. It is our intention to refine the model to adopt a multiple cell approach in which changes can be predicted at different depths in the pond. This will require further work on understanding and modelling the light penetration within the system and the mass transfer of gases and nutrients between different depth zones. The model could also be adapted to take into account of plug flow conditions by using cells in series with mass transfer in the longitudinal plane.

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Section A2: Matching waste stabilisation pond outputs to irrigation needs in a continental climate region

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Keywords: continental climates; irrigation; wastewater re-use

The Central Asian region has a sharply continental climate characterised by cold winters, hot summers and very low precipitation, making water a valuable resource. Waste stabilisation ponds (WSPs) offer an effective means of wastewater treatment, but uptake of the technology is hindered by the lack of appropriate design and operational protocols to facilitate water re-use. Existing protocols for pond systems that are subject to freezing in winter are mainly from north America, and recommend treatment followed by storage of 6 or 12 months. This provides a high standard of environmental protection, but in warmer and drier regions a more flexible approach may be appropriate. Ensink *et al.* (in review) noted the importance of climatic factors in the design of WSP systems for reuse in irrigation. Heaven *et al.* (in press) suggested that tailoring design more closely to local climate conditions can increase water availability and lead to a more economical solution.

The current paper describes an approach linking pond design with irrigation re-use, using an existing WSP model coupled with an irrigation software package. The WSP model developed by Heaven *et al.* (in press) uses a simple mass-balance approach assuming first order kinetics for Biochemical Oxygen Demand (BOD) and Faecal Coliform (FC) die-off: for the current work this was modified to include calculation of salinity. Irrigation scheduling was carried out using the FAO's CROPWAT package, which includes extensive databases of crop requirements and yield parameters. Output from each model is used iteratively to produce a suitable pond design. Modelling locations were chosen from Kazakhstan as the country covers a wide range of latitudes and climate types typical of central Asia, and includes large population centres in water-poor regions (Table 1). The models were run using average values for climate parameters, and results compared with multi-year data where available. Scenarios considered included irrigation of wheat, potatoes and small vegetable crops in the north and centre of the country, and cotton, maize and lucerne in the south. Data on current wastewater treatment and re-use was obtained from the Ministry of Environmental Protection Republic of Kazakhstan.

The results indicate that the intended re-use of wastewater can have a significant effect on pond design and operation, and that this varies according to local climatic factors, as illustrated in the following two examples. In the first, Figure 1 shows pond depth and FC concentration for a WSP in Petropavlovsk designed to provide water for irrigation of potatoes and wheat. In this northern region there is some precipitation during the summer, and a typical cold-climate design of a small facultative pond followed by a larger storage/maturation pond performs adequately. Potatoes have a small rooting depth and so require frequent smaller irrigations, while wheat is irrigated later and less often. Using a pond of the same area for these two crops gives a 10% difference in maximum water depth. The effect of the different irrigation regimes can also be seen in the modelled FC concentration, as the dilution of incoming water from the facultative pond is affected by the degree of emptying of the maturation pond.

The second example is from the south of Kazakhstan, where low rainfall and very high summer evaporation produce acute water shortages for agriculture. After the water needed for irrigation has been withdrawn, there is a period during which evaporation is greater than precipitation plus inflow into the storage pond, which thus dries up. This is not necessarily a problem for BOD or FC, which may be oxidised; but incoming salt accumulates. When evaporation falls in late summer, the pond starts to refill and a peak in salinity occurs that may affect treatment performance over several weeks (Figure 2). Additionally the standard 2-pond design is inefficient, as the large area of the storage pond leads to high evaporation, while the fact it runs at low depth for long periods represents waste capacity. From the modelling results it appears that WSP design in southern areas is likely to be evaporation and salinity-driven. One potential design alternative is a 3-pond system where the third pond is sized to allow it to dry up and thereby reduce evaporation losses, while water can be withdrawn from the second pond once it reaches the required standard.

The full paper discusses the potential scale of irrigation and the implications for technology and crop choice. The results confirm the power of such tools to assist in the development of rational design protocols for promoting wastewater reuse in continental regions.

