Experimental and Numerical Analysis of River Lake System and Non-traditional Water Usage in a new Eco-City

By FEI WANG

A thesis submitted to Cardiff University
in candidature for the degree of
Doctor of Philosophy
in
Civil Engineering

Cardiff, UK, 2013

DECLARATION

This work has not been submitted in substance for any other degree or award at this or any other university or place of learning, nor is being submitted concurrently in candidature for any degree or other award.

Signed …………………………… (candidate) Date …………………………

STATEMENT 1

This thesis is being submitted in partial fulfillment of the requirements for the degree of …………………………(insert MCh, MD, MPhil, PhD etc, as appropriate)

Signed …………………………… (candidate) Date …………………………

STATEMENT 2

This thesis is the result of my own independent work/investigation, except where otherwise stated.
Other sources are acknowledged by explicit references. The views expressed are my own.

Signed …………………………… (candidate) Date …………………………

STATEMENT 3

I hereby give consent for my thesis, if accepted, to be available for photocopying and for inter-library loan, and for the title and summary to be made available to outside organisations.

Signed …………………………… (candidate) Date …………………………

To my parents, nephew, niece

And

My husband

ACKNOWLEDGMENTS

I would like to express my gratitude to my supervisor, Professor Binliang Lin, giving me this opportunity to pursue my PhD study at Cardiff University. He provided support whenever I feel frustrated and guided me through the maze of doctoral research. Thanks are also due to Dr William Rauen and Dr Shunqi Pan for providing valuable suggestions on the laboratory work.

Many thanks to Yuheng Ren for being able to do the experiments with me and I will never forget those days filled with laughter and tear. I am also grateful to all members of the Hydro-Environmental Research Centre, for their help, especially Davor Kvocka - I am so lucky to know you and you made the last few months of my PhD much more enjoyable. Thanks also to technicians in the Hydraulic laboratory, particularly Paul Leach.

This work was supported by grants from Ove Arup & Partners, China Scholarship Council and Cardiff University. These supports are gratefully acknowledged.

Finally, to my dear husband, Zhidong Zhang, your patience and understanding over the past few years cannot be underestimated - thank you for being there constantly, for encouraging me, and for always making me calm down.

ABSTRACT

In recent years, Eco-City, which is designed with consideration for environmental impact and is inhabited by people dedicated to minimisation of required inputs of energy, air pollution and water pollution, has emerged as a way to address sustainability issues by adapting it to their local needs and context. The sustainability of urban water resources, water recycling and more efficient use of water resources will be the key features of the Eco-City. The current study takes Sino-Singapore Tianjin Eco-City as an example to investigate the sustainable use of water resources which focus on non-traditional water usage and ecological water requirements assessment.

Firstly, the potential non-traditional water supply was evaluated based on the data acquired from the gauging station and the Eco-City planning data. It was found that rainwater has a great potential for domestic use in the Eco-City from June to September. Differing from other water consumption, ecological demand of the river lake system in the Eco-City was analysed by minimum ecological water requirements determination. An improved wetted perimeter method was used in order to determine the minimum ecological water requirements in the river system. It was found that the current monthly flow rates, with the exception of January to March, are fairly satisfactory.

Secondly, an idealised river-lake system was assessed by hydraulics laboratory experimentation and 2D numerical modelling. The experimental and numerical investigations described in this study were undertaken to improve understanding of the hydrodynamic and flushing processes within such a river lake system. A water diversion scheme was implemented in order to study lake recharge by river water during dry periods and under augmented flows. Fluorescent tracer experiments and related computer simulations were conducted to assess the performance of different parts of the system before and after implementing the diversion scheme. The results showed that such measures improved flushing, as seen from the perspective of reducing the mean detention time. However, due to poor cross-sectional velocity distribution, recharge alone had little impact on the overall mixing level in the lake waters. The effect of inserting flow deflectors near the lake inlet combined with flow augmentation was then assessed and was found to positively affect the distribution of solutes, by mitigating the occurrence of dead zones.

Finally, an eco-hydraulic model was used to determine the levels of fish habitat suitability in the fluvial and lacustrine regions of a new Eco-City. This model has been developed by combining a depth integrated hydrodynamic and water quality model with a Habitat Suitability Index model. Carps were selected as the target species as they represent the major fish population in the study area. Hydrologic data recorded during 2001-2010 were analysed to determine the base flow, average flow and high flow rates, which were used to represent the discharges in the river for the three stages of the carp life cycle: overwintering, spawning and growth, respectively. Numerical model simulations were undertaken to determine the levels of habitat suitability for carps to live at these three life stages. The model results indicated that under the current flow regime the habitat suitability level in the lacustrine region is too low for carps at the growth and overwintering stages. DO depletion, overriding the role of velocity and depth, was attributed to the poorly suited habitat conditions in the lacustrine region. To

improve the suitability conditions in the lacustrine region, a DO enhancement scheme was used. Model results showed that the scheme has significantly enhanced the water quality in the lacustrine region. Due to the high flow requirement for carps to spawn in the fluvial region, further numerical model simulations were undertaken to investigate the effect of flow augmentation on the carp spawning habitat suitability.

Contents

List of Figures

List of Tables

CHAPTER 1 INTRODUCTION

1.1 Research Background

The world's population is growing at a rapid pace, and a significant percentage of the population is now moving from the countryside into cities. It is anticipated that in the next few decades there will be an unprecedented scale of urban growth in the developing world. By 2030, nearly 60% of the global population is projected to be urban with the developing world (Prasad, 2009). The major challenges for cities facing rapid population growth are the shortage of natural resource supply and the disposal of a growing volume of waste. Cities are affected by overcrowding, environmental degradation and inadequate housing infrastructure and services. In order to mitigate the problems of the urbanisation, city planners are becoming increasingly conscious of the need to consider environmental and sustainability issues when planning any development. Many countries tend to place an emphasis on the economic and social dimensions of sustainability, whilst allowing a more manageable and incremental approach to environmental sustainability. Making existing cities and new urban development more ecologically friendly and liveable is an urgent priority in the global push for sustainability. Urban sustainability can only be ensured by human understanding of the complex interactions between environmental, ecological, economic, political, and social factors, and with careful planning and management based on ecological principles. A particularly interesting and key strategy used by many cities is that of sustainable urban development, which is a mode of urban development in which resource use is aimed at meeting human needs while ensuring the sustainability of natural systems and the environment. In 1987, the United Nations released the Brundtland Report, which included the most widely recognised definitions of sustainable development: "Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (World Commission on Environment and Development, 1987). The United Nations World Summit Outcome Document (2005) referred to the "interdependent and mutually reinforcing pillars" of sustainable development as economic development, social development, and environmental protection. Based on the triple bottom line, numerous sustainability standards and certification systems have been established.

In recent years, Eco-City has emerged as a way to address sustainability issues by adapting it to their local needs and context. An Eco-City represents a type of city construction which takes into consideration ecological requirements combined with social and economic conditions. In other words, a city produces its own energy, food and water in a way which does not have detrimental effects on the world in forms such as waste, water pollution or damage to the air. Eco-Cities demonstrate that urban growth and development can be a sustainable process and that the concept of sustainable development can be applied to an urban setting. There are many cities in the world which are working on developing themselves as Eco-Cities, such as St. David in the UK, Calgary in Canada, Sino-Singapore Tianjin Eco-City in China and Manimekala Hightec Eco-City in India. Each individual Eco-City development has also set its own requirements to ensure that the city is environmentally sustainable, with these criteria ranging from zero-waste and zero-carbon emissions to simple urban revitalisation.

Concerning the sustainability of urban water resources, water recycling and more efficient use of water resources will be one of the key features in Eco-Cities, which means the use of non-traditional water has been seen to play an ever more important role in satisfying the increasing water requirement. Due to urbanisation and industrialisation, the runoff and waste water is increasing, which not only results in the deterioration of the ecological environmental and pollutes water sources, but also brings about a contradiction between water supply and demand to aggravate. Here we can take China as an example; indeed, among the country’s 661 cities, approximately 420 are facing the pressures of water resource shortage, wastewater treatment and aquatic environmental management (CCICED 2005). Some northern cities are forced to restrict water supply. Water shortage is further magnified by pollution, over-tapping of aquifers and wasteful use.

With this in mind, there are increasing opportunities to use water in a non-traditional manner, such as rainwater, desalinated water and recycled wastewater. This would lead to a reduction in the demand for potable water and the associated energy needed for treatment. It would also help mitigate climate change impacts of increased storm rainfall intensity on flooding. In many places, non-traditional water has already been utilised for various purposes such as washing, water cooling, toilet flushing, waterway restoration and the creation of recreational waterfront.

1.2 Key Objectives

A large number of studies on non-traditional water usage have been conducted over the past few years. With this said however, there is a lack of knowledge regarding how sustainable non-traditional water allocation should be attained and to what extent the non-traditional water should be utilised for various purposes such as typical domestic use, water cooling, ecological water requirements and the creation of recreational waterfronts. In this thesis the framework of a system analysis dealing with the above issues is described, with a focus on the assessment of non-traditional water usage and ecological water requirements. Ecological water requirements are a significant part of non-traditional water use which it is needed to provide the decision maker and the users with a concrete and clear answer regarding how an ecosystem can make use of non-traditional water resources in a way which is economically viable and environmentally non-degrading.

Therefore, the key objectives of this study are as follows:

- Determine the availability of non-traditional water supply and the potential water use in the Tianjin Eco-City;
- Estimate the minimum ecological water requirements to maintain the environmental water quality standard in the Eco-City;
- Develop an eco-hydraulic model to determine the fish habitat suitability in the river lake system in the Eco-City;

To achieve these objectives, the potential non-traditional water supply was first evaluated based on hydrological and hydraulic data acquired from gauging stations and the data in Eco-City planning Office, such as the planned population, monthly sewage and industrial effluent volumes etc. Secondly, detailed studies on a physical scale laboratory model and numerical modelling analysis were carried out to investigate the flow distribution and habitat suitability conditions in the river- lake system in the Eco-City.

1.3 Dissertation Organization

This dissertation is organised as follows: Chapter 2 presents a literature review of past studies related to the research on non-traditional water resources and ecological water requirement assessments. Chapter 3 presents the methods used to determine the non-traditional water supply and the potential water use in the Eco-City, particularly for ecological water requirements; Chapter 4 presents an idealised river-lake system assessed by hydraulics laboratory experimentation. A water diversion scheme was implemented to simulate lake flushing by river water. Chapter 5 presents the experimental results and numerical solutions for the flow augmentation and lake flushing improvement. Chapter 6 presents the assessment of habitat suitability for fish in the fluvial and lacustrine region in the Eco-City. Finally, Chapter 7 of this thesis provides a summary of the work presented and a discussion of limitations and followed by suggestions for future research.

CHAPTER 2 LITERATURE REVIEW

This chapter presents a review of existing studies on the Eco-City development and the relevant non-traditional water usage, ecological water requirements, and current methods for calculating the ecological water requirements. Most content in this section focusses on the theoretical research.

2.1 Eco-City

2.1.1 Eco-City Development

The urbanisation and increasing awareness of the ecological characteristics of cities has led urban planning and landscape scholars to find a new way of building and developing cities which appreciate and integrate a site's ecological and environmental conditions. Richard Register, one of the early advocates for linking ecological principles to the redesign of cities first coined the term "Eco-City" in his 1987 book, Eco-City Berkeley: building cities for a healthy future (Richard Register, 1987). The first Eco-City concept focussed primarily on urban metabolism, i.e. circles of energy, water, wastes, as well as the protection of the environment in an urban context. Over the past few decades, many Eco-City concepts can be found in planning theories. However, throughout the 1980s and 1990s, there was no commonly accepted definition of “Eco-City”. The term ‘Eco-City’ remained mainly a concept, a collection of ideas and propositions about sustainable urban planning, transportation, housing, public participation and social justice, with practical examples relatively few and far between (Roseland, 1997).

In recent years, the definition of Eco-City is becoming clearer and in some countries a campaign for Eco-City development is being spontaneously initiated. Kline (2000) suggests that, ‘‘an Eco-City encompasses four basic community characteristics, which are ecological integrity, economic security, quality of life and empowerment with responsibility’’. Similarly, Roseland (2001) indicates that a collection of apparently disconnected ideas about urban planning, transportation, health, housing, energy, economic development, natural habitats, public participation, and social justice all comprise a single framework, the eco city. An Eco-City is a city designed with consideration for environmental impact, and is inhabited by people dedicated to minimisation of required inputs of energy, water and waste output of heat, air pollution and water pollution. During the past decade, there has been renewed effort to apply the Eco-City concept to projects at the district and city scale. Many Eco-City projects are getting started in Europe, Asia, Africa, and South America. Eco-Town, a similar concept to that of Eco-City, is a government-sponsored programme which aims to have new towns built in England in the hopes of achieving sustainable development. In 2007, the department for Communities and Local Government (CLG) announced a competition to build up to 10 Eco-Towns across the UK (BBC, 2007). The city of Waitakere, the Western part of the greater Auckland urban region, was New Zealand's first Eco-City, working from the Greenprint, a guiding document which the City Council developed in the early 1990s. The Eco-City idea has evolved, mostly in the United States and Western Europe. Gothenburg and especially Älvstaden in particular are good examples of Eco-Cities in Sweden (Boslet, 2010). They have low environmental impact，contain passive houses, use a good recycling system for waste, etc. China, as a nation of 1.3 billion is urbanizing at a dizzying pace, powered by economic growth that is lifting millions out of rural poverty, is working with investment and technology to build Eco-Cities which aim to address several urban issues such as energy efficiency, water and waste management, economic vibrancy and social harmony. Figure 2-1 shows the main Eco-City projects in China, among which Tianjin Sino-Singapore Eco-City and Caofeidian Eco-City are now under construction, and the Dongtan Eco-City project has also just started recently.

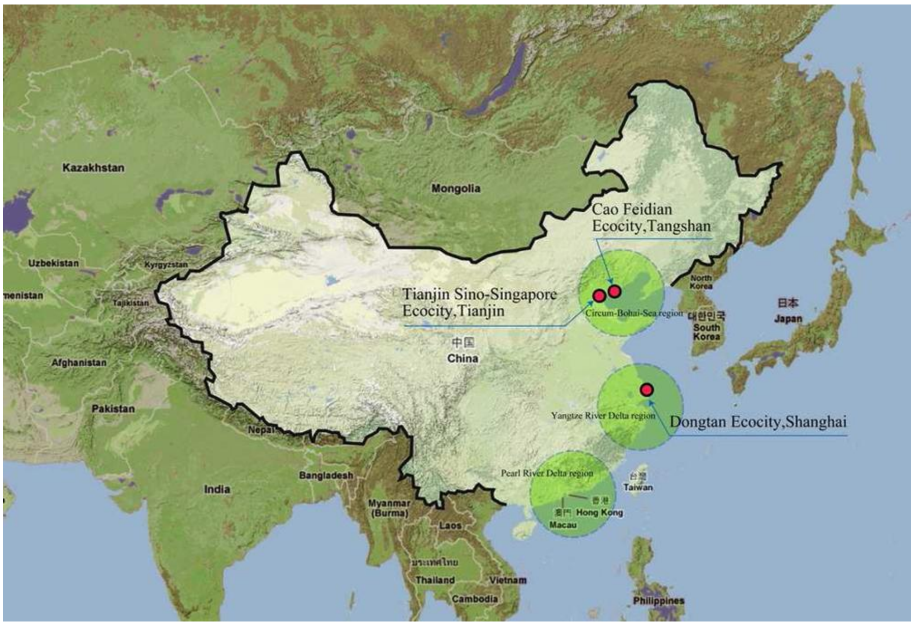


Figure 2-1 Eco-City construction in China (Ma, 2009)

With all of this said however, achieving greater sustainability in Eco-Cities requires an in-depth understanding of how or to what extent sustainable development can be applied to an Eco-City context. There are no existing indicator systems to measure how sustainability can be applied to Eco-Cities. It is therefore useful to identify several relevant Eco-City dimensions relating to: energy and climate; water quality, availability, and wastewater treatment; air quality; waste production and treatment; transportation; economic development and economic health; land use and urban form; and demographics and social health (Zhou & Williams, 2013). In terms of planning the concepts which underpin planning theory and practice are indistinct by nature and can often lead to disappointing outcomes for planning initiatives. There is a need to develop a concrete framework for Eco-City planning so as to help achieve operational sustainable cities which incorporate all aspects of environmental, social, and economic health through efforts to conserve natural resources, reduce and recycle waste streams.

2.1.2 Sustainable Urban Water Management (SUWM)

With Eco-City development, sustainable management of water resources is an increasingly pressing concern. How to make the water safe for use and see that it is reused and returned to nature has been a key feature of sustainable urban water management over the past few years. Dating back to 1999, the Swedish Foundation for Strategic Environmental Research (MISTRA, 1999) initiated a 6-year research program entitled "Sustainable Urban Water Management" with the aim of improving and raising knowledge with regard to sustainable water management. This is now an accepted concept in water resources management, incorporating integrated water management and total water cycle management. It is also considered highly desirable and a much needed trajectory for urban water management (Ashley et al., 2007).

More efficient and sustainable use of water resources will be one of the key features in the Eco-City. Here we can take Tianjin Eco-City as an example; the Administrative Committee developed a set of Integrated Water Management Guidelines (IWMG) for the Eco-City which bring together all facets of the water cycle covering water supply, sewage management, water treatment and storm water management. It is a process which promotes the coordinated development and management of water, land and related resources in order to maximise economic and social welfare. Water bodies in the Eco-City will be linked together for greater water circulation to enhance the ecology and to provide an attractive environment for waterfront development and water-based recreational activities. A wastewater pond will be rehabilitated and transformed into a clean and beautiful lake (Singapore Government, 2008).

