Collaborative Model for Regional
Water Quantity and Quality Management

by

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I hereby declare that, except where specifically indicated, the work submitted herein is my own original work.

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Technical Abstract

Global water resources are facing increasing pressure from rapidly growing population, economic growth, water pollution, land-use changes and climate change. The uncertain future of growing demand and external factors such as climate change provide a continuing challenge for the management of water resources to balance environmental and developmental needs. The future uncertainty has led to increasing interests in modelling of water resources particularly at a river-basin level.

The challenges facing policy-makers from competition for limited resources require adaptation strategies that take into account the trade-offs and concerns of different stakeholders. Several hydrological models are available for a wide range of applications. However, many of them require large data inputs and time to run often making them impractical for use as decision tools. There is a need to develop simple practical tools that enable water resource managers to better collaborate with stakeholders in decision-making processes to understand the outcomes of various management strategies under future uncertainties.

This project develops a simple catchment-scale water balance and quality model that can be used as a collaborative tool between multiple stakeholders. The model is applied to the Mekong River Basin in South-East Asia. The Mekong is facing several challenges as a result of rapid population growth and economic development. In particular, hydropower development throughout the basin and increasing irrigation demand for agriculture coupled with climate change are causing rapid changes that will affect future land and water resources. These could have significant consequences for the basin population who rely on these resources threatening livelihoods and food security in the basin.

The model is based on the long-term water balance concept combined with the rational method to calculate annual and seasonal runoff, which were calibrated against runoff data. The water quality is based on the mass balance concept for calculating the chemical loading in the runoff, using the calibrated flows from the hydrology model. These were calibrated and validated with water quality data for Chemical Oxygen Demand, Nitrates and Total Suspended Solids. The Chemical Oxygen Demand and Total Suspended Solids models were further improved by applying an exponential decay equation.

A water quality index (WQI) was developed based on international drinking water standards. The WQI provides a simple method of conveying water quality information in a

concise manner. Whilst the WQI means some of the quality model preciseness is lost, it has its advantages in representing the data in a simple manner allowing comparisons to be made readily. This is particularly important in using the WQI for decision-making purposes. It can easily be adapted for other water uses and purposes.

The project explores how the model can be used to consider various possible scenarios in the Mekong affecting water flows and water quality, including climate and land use changes and hydropower development. The hydropower scenarios demonstrate how the tool could be used by multiple stakeholders to communicate their perspectives of the impacts of the development on them. This was explored through the use of hypothetical WQIs, adapting the WQI for drinking water quality, to reflect the views of a few key stakeholders including subsistence farmers, local fishermen and hydroelectric engineers. These were based on evidence and opinions about the likely impacts of the dams on total suspended solids in the water and the consequences of the development on each stakeholder. This could allow them to collaborate in dialogue over the trade-offs required to achieve equitable and sustainable resource allocation.

The simplicity of the model structure with few parameters and limited data requirements allows the model to be readily calibrated. The nature of the calibration process and given the sparse data requirements means that more than one calibration is possible, with different combination parameters giving slightly different yet still valid calibrations. It was demonstrated how the final calibration can be influenced by the modellers' personal views and opinions and therefore vary between different users. The basic model structure means that it could be transferred to other study areas and be used to explore water quantity and quality issues. The model's simplicity means that it cannot provide precise results or answer specific questions about the catchment. However, it could be useful as a collaborative tool in decision-making processes to facilitate stakeholder participation in understanding the full impacts of the possible uncertainties and risks and to plan more informed future management strategies to minimise them.

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1. INTRODUCTION

1.1 Background and Motivation

Water resources are facing growing pressures as a result of global changes including climate change, population growth, economic development, pollution, land-use changes and management (Ludwig & Moench, 2009). These have a profound impact on hydrological systems and will continue to affect both water availability and water quality in the future.

There are many uncertainties that affect policy makers in deciding on water management strategies. Societies are constantly changing and adapting, thus there is great uncertainty in projecting socio-economic trends (Arnell, 1999). Additionally, there is great uncertainty in climate modelling. These changes can have wide consequences on hydrological systems. In some areas these anthropogenic factors have and will be miniscule compared with climatic changes, but it others they are critical and poses significant threats to local and regional water resources (Jones, 1997). This leads to challenges in deciding how to adapt resource management to be more resilient and minimise the associated risks of adverse impacts (UNESCO-WWAP, 2012).

However water managers lack clear guidance on how to respond to the challenges of future uncertainties and risks especially on a river basin level (Aerts & Droogers, 2004). Policy makers globally are faced with difficult decisions about how best to use their natural resources. Modelling of different scenarios using analytical tools enables policy-makers to better understand the range of uncertainties and risks so that they can plan, design and manage water systems to reduce the risks and adapt to the possible changes (UNESCO-WWAP, 2012).

The need to treat regional water resources holistically and more sustainably, distributing them equitably between multiple regions and sectors has led to the development of the Integrated Water Resources Management (IWRM) approach (UNESCO, 2009). This emphasises the need to manage water resources at the river-basin scale as a geographical and hydrological unit. Water resources management at a basin level facilitates the integration of management strategies for upstream and downstream issues relating to both water quantity and quality as well as addressing the social, economic and environmental development of the basin (UNESCO, 2009). An adaptive approach to water management requires active participation from all interested stakeholders involved in the decision process (Loucks, et al., 2005).

Water resources are rarely managed in an integrated way across transboundary borders and regions or across water-use sectors (UN-Water, 2008). With growing population, development and resource-use tensions between stakeholders are likely to increase. Hence, river basin management objectives need to balance the interests of the different stakeholders and users in enabling development in the basin, whilst also improving and preserving environmental quality (Gray, 2005). This requires practical modelling tools that allow policy-makers and other stakeholders to collaborate in planning short and long-term management strategies (UNESCO-WWAP, 2012) that allocate resources sustainably and equitably between all stakeholders (Loucks, et al., 2005).

1.2 Project Objectives

The objectives of the project are:

1. Develop a simple catchment-scale integrated water quantity and quality model that can be used as a complementary tool for decision-making.
2. Develop a water quality index as a simple method of conveying water quality information from within the model.
3. Explore how the model performs in different scenarios and establish its strengths and limitations in being used as a tool.
4. Illustrate how the model can be used to facilitate communications between the different stakeholders concerned about particular decisions in the catchment.

1.3 Existing Work

Numerous models have been developed for water resources management. The following section provides a brief overview of some of the modelling approaches that have been developed.

Models are invaluable tools for water resource management as they can help managers develop an increased understanding of complex natural systems allowing them to test different scenarios and predict the outcomes with high risk and prioritise management and adaptation strategies to deal with them (Caminiti, 2004). The purpose and aim for which the model is to be used determines the type of model that is developed. They are often developed to support management decisions related to current and future catchment hydrology and the wider catchment based on changes in climate, land-use, population and economic growth, irrigation demand and pollution (Pechlivanidis, et al., 2011).

Several hydrological models are available for a wide range of applications including water resources planning, development and management, flood prediction and infrastructure design. Some of these are coupled with water quality for ecological, pollution and climate change applications (Pechlivanidis, et al., 2011). River-basin level models are categorised into simulation models and optimisation models. The first simulate the behaviour of various hydrologic, water quality and other variables under different stationary and changing conditions, whilst optimisation models apply different water allocation and economic allocation objectives to the basin model and are also be used to support decisions about how to allocate water resources equitably between multiple sectors (McKinney, et al., 1999).

Several water balance models have been developed at various time scales ranging from hourly to yearly and to varying degrees of complexity (Xu & Singh, 1998) and are used to study a range of different hydrological problems. Lumped conceptual rainfall-runoff models are commonly used for modelling runoff in large areas such as an entire lake or river basin (van der Perk, 2006). They are generally used when spatial variation is either insignificant or irrelevant. Distributed models, take into account the spatial variation within environmental processes and can range from dividing the region into a few sub-regions as linked lumped models to dividing it up into a complex grid system (Ringman, 2002).

Numerous catchment runoff models have been developed for simulating watershed hydrology for which several commonly used hydrological models are given in Singh & Woolhiser (2002). Many of the current models have the capability to accurately simulate catchment hydrology and can be applied to address a wide range of environmental and water-resource problems, however they are not readily usable as practical tools for planning and decision-making (Singh & Woolhiser, 2002).

Many river-basin models are designed to look at water quantity issues relating to water use and availability (McKinney, et al., 1999) often without addressing water quality. But in recent decades the pressure on water resources has risen and increasing pollution of water resources are having adverse environmental effects. This has have resulted in a growing effort to develop water quality modelling tools for water resources management at a river basin scale (Pinho, et al., 2011)

Several water quality models exist ranging from the simple mass balance models to complex models capable of simulating three-dimensional dynamic flow for a range or organic and inorganic compounds. Kannel et al. (2011) and Cox (2003) review a number of water quality models in the public domain that simulate oxygen levels in rivers, ranging from the simplest such as SIMCAT based on a flow-solute mass balance to the complex models such as QUAL2EU and QUASAR simulating nutrients, algae and organic chemicals (Cox, 2003). The more complex models are able to provide realistic simulations of the numerous processes and their interactions affecting water quality and are necessary for understanding localised impacts of specific changes to water quality and to support decision-making to improve the water quality (Kannel, et al., 2011). However, these are generally too complex and very data intensive for understanding catchment-scale issues and develop appropriate policies.

Water resource quantity and quality issues are often treated separately and by different authorities with often conflicting objectives (Dai & Labadie, 2001). This is such that a number of models deal with issues related to each separately, so there is a lack of integrated tools for water quantity and quality assessment. Recently, several integrated models combining water quantity and quality have been developed such as Jia et al. (2007), Zhang et al. (2010) and Xia (2012).

Although several models exist for modelling catchments, a number of these have been developed for specific purposes and in some cases with personal or commercial interests, which determine the approach in the model development and the outcomes (Caminiti, 2004). There has been growing interest in developing generalized models for application for a range of issues in river systems and for various locations rather than being site-specific and customised to a particular system (McKinney, et al., 1999). Complex models are capable of simulating the impacts of dynamic natural processes on large watersheds on a short time-scale. However, their complexity and the time and effort required to run them makes them difficult for policy makers and resource managers to make effective use of them as decision tools (Fernandez, et al., 2002). In addition to the large data inputs and time required to run complex models, the underlying uncertainties inherent in these tools contribute to the uncertainties in the predictions. Thus the choice of model highly depends on the availability of data to the user, the system to which it is applied and application for which it is desired (Fernandez, et al., 2002).

As water management transitions towards a more adaptive, sustainable approach, there needs to be more communication and dialogue between the different stakeholders concerned in understanding and addressing their different values and perceptions of risks and concerns (UNESCO-WWAP, 2012). However, each stakeholder has their own objectives, interests and agendas which may be in conflict with others. Collaborative modelling methods can assist in this process to reach a common understanding and agreement among stakeholders, with greater likelihood of success if the stakeholders are themselves involved in the modelling and analysis processes (Serrat-Capdevila, et al., 2011). Simple models allow a better understanding of stakeholders' concerns, but also give them a common vision of how the system works, as represented by the model (Loucks, et al., 2005) Such decision support tools allow resource managers and other stakeholders to better engage in understanding the risks associated with various uncertainties in the catchment and develop more informed policies to overcome them (UNESCO-WWAP, 2012).