Table 1 Cities in Kazakhstan with population and current wastewater treatment

	City	Lat	Long	Population	Current wastewater treatment	Reuse
North	Petropavlovsk	54.8	69.2	203,000	full and tertiary then storage ponds	none
	Kustanai	53.2	63.6	221,000	mechanical then storage ponds	none
	Pavlodar	52.3	76.9	300,000	aeration tanks, then to ponds	<3%
Central	Ural'sk	51.2	51.3	195,000	mechanical then ponds	not reported
	Astana	51.2	71.4	313,000	full then storage ponds	not reported
	Semipalatinsk	50.4	80.2	270,000	mechanical then filtration fields	none
	Aktjubinsk	50.3	57.2	253,000	full then storage ponds	none
	Ust-kamenogorsk	50.0	82.5	311,000	biological then ponds	not reported
	Karaganda	49.8	73.1	437,000	full and tertiary ponds then river	not reported
South	Almaty	43.2	76.9	1,130,000	Full, to pond	small
	Turkestan	43.3	68.3	85,000	biological then filtration fields	not reported
	Shimkent	43.3	68.2	360,000	biological then storage ponds	<1%
	Taraz	42.9	71.4	330,000	filtration fields	none

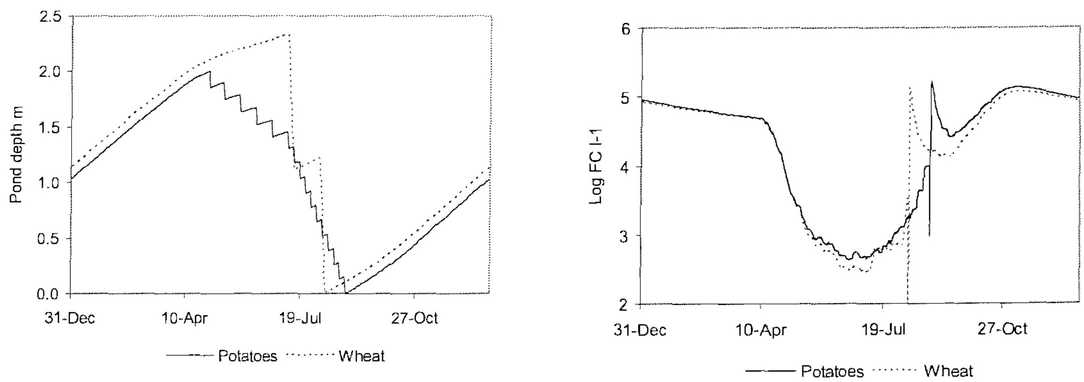

Figure 1 Pond depth and FC concentration for irrigation of wheat and potatoes in Petropavlovsk

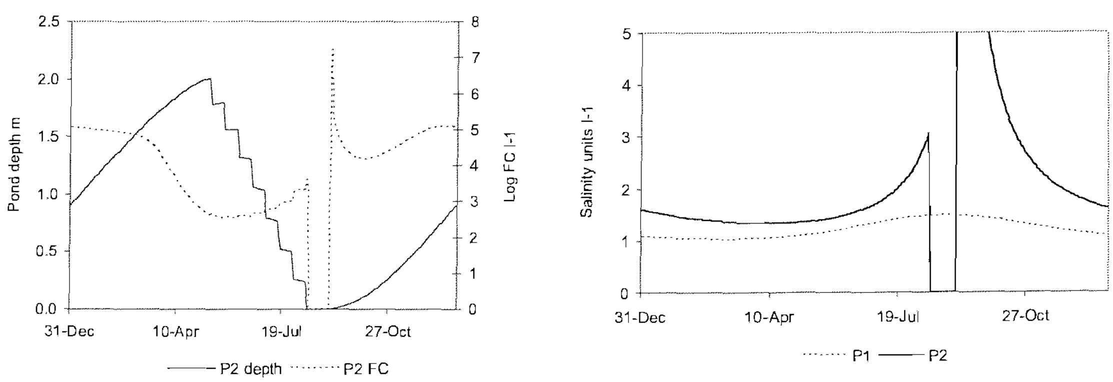

Figure 2 Pond depth and salinity for irrigation of lucerne in Turkestan.

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