2.2 Non -Traditional Water Resources

Traditional water resources generally include surface and ground water. Resources other than “good quality surface or ground water” are termed “non-traditional water resources”. Examples of such resources include rainwater, reclaimed domestic and industrial water, and desalinated seawater. Several driving forces can be identified in the worldwide practices of non-traditional water use such as water shortage caused by very low amounts of rainfall in combination with high evaporation, large freshwater demand from the population, or certain environmental and economic considerations. There are major opportunities to use non-traditional water. Using non-traditional water leads to a reduction in the demand for potable water and the associated energy needed for treatment. It can also help mitigate climate change impacts of increased storm rainfall intensity on flooding. Non-traditional water can be utilised by means of scientific and effective regulation and management of the existing reservoirs, construction of rainwater collection projects and a multi-channelled investment system for exploitation of non-traditional water resources.

The last decades has seen a number of studies carried out to understand the principles of non-traditional water for non-potable use, especially for agriculture production. For example, basic investigations, field surveys and experimental tests on the effects of municipal wastewater irrigation have been carried out in Sicily (Indelicato et al., 1988). In Turkey, several studies were conducted in order to obtain certain applicable parameters regarding the use of non-traditional water including wastewater and saline water in agriculture during more than 20 years (Kanber, 2003). The Gulf States were working together and developing non-traditional alternative water sources to bridge the gap between supply and demand (Alsharhan, 2001). In Egypt, where there are many important projects for the reuse of wastewaters, certain general technical guidelines have been provided concerning kinds of treatment and the quality of the effluents (Abdel-Ghaffar et al., 1985). In France the reuse of wastewater has been practised for more than a century and has covered almost all the territory, generally in areas spanning a few hectares (Soulie et al., 1991). In Tokyo, Japan, the reuse of treated wastewater has been highly promoted. A typical use of the reclaimed water is for toilet flushing, with approximately $97\times10^4 m^3$/year (Odeh & Jayyousi, 2003). According to the German market leader for prefabricated concrete tanks, Mall-Beton GmbH (1999a), during the last 10 years they have installed more than 100,000 decentralised rainwater storage tanks for service water purposes in Germany, providing a total storage capacity of more than 600,000 m^3. Seawater represents another theoretically unlimited water resource, particularly for coastal areas, and at present there are approximately 4,000 desalination plants in 120 countries worldwide with a combined capacity of over 5750 M m^3/year (Mielke, 1999). In arid and semi-arid regions, as in most of the Mediterranean countries, the use of non-traditional water resources has been seen to play an increasingly important role in satisfying the escalating water requirements.

As the main non-traditional water resources, rainwater, reclaimed domestic and industrial water, and desalinated seawater will be discussed in detail in Section 2.2.1, Section 2.2.2, and Section 2.2.3 respectively.

2.2.1 Rainwater

The most cost-effective and longest serving form of non-traditional water is harvested rainwater, which is actually nothing new as it has been in widespread use for many years. The rapid expansion of the city has led to the sealing off of large surface areas, and increasing the speed and volume of storm water run-off. Therefore, the collection and management of rainwater can contribute to water conservation while reducing the risks of flooding after heavy rainstorms. In 2003, the Scottish executive issued a national flooding framework to implement of Sustainable Drainage Systems and rainwater harvesting and reuse (CIRIA, 2007). There is also increasing support from governments and councils when it comes to the rainwater harvesting system. In 1996, urban ecologist innovators, Ole & Maitri Ersson (1996) built Portland's first permitted rainwater harvesting system to significantly supplement their residential water needs. The system is designed to harvest and purify rainwater for all of their water-related needs except during long dry summers when they are able to switch back to city water. The Queensland Water Commission (2010) has proposed the reuse of rainwater from household roofs to supply typical domestic use, such as, toilet flushing, laundry and outdoor uses as a means to reduce the demand for potable water.

The components used in a rainwater harvesting system may vary from one system to another, although generally speaking every rainwater catchment system consists of three main components: (1) a catchment surface for collecting rainwater; (2) a storage reservoir or storing rainwater; and (3) a delivery system for transporting water from the catchment to the storage reservoir. For example, a typical rainwater harvesting system is shown in Figure 2-2.

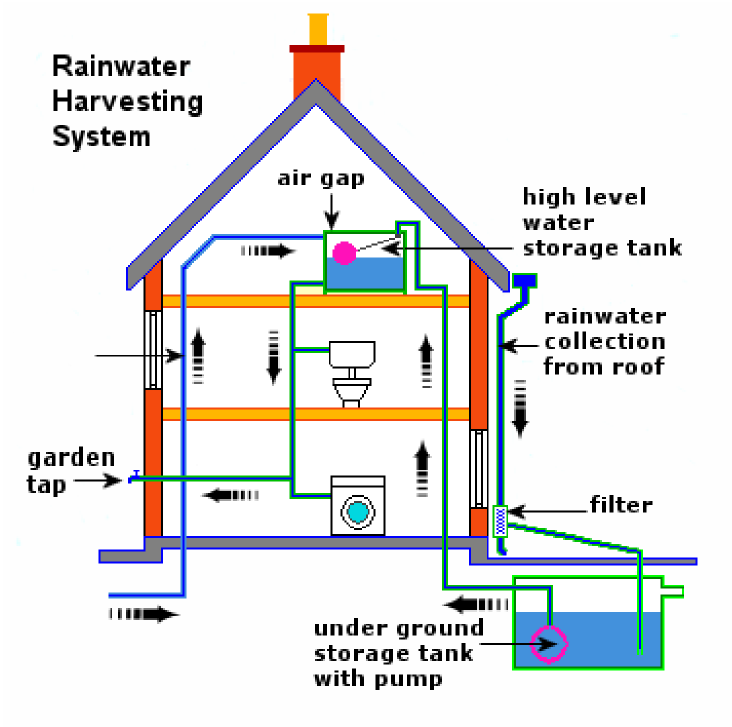


Figure 2-2 Typical rainwater harvesting system

(Quality Domains Ltd, 2010)

2.2.2 Reclaimed Domestic and Industrial Wastewater

Reclaimed domestic and industrial wastewater is emerging as an established water management practice in several water-stressed countries. Over the past decades reclaimed wastewater has been increasingly used for landscape irrigation in urban areas and for groundwater recharge. Domestic sewage is the largest source of reusable water resource, followed by industrial effluents. Indeed, industry should be encouraged to invest in better water efficiency, more recycling and management. Water use normalised indices can be developed for each industry in order to allocate only as much water as is necessary to achieve their production targets. Beneficial use of reclaimed wastewater has been practised in California since the 1890s, when raw sewage was applied on 'sewer farms' (EPA, 1977). The International Water Management Institute (IWMI, 2008) released a report at the World Water Week and highlighted the need to develop practical measures when utilising wastewater while avoiding potential environmental and health risks. Treated municipal wastewater is widely available in communities throughout the United States in sufficient volumes and is reliable enough to supply power plant cooling water. Reclaimed wastewaters are already being used in more than 50 U.S. power plants and are subject to Federal and state regulations in order to protect worker and public health.

2.2.3 Desalinated Seawater

Independent of rainfall, droughts, and other natural phenomena related to irregular water supplies, desalination enables the seawater to be explored, with the aim of producing high volumes of high quality non-potable water and even potable water. Approximately 75 million people worldwide obtain their potable water from desalination plants, and that number is due to increase with the continuous growth in demand for water. According to the International Desalination Association, in 2009, 4000 desalination plants operated worldwide, producing 59.9 million cubic metres per day, a year-on-year increase of 12.3% (Henthorne, 2009). The production was 68 million m^3 in 2010, and is expected to reach 120 million m^3 by 2020.

Most of the modern interest in desalination of seawater is focussed on developing cost-effective ways of providing fresh water for human use. Factors which determine the costs for desalination include capacity and type of facility, location, and concentrate disposal. However, improved technology has cut the cost of desalination in half over the past decades. Major desalination technologies are based on evaporation methods, Electro Dialysis (ED), and Reverse Osmosis (RO), the latter of which is responsible for 43.5% of the global desalination capacity (Crittenden et al., 2005; Al-Subaie, 2007).

2.2.4 Potential Usage of Non-traditional Water Resources

In recent years, scientists, engineers and environmentalists have combined their skills in the development of a comprehensive non-traditional water use scheme for the sustainable exploitation of this valuable water resource. The scheme deals with all aspects of wastewater management with the objective of maximising utilisation of the water resources. Figure 2-3 shows a schematic description of the non-traditional water use procedure.

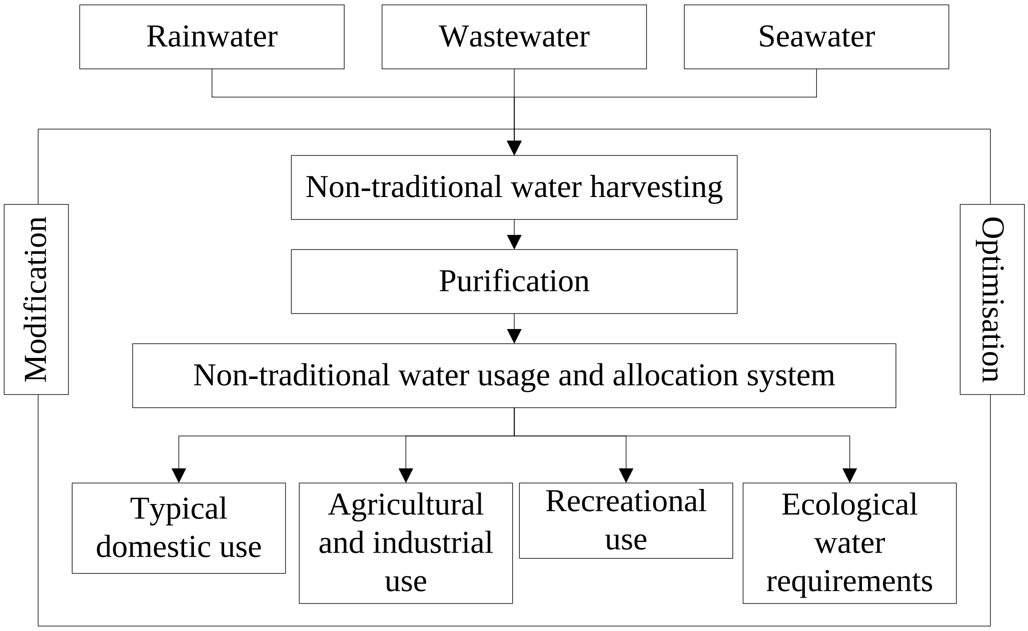


Figure 2-3 Schematic description of the non-traditional water use procedure

As shown in Figure 2-3, non-traditional water mainly focuses on the following non-potable applications:

• **Typical Domestic use**: e. g. toilet flushing, washing clothes, watering garden

•**Agricultural and industrial use:** e. g. irrigation, water cooling, diluting, incorporating water into a product;

•**Recreational water use:** e. g. Filling ornamental ponds, water features and fountains;

•**Ecological water use:** e. g. River and lake recharge to meet ecological water requirements.

It is important to first consider what standards of water quality are required for different types of non-traditional water usage. Different uses raise different concerns and therefore different standards are considered. Thus, based on variables which characterise the quality of water, water quality criteria have been developed by scientists on a specific water use. Many water quality criteria set a maximum level for the concentration of a substance in a particular medium (i.e. biota) which will not be harmful when the specific medium is used continuously for a single, specific purpose. For some other water quality variables, such as dissolved oxygen, water quality criteria are set at the minimum acceptable concentration to ensure the maintenance of biological functions. The European water Framework Directive (WFD, 2000) was established to improve and protect the surface water quality of the European countries. The Framework Directive sets standards for the quality of the surface water, for example, what the oxygen level in the water should be, and what types of fish should be found in the water. Table 2-1 roughly lists the water quality criteria by intended use.

2.2.5 Summary of Non-traditional Water Usage

A review of the literature indicates that there are major opportunities to use non-traditional water as an additional water resource. While a number of alternative non-traditional water sources and new technologies are theoretically available, the use of non-traditional water has a multidisciplinary nature as it can inter-link with the environment, health, industry, agriculture, and water resource policy. In this regard, significant challenges still remain in the area of technological, information transfer and social and environmental considerations.

Table 2-1 Water quality criteria for different uses (UNEP, 2008)

Designated use	Class of	Criteria
Drinking water	A	•Total Coliforms Organism MPN/100ml shall be 50 or less •pH between 6.5 and 8.5 •Dissolved Oxygen 6mg/l or more •Biochemical Oxygen Demand 5 days 20°C 2mg/l or less
Domestic use	C	•PH between 6.0 to 9.0 •Turbidity 5 NTU or less •Disinfection < 4.0 mg/L •Biochemical Oxygen Demand 5 days 20°C 10 mg/L or less
Agricultural and industrial use	E	•pH between 6.0 to 8.5 •Electrical Conductivity at 25°C micro mhos/cm Max.2250 •Sodium absorption Ratio Max. 26 •Boron Max. 2mg/l
Recreational use	B	•Total Coliforms Organism MPN/100ml shall be 500 or less pH between 6.5 and 8.5 Dissolved Oxygen 5mg/l or more •Biochemical Oxygen Demand 5 days 20°C 3mg/l or less
Ecological use	D	•pH between 6.5 to 8.5 Dissolved Oxygen 4mg/l or more •Free Ammonia (as N) 1.2 mg/l or less

2.3 Ecological Water Requirements (ERWs)

Among the main non-traditional water usages, most emphasis is placed on ecological water use, due to seasonal and annual variation. To protect the ecological functioning of water resources, some water must be left in rivers and lakes, and this is known as the ecological reserve, occasionally called Ecological Water Requirements (EWRs). EWRs describe water regimes needed to sustain the ecological values of water dependent ecosystems at a low level of risk (ARMCANZ / ANZECC, 1996). Other definitions and terms regarding environmental flows do exist in the literature (Acreman, 2004), including minimum ecological flow, optimal ecological flow and instream flow requirements. The impacts of river regulation and land use change on the ecological flow of Scottish rivers ecosystems were discussed by Gilvear et al (2002). However, to date, EWRs are the only regimes which truly encompass the holistic nature of the concept. EWRs serve to represent water allocation for ecosystems. As ecosystems, in turn, provide services to people, providing for environmental flows is not exclusively a matter of sustaining ecosystems, but is also a matter of supporting humankind/livelihoods, particularly in developing countries. Water dependent ecosystems need particular flow regimes which determine both the physical structure of the stream and the species which are found. These flow regimes include the timing, frequency, duration, volume and quality of water they receive. Fish and other stream life within these systems have adapted to the particular water regime. If the flow regimes were changed, this could affect habitat availability, food supplies, chemistry and nutrient processing. This can in turn lead to a loss of biodiversity, a decline in the ecological condition and a decline in river water quality. For a lake or reservoir the flow regime is considered here as the depth and time variations in depth of the water body. For rivers the flow regime is considered here as discharge and time variations in discharge. Changes in discharge alter flow depths and velocities, which alter the nature and extent of aquatic habitats in rivers, as well as physical and chemical processes. It is recognised that both water quantity and quality must be protected in order to contribute to the conservation and restoration of water-dependent ecosystems.

2.3.1 EWRs Research

Ever since the mid-70s, the recommended ecological flow for a given river has been defined (Tennant, 1976). Indeed, early EWRs studies were focussed on the concept of the minimum flow requirements based on the idea that the river ecosystem will be conserved as long as the flow is kept at or above a minimum level. For example, Tennant (1976) developed a method using data taken from hundreds of sites on rivers in the mid-western states of the USA to specify minimum flows to protect a healthy river environment. Gradually, it became apparent that all the elements of a flow regime, including floods, medium and low flows are important to the ecosystem (Junk et al., 1989). Any alteration of the regime will lead to ecological changes. The ecological flow will have to be very close to the natural flow regime in order to maintain a pristine natural river ecosystem.

The importance of ecological flows was highlighted in the early 1990s when many freshwater ecosystems had been degraded to the point that they could no longer support biodiversity. The Australian Capital Territory (ACT) government in Australia even published Environmental Flow Guidelines specifying the flows required to maintain aquatic ecosystems in 1999 (ACT, 1999). Since then water allocation and licensing in the ACT has been based on the Guidelines, ensuring that sufficient water is allocated for the environmental needs of aquatic ecosystems in the ACT. The volume of water and the timing of flows specified in the 1999 Guidelines were determined to protect the health and viability of the river systems. Indeed, many methods have been developed and used to estimate EWRs internationally. These methods differ significantly in terms of the data requirements. For example, some methods require only hydrological data, while others require hydraulic and biological information.

With the advancement of computing power, the use of numerical models for conducting EWRs has become widespread for rivers, lakes, estuaries and coastal waters. For example, one-dimensional (1D) flow models are often used to analyse a river reach by breaking it into discrete cells, with each being assigned a single depth and velocity value. The HEC-RAS software is an example of the type of models capable of modelling a network of channels, a dendritic system or a single river reach and several simulation features have been added to the new version 4.1 program since it was released (USACE, 2010). However, the inability of the 1D model to describe two-dimensional (2D) flow processes favours the use of 2D hydrodynamic models as predictive tools in ecological flow studies for lakes and reservoirs. 2D numerical modelling is a very effective method through which to study the ecosystem habitat requirements. The most promising aspect of 2D models in ecological flow studies is their potential to accurately and explicitly quantify spatial variations and combinations of flow patterns important to stream habitat (Bovee, 1996).

Recent developments in the fields of EWRs have provided an opportunity to link 'biological' and 'physical' processes of aquatic habitat. Linking physical habitat conditions in rivers to their ecological characteristic is now a fundamental requirement in river management and river restoration (Bockelmann et al., 2004; Clifford et al., 2010). Many researchers have combined hydrodynamic and ecological models. For example, Nagaya et al. (2008) used a horizontal 2D numerical model to predict the EWRs of ayu (*Plecoglossus altivelis*) with the preference curves of the flow depth and velocity. Yi et al. (2010) developed a mathematical model to predict the minimum in-stream flow and suitable daily discharge during the reproduction season for the carp species in the Yangtze River, in which the habitat suitability curves were coupled with the mathematical model.

Overall, a major trend in EWRs assessment has been a shift from narrow studies which concentrate on one single method to a holistic approach. The categorisation and details of the EWRs assessment methods will be discussed in 2.3.2.