2. DATA AND METHODS

2.1 Data and Study Location

2.1.1. Study Location

Geography **-** The Mekong River in South-East Asia is one of the world's largest river basins with an area of 795,000 m^2 and is shared between six countries along its 4,800 km length. It originates in the Tibetan Plateau at an elevation of 5160 m flowing through the Yunnan Province of China, Myanmar, Lao PDR, Thailand, Cambodia, and Vietnam and into the South China Sea, as shown in Figure 1. The Lower Mekong Basin (LMB) covers 77% of the catchment area and is defined as beginning at the common border of Myanmar, Thailand and Laos (Ringler, 2001). It is the only large river globally that still flows freely to the sea through five of the six riparian countries that share it (Chheang, 2010). The Mekong River Commission (MRC) was established by Cambodia, Lao PDR, Thailand and Vietnam in 1995 with the four countries agreeing to cooperate in a mutually beneficial manner for sustainable development of the Mekong River and its resources. China has significant influence over all the countries in the LMB which are downstream, but is not a member country of the MRC. The LMB is considered the most important part of the basin, both economically and environmentally (Hoanh, et al., 2003), and several issues are emerging relating to the basin's water usage including: irrigation development in Northeast Thailand which is resulting in dry season water shortages, saltwater intrusion in the delta, flooding risks, hydropower development and growing populations and increasing economic development affecting water availability and water quality (Hoanh, et al., 2003).

Figure 1: Map of the Mekong River Basin (MRC, 2007a)

Climate - The Mekong experiences a tropical monsoonal climate which dominates the hydrology. The southwest monsoon brings very high rainfall from May to September and the northeast monsoon gives low rainfall from November to March, with a transition period between each. Regional rainfall varies significantly with an annual rainfall of 1,000 – 1,600 mm in the driest regions in Northeast Thailand to 2,000-3,000 mm annually in the Northern and Eastern Highlands in Laos and Vietnam (Hoanh, et al., 2003). The temperature is quite uniform throughout the year and the basin varying from 20 to 30°C with the lowest temperatures during the middle of the dry season. The Tonle Sap Lake in Cambodia has a large impact on the hydrology in Cambodia and Vietnam. During the wet season the river flows upstream into the lake increasing its area from 3,000 km^2 to 13,000 km^2 forming wetlands and productive fisheries; in the dry season the flow is released back into the Mekong.

Water Resources **-** The Mekong Basin has abundant water resources; however availability varies widely by country, by region within countries, and by season. Groundwater resources in the basin have not been adequately assessed, but they are mainly used for domestic consumption and industry and some for irrigation (Hoanh, et al., 2003). The

Mekong River Delta in Vietnam has abundant groundwater resources allowing three cropping seasons of rice each year.

Water Quality - The water quality in the mainstream is generally good, although the concentration of suspended solids is high, especially in the wet season (MRC, 2010). Poorer water quality has been observed at a local level in particular tributaries, but not at a basin-wide scale (MRC, 2007a). While no obvious signs of transboundary pollution have been observed between the Lower Basin countries, there has been some evidence of changes to water quality in China as a result of economic growth and hydropower development which could affect water quality for the entire LMB downstream (MRC, 2007a). Salinity is a particular issue, especially in Northeast Thailand due to rock salts. The delta in Vietnam also experiences saltwater intrusion, which could be exacerbated by rising sea levels and a reduction in flow rate from upstream for irrigation abstractions.

***Natural Resources* -** The Mekong Basin has five key areas of natural resources and development: agriculture, fisheries, hydropower generation, fisheries, forest resources, conservation and tourism (Hoanh, et al., 2003). During the last fifty years high deforestation rates due to excessive logging, shifting cultivation, farming and infrastructure development have led to high sediment loads entering the river in all of the basin countries.

Agriculture and fisheries are the dominant economic sectors in the basin with 75% of the population dependent on them for their livelihoods with almost half the basin population living within 15km of the river and relying on the basin's fisheries, riverbank gardens, and fertile land for agricultural productivity and food supply (Pearse-Smith, 2012) where rice is the main crop grown and fish is the primary source of protein. The population is expected to rise from 60 million at present to 90 million in 2050 (Kirby & Mainuddin, 2009), which could have impacts on future food security. Agriculture, fisheries and forestry sectors contribute between 15-60% of the GDP of the countries sharing the Lower Mekong (Hoanh, et al., 2003); hence reduced productivity will significantly affect the economies of the LMB. The Mekong Basin is the largest inland freshwater fishery in the world (Pearse-Smith, 2012) in which over 850 species have been recorded (MRC, 2010). Rapid development threatens to dramatically affect several species in the river including the rare Mekong Giant Catfish and the Irrawaddy Dolphin, which could become extinct. The impacts will be especially noticeable in the Tonle Sap Lake in Cambodia where the production of fisheries is correlated with the magnitude of the flood (Kirby, et al., 2010).

Hydropower development - Rising energy demands and an estimated 30,000 MW of stored potential hydropower (MRC, 2010) in the LMB, make hydropower development in the basin an attractive solution. The constructions of several dams are planned over the next few decades. These will mostly be built in Laos and Cambodia providing economic stimulus for these countries; however most of the power will be exported to the cities in Thailand and Vietnam. These will have major impacts on the basin, with some regions and sectors benefitting and others losing out, and will continue to affect water resources for all the riparian countries. They have attracted large amounts of opposition within the basin and also internationally, especially the Xayaburi Dam in Lao PDR which is the first out of eleven proposed dams to begin construction of the Lower mainstream. The start of construction of the dam in Laos has led to rising tensions between the riparian countries which share the Mekong.

2.1.2. Model Data Inputs

The model was calibrated using long-term average monthly and annual runoff data for 1960-2004 from the MRC (2005) for seven monitoring stations along the Lower Mekong mainstream river. The upper basin is not modelled due to the complexities involved in the upstream hydrology; however the changes in China can affect the whole LMB and must nevertheless be taken into consideration during the model testing. It would also require data from alternative sources, which may not be compatible with the data used for the LMB.

Monthly precipitation and temperature data was obtained for most of the monitoring stations, with some data available at other locations from the MRC (2005). Temperature was assumed to be the same across the basin varying with latitude. These were compared with a rainfall map of the basin (Appendix A) to estimate the average precipitation over the land area contributing to the runoff between each monitoring station. Data for the larger sub-basin areas and flows between the seven monitoring stations were obtained from the MRC (2007b) and aggregated to find the total land area from which runoff contributed to the mainstream flow between each location.

The water quality model was calibrated using data from the United Nations Environment Programme (UNEP) GEM/Water project for the time period 2000-2008 and validated with data from the time period 1985-1994. Land use distribution data was obtained from the ADAPT project (Hoanh, et al., 2003) for different land uses from 1992-1993 (Appendix B). These were combined into six land uses as given in Table 1. Each land use type has given typical runoff coefficients and a pollutant loading profile for COD, NO3 and TSS based on typical literature values combined from various sources.

The year 2000 was set as the baseline for the calibration. However, limited data availability meant that time periods do not completely overlap, which introduces bias into the model. The spatial and temporal variability in the rainfall and temperature and the aggregation of the data to fit into the model structure adds uncertainty to the data (Pechlivanidis, et al., 2011).

Table 1: Summary of land uses, runoff coefficients and concentration loadings used in the model[1]

Land Use		Runoff Coefficient	Unscaled Concentration Loadings (mg/l)		
			COD	**NO3**	**TSS**
1	Urban	0.5	65.0	0.6	67.0
2	Industrial	0.7	57.0	0.6	69.0
3	Forest	0.2	15.0	0.3	109.9
4	Shrubland	0.25	15.0	0.6	723.6
5	Agricultural	0.3	32.5	0.9	804.0
6	Wetlands	0.1	16.0	0.2	110.0

2.2 Water Balance Model

Lumped conceptual rainfall-runoff models have been widely used in hydrology to describe the most important processes in a catchment. These require calibration of parameters which can't be measured directly in the catchment. The Rational Method is a widely used method and the simplest for calculating runoff:

$$Q = C*P*A \tag{1}$$

where C is the runoff coefficient constant of proportionality relating the runoff, Q, with precipitation, P, and A is the catchment area (Jones, 1997).

Although this is generally more suited to small urban catchments for peak flows (Shaw, 1999) the principle can still be applied for the rural catchment with different runoff coefficients reflecting the land use distribution in the catchment, as given in Table 1.

[1] Runoff coefficients based on Jones (1997) and Gray (2005), and concentration loadings based on Laws (2000), Elrashidi et al. (2005) and Udeigwe & Wang (2010).

Water balance models are based on the long-term average water balance equation:

$$Q = P - ET \pm \Delta S \qquad (2)$$

Where *Q, P* and *ET* are the long-term average annual runoff, precipitation and actual evapotranspiration respectively, with the change in storage (*ΔS*) is considered negligible (Jones, 1997) in the long-term.

Whilst water balance models are based on this equation they differ widely in how they simulate evapotranspiration and storage terms. The actual evapotranspiration is related to the potential evapotranspiration for which several empirical methods exist for calculating.

Equations (1) and (2) were combined to produce an annual water balance incorporating precipitation, temperature and land use:

$$Q = (P - k_1 * PET) * C * k_2 * A \qquad (3)$$

where the effective precipitation is given by:

$$P - k_1 * PET > 0 \qquad (4)$$

k_1 and k_2 were used to calibrate the modelled runoff to observed data. The overall model structure of the water balance and water quality model is given in Figure 2.

Although it is possible for the evapotranspiration to exceed precipitation during the dry seasons, on an annual basis the precipitation will be greater.

Potential evapotranspiration (*PET*) is one of the main inputs of hydrological models for which a number of different empirical formulae exist. Several studies have been carried out to compare these methods but there is limited consensus on which equation is most applicable in hydrological climate impact assessments (Sperna-Weiland, et al., 2012). PET can be calculated by various methods, which each have a different sensitivity to temperature and climate change and hence can greatly influence the conclusions about the implications of climate change and other variables (Cohen, et al., 1996). Thornthwaite's (1949) PET method a temperature-based method with relatively little data inputs, was used for calculating the monthly PET (Shaw, 1999).

Surface water and groundwater, particularly shallow resources are linked through recharge processes from infiltration and surface-water bodies, which affect both the quantity and quality of the water (Ringman, 2002). The inclusion of these processes was beyond the scope of the project. The groundwater resources were not included in this work due to the lack of knowledge about the resources in the Mekong Basin (Hoanh, et al., 2003). However, in some river basins groundwater resources can be very significant in the overall hydrology of the basin, particularly when water shortages are faced and pumped groundwater abstractions are high. These therefore need to be considered in assessing the water resources.

The Nash-Sutcliffe (1970) efficiency (NSE) and the root mean square error (RMSE) were used to quantify the model's performance. The model flows and concentrations were calibrated to maximize the Nash-Sutcliffe efficiency between the modelled and observed values. The NSE ranges between $-\infty$ and 1. An efficiency between 0 and 1 indicates that the model performance is acceptable and the higher the efficiency the better it performs, whereas an efficiency of less than 0 means that the observed data is a better predictor than the model (Krause, et al., 2005). The lower the RMSE the better the model performance is.

2.3 Water Quality

Traditionally, water balance models have only been interested in water quantity. However, the movement of pollutants from land into aquatic environments is intrinsically linked with the hydrological processes in the catchment (Davie, 2003). Human activities are having an increasing impact on the chemical loading in water bodies so there is a need to include water quality assessment in simple hydrological models.

The water balance model developed here includes a very simple water quality component based on the mass balance to determine the water quality in the flow for a few widely used constituents: Chemical Oxygen Demand (COD), Nitrates (NO3) and Total Suspended Solids (TSS).

The most important indicator in assessing river water quality is dissolved oxygen levels as this has a significant impact on aquatic life and provides an indicator of the general quality of the water (Davie, 2003). The variation in concentration of dissolved oxygen in the river depends on a number of factors, but is dominated by two factors, the oxygen demand and the reaeration rate (Droste, 1997). Chemical oxygen demand (COD) is a measure of the effects of pollution in the water. The higher the COD the more the dissolved oxygen is depleted in the water and consequently the water quality reduces (Davie, 2003).