2.3.2 EWRs Assessment Methods

2.3.2.1 Hydrologic-based methods

Hydrologic-based methods are the methods which rely primary on historical flow records. They are the earliest and most widespread EWRs assessment methods and are generally used as preliminary estimates. The most frequently used methods include the Tennant Method (Tennant, 1976; Hughes & Hannart, 2003) and Flow Duration Curve Method (FDC) (Arthington et al., 1992a). A major assumption of Hydrologic-based methods is that the most frequent conditions over a period of record are suitable for all life history stages without any examination of short-duration perturbations and species responses. As illustrated by Table 2-2, the general flows for different habitat quality are specified by the Tennant method: 10% of the average discharge is the minimum flow needed to sustain short-term survival habitat; 30% maintains a good quality habitat for aquatic life, whilst optimum habitat occurs at 60%-100%. The FDC method was developed by constructing the modified flow based upon set cut levels in each month. Arthington et al. (1992a) suggested that the 50% flow for each month represented the ideal, whereas the 20% flow should be regarded as the lowest flow permissible in drought years. The improved aspect of the FDC method is that incorporation of monthly flows allows for the maintenance of the natural temporal pattern of intra-annual variation. Furthermore, additional volumes could be added to monthly allocations to achieve specific ecological purposes or to accommodate for downstream abstraction or diversion.

Hydrologic-based methods are essentially desktop methods which use historical flow records, to evaluate suitable flows, often for fish habitats. Their major benefit was considered to be limited to the early reconnaissance phase of the allocation process and had only moderate data requirements.

Table 2-2 Flow requirements for different habitat quality by Tennant Methods

Flow category	Flow in proportion of average discharge	Habitat quality level
Optimum	60-100	optimum
Good	30	good
Minimum	10	Survival
Poor	<10	Severe degradation

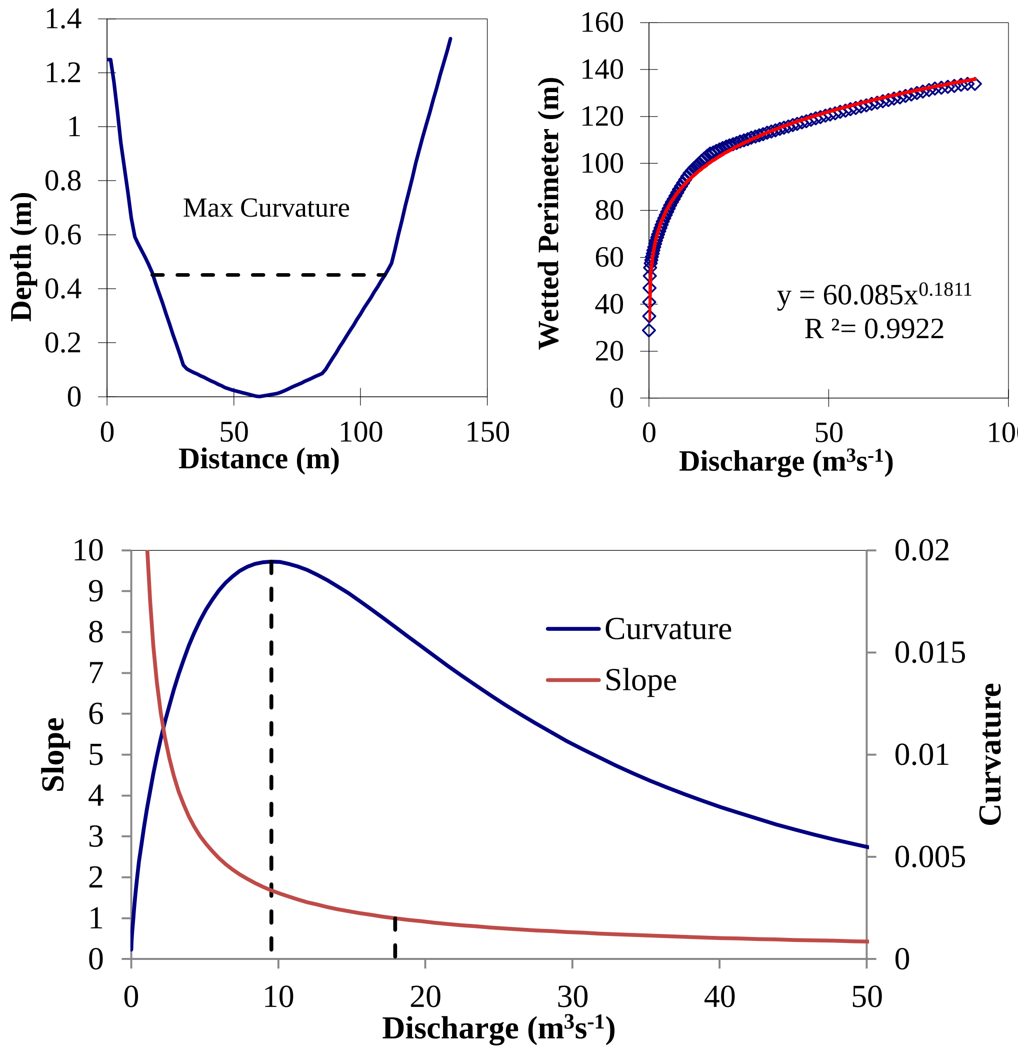
2.3.2.2 Hydraulic-based methods

Hydraulic-based methods are designed to assess the relationship between various conditions of physical habitat structure and discharge. Over the past few years the Wetted-Perimeter and R2Cross methods have represented two commonly applied hydraulic methods, especially in Australia (Anderson & Morison 1989; Davies & Humphries 1995). Hydraulic-based methods require site-specific physical and hydraulic data at a riffle section, such as channel geometry, average velocity, and mean depth over a range of discharges. An advantage of Wetted-Perimeter methods is that they are based upon field observations and do not require data from a stream gauging station, thus meaning that the EWRs obtained by these methods can be applied in hydrologically disturbed drainage basins and at gauged or ungauged sites.

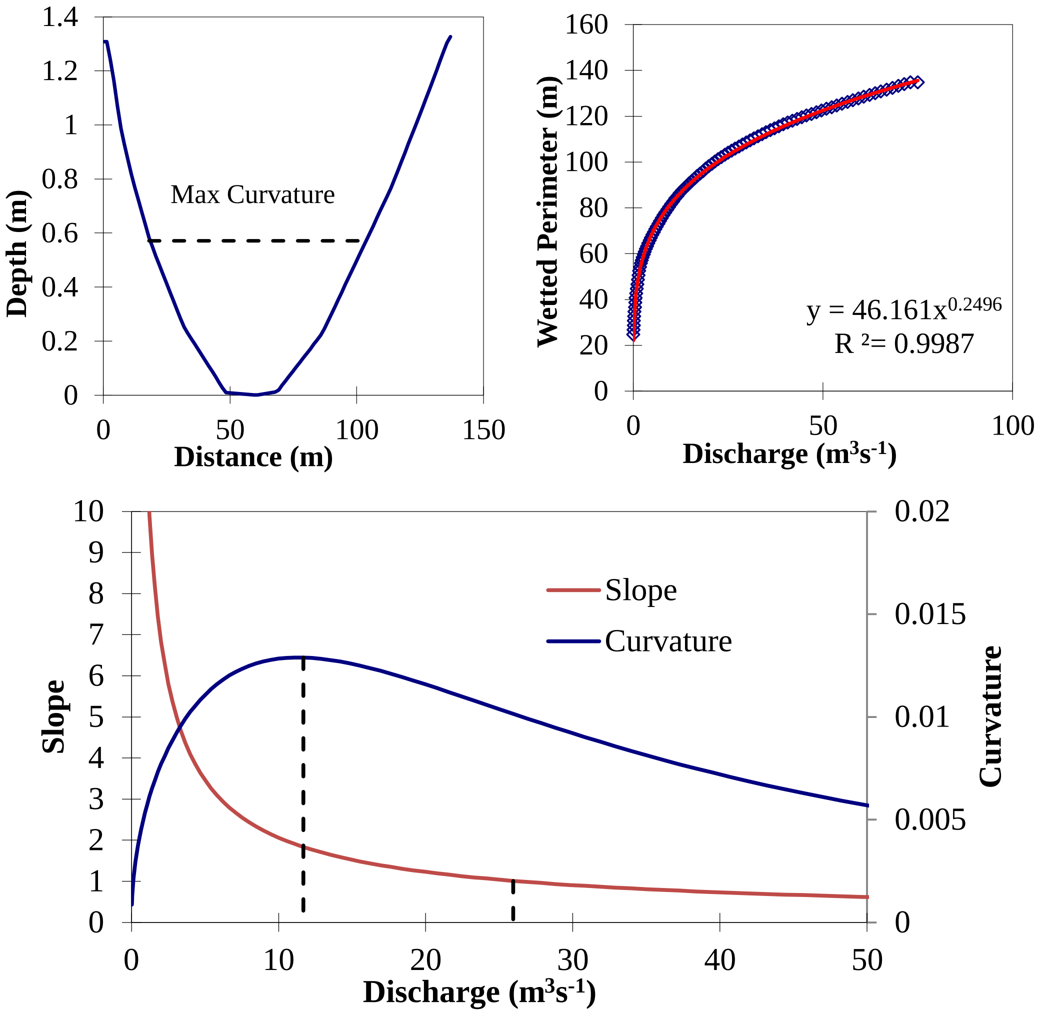
The wetted perimeter method has been used to define minimum flows for fish rearing in the US and Australia since the 1970s (Collings, 1974; Richardson, 1986). This method is based on a plot of the relationship between wetted perimeter and discharge. One procedure is to derive the relationship from channel cross-section surveys at several discharge levels. The cross-sections are often located only at riffle sites, or at sites where fish passage is likely to be limited. The wetted perimeter increases rapidly with increasing discharge, from a base level of zero flow before reaching an inflection point where there is a break in the shape of the curve (usually a logarithmic or power function). Following this, increases in wetted perimeter occur much more slowly until the bankfull stage is reached. This inflection point is taken to represent the minimum discharge. Alves (1994) defined the breakpoint as corresponding to a threshold discharge below which habitat quality becomes significantly degraded for a river in Portugal. The important break in the shape of the curve can be systematically defined by the point where the slope equals 1 or where the curvature is maximised (Gippel & Stewardson, 1998). Once data is collected in the field, according to hydraulic formulas (Table 2-3), the anticipated stream conditions at various flow rates, such as velocity, depth, and percent of wetted perimeter can be obtained. The wetted perimeter–discharge relationships and slope of wetted perimeter relationship will be determined. Generally speaking, there are three types of channels: “U” channel, triangular channel and trapezoidal channel. Figure 2-4 illustrates the inflection point determination for these three types of channels using hypothetical data.

Hydraulic data from past EWR studies were used to estimate the hydraulic parameters. A hydraulic sub-model was designed to produce a realistic representation of the hydraulic conditions using hydraulic parameters from readily available information for any part of South Africa (Desal, 2012). These estimated hydraulic parameters were used to develop hydraulic estimation relationships and these relationships were developed based on a combination of regression and rule-based procedures.

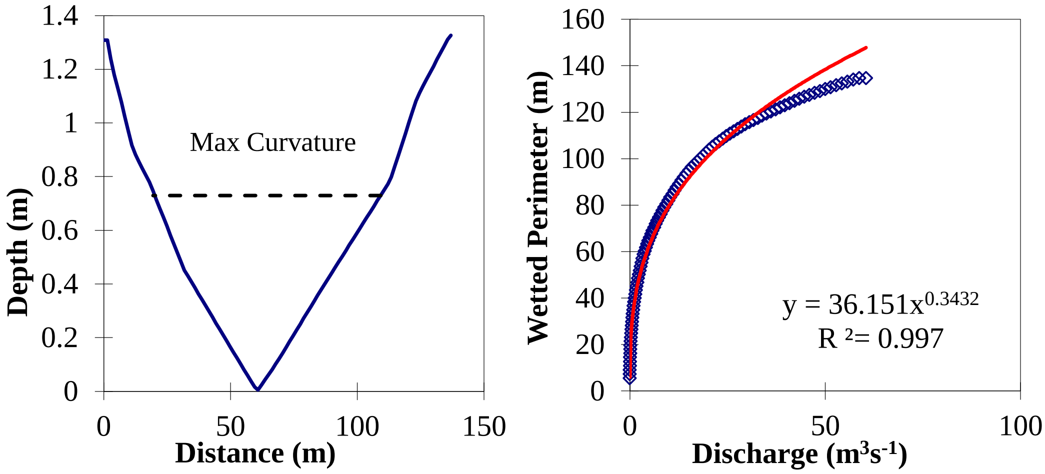
(a) “U” type cross section



(b) “trapezoid” type cross section



(c) “V” type cross section



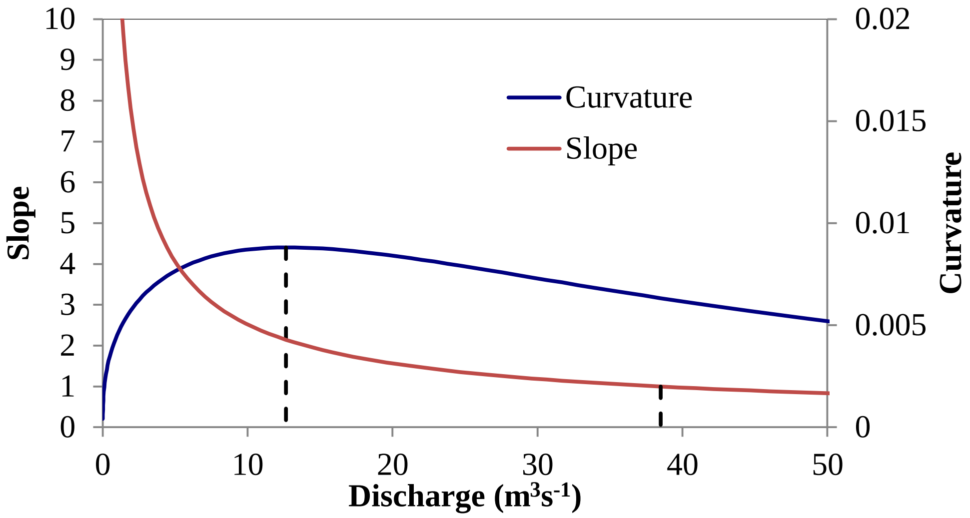


Figure 2-4 Inflection point determination process for three types of channels (a) "U" type, (b) "trapezoid" type, (c) "V" type using hypothetical data

Table 2-3 Hydraulic Formulas used in Wetted-Perimeter methods

Parameters	**Formulas**
Area	$\sum_{i=1}^{n} A_I$
Wetted Perimeter	$\sum_{i=1}^{n} WP_i$
Percent Wetted Perimeter	WP *100%/Bankfull WP
Percent Flow	Q*100%/Bankfull Q
Hydraulic Radius	A/WP
Velocity	$V = \frac{R^{2/3}}{n} \times J^{1/2}$
Discharge	$Q = A \times V$

2.3.2.3 Habitat simulation modelling methods

Rather than putting arbitrary limits on what is considered 'good' or 'optimum' maintenance of habitat based on proportions of the average discharge, habitat modelling methods seek to define the relationship between the discharge and the amount of habitat which is provided. Indeed, a habitat simulation model can be used to simulate a relationship between stream flow and physical habitat for an aquatic species (Marsili-Libelli, 2013). The habitat simulation model is generally categorised into two types: microhabitat model and mesohabitat model. The Physical Habitat Simulation System (PHABSIM) is a microhabitat model, which requires detailed hydraulic and morphological surveys, and knowledge of habitat preferences for the species of interest. Developed by the US Fish and Wildlife Service, it has been used throughout the USA since the 1970s (Milhous et al., 1984) and is currently in use worldwide. The hydraulic and habitat simulations of a stream reach using defined hydraulic parameters and habitat suitability criteria are the two basic components of PHABSIM. It requires the collection of field data on stream cross-sections and fish habitat features, hydraulic simulation to evaluate habitat variables at different flows, and species suitability criteria to calculate stream characteristics with available habitat at alternate flows. The output of the habitat modelling in PHABSIM is an index, known as Weighted Usable Area (WUA), with the dimension of an area (Waddle, 2001). It represents an area weighted for the aquatic species preference. The function WUA versus the discharge can be considered as the transfer function which transforms the hydrologic information into biological information (Figure 2-5). In actual fact, the main result of the application of habitat simulation is not the definition of a value for stream flow, but rather an estimation of the response of the aquatic ecosystem to different flows.

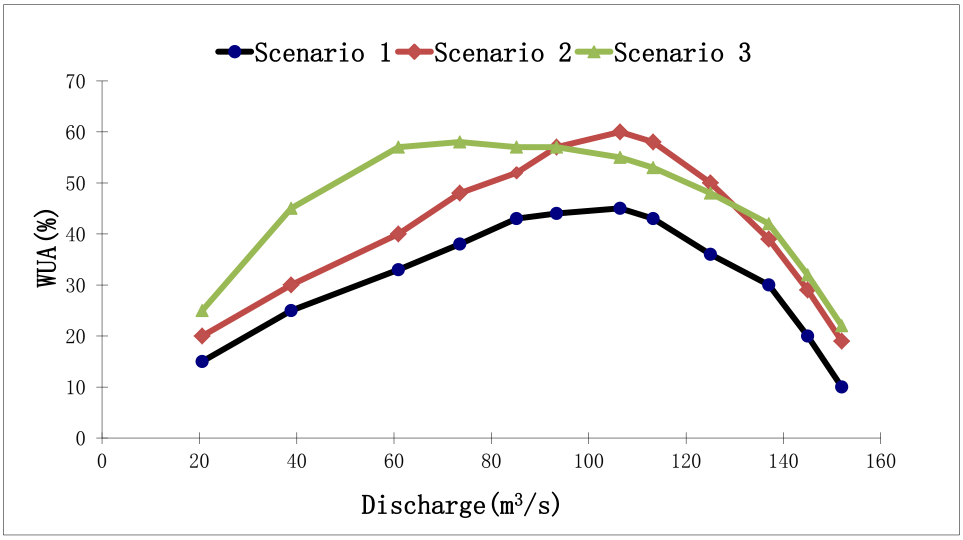


Figure 2-5 Variation of WUA with discharge for various scenarios

Based on PHABSIM, MesoHABSIM, which modifies the data acquisition technique and analytical approach of similar models by changing the scale of resolution from micro- to meso scales, was developed during a restoration study of the Quinebaug River (Parasiewicz, 2008). The greatest advantage of the MesoHABSIM method is its ability to quickly collect detailed information about physical conditions from long river sections. This allows a reduction in second stage error caused by extrapolation, when compared with hydraulic simulations using microhabitat approaches (Parasiewicz, 2007).

2.3.2.4 Holistic methods

Hughes & Louw (2010) pointed out that accurate hydrological and hydraulic data, together with a sound understanding of the ecosystem dynamics is referred to as the 'holistic' method. It should be emphasised that the holistic method is not a set of well-defined methods, but rather a framework capable of incorporating a range of methods. This method has provided an opportunity to combine data and knowledge from all the relevant disciplines so as to produce flow-related scenarios. In recent years, Downstream Response to Imposed Flow Transformation (DRIFT) has become the most promising holistic method for EWRs consideration (King et al., 2003). It is a structured process for combining data and knowledge from a variety of different disciplines to produce flow-related scenarios for the consideration of water managers.

As illustrated by Figure 2-6, the procedure consists of four modules: biophysical module, sociological module, scenario development and economic module. DRIFT has been used to assess the EWRs in a semi-arid region in South Africa where water-supply problems are pressing (King et al, 2003). The trade-offs are thoroughly considered by decision makers along with the macro-economic assessment. DRIFT provides a tool for holistic methods which links the natural and subsistence components of river ecosystems which are rarely considered in water-resource developments, to potential human and ecosystem costs.

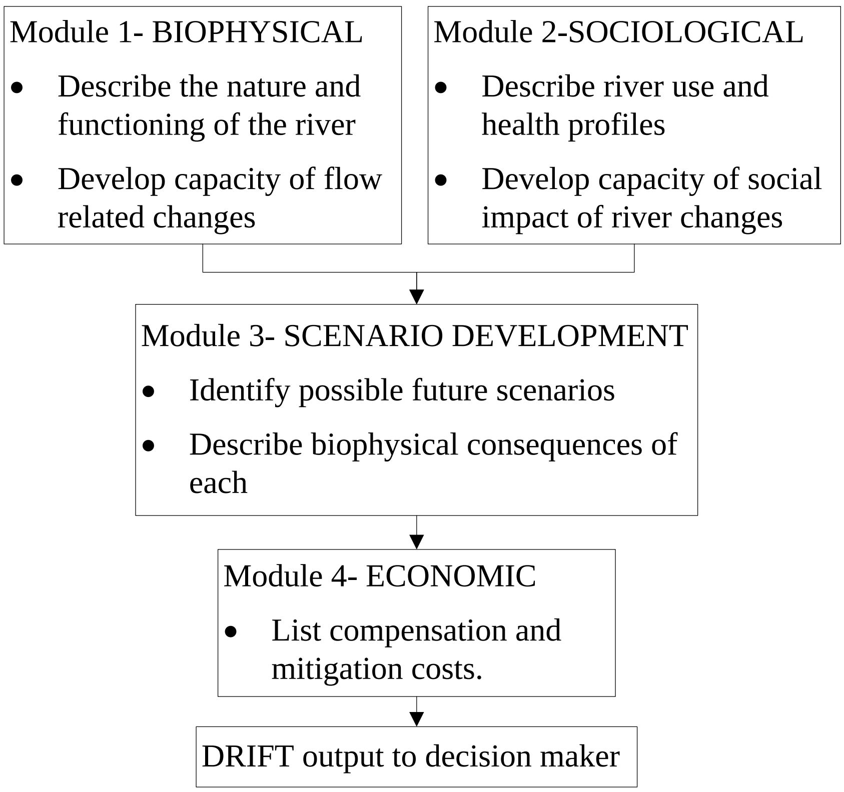


Figure 2-6 Schematic description of DRIFT procedure

2.3.2.5 Summary of the EWRs assessment methods

Each method has its advantages and disadvantages. Hydrologic-based methods are inexpensive and easy to implement. In contrast, hydraulics-based methods provide river-specific data (water depth, velocity and wetted perimeter) and are relatively simple to apply, although they fail to indicate the significance of changes in the physical conditions for the aquatic biota. Habitat simulation model methods and holistic methods involve high resolution characterisation of habitat availability for target organisms and are flexible when it comes to the assessment of different flow scenarios, although they are resource and time intensive (King et al., 2003). Overall, there is no straightforward way to select a method by which to determine EWRs and evaluate the validity of the results from a particular method. The choice of a method is mainly determined by the data available and the type of issues to be addressed. However, the holistic methods which incorporate a range of different methods mark an advance for EWRs consideration. The development of the EWRs will be towards a more holistic consideration of the interactions between landscape, hydrology, the riverine biota and even economics.

2.4 Summary of Literature Review

The review of past Eco-City work reveals that the Eco-City concept has been proliferating around the world, including USA, Europe, and Asia. To apply sustainability to Eco-Cities, theoretical research has been conducted in order to develop a better understanding of the city concept. It was also found that in terms of environmental sustainability, there has been a considerable amount of research conducted over the years related to the EWRs. Previous studies have exerted significant effort when it comes to the water quantity aspect of EWRs research. Although water quantity plays an important role in the diversity of freshwater species, the influence of water quality on habitat availability should also be considered in EWRs. Overall, both water quantity and quality have to be protected in order to promote sustainable water use and to contribute to the conservation and restoration of water-dependent ecosystems.

Water supply from non-traditional sources could be considered as an additional water resource to meet the ERWs in streams during low flow periods. Over the past decades non-traditional water has been increasingly used for landscape irrigation in urban areas. Ongoing research is being conducted to evaluate the expanded use of non-traditional water in many aspects. This can be from collecting and storing water run-off in cities and using it for secondary purposes to provide efficient flow augmentation of surrounding landscapes.

Although much work on the assessment of ERWs has already been conducted, this study has identified that the water quality aspects of ERWs still require further research while improvements on the expanded use of non-traditional water resources are still possible. There is continuing research in all areas relating to sustainable water management in Eco-Cities and understanding both past and present research will enable the thesis objectives to be achieved.

CHAPTER 3 NON-TRADITIONAL WATER SUPPLY AND EWRs IN TIANJIN ECO-CITY

As mentioned in Chapter 2, there are major opportunities to use non-traditional water in Eco-Cities. Exploring ways in which to reduce demand for surface water is essential to ensuring a sustainable future for the Eco-City. This Chapter takes Tianjin Sino-Singapore Eco-City, shortly to be Tianjin Eco-City, as an example to present the determination of non-traditional water supply and the evaluation of the EWRs in the Eco-City. Firstly, in order to estimate the potential non-traditional water supply in Tianjin Eco-City, 1) historical annual rainfall records from 1956 to 2000 of the Eco-City area were obtained from the Fangchaozha gauging station and were analysed to determine the potential rainwater harvesting in the Eco-City; 2) monthly industrial and domestic effluent data from Tianjin Eco-City Management Committee (TEMC, 2008) were used to estimate the reclaimed wastewater collection. Secondly, differing from regular volume of typical domestic and industrial water consumptions in the Eco-City, the minimum EWRs of the Eco-City river system should be estimated with a concrete and very clear answer regarding how to maintain basic ecological characteristics which are capable of providing environmental use and recreational services. In this study, an improved wetted perimeter method, multiple transect method, were used to estimate the minimum EWRs based on the available data for the 8 selected cross sections of the river system.

3.1 Introduction of Tianjin Eco-City

3.1.1 Location of Tianjin Eco-City

The Eco-City, which is located 40 km from Tianjin city centre and 150 km from Beijing city centre has a total land area of 34 km^2. Prior to the development of the Eco-City, the site comprised mainly saltpans, barren land and polluted water-bodies, including a 2.6 km^2 wastewater pond. The Singapore and Chinese government jointly started this Eco-City project in 2007. When fully completed in around 2020, it will have 350,000 residents. The start-up area is scheduled for completion by the end of 2013. Figure 3-1 shows the location and master planning diagrammatic sketch of Tianjin Eco-City. The main centre of the Eco-City is located along the lower reach of Jiyun River, Northern China. It has been planned for a variety of uses, including commercial, environmental and recreational. The study area in the Eco-City covers an old reach and a new reach of the Jiyun River as well as an artificial lake (latitude 39°06´N to 39°11´N, longitude 117°43´E to117°47´E , see Figure1). The old reach is a historical 1000-year-old river course with a total length of 10.7 km and an average water depth of 2.1 m. The artificial lake was once a wastewater pond with an average water depth of 2.0 m. The vision is to build "a thriving city which is socially harmonious, environmentally-friendly and resource-efficient - a model for sustainable development". Environmental sustainability will be the key feature of the Eco-City.

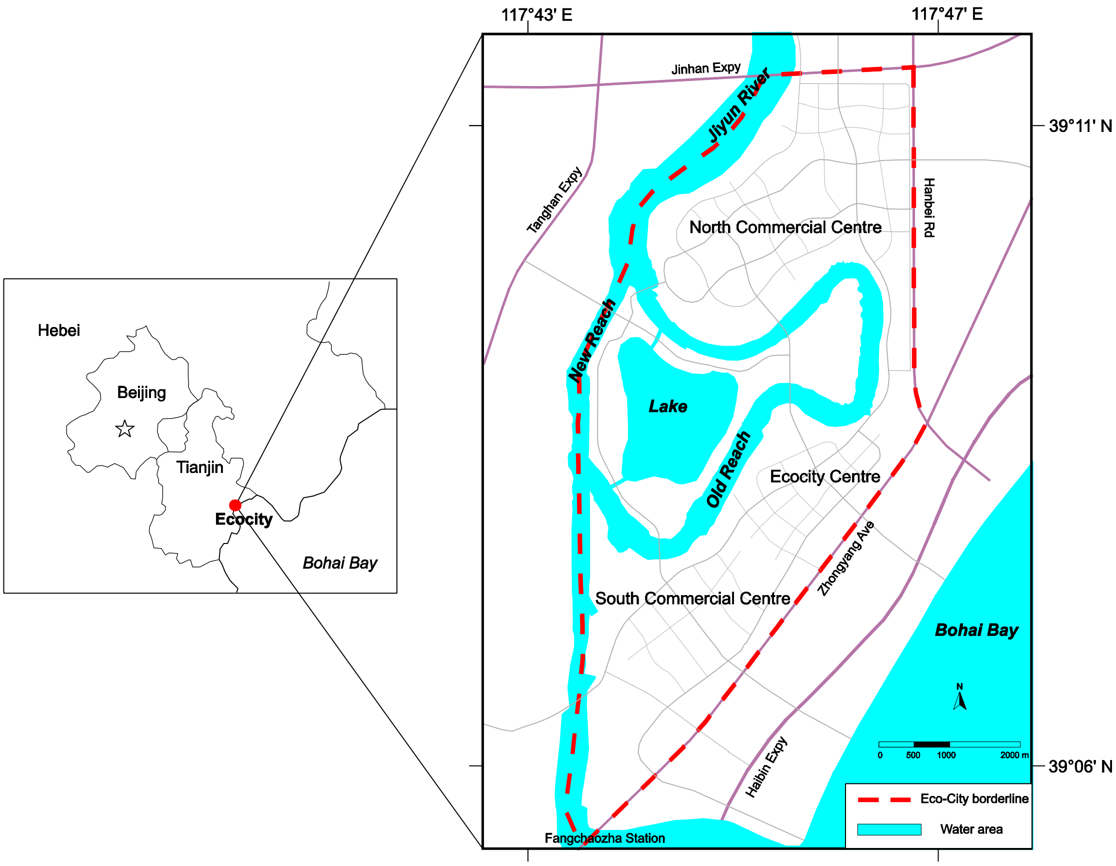


Figure 3-1 Location of the Eco-City and master planning diagrammatic sketch

3.1.2 State of Municipal Water Supply and Consumption in Tianjin

A survey on urban water supply and consumption is judged as a necessary starting point to achieving efficient water planning and management in the Eco-City. The municipal water supply and consumption status in Tianjin was achieved by the Tianjin Water Authority (2010). The state of municipal water supply and consumption in Tianjin is presented in Figure 3-2. The surface water, which provides more than 80% of water for urban use is the main water resource, while recycled water constitutes only 4% of total water supply. Industrial demand and public use consumes over half of total municipal water use. Domestic demand is approximately 18%, with the environmental use representing the remaining 6% of the total supply. However, as the Eco-City is located in a water-deprived area with salty land, scarce vegetation, unfavourable natural conditions and fragile ecology, surface water from rivers flowing through the region will not be able to meet the needs of the Eco-City. Indeed, careful future planning is needed so as to ensure reliable water supplies are available for everyone whilst protecting the natural environment in the Eco-City. To reduce its reliance on surface water sources, the Eco-City will draw a significant part of its water supply from other water resources. As shown in Figure 3-2, the recycled water supply is at an embryonic stage and therefore has a huge potential for water use in the Eco-City. The target is that at least 50% of water supply in the Eco-City will be from non-traditional water sources such as rainwater, recycled domestic and industrial wastewater, and desalinated water by 2020. Among all the water consumptions in the Eco-City, adequate water allocation to the fluvial and lacustrine ecosystem is particularly important with regard to maintaining ecological characteristics which are capable of providing environmental use and recreational services. The content of the following sections of Chapter 3 will cover non-traditional water supply and consumption estimation, with particular focus on the water requirements of the fluvial and lacustrine ecosystem in the Eco-City.

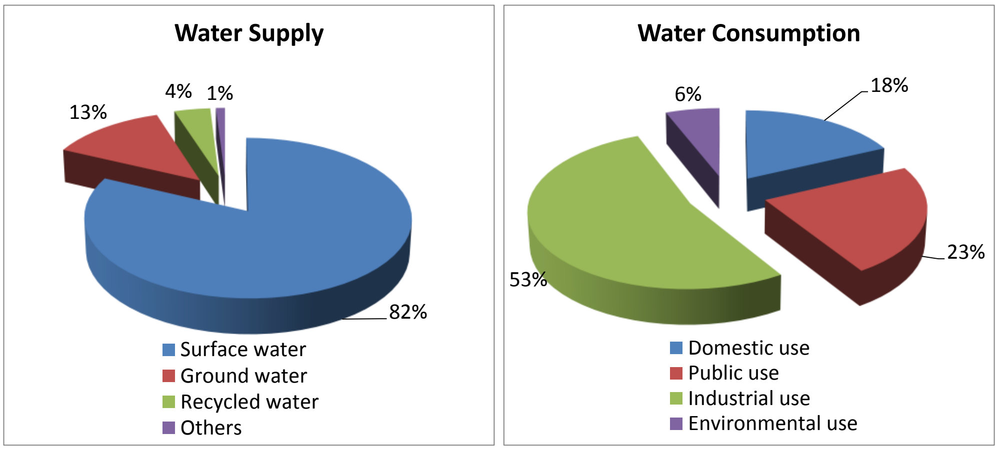


Figure 3-2 municipal water supply and consumption in Tianjin in 2010

3.2 Non-traditional water supply

3.2.1 Estimation of Rainwater Harvesting

The Eco-City is located in an area which has semi-humid continental monsoon climate, low rainfall in most parts of the area in winter and high rainfall and temperatures in summer. The long term series of historical annual rainfall records from 1956 to 2000 of Tianjin city and every district have been obtained from Fangchaozha Gauging Station. Missing annual data were estimated by considering data corresponding to neighbouring gauges. Annual rainfall of the Tianjin city and the Eco-City area were compiled and plotted in Figure 3-3a. It can be seen that the distribution of annual amounts of rainfall in the Eco-City area varies. The maximum rainfall was recorded in the year 1964 with values reaching 1170mm while the minimum rainfall, 318mm, was recorded in the year 1968. The average annual rainfall of 639 mm, was then obtained and presented in the figure with the dashed line. Besides the annual distribution of rainfall, the monthly amounts of rainfall within one year were analysed and are shown in Figure 3-3b. The most significant feature, as can be noted, is that the rainfall is unevenly distributed within one year, with 82.7% of rainfall amounts concentrated in the rainy season, which generally begins in June and lasts until September. The largest monthly rainfall amount of 224 mm was observed in July, followed by August and June with the rainfall measuring 170mm and 87mm respectively.