High levels of nitrates entering water bodies from agricultural areas, through the application of fertilisers, can lead to lead to eutrophication which can rapidly reduce oxygen levels in the long-term (Davie, 2003). They can also affect human health at high concentrations.

One of the largest water quality issues is the amount of suspended solids in a river, for which high levels can often be a result of natural processes (Davie, 2003). The concentration of suspended solids varies significantly due to local variations is morphological processes and biochemical reactions. Suspended solids can have very different physical and chemical properties, even in the same location and time. Organic solids have an associated sediment oxygen demand, whereas inert solids such as clay do not (Davie, 2003). They can also adsorb various compounds to different degrees.

Each type of land use is given a pollutant loading profile for COD, nitrates, and suspended solids based on literature as given in Table 1. The concentration at each monitoring station is calculated using the mass balance equation (Droste, 1997) for the upstream river flow and runoff from the left and right side catchment areas of the mainstream rivers:

$$C = \frac{Q_{up} * C_{up} + Q_{left} * C_{left} + Q_{right} * C_{right}}{Q} \tag{5}$$

where Q is the flow (m^3/s) and C is the concentration (mg/l).

The mass balance assumes perfect mixing of flows throughout the reach. The concentration at each monitoring station was calculated using the mass balance for the chemical oxygen demand, nitrates and total suspended solids. Each water quality parameter was calibrated by scaling the concentration loadings. This assumes conservative movement of pollutants where the concentration is diluted. The nitrate concentration is principally governed by dilution in aerobic conditions and therefore considered a conservative species. For decaying pollutants such as COD and TSS, the mass balance predicts higher than actual concentrations (Loucks, et al., 2005).

A first-order exponential decay can be applied to the COD and TSS decaying with distance travelled downstream to further improve the model. This assumes mixing occurs immediately downstream of the previous monitoring station and a first-order decay along the distance until the next downstream monitoring station. A decay constant as a function of time would require information about the channel cross-sectional area which would add further uncertainty to the model. The deoxygenation decay constant for the chemical oxygen demand is temperature-dependent varying as $k_T = k_{20} \times 1.047^{(T-20)}$ (Droste, 1997). Given the localised contributions to the high variation in solids concentration and the unknown nature of the solids' chemical and physical compositions the mass balance decay equation was calibrated with a single decay parameter to match the observed data. The solids also have an associated sediment oxygen demand which was not taken into account in the model development (Davie, 2003).

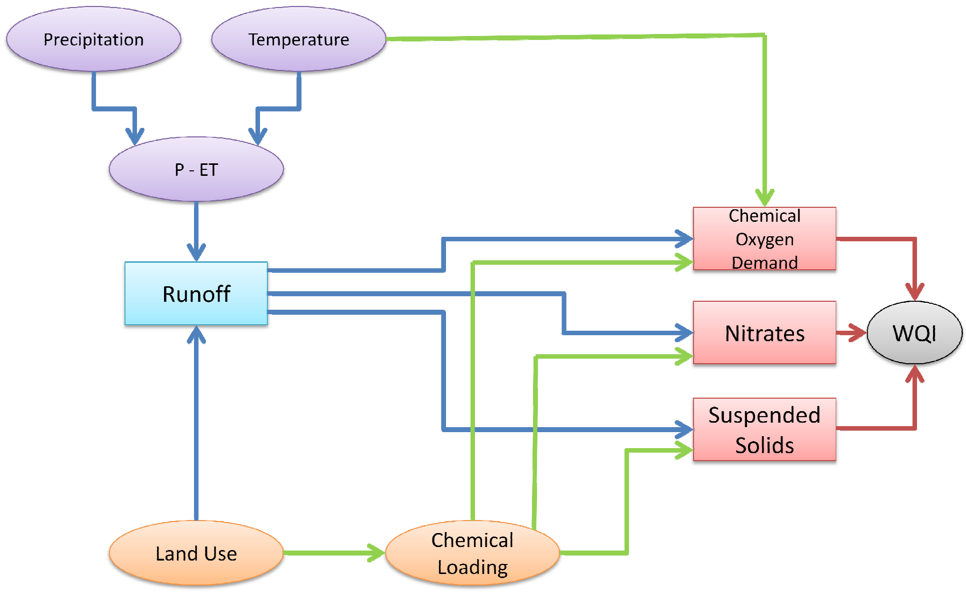


Figure 2: Conceptual water balance and quality model structure

2.4 Sensitivity Analysis

A sensitivity analysis enables the performance of the model to be tested and evaluated by determining which parameters have the most influence on the model outputs by quantifying their relative importance.

A case matrix of single parameter and multiple parameter combinations was created to study how changing each variable affects the output (Appendix C). Each input variable was changed one at a time with an arbitrary 25% (or 4°C for temperature) increase or decrease keeping the other variables as their base values. This allows for comparability of the results of the model; however, it doesn't explore the full variability in the inputs. Multiple-parameter runs and exploratory runs looking at simultaneous variation of input variables allow the presence of interactions between input variables to be identified.

2.5 Water Quality Index

Given the simplicity of the water quality model and the uncertainty in the parameters, it is expected that the resulting concentrations won't be precise, but they are sufficient for the model's aim. For the purposes of decision-making, a simple method of assessing the water quality can be used in the form of a Water Quality Index (WQI), which provides a concise method for expressing the quality of the water. Several WQIs have been developed such as Prati et al. (1971), Cude (2001) and Thi Minh Hanh et al. (2011), which apply different weights to various parameters based on the intended use of the index

A WQI was created based on international drinking water standards, indicating the level of treatment required to achieve water suitable for potable uses. It uses a scale for COD, NO3 and TSS to produce an index for each parameter which were averaged and given equal weighting. The WQI at each monitoring station lies along the water quality scale from 0 to 1 based on the concentration of the constituent. The maximum concentration corresponds to the lower limit on the scale for that band with a lower concentration giving a higher index indicating a higher water quality.

The water quality scale was based on river quality classifications and treatment requirements required for the water to be of acceptable drinking standard based on concentration limits, where given, from the World Health Organisation (WHO, 2011), US Environment Protection Agency (EPA, 2009) and EC Surface water guidelines (Gray, 2005) Figure 3 and Table 2 summarise the water quality scale and the concentrations used for each band. However, it can also be adapted to be used as a scale for other criteria such as water quality for aquatic life. The scale was made such that the more polluted the water, the lower the WQI and the more treatment required to reach acceptable drinking water standards.

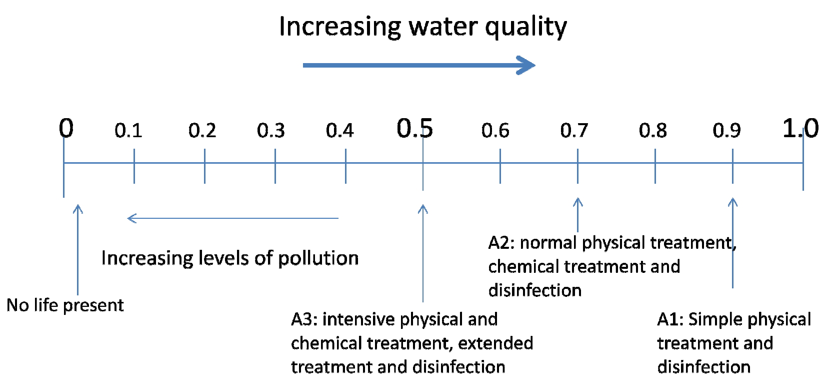


Figure 3: Diagram of the water quality scale based on international drinking water standards

Table 2: Maximum concentration, in mg/l, of each constituent within each quality band

WQI bands	COD	NO3	TSS
0.9 – 1	5	10	20
0.7 – 0.9	8		50
0.5 – 0.7	12	50	100
0.3– 0.5	25	>50	500
0 - 0.3	>25		>500

3. RESULTS AND DISCUSSION

3.1 Model Calibration Results

3.1.1 Water Balance Model

The calibrated flows at each monitoring station are compared with observed data for long-term mean annual runoff in Figure 4. The runoff was scaled by a factor of 1.8 for k_2 to give the calibrated runoff with an NSE of 0.98, as given in Table 3. This factor is due to the fact that the rational method for which typical runoff coefficient values were taken does not include an evaporative loss, so the effective precipitation used in this model is lower than the actual precipitation. Therefore, one would expect the scaling factor to be greater than one and higher where temperature and hence evapotranspiration are higher.

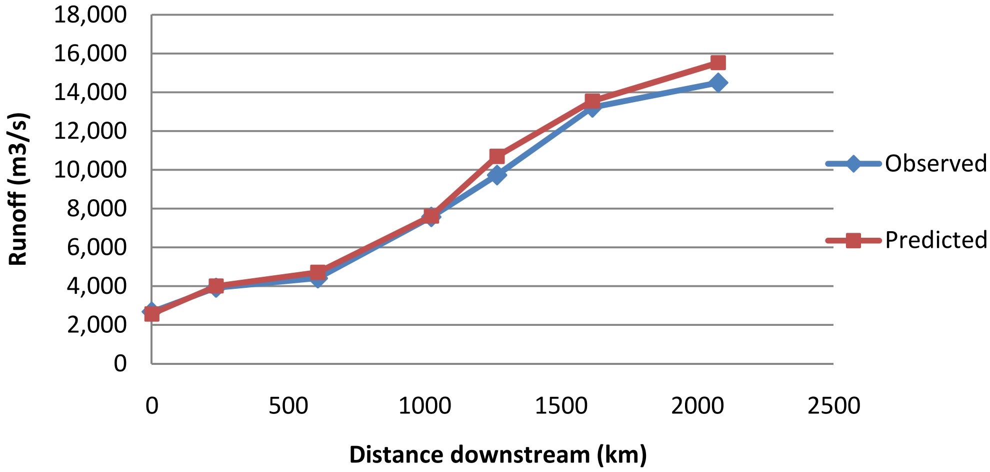


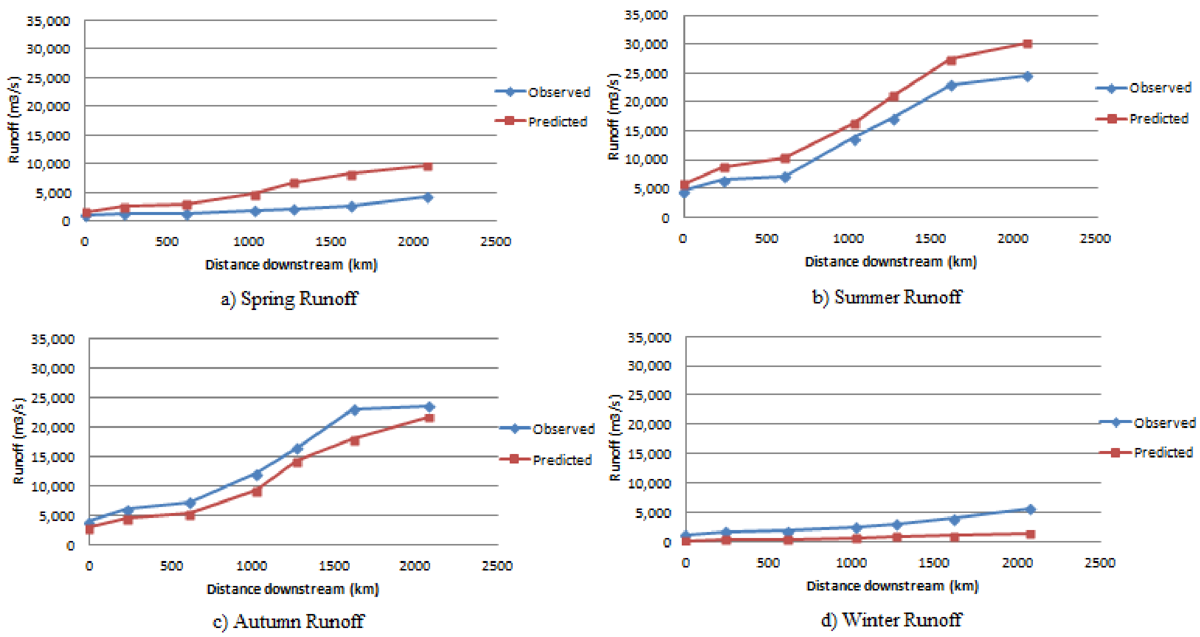
Figure 4: Predicted and observed long-term mean annual runoff

The source of the Mekong originates in the Tibetan Plateau where the initial runoff is significantly affected by glacier runoff and snowmelt. The variation in snowmelt will depend on snow accumulation, heat balance, precipitation amount and type (snow or rain) and snowpack movements (Ringman, 2002). Variations in precipitation amount and type can also affect glacier meltwater produced. This can have greater inter-annual variations than lowland runoff which is influenced predominantly by precipitation. For example they can lead to sudden release of a large quantity of stored water into the runoff. The influence of meltwater affects the runoff during the spring season at the most upstream locations in the LMB where melt runoff dominates over local precipitation. This means that the model can simulate evapotranspiration as greater than precipitation, although there is still runoff; hence the model was calibrated such that the condition in (4) is satisfied. The same scaling factor for the actual evapotranspiration, k_1*PET, was used for the whole basin; however it is likely to vary depending on local variations such as soil conditions and micro-climatic influences across the basin.