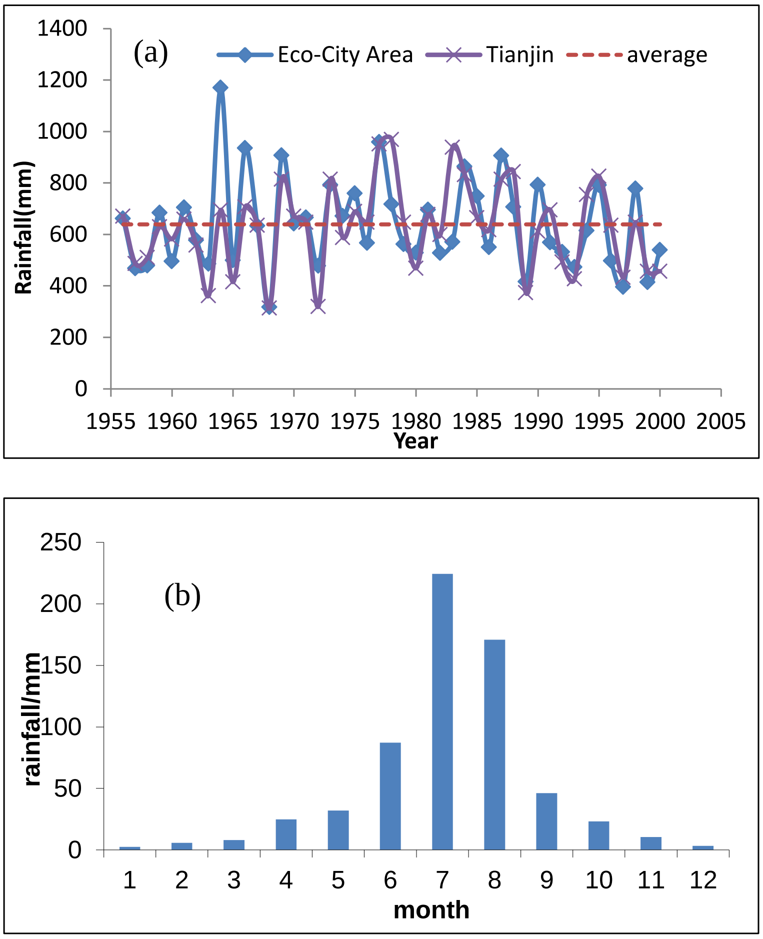


Figure 3-3 Annual (a) and average monthly (b) rainfall of the Eco-City area

The rainwater harvesting from a catchment surface is estimated using the following equation (Environment Agency, 2010):

$$Q = F \times \varphi \times q \qquad (3\text{-}1)$$

where Q - Harvested rainwater, L/s

F - Catchment area, hm^2

φ - Runoff coefficient

q - Rainfall intensity, L/(s hm^2)

The equation of rainfall intensity for Tianjin is given as (Tianjin Water Authority, 2010):

$$q = \frac{3833.34 \times (1 + \lg p)}{(t+17)^{0.85}} \tag{3-2}$$

where p - Return period, a

t – Duration, minute

Given the master plan of the Eco-City which measures 34.2km^2, the volume of harvested rainwater in an average year could be determined based on Equation 3-1 and Equation 3-2.

According to the master plan of the Eco-City, the catchment area for roofs, lawns, downtown areas, drives and walks, and waters is 10.96 km^2, 8.49 km^2, 5.39 km^2, 2.64 km^2, and 6.69 km^2 respectively. Figure 3-4 illustrates the percentage of each type of catchment area. It can be seen that the area of roofs accounts for 32% of the entire Eco-City area, followed by lawns at 25%. Impervious catchment surfaces such as roofs or non-porous pavement can lose 5% to 25% of the rain which falls on them due to evaporation and minor infiltration into the catchment surface itself (Tianjin Water Authority, 2010). The more porous or rough the catchment surface, the more likely it will be to retain or absorb rainwater. Herein, runoff coefficient, specifically the percentage of rainfall which appears as storm water runoff from a surface runoff, is used to reflect the ratio of rainfall to surface. Table 3-1 below shows some runoff coefficients φ from GB50015-2009, for Water and Drainage Design Standard for Building in China, although these are just rough estimates since runoff rates are also affected by rainfall intensity and duration. Generally speaking, larger areas with permeable soils, flat slopes and dense vegetation should have the lowest runoff coefficient values. Smaller areas with dense soils, moderate to steep slopes, and sparse vegetation should be assigned the highest runoff coefficient values.

Table 3-1 Values of Runoff Coefficient (φ) for Rational Formula (GB50015, 2009)

Land use	**Runoff Coefficient (φ)**
Downtown areas	0.70 - 0.95
Lawns with Sandy soil	0.10 - 0.15
Lawns with Heavy soil	0.18 - 0.22
Playgrounds	0.20 - 0.35
Drives and walks	0.75 - 0.85
Railroad yard areas	0.20 - 0.40
Roofs	0.75 - 0.95

Table 3-2 Average rainfall intensity in Tianjin Eco-City (mm)

Month	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
Rainfall	2.6	5.8	8.1	24.8	32	87.2	224.3	170.9	46.2	23.3	10.6	3.4

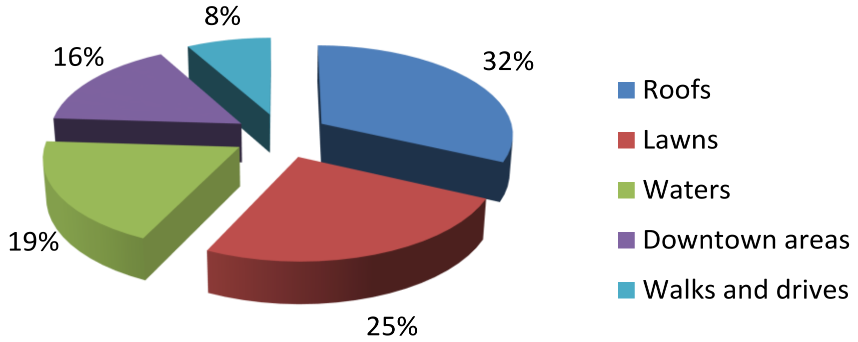


Figure 3-4 Master plan of the Eco-City

Based on the values of runoff coefficient for the rational formula shown in Table 3-1and rainfall intensity in Table 3-2, a realistic estimation of the volume of rainwater could be calculated from Tianjin Eco-City in an average year through the use of Equation 3-1. Assuming that the loss of water which occurs on the catchment surface is at the high end of the range, then a runoff coefficient of 0.75 for roofs, 0.10 for lawns, 0.70 for downtown areas and 0.75 for drives and walks, were used for the minimum rainwater harvesting, while the maximum rainwater harvesting was obtained by using the highest value of runoff coefficient in this study. The maximum and minimum monthly rainwater harvesting amounts are presented in Table 3-3, from which it can be seen that uneven temporal distribution of rainfall results in a significant difference in terms of monthly rainwater harvesting amounts. More than 80% of rainwater that can be harvested from June to September with the highest amounts reaching $4027\times10^3m^3$ in July.

Once collected and stored, harvested rainwater can be used for non-potable domestic purposes. With regards to rainwater, typical domestic use is taken into account in the Eco-City. This includes toilet flushing, garden watering and clothes washing using a washing machine. In order to evaluate the extent to which the harvested rainwater should be utilised for domestic purposes, the monthly water consumption in the Eco-City is also estimated using the following equation:

$$\mathrm{V} = \mathrm{P} \times \mathrm{C} \tag{3-3}$$

where V – total water consumption, m^3;

P -number of people in residence, 35×10^4

C -water consumption per capital per day, L/d

The estimated results are listed in Table 3-4 based on the statistical water consumption per capital per day by TEMC (2008). It can be seen from the table that the monthly water consumption in the Eco-City varies over a range of $1008\times10^3m^3$ to $1638\times10^3m^3$ which occurs in June. Generally speaking, the water consumption in summer is larger than that in winter. People use more water for cooling and recreational services due to the high temperature in summer. Figure 3-5 compares monthly harvested rainwater and domestic water consumption. It was found that the total harvested rainwater from June to September is $8.66\times10^6 m^3$ while the domestic water consumption is $4.66\times10^6 m^3$. The figure demonstrates that rainwater has a great potential for domestic use in the Eco-City from June to September which not only mitigates the impacts of flooding during the rainy season but also reduces the need for general surface water resources.

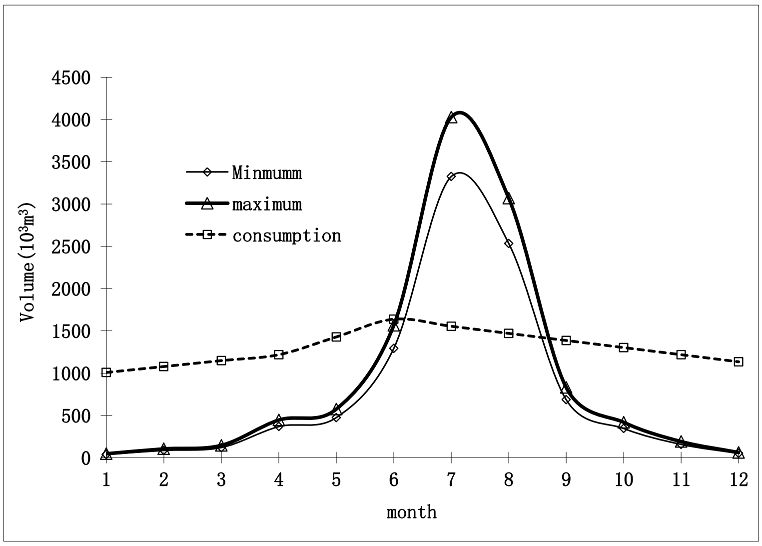


Figure 3-5 Comparison of harvested rainwater and water consumption in the Eco-City

Table 3-3 Monthly rainwater harvesting in the Eco-City ($10^3 m^3$)

Month	1	2	3	4	5	6	7	8	9	10	11	12
Minimum	38.5	85.9	120.1	367.6	474.3	1292	3325	2533	684.8	345.4	157.1	50.39
Maximum	46.7	104.1	145.4	445.3	574.5	1566	4027	3068	829.5	418.3	190.3	61.04

Table 3-4 Monthly domestic water consumption in the Eco-City ($10^3 m^3$)

Month	1	2	3	4	5	6	7	8	9	10	11	12
L.Cap.d	96	102.7	109.3	116	136	156	148	140	132	124	116	108
Total	1008	1078	1148	1218	1428	1638	1554	1470	1386	1302	1218	1134

3.2.2 Estimation of Reclaimed Domestic and Industrial Water

As mentioned in Chapter 2, wastewater reclamation and reuse is an essential component of water resource management plans in the Eco-City. The volume of the reclaimed wastewater can be estimated using the following equations (GB50015, 2009):

$$U_d = P \times W_d \times k_d \quad (3\text{-}4a)$$

$$U_i = W_i \times k_i \quad (3\text{-}4b)$$

where U_d - reclaimed domestic wastewater volume;

P - Population served;

W_d - Average water consumption, litres per person

k_d - Domestic wastewater return factor (≈0.8)

U_i - Reclaimed industrial wastewater volume

W_i - Industrial water consumption

k_i - Industrial wastewater return factor (≈0.9)

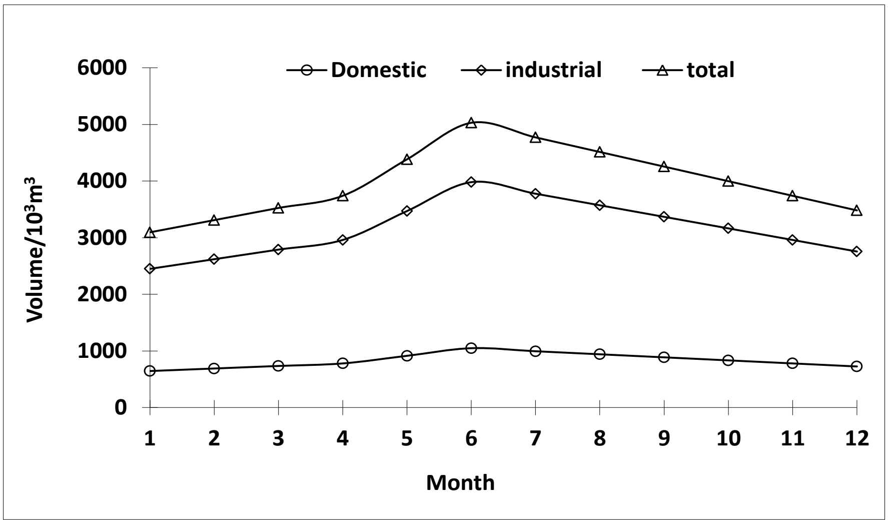


Figure 3-6 Monthly Reclaimed water volume

Based on the data from the TEMC (2008), the monthly domestic and industrial effluents amounts are listed in Table 3-5. Return factor is generally used to reflect the reclaimed ratio of wastewater to reclaimed water. The values for k_d =0.8 and k_i =0.9 have been used in Tianjin Eco-City (GB50015, 2009). Reclaimed wastewater volume are then determined from the domestic and industrial water consumption and presented in Table 3-5 and Figure 3-6. Reclaimed wastewater is commonly used to maintain constructed wetlands, enhance natural wetlands, and sustain or augment stream flows. In this study, reclaimed wastewater is considered to augment stream flows during the low flow periods in the Eco-City.

3.2.3 Estimation of Desalinated Seawater

In Tianjin, a combination desalination and coal-fired power plant is designed to alleviate Tianjin's critical water shortage. The facility has the capacity to produce 200,000 m^3 of potable water per day (Watts, 2011). Therefore, in the Eco-City, which is located in the coastal area, seawater desalination offers a promising option to enable the exploration of seawater with a view to producing high quantity and quality non-potable water as regular water supply.

Table 3-5 Monthly Reclaimed water volume ($10^3 m^3$)

Category		Month											
		1	2	3	4	5	6	7	8	9	10	11	12
Domestic	Domestic effluents	806.4	862.4	918.4	974.4	1142	1310	1243	1176	1109	1042	974.4	907.2
	Return factor	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8
	Reclaimed volume	645.1	689.9	734.7	779.5	913.9	1048	994.6	940.8	887	833.3	779.5	725.8
Industrial	industrial effluents	2722	2911	3100	3289	3856	4423	4196	3969	3742	3515	3289	3062
	Return factor	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
	Reclaimed volume	2449	2620	2790	2960	3470	3980	3776	3572	3368	3164	2960	2756
	Total	3095	3309	3524	3739	4384	5029	4771	4513	4255	3997	3739	3481

3.3 Determination of EWRs

3.3.1 Minimum EWRs Determination

The urban river-lake system is one of the most important components of the Tianjin Eco-City. Maintaining particular ecological characteristics which are able to provide environmental use and recreational services is of significant importance to citizens living in the Eco-City. As shown by Figure 3-1, the Eco-City is located along the Jiyun River in a coastal area, which is 5km from the Bohai bay. The study area includes an old reach and a new reach of the Jiyun River which is 10510m, and an artificial lake. This section focusses on the issue of minimum EWRs. Indeed, a certain level of water must be left in the Jiyun River and the lake to protect the basic ecological function of water resources. The assessment of the minimum EWRs of the river system in the Eco-City is a precondition for the deployment of non-traditional water for river rehabilitation. The adequacy of the calculated minimum flow was then compared with the historical record flow collected from the gauging station. The ecological function could then be retrieved via non-traditional water augmentation.

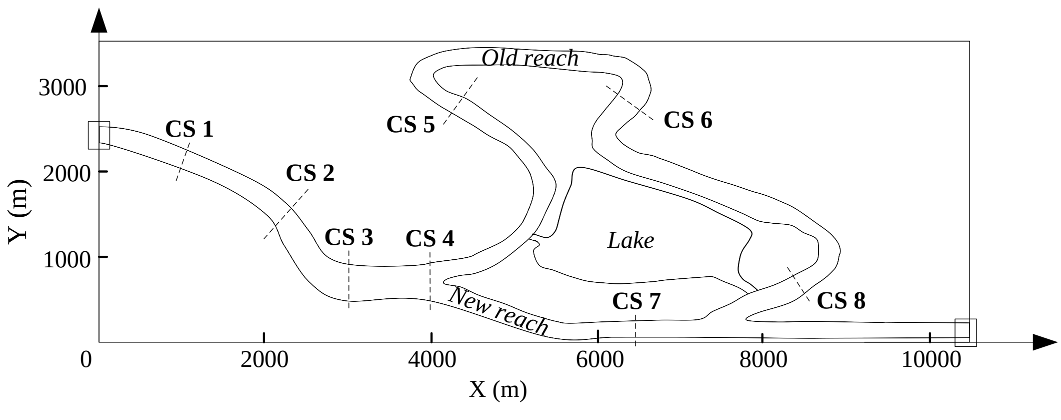


Figure 3-7 Location of the selected 8 cross sections in the Jiyun River

As pointed out in Chapter 2, hydrologic based methods, hydraulic based methods, habitat simulation modelling methods and the holistic methods are the main methodologies used to estimate the EWRs. However, the choice of a method is mainly determined by the data available and the type of issues to be addressed. In this study, an improved wetted perimeter method, multiple transect method, was used based on the available data for the 8 selected cross sections. The relationship between wetted perimeter and discharge is often used as an expedient technique for determining the minimum flow allowable for environmental purposes, as pointed out in Chapter 2. The locations of the cross sections are shown in Figure 3-7. The multiple transect method is an attempt to rectify the problems associated with a reliance on a single transect. A series of transects were implemented within the Jiyun River and variables such as velocity, depth, and wetted perimeter were obtained. The wetted perimeter–discharge relationships and the maximum curvature (K_{max}) were determined for each cross section (see Figure 3-8). A power function could be fitted to the wetted perimeter data. The correction coefficients R^2 were calculated and it was found all of the cross sections showed power relationships between discharge and water perimeter with the value of R^2 being above 0.99. The discharges at the breakpoints corresponded to the minimum flow levels required to maintain the ecological function for the water dependent ecosystems. Below this discharge, riffle areas became exposed and unproductive, stream bank cover for fish diminished, the water quality decreased and fish overcrowding was possible.