Table 3: Model performance for annual and seasonal runoff

Flow Timescale	Model Metric	Before Seasonal Calibration	After Seasonal Calibration
Annual	NSE	0.983	**0.983**
Annual	RMSE	562.663	562.663
Spring	NSE	-12.227	**0.897**
Spring	RMSE	3670.109	324.389
Summer	NSE	0.777	**0.973**
Summer	RMSE	3546.881	1225.848
Autumn	NSE	0.871	**0.920**
Autumn	RMSE	2684.290	2118.204
Winter	NSE	-1.790	**0.912**
Winter	RMSE	2378.187	422.958

The seasonal runoffs can be calibrated by varying the scaling factor k_2 to match the data. The Mekong has a large seasonal variation in rainfall with the wet monsoonal season having several times more rainfall and runoff than in the dry season. The model was used to simulate the seasonal runoff using the monthly climatic data, but also as a means of validating the model calibration. The annual calibration factors were maintained giving the results seen in Figure 5a-d.



a) Spring Runoff

b) Summer Runoff

c) Autumn Runoff

d) Winter Runoff

Figure 5a-d: Predicted and observed seasonal runoff at monitoring stations along the Lower Mekong River

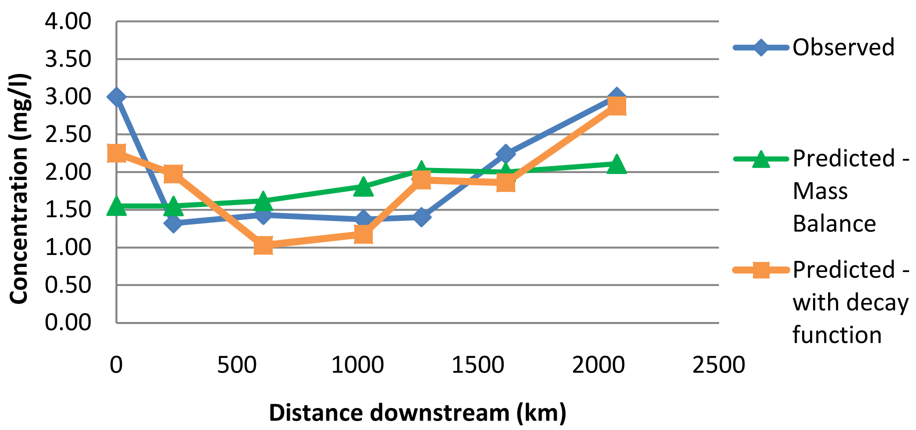
The seasonal runoff is over-predicted in Spring and Summer and under-predicted in Autumn and Winter using the annual calibration factors. This is likely to be due to the seasonal changes in soil moisture characteristics and air saturation levels arising from the greatly differing levels of rainfall throughout the year resulting in a hysteresis loop in evaporation losses. During Autumn and Winter there is likely to be greater levels of stored water in the catchment which reach the main river by throughflows and baseflows but have relatively longer time-scales. This results in a time lag seen between the precipitation and the runoff at the mainstream monitoring stations, which the model doesn't take account of. The hydrology of the Tonle Sap Lake was not taken into consideration which could explain the greater discrepancy at the most downstream monitoring station.

The model performs best in the Summer and Autumn when the rainfall is highest and least well in the winter where rainfall is lowest. Table 3 summarises the Nash-Sutcliffe efficiency values and root mean square values calculated for the annual and seasonal runoff. After using seasonal calibration factors, the NSE was greater than 0.89 for annual and seasonal flows, showing that the simple model is able to simulate long-term flow well for the Lower Mekong on an annual and seasonal basis.

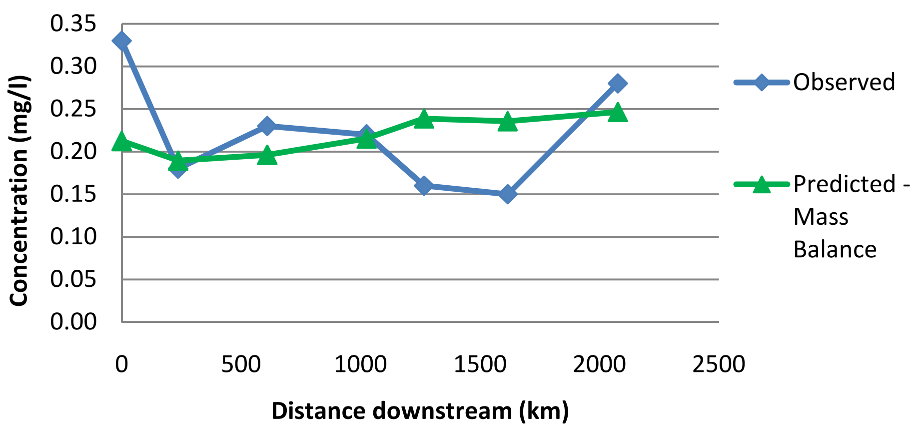
3.1.2. Water Quality

The pollutant loadings for COD, NO3 and TSS were initially calibrated to optimise the mass balance for the nitrates as this varies mostly as a result of dilution in the mass balance. The mass balance model gives a higher COD than observed data. This is expected as the oxygen demand reduces with time so would be lower when measured downstream. The model gave a fairly uniform output for suspended solids unlike the large variation in the observed data. These match the observed values further downstream but were significantly lower than the very high levels of solids observed at the upstream stations. Most of this area is shrubland and forest (Appendix B), which appear to contribute a much higher level of solids to the runoff than used in the model. The NSE and RMSE are summarised in the first column of Table 4. These were be improved by adjusting the scaling factors in the mass balance.

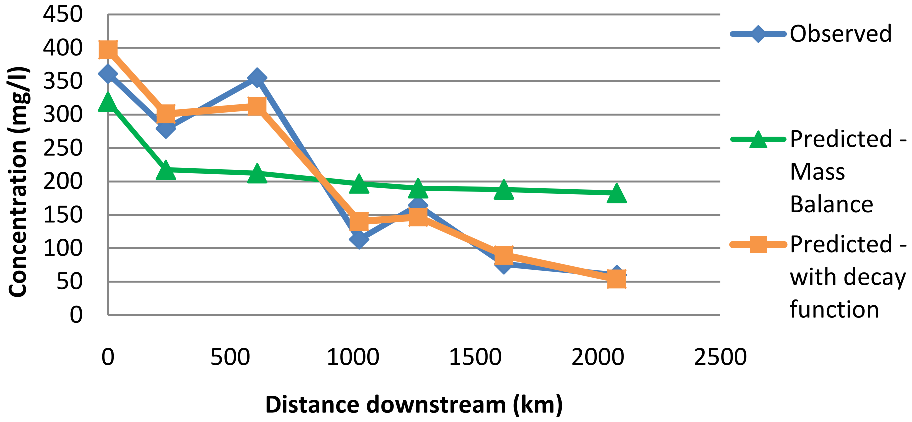
The concentration loadings for the mass balance after adjusting the loading scaling factors were calibrated for each parameter to give a concentration loading of 10% of the literature values for COD, 40% for nitrates and 20% for suspended solids with each land use then scaling factor adjusted to improve the calibration. The results are shown in Figure 6a-c and the second column of Table 4 with NSE values of -0.01 for COD, -0.30 for nitrates and 0.39 for TSS. These concentration loadings seem reasonable for nitrates and suspended solids as the literature sources were taken from mostly European and American data where there are higher population densities and heavier use of fertilisers is wide-spread. However the observed data for the COD is quite low, especially given the high level of suspended solids, which are likely to include some organic solids and therefore have an associated sediment oxygen demand. Typical levels for BOD in lowland rivers are 2-7 mg/l (Gray, 2005), so a higher COD concentration would be expected.



a) Chemical Oxygen Demand



b) Nitrates



c) Total Suspended Solids

Figure 6a-c: Predicted and observed annual pollutant concentrations for the mass balance and the calibration with decay function for COD and TSS

The decay equations applied to the original loading profile improved the calibration, as shown in Figure 6 with the NSE for COD increasing to 0.56 and NSE for TSS to 0.95. The deoxygenation constant after calibration was 0.002/km. Assuming the flow velocity in the middle stretches of the river to be 0.8m/s this gives a deoxygenation constant of about 0.15/day at 20^{o}C. This is comparable to typical values as given in Gray (2005). The variation in flow velocity in the upper basin and in the low-lying floodplains and delta region mean that this factor varies.

The types of suspended solids can vary significantly in the river depending on their source and erosion processes. The level of TSS also varies throughout the year, rising during flooding as more sediment is washed into the river. Hence the rate at which these decay varies, so the decay constants were also adjusted to maximise the NSE between the model and the data giving a decay constant of $2x10^{-5}$/km for the first three monitoring stations and 0.0022 for the downstream stations. The nitrate concentration gives a NSE of -0.15 showing that the observed mean is a better predictor than the model, but it gives a similar nitrate concentration to the data along the river. Given the very low concentration of nitrates to which the NSE is applied, and the simplicity of the model the calibration seems reasonable for the purpose of the model; however further decay equations could be included to take account for the nitrification processes. The model also doesn't account for variations in concentration loadings in runoff from the same land-use in different regions.