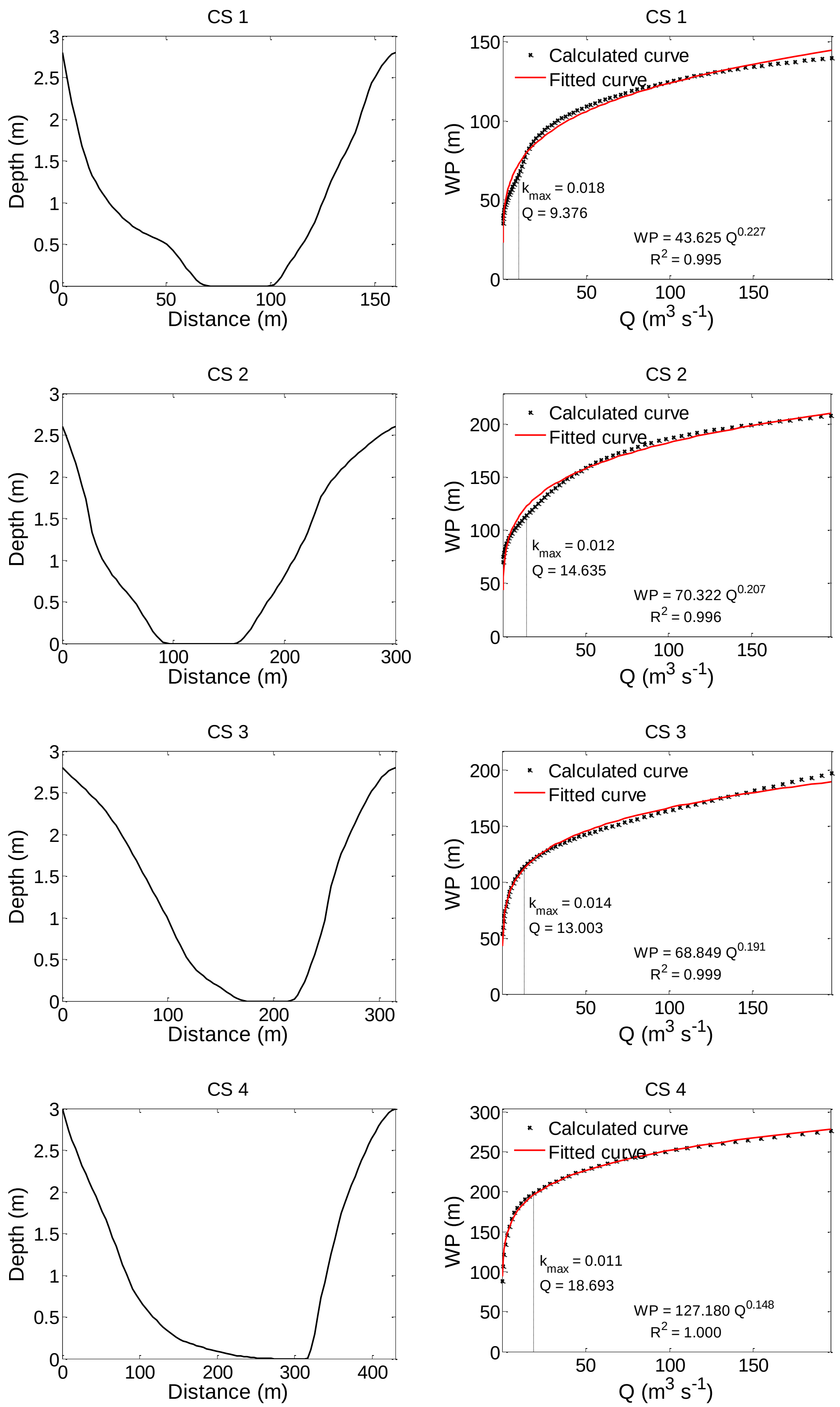


Figure 3-8 Wetted Perimeter–Discharge relationships and the maximum curvature for 8 cross sections of Jiyun River （continued）

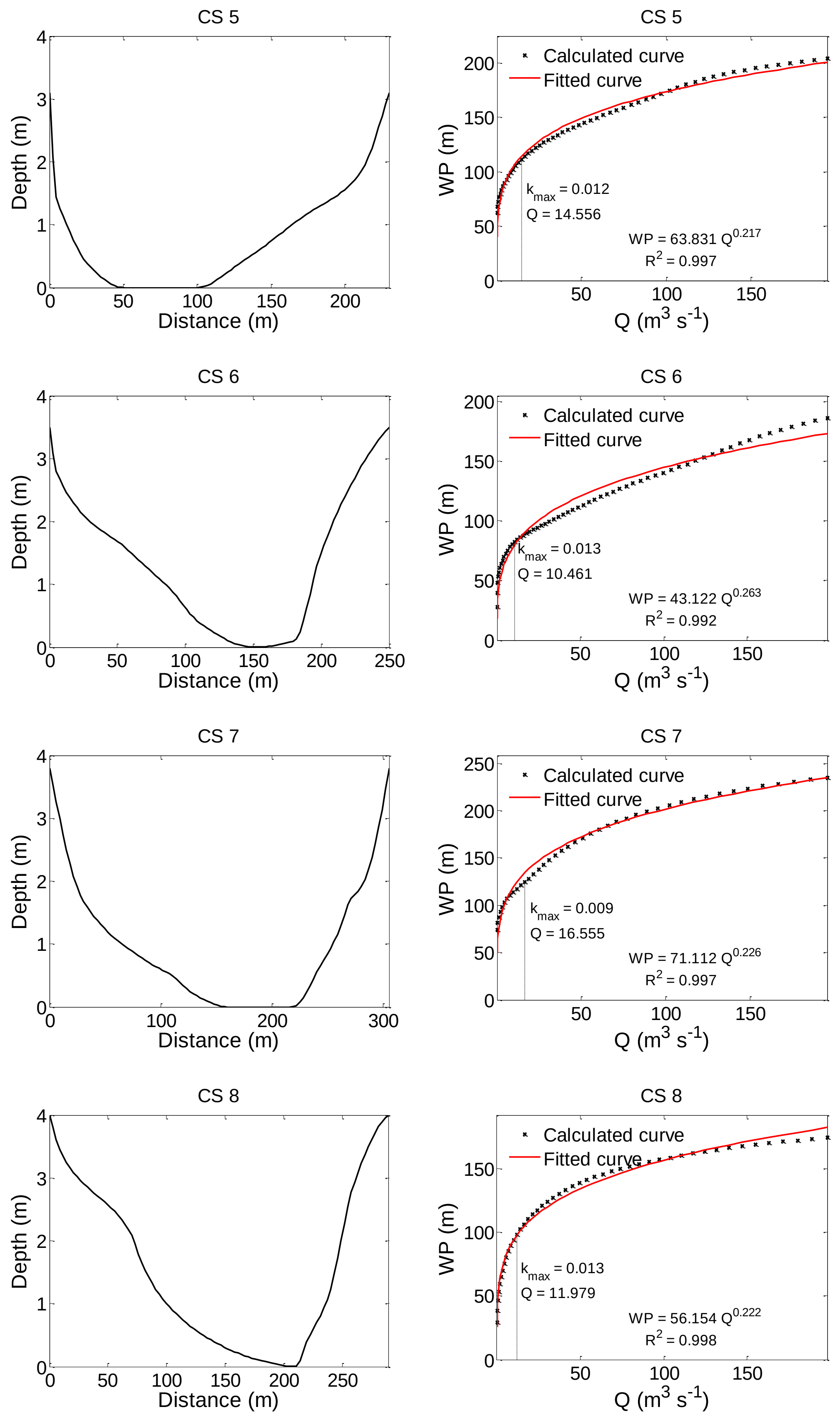


Figure 3-8 Wetted Perimeter–Discharge relationships and the maximum curvature for 8 cross sections of Jiyun River

Table 3-6 Minimum EWRs suggested by breakpoints of the wetted perimeter

Cross sections	Distance from upstream (m)	Max curvature	Minimum flow(m^3/s)
CS1	1000	0.018	9.37
CS2	2180	0.012	14.60
CS3	3000	0.014	13.00
CS4	4000	0.011	18.69
CS5	4500	0.012	14.56
CS6	6200	0.013	10.46
CS7	7400	0.009	16.56
CS8	8400	0.013	12.00

The minimum EWRs determined from the calculations are listed in Table 3-6. It was found that the wide range of cross-sectional shapes along the Jiyun River resulted in different breakpoints for individual transects and therefore occurred over a range of discharges. As shown in Figure 3-7, CS5, CS6 and CS8 are located in old reach while the left 5 transects are located in the new reach. The minimum flow was calculated for the old reach and new reach respectively. In order to meet the ecological functions for the whole river system, the minimum EWRs should be described by the cross sections of all the above EWRs:

$$MinEWRs = \max(MinEWRs(CS1), MinEWRs(CS2) \ldots MinEWRs(CSn))$$

where $MinEWRs$ is the minimum EWRs for the river system; $MinEWRs(CSn)$ is the minimum EWRs for cross section n. From Table 3-6, it can be concluded that the minimum EWRs for the old reach and the new reach are slightly different with a discharge of 14.56m^3/s for the old reach while 18.69 m^3/s for the new reach of Jiyun River.

3.3.2 Flow Augmentation to meet the Minimum EWRs by Reclaimed Water

Once the minimum EWRs were determined, the balance between the minimum EWRs and the natural flow could be evaluated, following which the amount of augmented flow required for the ecological rehabilitation by reclaimed water was estimated. The flow conditions for old reach and new reach are provided in Table 3-7. In this table, the measured monthly flow is acquired from Fangchaozha hydrologic gauging station. The monthly ET flow was calculated by dividing the monthly evapotranspiration (TEMC, 2008) by time (see Figure 3-9). The total annual water percolation volume for the Jiyun River was $43.4\times10^4 m^3$ (TEMC, 2008) and therefore the average monthly percolation flow was obtained and listed in Table 3-7. Herein, the final minimum EWRs are the calculated EWRs by multiple transects method added by the ET flow and percolation flow. Figure 3-10 compares the monthly measured flow and the total minimum EWRs for the old and new reach of Jiyun River. It can be seen from the figure that the current flow conditions were satisfactory, with the exception of the flow in January to March. The total Min EWRs in January, February and March were 33.2 m^3/s, 33.3m^3/s, and 33.6m^3/s respectively, while the measured flows from January to March ranged from 15.24 m^3/s to 30.29 m^3/s. The amount of reclaimed water needed, 82.9×10^6 m^3, was determined by multiplying the time by the balance between the measured flow and the total minimum EWRs. As shown by the reclaimed water calculation in Section 3.2.2, although the potential reclaimed water collected in January could not meet the EWRs, the ecological function could be retrieved by flow augmentation from the volume stored in November and December.

Table 3-7 Monthly flow conditions for the old and new reach of Jiyun River (Unit: m^3/s)

Month	ET flow	Percolation flow	Measured Flow	min EWRs-old reach	min EWRs-new reach
1	0.07	0.014	15.24	14.64	18.77
2	0.10	0.014	22.51	14.67	18.80
3	0.18	0.014	30.29	14.75	18.88
4	0.29	0.014	35.67	14.87	19.00
5	0.33	0.014	48.18	14.91	19.04
6	0.32	0.014	60.70	14.90	19.03
7	0.27	0.014	66.74	14.84	18.97
8	0.24	0.014	60.70	14.82	18.95
9	0.23	0.014	35.40	14.80	18.93
10	0.17	0.014	37.43	14.75	18.88
11	0.11	0.014	41.13	14.68	18.81
12	0.07	0.014	30.95	14.64	18.77

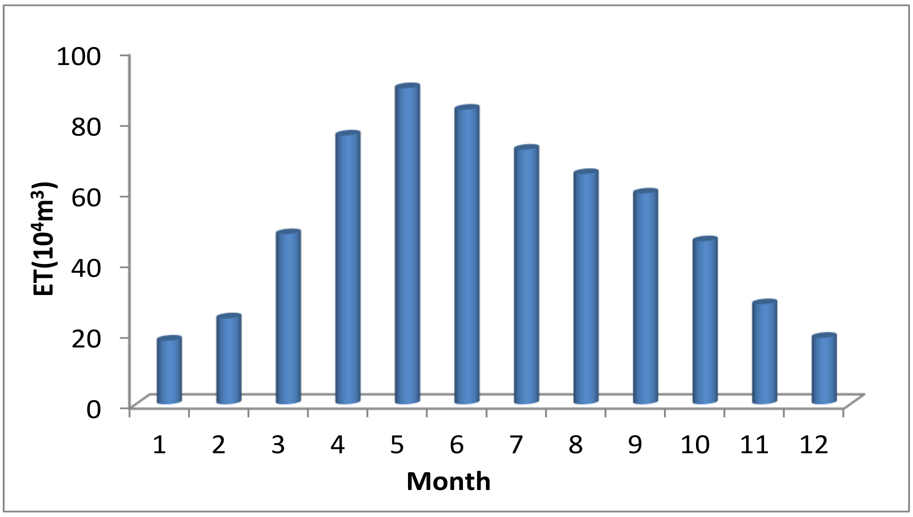


Figure 3-9 Monthly evapotranspiration of the Jiyun River (TEMC, 2008)

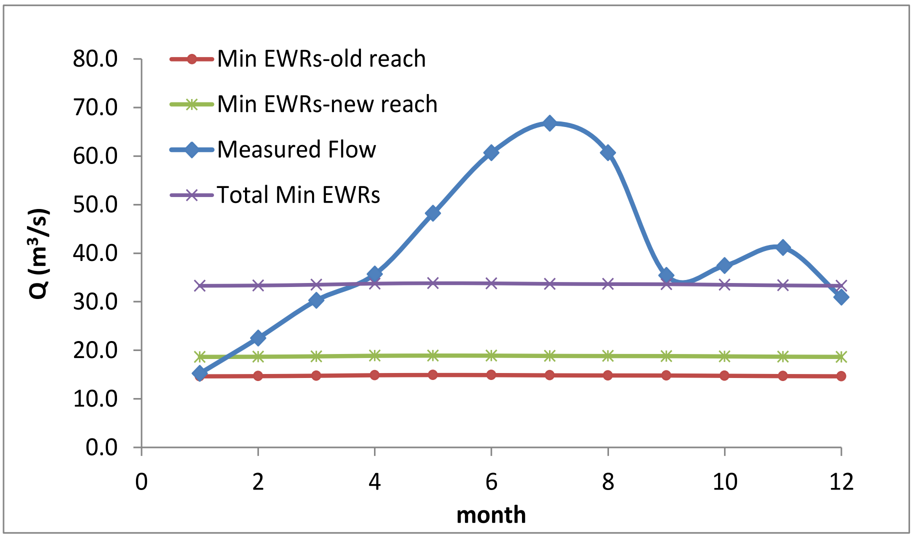


Figure 3-10 Comparisons of the monthly measured flow and the minimum EWRs

3.3.3 Water Quality Requirements

According to the China National Environmental Quality Standard for Surface Water, there are five classes of surface water quality standards (GB3838-2002). Grades III, IV, and V are suitable for ecological water use. The reclaimed water needed for the EWRs should meet these standards (see Table 3-8).

Table 3-8 The major water quality standards for ecological water use (mg/l)

Criteria	**Class of water**		
	Grade III	Grade IV	Grade V
DO≥	5	3	2
BOD_5≤	4	6	10
COD≤	20	30	40
COD_{Mn}≤	6	10	15
NH_3-N≤	1.0	1.5	2.0
TP≤	0.2	0.3	0.4
TN≤	1.0	1.5	2.0
TC(MPN/100ml) ≤	200	500	1000

3.4 Summary

In order to explore ways in which to reduce demand for surface water resources, using the Tianjin Eco-City as a case study, the potential of non-traditional water resources was studied in this chapter. An initial estimation was made regarding how much potential non-traditional water could be attained from the Eco-City.

It was found that rainwater has a great potential for domestic use in the Eco-City from June to September. Secondly, and differing from other water consumption, ecological demand of the river system in the Eco-City was emphasised and analysed by minimum EWRs determination. An improved wetted perimeter method was used in order to determine the minimum EWRs in the river system. The estimated results showed that current monthly flow conditions were satisfactory, with the exception of January to March. As a major non-traditional water resource, the reclaimed domestic and industrial water needed to fulfil the basic ecological function was determined. The reclaimed water allocated to the Jiyun River system should meet the key quality standards so as to maintain the river quality. In reference to this point the ecological water quality requirements will be discussed in Chapter 6.

CHAPTER 4 EXPERIMENTAL INVESTIGATION

4.1 Introduction

As illustrated by Figure 3-1 in Chapter 3, the artificial Lake, old reach and new reach of Jiyun River are the three water features which provide habitats to wildlife, allow for recreational activities for the residences, and add to the aesthetic value of the entire Eco-City. The quantity and quality of the water is of vital significance when it comes to maintaining these functions in the Eco-City. The minimum EWRs of the Jiyun River in the Eco-City were assessed in Chapter 2 and the non-traditional water resources, which have a great potential for environmental usage during low flow periods, were taken into account. However, with the overall goal of maintaining healthy ecosystems and fostering a more sustainable use of water resources, efforts should also be put into controlling and preventing water quality problems in the artificial lake which measures 1.47km^2 in the Eco-City. Long residence times and low mixing of water in lakes has been known to result in severe water quality problems, such as overgrowth of phytoplankton. There would be a danger of oxygen depletion when water visibility is limited to less than 12 inches, which is caused by overgrowth of phytoplankton (Allen et al, 2013). These dense blooms use large amounts of dissolved oxygen, which usually leads to population reductions of fish species and other animals.

In this chapter, an idealised river-lake system was assessed by hydraulics laboratory experimentation. The river-lake system of this study was inspired by the Jiyun River and the artificial lake at the site of the Tianjin Eco-City (www.tianjinecocity.gov.sg) (see Figure 3-7). A water diversion scheme was implemented to simulate lake flushing by river water in dry periods and under augmented flows. Connectivity with the Jiyun River through a controlled water diversion scheme can potentially enhance lake water quality, as an increased flushing rate will assist, for example, the washout of phytoplankton and nutrients. In this study, the hydraulic performance of the river- lake system could be analysed using hydraulic tracer studies. Fluorescent tracer experiments were conducted using rhodamine WT, a fluorescent dye which has a particularly low rate of decay, to assess the performance of different parts of the system before and after implementing the diversion scheme. Tracer experiments provide tracer passage curves from which characteristics of stream transient storage, and residence time distributions (RTDs), can be estimated. The use of tracers to infer hydrological processes has become increasingly common in catchment hydrological and hydraulic studies. A basic assumption of hydraulic tracer studies is that the chosen tracer is conservative and thus represents the water flow through a river, lake or other water body. For instance, Laenen & Bencala (2001) summarised a number of tracer experiments undertaken in streams to estimate key water quality parameters throughout the Willamette River basin, in the USA. Bernhard et al. (2010) took a similar approach to characterising solute transport of a small stream in Austria with transient storage, across a range of flow rates. The presence of preferential flow paths and dead zones can be identified through the introduction of a tracer at the inlet followed by spatial monitoring of tracer concentrations throughout the streams. The tracer is likely to be detected in preferential flow paths before it arrives in low-flow areas or dead zones within the streams. The tracer is also likely to remain in back-waters and low-flow zones for longer periods of time when compared to preferential flow channels. Spatial tracer monitoring can be conducted along longitudinal, lateral and vertical profiles to gain an insight into the distribution of flow velocities, preferential flow paths and mixing characteristics as water moves through the system (Grismer et al., 2001).