Table 4: Model performance for water quality parameters

		Calibration Data (2000-2008)			*Validation Data (1985-1994)*
Concentration	**Model Metric**	**Mass Balance**	**Adjusted Mass Balance**	**First-Order Decay Calibration**	**First-Order Decay (Mass Balance for Nitrates)**
COD	NSE	-55.114	**-0.006**	**0.556**	**0.195**
COD	RMSE	5.367	0.719	0.477	0.625
NO3	NSE	-0.591	**-0.150**	--	**-0.133**
NO3	RMSE	0.077	0.065	--	0.121
TSS	NSE	-0.401	**0.389**	**0.951**	**0.758**
TSS	RMSE	141.507	93.416	26.370	98.749

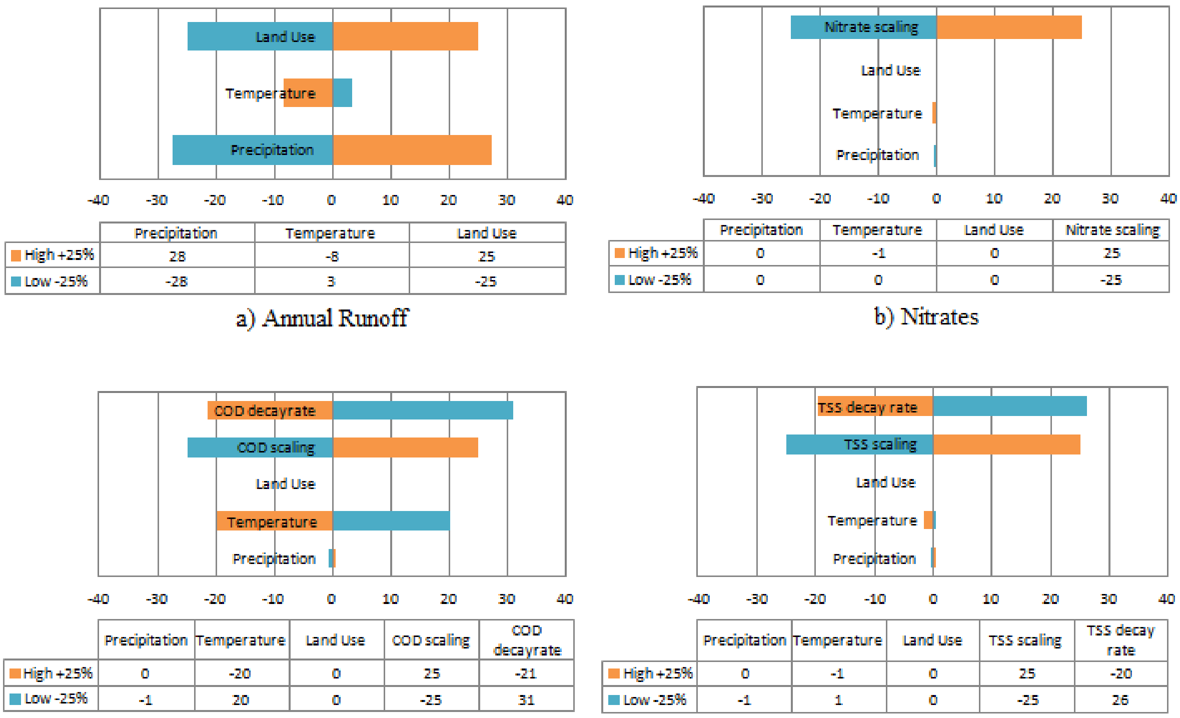
3.1.3. Model Validation

The quality model was validated against data for the time period 1985-1994 by varying the loading concentration scaling factors to maximise the NSE. The data for nitrates were 10-20% lower and for TSS was about 30-35% higher during this time period. The COD data was very similar in both time periods and still very low for both time periods which is surprising, especially given that the nitrate and suspended solids concentrations vary. The results are also summarised in Table 4. The NSEs for the validation period are lower than for the calibration period for COD and TSS (although higher for NO3) as the loading profile for the calibration period was uniformly scaled to match the validation data. The increased nitrates are likely to be due to a higher use of fertilisers in agriculture. The lower level of solids in the upstream locations is a result of the construction of the dams on the Lancang in the Upper Basin.

3.2 Sensitivity Analysis

The results from the single-parameter runs of the sensitivity analysis are summarised in Figures 7a-d. Figure 7a shows that the runoff is sensitive to all the inputs being most sensitive to precipitation, followed by the land-use runoff coefficient. It is also sensitive to a change in temperature, especially to an increase but to a lesser extent. The most uncertainty in the water balance model is likely to be in the PET calculation and the conversion to actual evapotranspiration, and therefore should be taken into consideration in scenario analyses.

The water quality parameters COD, NO3 and TSS are all sensitive to variation in the concentration loadings and the outputs for COD and TSS are similarly sensitive to the decay constant. The quality parameters have negligible sensitivity to the water balance components, except COD which is also sensitive to temperature due to the temperature dependence of the deoxygenation constant. The land-use runoff coefficient shows no effect on water quality. However, each land use has a different concentration loading profile, so a change in runoff from one land type to another would affect the water quality in the model.



	Precipitation	Temperature	Land Use
High +25%	28	-8	25
Low -25%	-28	3	-25

a) Annual Runoff

	Precipitation	Temperature	Land Use	Nitrate scaling
High +25%	0	-1	0	25
Low -25%	0	0	0	-25

b) Nitrates

	Precipitation	Temperature	Land Use	COD scaling	COD decayrate
High +25%	0	-20	0	25	-21
Low -25%	-1	20	0	-25	31

c) Chemical Oxygen Demand

	Precipitation	Temperature	Land Use	TSS scaling	TSS decay rate
High +25%	0	-1	0	25	-20
Low -25%	-1	1	0	-25	26

d) Total Suspended Solids

Figure 7a-d: Summary of results obtained from the single-parameter runs to a 25% (or a 4°C temperature) increase and decrease in each input variable.

The exploratory runs looked at some of the interactions between multiple water balance and quality parameters for likely scenarios. For example, deforestation is likely to result in an increase in runoff as there is less vegetation to infiltrate rainfall. It is also likely to lead to an increase in solute concentrations in the runoff, especially from suspended solids. A climate change scenario with a temperature and precipitation increase because of the higher sensitivity to precipitation would cause an overall increase in runoff under the variations above. More intense rainfall would increase the TSS concentration loading as the heavy rainfall causes more solids to be leached in the river.

3.3 Water Quality Index

The water quality index for the model outputs of the mass balance and the calibrated quality with decay functions are compared with the WQI for the observed data in Figure 8. Both representations give a WQI gradually increasing going downstream within a similar range, between 0.75 and 0.9, to the WQI for the observed data. This indicates that the water quality in the Mekong is good enough such that normal physical treatment, chemical treatment and disinfection would be sufficient to ensure that the water meets of drinking standards. The increase is much greater for the calibrated model with the decay functions showing the most agreement at all points, with an NSE of 0.93 compared with 0.07 for the mass balance.

The increase in quality is due to the WQI having a much higher sensitivity to being far more sensitive to TSS than the COD or nitrates due to its high concentration and the very low concentrations of COD and NO3 in the equally weighted average; so the significant decrease in TSS downstream dominates the WQI score. Figure 9 shows the WQI for each of the constituents where it decreases slightly for COD and NO3 and increases for TSS.

The WQI was also incorporated into the mass balance mixing equation where each concentration was substituted with its WQI. For the mass balance and WQI mixing in the mass balance the trends in the individual parameters almost cancel out to give a fairly similar WQI across the entire river length. This is due to the quality model for TSS having a much higher concentration downstream for the mass balance than the TSS calibrated with the decay function (as seen in Figure 6c). This is reflected in Figure 9, with the WQI for the calibrated model increasing significantly from 0.35 to 0.7. This shows the nature of the data that can be lost from using the WQI especially from averaging, which could lead to potential misinterpretations of water quality from the model.

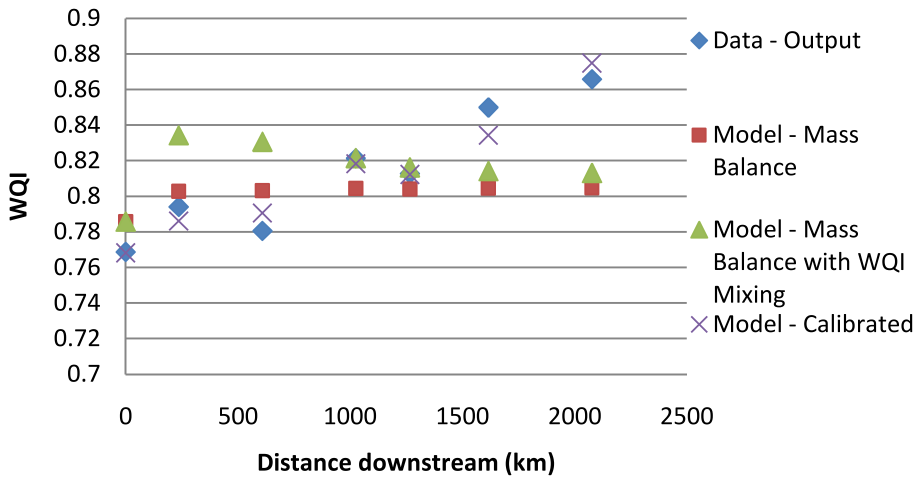


Figure 8: WQI based on drinking water standards along the river for different model output methods

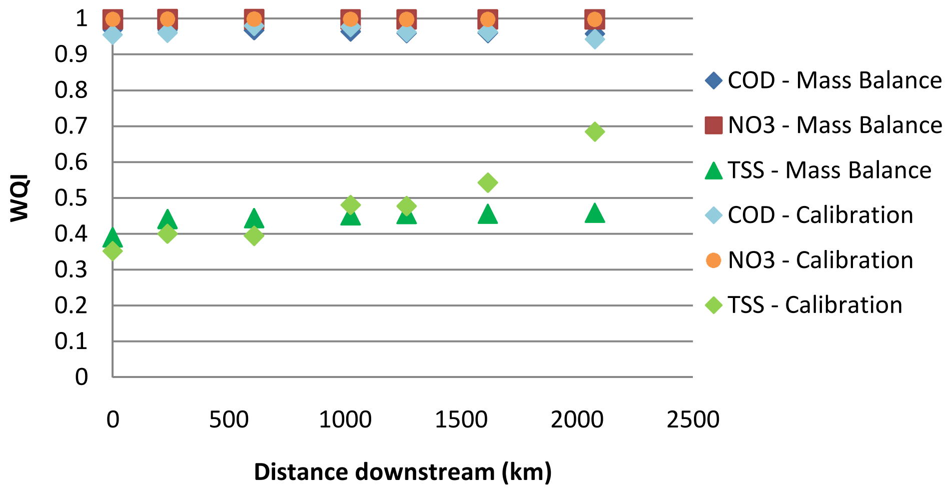


Figure 9: WQI broken down into its individual constituents for the mass balance and calibration with decay

Many stakeholders concerned about the quality of the water generally don't have the time or expertise to evaluate and understand large amounts of raw data, when available, from water quality monitoring. A water quality index thus summarises the water quality data as a single number on a scale that can be easily understood by those without a technical background in water quality by aggregating multiple parameters for each monitoring station, with a higher number indicating better water quality.

The WQI gives an imperfect answer to non-technical questions about water quality. It can show how "good or bad" the water quality is giving an indication of whether the water quality is less than what is expected or necessary for designated uses from the river such as for drinking water. For some purposes this can be sufficient and is better than having no information about water quality.

The WQI is useful for comparisons and for general questions rather than site-specific questions. It's simplicity in expressing the water quality can help answer questions such as whether the water is drinkable or whether it can be used for irrigation and is thus useful as a means of easily understanding the impacts of different scenarios and evaluating the effectiveness of various management strategies. However, this requires information about the level of water quality required to support these uses. Also, comparing scores at different stations does not necessarily indicate that the water quality is better at one than the other unless the expectations from both stations are the same. Upstream and downstream users can have widely differing views about what quality is necessary for their own uses. The WQI is subjective based on the requirements of the use of the water from the particular river segment. Different stakeholders will have different requirements at different locations along the river. Thus the WQI does not show the absolute water quality, but the water quality relative to that for the particular requirements given by the different bands of scores used in the scale. Moreover, the lower preciseness of the WQI means that extensive calibration with the uncertainties associated with parameter estimation isn't required in order to be useful. Even when there are insufficient records of water quality data in order to calibrate the model precisely, the WQI could still be developed more subjectively based on professional judgement.

However, the WQI still needs to be used with caution as it could give a good score, but a parameter not included in the index may be degrading the water quality. The aggregation of data may also mask some problems, as demonstrated Figure 9, with the much greater variation in TSS being averaged out by the high scores for COD and nitrates.