This experimental study was aimed at characterising an idealised river-lake system, and more specifically its ability to divert water. The study also aimed to assess the impacts of water diversion and flow augmentation on the mixing and flushing processes in the lake. Detailed descriptions of the experiment procedure and the general tracer data observed will be provided in this chapter. The chapter will conclude with a discussion of the experimental data processing and analysis.

4.2 Experiment Setup

4.2.1 Physical Model Design

For the physical model, Froude Number similarity must be maintained. This is accomplished by letting $Fr_m = Fr_p$ when scaling the flow dynamics. Because the properties of most turbulent flows do not change significantly with changes in Reynolds number Re, the similarity constraint on Re is often relaxed in hydraulic model studies of flows with free surfaces.

The specific scales for the physical model are as follows.

$$\frac{Fr_m}{Fr_p} = 1 \tag{4-1}$$

where Fr_m, Fr_p are the Froude Numbers of the physical model and the prototype respectively.

The schematised prototype and physical model are shown in Figure 4-1.

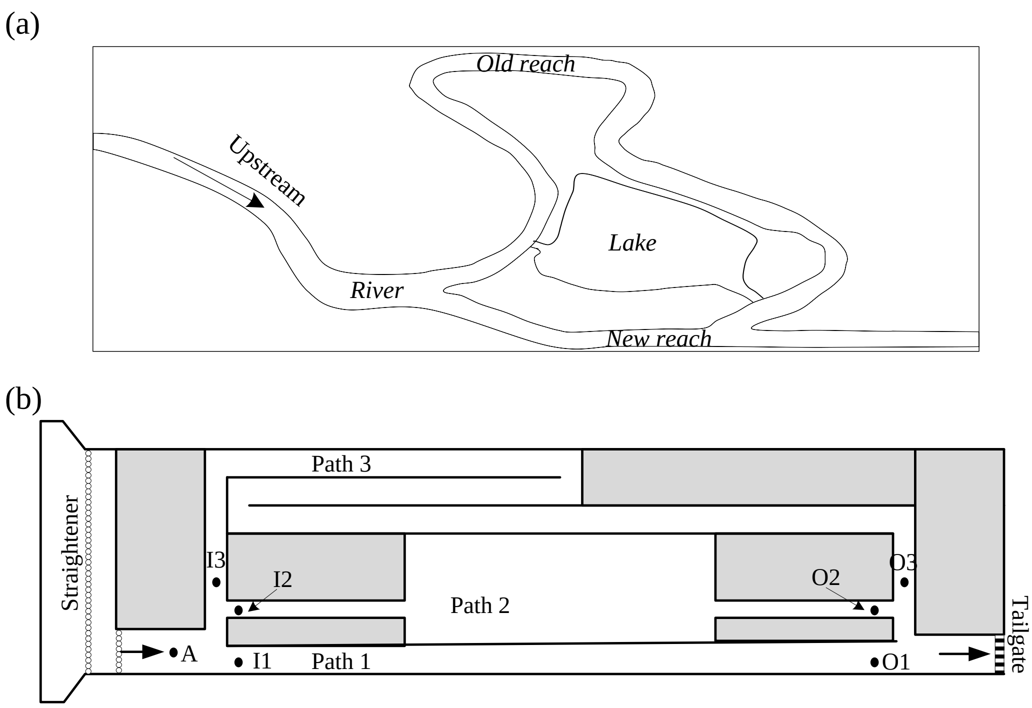


Figure 4-1 Schematic illustration of a) real and b) idealised version of the river-lake system

A physical model was then designed and constructed in a 10 m long, 1.2 m wide flume of the Hyder Hydraulics Laboratory at Cardiff School of Engineering, UK (see Figure 4-2). Flow straighteners were placed at the inlet section to mitigate the occurrence of swirls and inflow asymmetry. An adjustable-height rectangular sharp-crested weir was used at the flume outlet section to control the water depth in the model. A closed hydraulic circuit was used, which included a reservoir, centrifugal pumps, return pipes and flow metres. The model was built out of 12 mm thick PVC sheets. As shown in Figure 4-1b, Paths 1 and 3 corresponded to the two river reaches, while Path 2 was the simulated lake. The lengths and widths of the corresponding channels were determined based on the available flume space and by preliminary numerical model testing. The lengths of Paths 1 and 3 are 776cm and 1626cm respectively. The length of the lake is 404cm and the width is 64.8cm, which is ten times that of the lake inlet. This also involved different water depths and flow rates, targeted at meeting the design conditions mentioned above.

The serpentine configuration of Path 3 was generally associated with the meandering “old reach” of the Jiyun River, while Path 1 was relatively straight, as the corresponding “new reach” of the river. Figure 4-3 shows the top view of the physical model in the flume. The bed slope was set to 1/2500 and the bed surface is smooth glass. The feasibility and applicability of the distorted model has been discussed in the following section of this chapter.

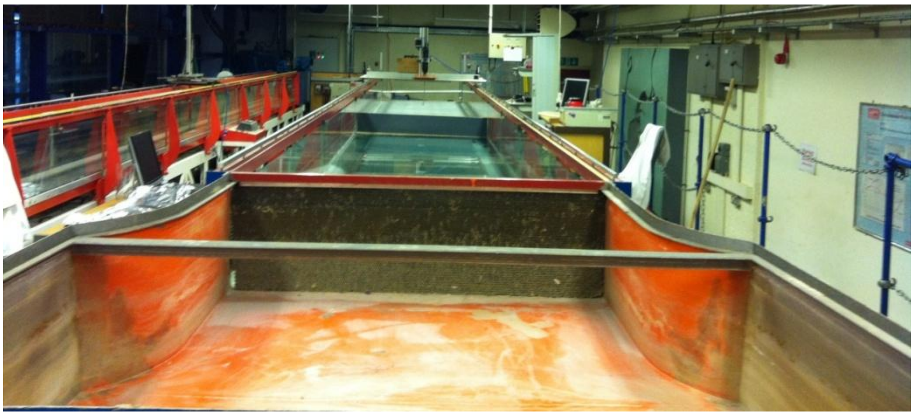


Figure 4-2 Flume used in the experiment

Figure 4-3 Top view of the physical model built in the flume

4.2.2 Data Measurement Methods

In this study, the designed experiments included the hydraulic parameters measurements and the tracer concentration measurements. The data collection equipment used in this study will be discussed in the following section.

4.2.2.1 Flow Metre

The total inflow rate to the river-lake system model was obtained from the Ultrasonic flow metre (Figure 4-4) which measures flow based on the difference in time it takes for an ultrasound wave to travel upstream versus downstream. The water in the flume is supplied by the motor-driven pump and its output frequency is determined by the SIMENS Micromaster 430. The outflow from the flume flows to the ground tank and then back to the head of the flume again.

Figure 4-4 Ultrasonic flow metre

4.2.2.2 Acoustic Doppler Velocimetre

The velocity measurement was carried out by the Acoustic Doppler Velocimetre (ADV) (Figure 4-6a) which is designed to record instantaneous velocity components at a single-point with a relatively high frequency. The advantage of using this device relies on its ability to adequately measure the velocity components (x, y, and z) of flowing water. The ADV was positioned on a movable truss in the direction of the X-axis arrow. The vertical movement was enabled by a sliding scale, thus making it possible for the probe to be placed at any depth with in the flow. As shown in Figure 4-5, the arm with the marking defines the x-direction. The Z-direction is towards the electronics of the vectrino. In XYZ coordinates, a positive velocity in the X-direction goes in the direction of the X-axis arrow. Measurements are performed by measuring the velocity of particles in a remote sampling volume based upon the Doppler. The manufacturer's specifications state that the measuring accuracy of the ADV is ±1% of the average velocity. Velocity range ±0.3m/s is set to cover the range of the velocities anticipated during the data collection.

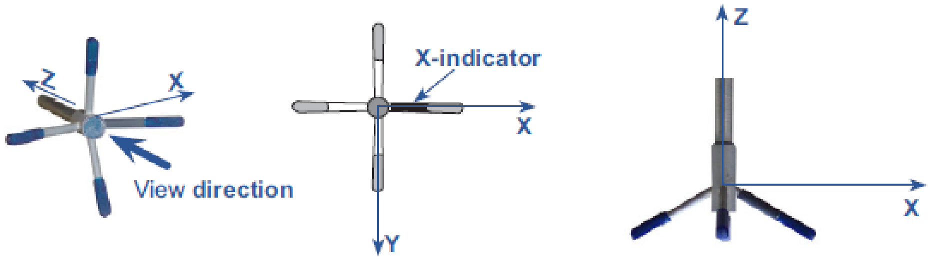
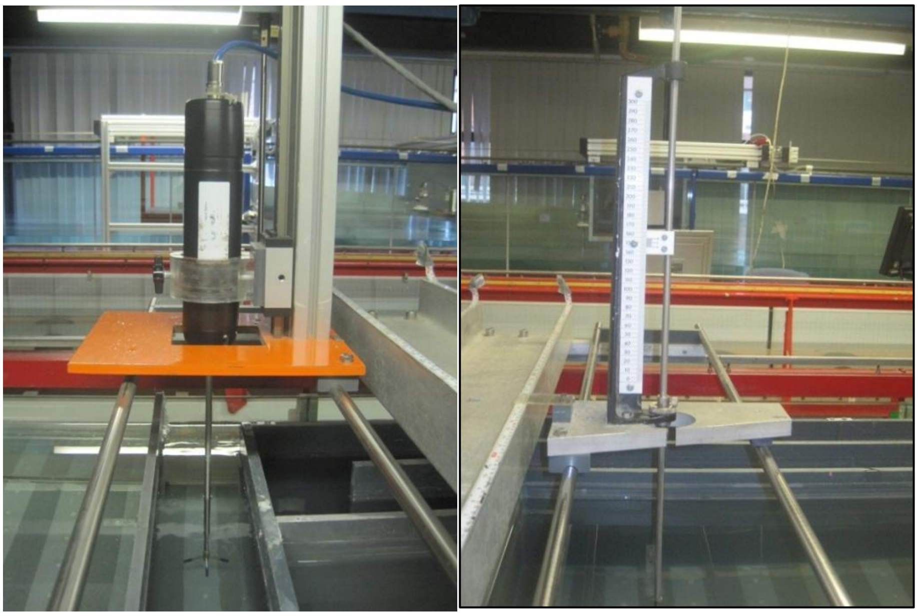


Figure 4-5 XYZ coordinates of the probe

Correlation was monitored during data collection and gave a numeric value to the quality of the velocity data. Poor correlation can be due to many reasons, such as, the velocity range being too high or the probe not being submerged. The Correlation and Signal-to-Noise ratio (SNR) was used to measure the strength of the echo from the acoustic pulse. The velocity will show significant short time variability when the echo is not sufficiently powerful to allow proper calculation of the frequency shift. Ideally, the correlation should be above 70% and a level of 15% SNR must be maintained to give satisfactory velocity values.

4.2.2.3 Point Gauge

The measurement of steady state water depth was needed during this experiment. This was done using a point gauge with an accuracy of 0.1mm. A small point was manually adjusted to slide up and down to touch the water surface, and a reading was taken of the vertical movement using a vernier which eliminates observation errors (Figure 4-6b).



(a) (b)

Figure 4-6 Velocity and Water Depth measurement apparatus (a) ADV (b) Point Gauge

4.2.2.4 Fluorometre

The tracer concentration was measured by using the CYCLOPS-7 Submersible Fluorometre (Figure 4-7). From September of 2011 to April of 2012, a series of tracer experiments were conducted on the river-lake network model described above. Since the water in the flume was recirculated, the background concentrations at the inlet of the channel were measured during each experiment in order to take its concentration away from the measured value for data analysis.

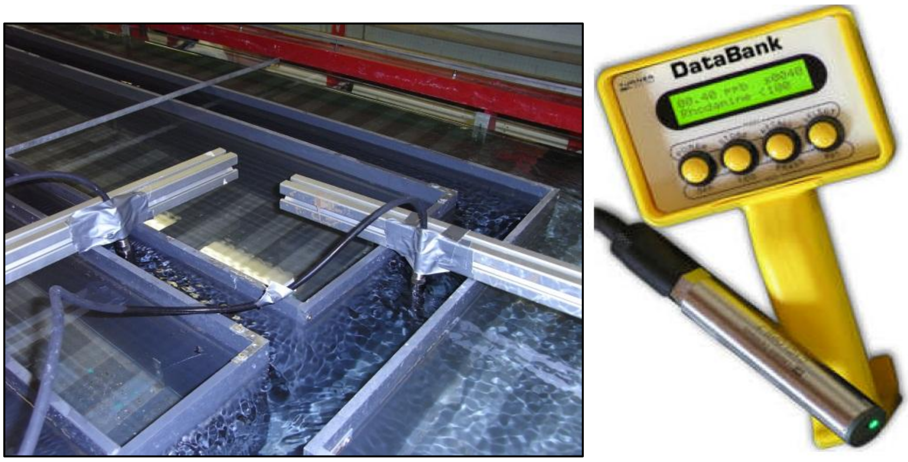


Figure 4-7 CYCLOPS-7 Submersible Fluorometre

Sensors connected to the DataBank require calibration and auto setup to function properly. Instrument reading for dyes is proportional to standard solution concentration (linear) from the lowest detectable level up to a certain concentration. Above this concentration, a multipoint calibration curve may be used to obtain concentration. Following this, at a certain concentration, the curve flattens out and eventually became nonlinear. Three fluorometres were used in the experiments, and the calibration results are shown in Figure 4-8. It can be easily seen from the graph that Rhodamine WT is linear to around 100ppb, thus meanings that the maximum measured concentration during the experiment should be around 100ppb.

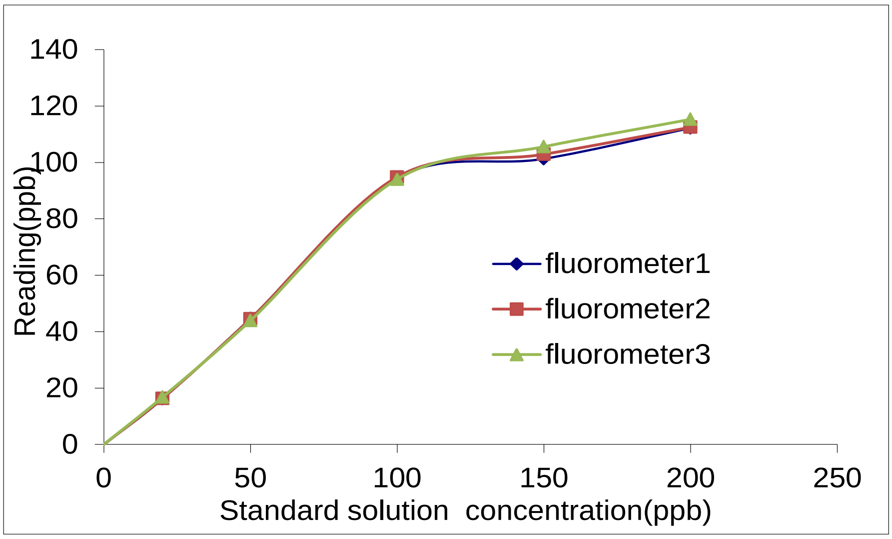


Figure 4-8 Fluorometers calibration

4.3 Tests

Figure 4-1b illustrates the schematised view of the flume with designed channels. In order to obtain the uniform flow, straighteners were adopted at the inlet to maintain uniform flows entering the flume. The water flows through the straightener into the flume and an adjustable weir was set up at the end of the flume to control the water level in the flume. Point A is located at the centre of the upstream main channel, measuring 70cm distance from the beginning of the network. I1, I2, I3 represent the inlet of branch channel with 1, 2 and 3 denoted as path1, 2 and 3, with the discharge Q1, Q2, and Q3 respectively. The water depth could be controlled by the tailgate in the end of the model. In the river-lake network model, the Froude and Reynolds numbers have to satisfy certain conditions. In order to keep the flow turbulent, the Reynolds number should be larger than 2000, and to avoid influences of surface disturbances on the flow, the Froude number should be much smaller than 1. In this experiment, Flow in the upstream main channel was maintained subcritical, with the minimum Re number totalling 15400 and the Fr number 0.06.

A series of tests with varying experimental parameters were conducted in this study. Tracer experiments were conducted using rhodamine WT, which involved 4 injection points and 12 monitoring locations, comprising 6 distinct tests. The tracer solution was prepared in a volumetric flask by adding 4g of 20% rhodamine WT solution to 2 litres of water before mixing thoroughly so that the tracer was completely dissolved. The water flows (depth and velocity) were measured directly after the tracer solution injection. Before the tracer injection procedure, the model would be run for half an hour so that the water was thoroughly mixed and a homogeneous distribution of the tracer solution was ensured.

6 tests were planned and performed in the experiments. The following part presents the details of the general flow characteristics and tracer passage curves observed from the 6 tests.

4.3.1 Test 1

In Test 1, the discharge Q=5.50l/s was held constant and the tailgate was set up to keep the water depth at 25cm. This constant flow rate and depth produced a constant Froude number and average velocity. A single injection took place at position A. The hydraulic and injection conditions are listed in Table 4-1.