This WQI does not evaluate several water constituents that may be found, but only considers a few important constituents used in the water quality model that can give a reasonable idea of the water quality. It is beyond the scope of this project to include the various other water constituents that can be found in water bodies. However, these should nevertheless be considered during the planning of management schemes.

3.4 Stakeholder Scenarios

The following section illustrates how the model could be used by different stakeholders to consider scenarios affecting water quantity and water quality in the basin. The scenarios demonstrated are possible scenarios, but not necessarily the most likely scenarios that will occur in the river basin. Given the uncertainty in possible changes and the lower preciseness of the model compared with other models it would be difficult and unwise to try and accurately predict their impacts here. However, even with limited understanding of future outcomes, 'what if' scenarios can still be created to consider the possibility of outcomes and whether particular decisions may lead to such outcomes.

The impacts of various future changes are likely to affect many stakeholders in a number of ways. They are likely to have several questions which models can be used to address such as:

- How will climate change affect water availability and quality?
- What impacts will it have on upstream and downstream locations and how can they be mitigated?
- Can adverse effects of hydrology and water quality be better mitigated by upstream or downstream management approaches?
- To what extent can land-use management mitigate the impacts of upstream impacts on water resources?
- What effect will population growth have on land and water resources and how will this affect food and water security?
- What are the hydrological and ecological consequences of hydropower development?
- What are the economic and environmental benefits and trade-offs associated with the hydropower development? Which will have the greatest impacts?
- What will be the ecological impacts of river fragmentation?
- To what extent will they mitigate flood risk and how will climate change affect their operation?

This model cannot answer all of these questions, but it can provide some useful insight into some of them that help to better understand and predict the interactions between different sectors and users so that policy-makers and stakeholders can better manage the land and water resources in the region.

3.4.1. Climate change and land use scenarios

Global circulation models (GCMs) are used to make climate change projections and are frequently used to quantify the hydrological impacts of climate change; however their outputs differ greatly sometimes disagreeing and are too coarse to use in regional hydrological models (Sperna Weiland, et al., 2012). The impacts of climate change add further risk to the considerations of management approaches addressing socio-economic requirements. Climate change could have a number of impacts in the Mekong that would affect water resources. It is likely that the dry season will lengthen and intensify and the rainy season will shorten and intensify (Kirby, et al., 2010) accompanied by a temperature increase.

Various possible climate scenarios can be explored to see how they could affect river flows. The distributed nature of the model can give some sense of the relative differences between upstream and downstream locations.

The Mekong Delta in Vietnam, which is the most productive rice growing area in the basin, is likely to experience the greatest impacts from greater flooding risk and saltwater intrusion. The saltwater is more likely to flow further upstream in the dry season if considerable amounts of water are diverted to irrigation (Kirby, et al., 2010) or lower precipitation and higher temperatures reduce flow rates from upstream. The upstream hydropower development may alleviate some of the potential salinity by increasing dry season flows, and thus offsetting the potential impacts of irrigation and climate change to some degree.

Climate change can also affect water quality, particularly through short-term effects, such as low flows and storm flows which can affect oxygen levels, suspended sediments, coliforms, nutrients and salts (Orlob, et al., 1996). Modelling the water quality was considered, however the sensitivity analysis shows that climatic changes have an insignificant effect on water quality in the model, except for COD which increases as temperature rises due to the higher rates of biochemical reactions. However, given the very low COD concentration in the river currently a temperature rise would have little impact. An increase in average suspended solids is likely if heavy rainfall events increase in frequency or intensity.

Figure 10 shows how the annual runoff can be modelled under different climate scenarios in 2050. Given the uncertainty in future climate it could be used to look at upper and lower bounds of likely runoff in the basin. For example, a high precipitation scenario with 10% increase and low precipitation scenario with a 10% decrease by 2050, both with a 1^{o}C temperature increase are shown. The results show how the runoff will vary at different locations along the river under different climate scenarios. The effects of climate change on snowfall and glacier melt will also have a significant impact on flow volumes and timings especially in the upstream locations; however these have complex interactions with the flow that are not taken into account in the model.

The impacts from climate change combined with increased irrigation demands are likely to be especially critical during the dry seasons when the lowest flows are experienced. The effects of the different changes need to be viewed together in order to understand the full impacts. The model could be used to compare different combined climate and management scenarios and how each impacts water availability and saltwater intrusion in the delta.

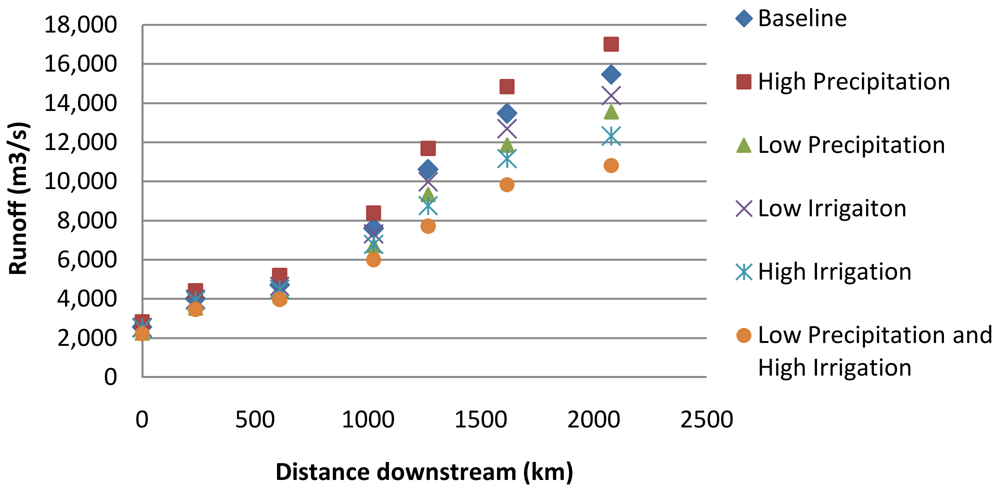


Figure 10: Comparison of annual runoff under wetter and drier climate scenarios and different irrigation scenarios in 2050

Similar studies can be done for land use management scenarios. Different rates of deforestation and afforestation can affect TSS concentration. The model could be used to compare the effects of continued deforestation on TSS in the river and what order of scale of afforestation schemes would be required to improve sediment loading in the river as shown in Figure 11. The model could also be used to investigate agriculture scenarios comparing the impacts on water quality of different rates of fertiliser application with increasing the area of land used. These can also be combined with climate scenarios. Impacts of urbanisation and increased industrialisation can also be investigated, however given the high water quality in the Mekong, any realistic increase in these are unlikely to have major negative impacts that would be detected by the model.

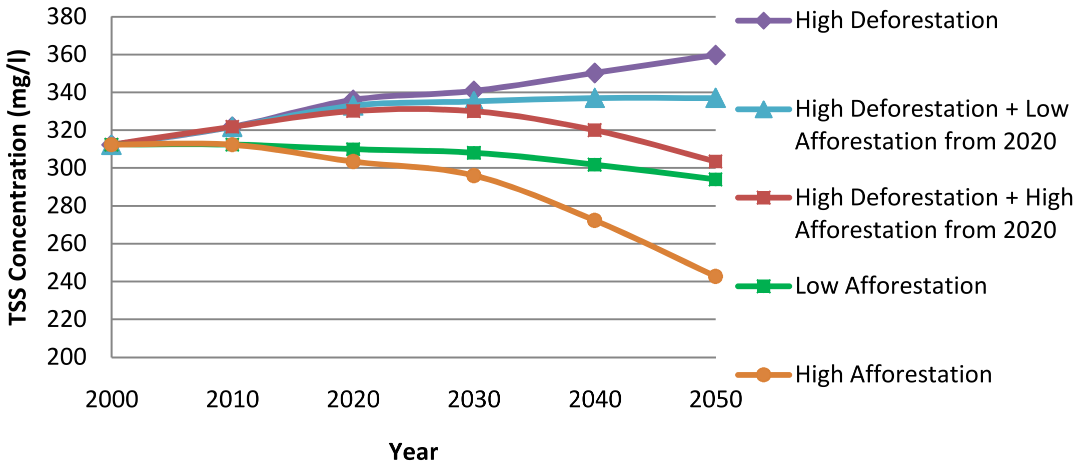


Figure 11: Comparison of total suspended solids concentration under different land management scenarios

3.4.2. Multiple Stakeholder Calibrations

The model calibration as described in the previous section was done from an impartial viewpoint, but it is not unique and more than one combination of parameters could be used. Hence different calibration results will be obtained by different modellers, especially given the sparseness of the data set. In the river basin, each stakeholder could potentially use the tool to do their own calibration. These may be bias based on their own personal experiences and perspectives but would be equally valid calibrations provided they support the available data. Yet only a particular range of values for each parameter will give a valid calibration.

For example, a drinking water supplier could argue that the TSS concentration in the water doesn't decay as fast as the model shows and that a higher concentration is actually present which is negatively affecting the water treatment process. Thus they would produce a calibration with a lower decay constant of say 0.15. Similarly, a downstream farmer could argue that the calibration over predicts the sediments reaching downstream when he thinks that there is less sediment in the river to fertilise the fields during flooding and therefore has to use more artificial fertilisers for his crops. The farmer would therefore produce a calibration with a higher decay constant of 0.3 to show the reduced load downstream. Both of these values would still produce a valid calibration. Another stakeholder may vary the scaling factor instead.

From Figure 12a-e it can be seen that the loading scaling factor and the decay constant can be varied independently between a range and still give a valid calibration. For example, a range from between 0.3 and 0.5 could be used for the COD scaling and between 0.15 and 0.3 could be used for the COD decay constant, or 0.55 to 0.85 for the TSS scaling factor or 0.12 to 0.32 for the decay constant. A calibration outside these ranges, however, would not support the data. Each value will give a calibration that is closer to the data at some points and further from the data at others, hence upstream and downstream representatives could each produce different calibrations depending on their view at a particular location.

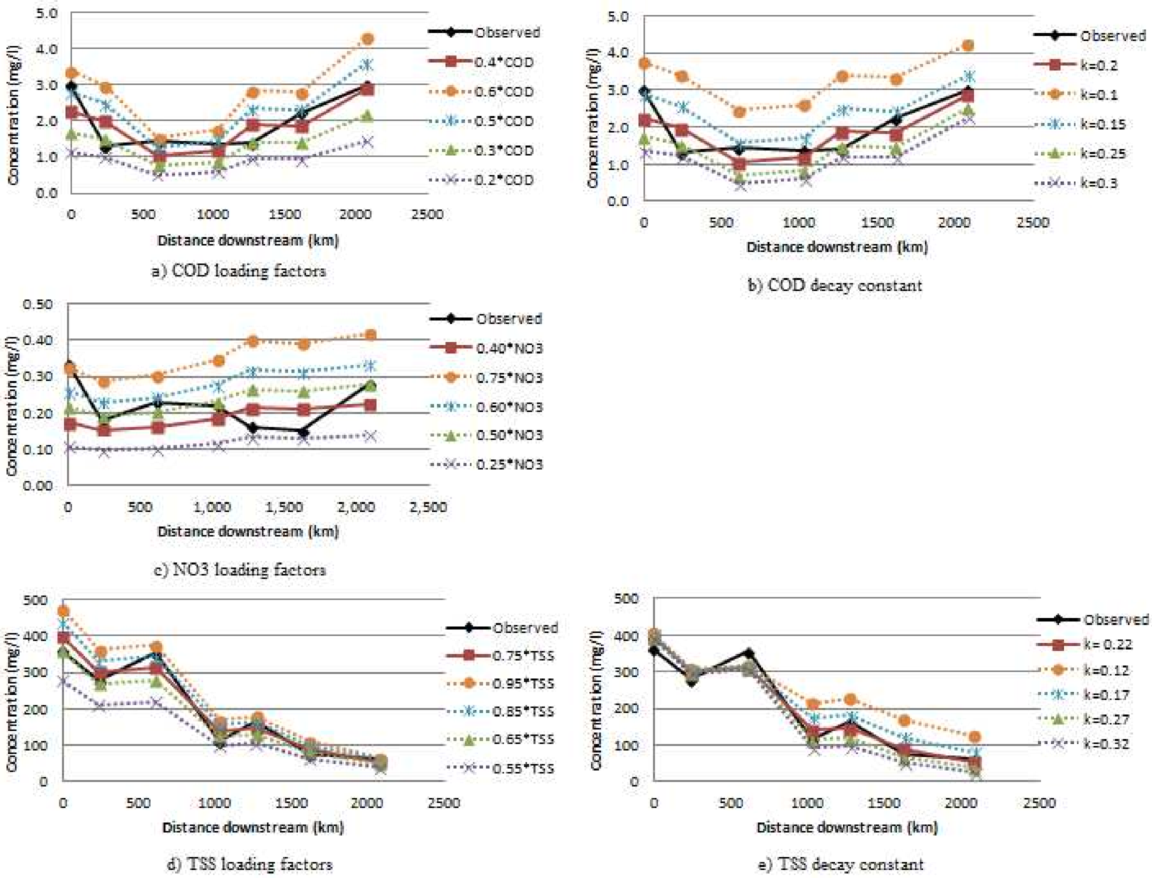


Figure 12a-e: Multiple calibrations are possible with different calibration parameter values

Most models developed for water resource management allow little or no stakeholder input. The simplicity of this model allows stakeholders to understand the conceptual model of the system and also enables them to gain insight into the interests and concerns of each stakeholder and the institutions being represented. This allows them to be able to identify potential strategies by negotiating trade-offs between opposing groups. The views and understanding of the system of any stakeholder involved in the decision-making process are likely to be influenced by their background and limited individual experience. This would lead to their own individual partial conceptual model of the system which differs from those of other stakeholders (Serrat-Capdevila, et al., 2011), but each of these could include several parameters with great uncertainty in each. The simple model with only a few parameters condenses each individual model into a single conceptual model giving a shared understanding of how the system works and where the uncertainties and unknowns are. This enables stakeholders and decision-makers to understand what the main issues and challenges are at the basin scale.

3.4.3. Hydropower Development Scenario and Stakeholder WQIs

An important change in the Mekong is the planned hydropower development throughout the Lower Basin. These could have profound and wide-ranging socio-economic and environmental impacts in all four riparian countries (ICEM, 2010). Hydropower development is an important example of the need to integrate management across multiple sectors with implications for water, food and energy security in the region. This section illustrates how the model could be used to look at the viewpoints about the impacts on water quality as a result of hydropower development in the basin.

Four possible hydropower scenarios are considered with hypothetical WQIs for four different stakeholders concerned about hydropower development in the Mekong:

a) Upper Basin Development – Lancang cascade on the mainstream in China and the first of the mainstream dams, the Xayaburi Dam, in the Lower Basin.
b) Mainstream Development – Lancang cascade and all the LMB mainstream dams
c) Tributary Development – Lancang cascade and planned LMB tributary dams
d) High Basin-wide Development – Lancang cascade, LMB mainstream dams and tributary dams

The stakeholder WQIs created here only look at the impacts of suspended solids as this is likely to have one of the greatest impacts on water quality, whilst maintaining the same WQI for COD and NO3 as used for the drinking water standards. There is a large amount of uncertainty in how much of the incoming sediment can be trapped by the dams. Figure 13 shows the extent to which the trap efficiency of the dams influences the level of solids reaching downstream. The first point shows the potential effects of the Lancang cascade alone. The trapping efficiency of the dam is just one variable that affects the water quality. Other effects such as residence time, reservoir storage volume and dam operation also affect water quality parameters. A trapping efficiency of 50% is assumed for each section of the river between monitoring stations. The cumulative impacts on TSS of multiple dams are not included.

Different stakeholders value the water quality differently for various reasons and have different views of what 'better' water quality is. They can thus create their own water quality scale accordingly, which can also vary between upstream and downstream stakeholders. They could then use their own WQI to support their view regarding possible impacts of management scenarios on water quality. The baseline scenario with no further development using the WQI for drinking water developed earlier is compared with WQIs created for three different stakeholders: a subsistence farmer, fisherman and a hydroelectric engineer.

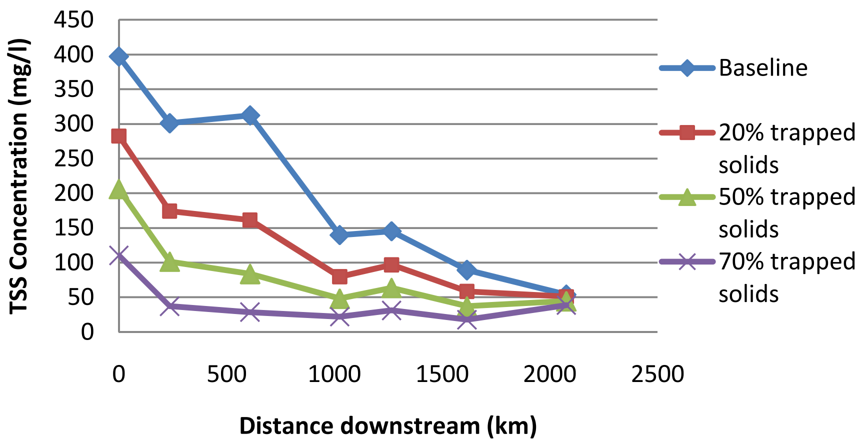


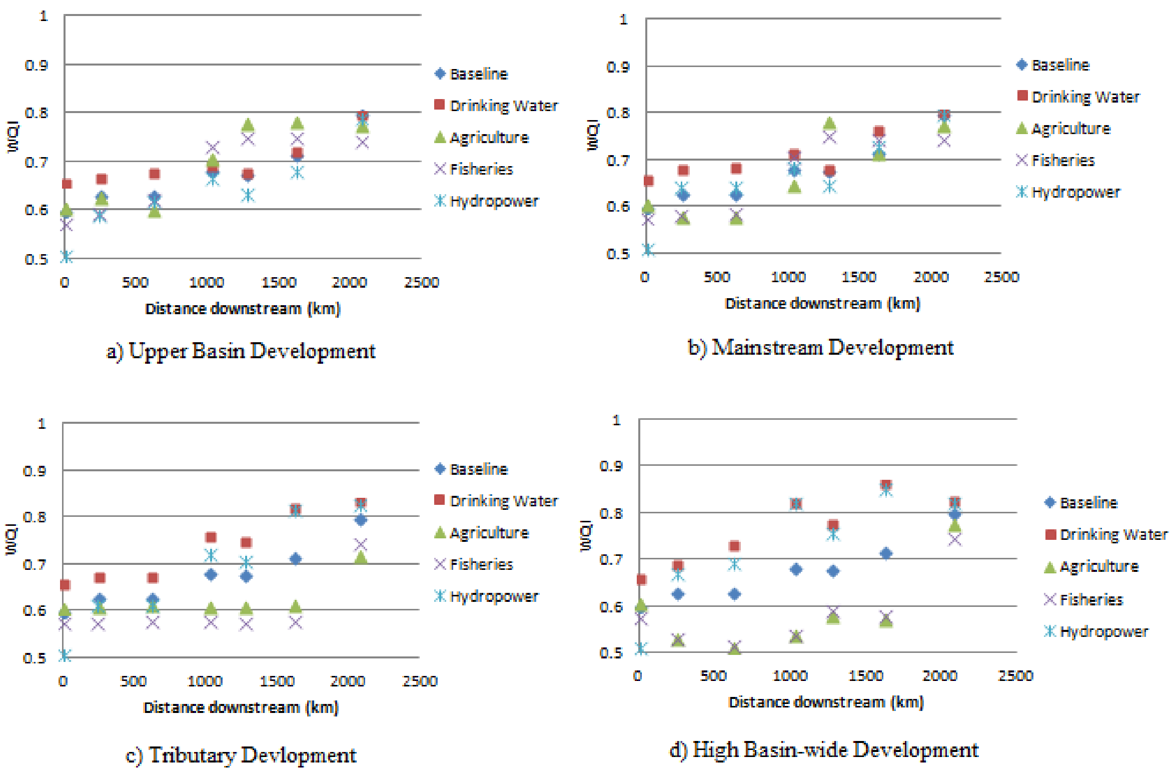
Figure 13: Variation in TSS as a result of different trapping efficiencies for the high basin-wide development scenario

The rural farming population living near the river rely upon the annual flooding to naturally provide nutrients for their crops, especially in riverbank gardens (Pearse-Smith, 2012). A reduced nutrient-laden silt layer on their fields will lead to a greater reliance on using artificial fertilizers, which are more costly and time-consuming and can leach into the river causing further water quality problems. This is likely to have the greatest impact on downstream Cambodia and especially in the delta in Vietnam, where the fertile soils resulting from flooding allow three rice crops to be grown a year. Hence, the reduced load would affect the food production yield. Therefore, for a farmer a greater reduction in TSS results in less beneficial water and therefore to a lower WQI. The WQI scales the reduction in solids from the original baseline levels with a decrease in WQI. This assumes farmers won't be directly affected by inundation of their land as a consequence of the reservoirs behind the dams.

Fishermen require water quality that provides the best habitats for the fish species in their fisheries. These too adapt to the seasonal changes in the river hydrology and often have optimal water flow and quality conditions. Any significant changes in these can put fish species under great stress. Not all fish are susceptible to suspended solids and neither are all solids harmful, hence different solids criteria are required for different fish species (Alabaster & Lloyd, 1982). One impact resulting from the dams is that the reduced sediment load disrupts spawning grounds for fish. The reduction in solids may also have a number of other impacts, both positive and negative, but are not considered here. Thus a decreasing WQI scales with the reduction in solids from the original baseline levels. A number of other water quality issues could affect the fish including low dissolved oxygen levels. The fishermen will also be concerned about the dams fragmenting the river and the consequences for migratory fish (Marmulla, et al., 2001). This could have a much greater impact on fish yield than the change in flow or quality. The fisherman could easily adapt the WQI to incorporate the degree of fragmentation on migratory path restriction.

The hydroelectric engineer is concerned about the efficient long-term operation of the dams and hydroelectric turbines. It has been observed that many of the dams built on Mekong tributaries are filling with silt from sediment deposition shortening the lifetime to much less than 100 years (MRC, 2011).. This would adversely affect the performance of the turbines and sluice gates reducing the operating efficiency and may require expensive dredging equipment to clear the sediment. Therefore, a 'better' water quality for the hydroelectric engineer would be with the least suspended solids; hence an adjusted version of the WQI for drinking water standards can be used. Additionally the economic benefits from the dam construction and operation could also influence their opinion and how they define their WQI. They could also argue that the first dam constructed will have a beneficial impact for subsequent dams as the resulting downstream flow will have lower sediment level and be of better quality. Whilst the model takes into account the overall sediment loss downstream as a result of the dam trapping the solids, during construction, a lot of sediment can be washed into the river from destabilisation of riverbank slopes which can lead to increases in local solids concentration.

Figures 14a-d compares different WQIs for the four hydropower scenarios. Table 5 summarises how each stakeholders' WQI varies with TSS. The number of dams increases progressively with each scenario. They show that as development increases the more the WQI for the different stakeholders diverge. The water quality improves for the drinking WQI and the hydropower engineer, but decreases even more for the farmer and the fisherman. The changes in WQI between each scenario are different. For example, the construction of the subsequent mainstream dams after the first dam appears to have much less impact than the tributary dams. This shows how the model could be used to compare the effects of different dams and demonstrate which are likely to have the least and the most impact.



a) Upper Basin Development

b) Mainstream Development

c) Tributary Devlopment

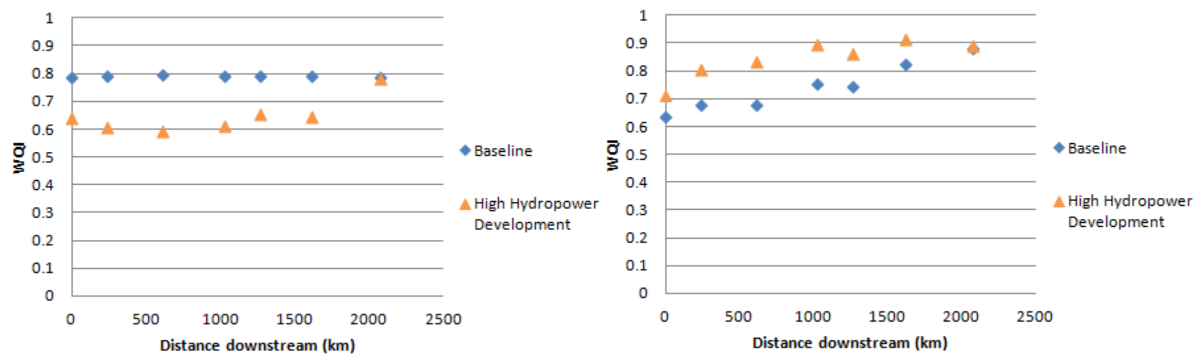
d) High Basin-wide Development

Figure 14a-d: Different stakeholder WQIs applied to four hydropower development scenarios with a 50% trapping efficiency, with a higher weighting given to TSS than COD and NO3

Table 5: Maximum concentration limits (in mg/l) for total suspended solids for each stakeholder in each quality band. WQI for fisheries and agriculture use the percentage of the baseline concentration.

WQI bands	Drinking Water Supply	Fisheries	Agriculture	Hydropower Turbine
0.9 - 1	20	>80%	>80%	10
0.7 - 0.9	50	80%	80%	50
0.5 - 0.7	100	60%	70%	100
0.3- 0.5	500	50%	40%	200
0 - 0.3	>500	30%	30%	>200

This demonstrates that the hydropower development is likely to have positive benefits for some stakeholders but have negative impacts on others. Thus the model serves as a dialogue tool between the different stakeholders to communicate their views about the hydropower development and other concerns in the basin and initiate dialogue to negotiate between the different trade-offs in socio-economic and environmental benefits and consequences.



a) Fishermen's perspective b) Alternative perspective

Figure 15a-b: Different perspectives of WQI for fish habitats.

The hypothetical fisherman's WQI was compared with a WQI based on a general water quality criterion for freshwater fisheries as given in Alabaster & Lloyd (1982) in Figure 15. They show that the fisherman's WQI indicates that the hydropower development will negatively affect TSS, whereas the alternative WQI indicates that the reduces solids will improve the WQI for the fisheries. This shows the different perspectives and criteria that can exist for assessing the same impact and therefore the need to involve all interested stakeholders to understand the impacts specific to the regional location.

The same WQIs have been applied for the entire river length; however different stakeholders along the length of the river at different points downstream may have different requirements for water quality and differing limits in their WQI that would shift their results. The WQIs used here aren't necessarily how each stakeholder sees water quality, but it shows the flexibility of the tool in creating their own WQIs based on constituents and their concentrations. Each stakeholder in the basin will ask different questions about how changes in the catchment will affect them. Not all of them can be answered by the tool but it can be used by the different stakeholders to address some of these. The above analysis bases the WQIs on the outputs from the original calibrated baseline model. Further investigation could be carried out to compare how different calibrations of the model by the stakeholders affect the resulting WQIs from the same scenarios.

The water quality indices can be further developed in several ways. They could be further developed to look at a number of other parameters, such the effects of thermal stratification in the reservoirs on oxygen levels or the increase in nutrient loading on the rate of algae growth where each parameter are given different weightings based on their importance.

The multiple WQIs developed here only looked at the total suspended solids and were based on subjective information about how the change in TSS would affect each stakeholder. These could be expanded to include criteria for the other quality constituents and to be produce more informative WQIs with the quality within each band representing different requirements. They could also incorporate other water quality parameters such as dissolved oxygen, electrical conductivity and coliforms, but these would require further model development and calibration making the model more complex with more parameters.

4. CONCLUSIONS

A regional water balance and quality model was developed and applied to the Lower Mekong Basin. The model was shown to adequately predict annual and seasonal flows after calibration, and also model the quality parameters: chemical oxygen demand, nitrates and total suspended solids using a simple mass balance equation with COD and TSS models extended with first-order decay models. Although it does not predict exact flow rates or concentration loadings, the simplicity of the model and the limited data requirements and parameters requiring calibration mean that it could be useful for water managers and policy makers to use as an aid in decision-making and as a collaborative tool for stakeholder participation.

The model includes the development of a water quality index based on international drinking standards as a way of presenting water quality information in a concise manner. Whilst this means the results lose some of their preciseness in being condensed in the WQI, it can be used to make comparisons and is easily adaptable for other purposes as demonstrated in the hydropower scenario discussion.

The model was used to explore various climate, land-use and hydropower scenarios and some of the impacts they can have on water quantity and quality in the basin. The report illustrates how stakeholder inputs can be used in the model to facilitate multi-stakeholder communications about the sustainable and equitable water resource management within the basin. The tool has its limitations and cannot answer all questions nor provide accurate answers to specific questions necessary to make management decisions. It can however be used to gain useful insight into how macro-scale changes within the basin can affect different regions and stakeholders. It can hence be used in addition to other tools to support discussions between stakeholders about issues faced at a catchment-level.

The nature of the project means that further work can be done to expand and develop on ideas presented in this report. The work done on the water quality index can be developed in several ways to assess water quality for different uses. The model is developed with input data for the Mekong River Basin. The simple model structure means it could be transferred to other basins and study areas with different input datasets and would only require calibration of a few model parameters. It could then be used to further explore various scenarios of significance and understand how they affect the water availability and water quality and the impacts on stakeholders such that adaptation strategies can be planned.

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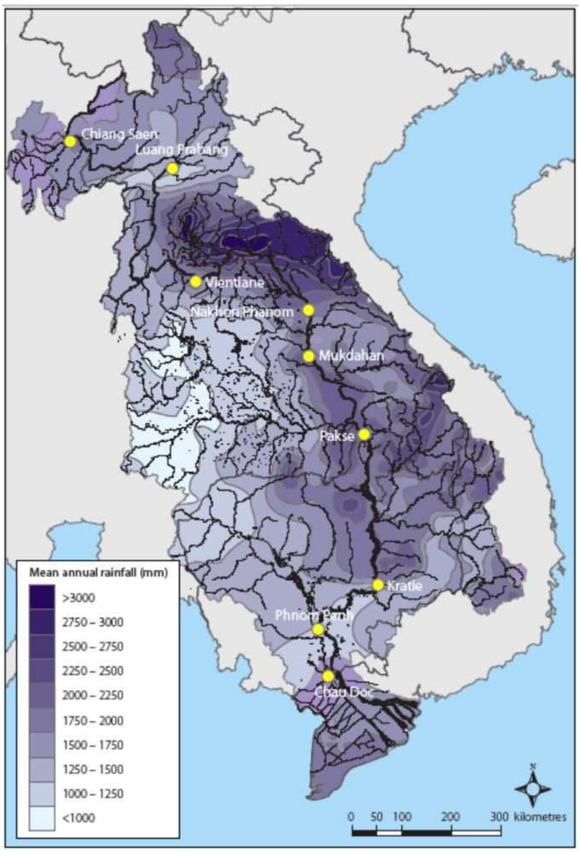
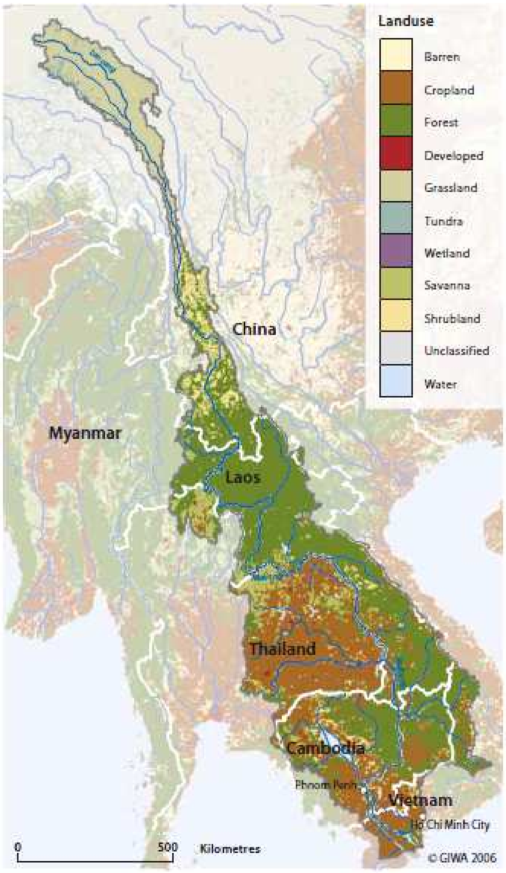
Appendix A

Map showing spatial distribution of average annual precipitation in Lower Mekong Basin (MRC, 2010).



Appendix B

Land use map shows distribution of different land uses (Snidvongs & Teng, 2006)



Appendix C

Table summarising single parameter runs used to test the sensitivity of the model.

Run ID	Run Description	Precipitation	Temperature	LandUse Runoff coefficient	COD - scaling factor	COD - decayrate	Nitrate-scaling factor	TSS- scaling factor	TSS- decayrate
Mekong Basin Case Matrix									
	Single parameter sensitivity runs								
001	BASE CASE (CALIBRATED to Yr 2000)	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000
002	High Precipitation	**125% 2000**	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000
003	Low Precipitation	**75% 2000**	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000
004	High Temperature	100% of 2000	**+4 C of 2000**	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000
005	Low Temperature	100% of 2000	**-4 C of 2000**	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000
006	High LandUse Runoff Coefficient	100% of 2000	100% of 2000	**125% 2000**	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000
007	Low LandUse Runoff Coefficient	100% of 2000	100% of 2000	**75% 2000**	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000
008	High COD loading scaling factor	100% of 2000	100% of 2000	100% of 2000	**125% 2000**	100% of 2000	100% of 2000	100% of 2000	100% of 2000
009	Low COD loading scaling factor	100% of 2000	100% of 2000	100% of 2000	**75% 2000**	100% of 2000	100% of 2000	100% of 2000	100% of 2000
010	High COD decay rate constant	100% of 2000	100% of 2000	100% of 2000	100% of 2000	**125% 2000**	100% of 2000	100% of 2000	100% of 2000
011	Low COD decay rate constant	100% of 2000	100% of 2000	100% of 2000	100% of 2000	**75% 2000**	100% of 2000	100% of 2000	100% of 2000
012	High Nitrate loading scaling factor	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	**125% 2000**	100% of 2000	100% of 2000
013	Low Nitrate Loading scaling factor	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	**75% 2000**	100% of 2000	100% of 2000
014	High TSS loading scaling factor	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	**125% 2000**	100% of 2000
015	Low TSS loading scaling factor	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	**75% 2000**	100% of 2000
016	High TSS decay rate constant	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	**125% 2000**
017	Low TSS decay rate constant	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	100% of 2000	**75% 2000**

Appendix D - Retrospective Risk Assessment

The only risks associated with the project were health related risks associated with computer use as detailed in the initial risk assessment. These include poor posture whilst using computer equipment resulting in neck and back aches, visual problems through incorrect position of screen and looking at the computer screen for long periods of time, and fatigue through insufficient breaks. This reflects the risks encountered during the project, which whilst significant could easily be overlooked.