Table 4-1 The hydraulic and injection conditions for Test 1

Test	Discharge(l/s)	Injection point	Injection mass	Observation point
1	5.50	A	8mg	O1, O2, O3

As illustrated by Table 4-1, in Test 1, the tracer was injected at point A and monitored at O1, O2, and O3 simultaneously. Instantaneous injections, 20ml at a concentration of 400ppm were conducted during a steady flow regime via a syringe. The injection locations were 2 mm below the water surface. Tracer solutions were injected within 2s, which was much less than the travel time to the fluorometre, and as such the injections were assumed to be instantaneous. The injection velocity was considered to match to the ambient velocity. The recording fluorometres were positioned at O1, O2, and O3 separately following which the concentrations were measured at time intervals of 1s. Losses of mass of rhodamine during tracer experiments might occur due to photo-degradation and sorption. However, because of the short duration of the experiments, such effects would be very small and ignored in the present study. The model was run until the value on the databank was stable, following which the data could then be exported to the computer. The tracer concentration raw data were compiled and plotted in Figure 4-9. Three tracer passage curves corresponding to O1, O2, and O3 were derived from the raw data. Tracer concentration at O1 reached the peak value at 71s, followed by 256s at O2 and 469s at O3.

The flow velocity was measured directly after the tracer solution injection. In this test, velocity at position A was measured first. The velocity was measured at 13 points along the vertical profile with the near bed points more closely spaced. The lowest point was located at 2 mm above the channel bed; the highest point, 70 mm below the water surface. The measured vertical distribution of water velocity at position A is shown in Figure 4-10. Generally speaking, velocity in the water column varies logarithmically with depth, with the mean velocity (depth-averaged) usually at 0.4 times the depth above the bed, maximum velocities at or just below the water surface, and near-bed velocities close to zero (Jowett, 2003). The measured velocity at 10cm for point A was 0.095 m/s whilst the calculated mean velocity was 0.091 m/s ($5.50\times1000/(0.25\times0.24)$) and the accuracy index was 4.4%. This meant that 10cm above the bed depth velocity can be represented as the depth averaged velocity in this study. Thus the velocity measurement in the following experiments generally measures the velocity at 10cm above the bed. Velocity measurements were recorded at 50 Hz for a sampling period of 120 s for each point. It was found that velocity became almost constant for sampling periods greater than 30s, and as such, a sampling time of 120s was considered to be representative for an appropriate determination of mean velocity. The time series of velocity at data points O1, O2, and O3 were measured and analysed to produce a normalised average velocity (Figure 4-11).

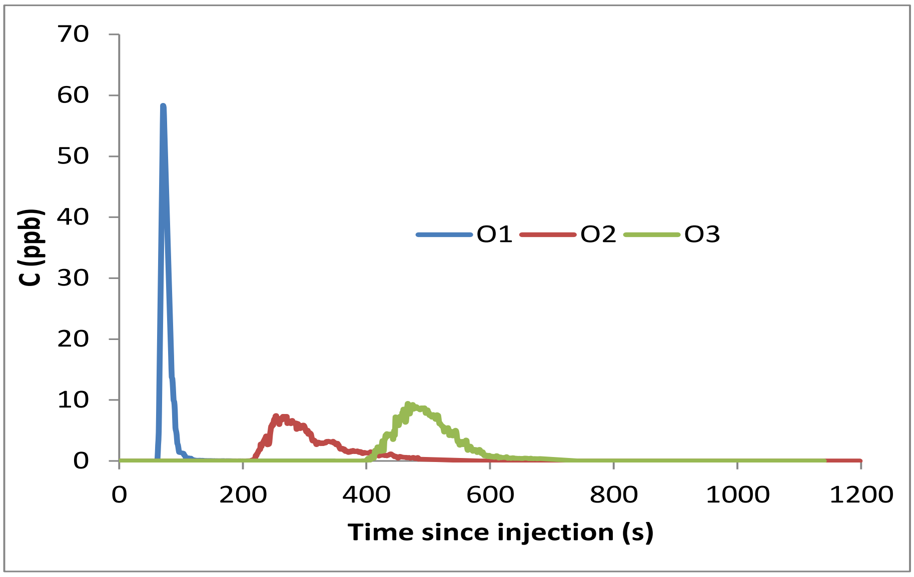


Figure 4-9 Time series of tracer concentration of Test 1

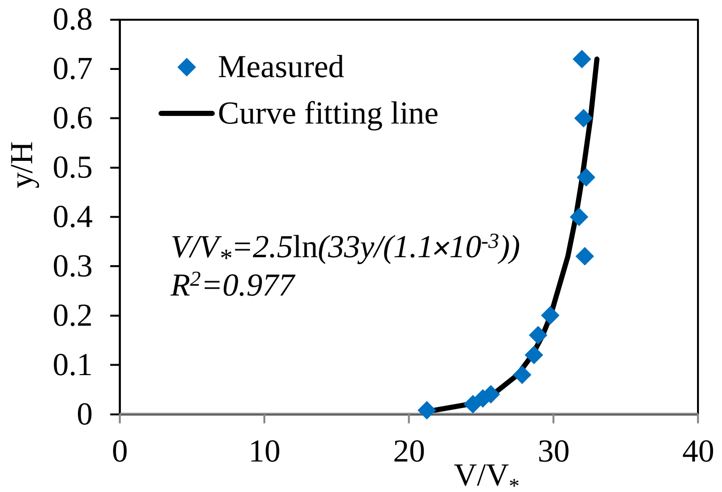


Figure 4-10 Velocity profile over a discharge of 5.50l/s

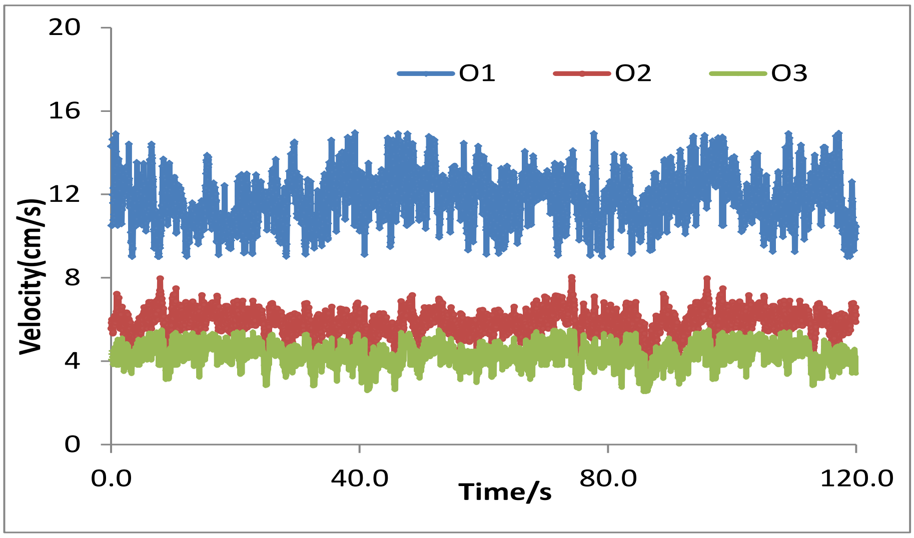


Figure 4-11 Time series of velocity at data point O1, O2, and O3

Table 4-2 The flow distribution results for Test 1

Channel	Channel width(cm)	Discharge (l/s)	Mean velocity(cm/s)	Bed slope	Froude Number
Path 1	12	3.57	11.2	0.0004	0.16
Path 2	65	0.94	5.80	0.0004	0.11
Path 3	12	0.99	4.30	0.0004	0.06

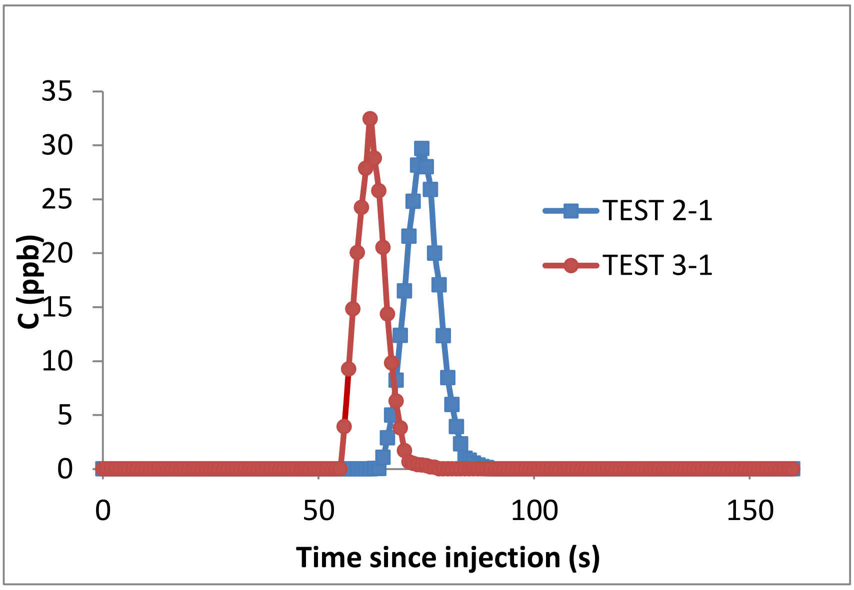
The discharge of Paths 1, 2 and 3 were calculated by multiplying the area of each path's cross section by the mean velocity. The result is listed in Table 4-2, from which it can be seen that more than 65% of total water was distributed to Path 1, with Path 2 and Path 3 totalling 17% and 18%. To further investigate the flow characteristics of the three paths, Test 2, with three different injections, was carried out.

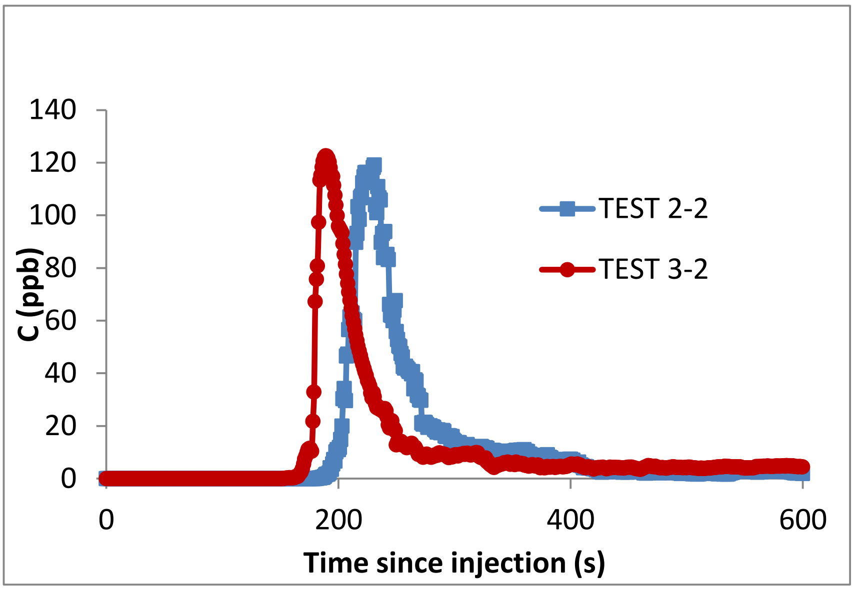
4.3.2 Tests 2 and 3

In Test 2, the upstream total discharge and water depth setup was identical to that of Test 1 but with three different injections. Test 3 was carried out under a higher discharge of 7.20l/s. The hydraulic and injection conditions are listed in Table 3. In Test 2, three injections (Test 2-1, 2-2, 2-3) which were injected from inlet I1, I2, I3 respectively and concentrations measured at O1, O2, and O3 correspondingly, at time intervals of 1s, took place. Likewise, Test 3 was performed with a higher upstream discharge. Three runs were repeated for each injection to ensure the accuracy of the experiments. The injected tracer mass per experiment was 8 mg, apart from the test for Path1, where 0.8 mg was injected. Further details of the tracer experiments are listed in Table 4-3.

Table 4-3 Injection conditions for Tests 2 and 3

Test	Discharge	Injection	Injection mass	Observation
2-1	5.50	I1	0.8mg	O1
2-2	5.50	I2	8mg	O2
2-3	5.50	I3	8mg	O3
3-1	7.20	I1	0.8mg	O1
3-2	7.20	I2	8mg	O2
3-3	7.20	I3	8mg	O3





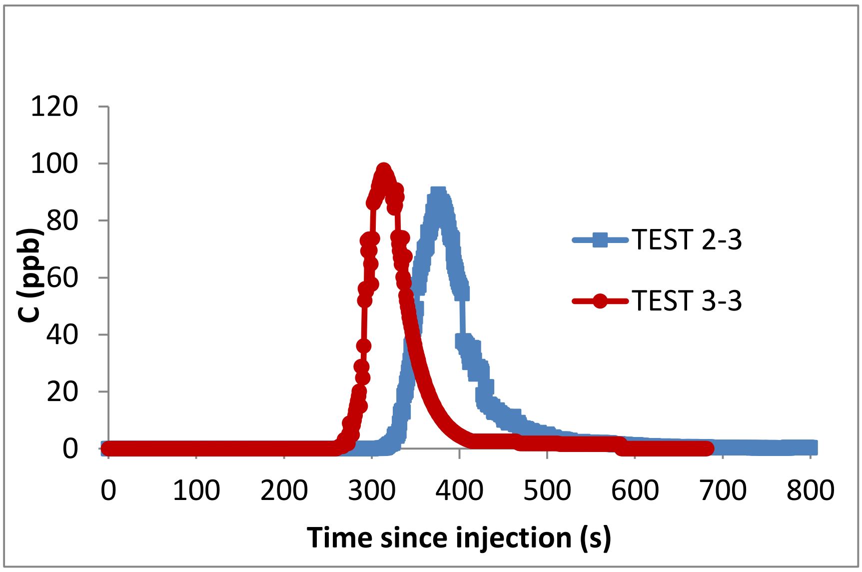


Figure 4-12 Time series of tracer distribution in Tests 2 and 3

The tracer passage curves for Test 2 and 3 are plotted in Figure 4-12 from the data obtained from the three fluorometers. For Path1, the curve shows typical shapes and trends, with the tracer plume increasingly spreading out to a peak value before then decreasing symmetrically. In contrast, for Path 2, its morphology was characterised by a relatively rapid increase in concentration values until they peaked; the subsequent decrease in concentration was slower and the tails of these curves remained for a considerable period of time. The asymmetrical shape of observed solute concentration profiles, characterised by steep leading edges and prolonged tails resulted from the transient storage which occurs in stagnant water zones in the lake (Nordin & Troutman, 1980).

The long tail in tracer passage curve of Test 2-2 and Test 3-2 illustrates that there are certain regions which slow water exchange in the designed lake, such as dead zones. The existence of the dead zones increased the water residence times which could be deduced from tracer passage curves. In Tests 4-6, a water diversion scheme was implemented to increase water flushing in the lake area.

4.3.3 Tests 4-6

Water diversion was achieved for Tests 4-6 through the use of a submerged weir placed at the inlet section of Path 1(see Figure 4-13). The weir crest was situated 0.17 m above the model bed, while the average water depth was H = 0.25 m in the experiments. To investigate the effect of flow augmentation on Paths 2 and 3, the total discharge in Test 5 and Test 6 was increased to 8.80l/s and 12.30l/s respectively from 5.50l/s in Test 4. Figures 4-14 shows the measured vertical distribution of water velocity at position A. Prior to each dye tracer experiment, the flow rate was set to a predetermined level. The tracer was injected from position A. Further details of the tracer experiments are listed in Table 4-4. For a given experiment, the values of the flow rate in Paths 1 and 3 were determined as Q = V/T, where V is the wetted volume of the path. The flow rate in Path 2 was obtained by subtracting the discharge values calculated for Paths 1 and 3 from the experimental discharge, Q_t.

Table 4-4 Details of the tracer experiments conducted in Tests 4-6

Test	Discharge Q_t (l/s)	Injection point	Monitoring point(s)	Mode of operation
4	5.50	A	O1, O2, O3	with water diversion
5	8.80	A	O1, O2, O3	with water diversion
6	12.3	A	O1, O2, O3	with water diversion

Table 4-5 Discharge ratios (Q/Qt) in different parts of the river lake model for the tracer experiments

Test	Path 1	Path 2	Path 3
1-2	0.65	0.17	0.18
4	0.37	0.35	0.28
5	0.38	0.34	0.28
6	0.44	0.27	0.29

As shown in Table 4-5, prior to implementing the diversion mechanism, 65% of the total discharge went through Path 1, with Paths 2 and 3 receiving 17% and 18% of the discharge respectively. The relative impact of the mechanism varied in Tests 4-6. In absolute terms, the flow rate in Path 2 increased from 0.94 l/s in tests 1-2 to 2.0 l/s in Test 4, 3.0 l/s in Test 5 and 3.3 l/s in Test 6, which reflected scenarios of gradual flow augmentation in the lake. Therefore, a water diversion project is necessary in order to temporarily solve the low water mixing problems in the designed lake and is also useful when it comes to improving the efficiency of water resource distribution.

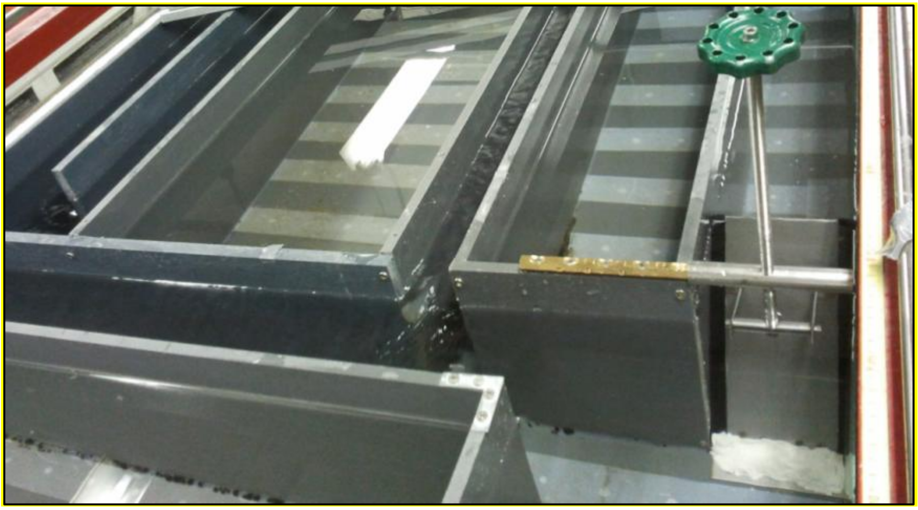
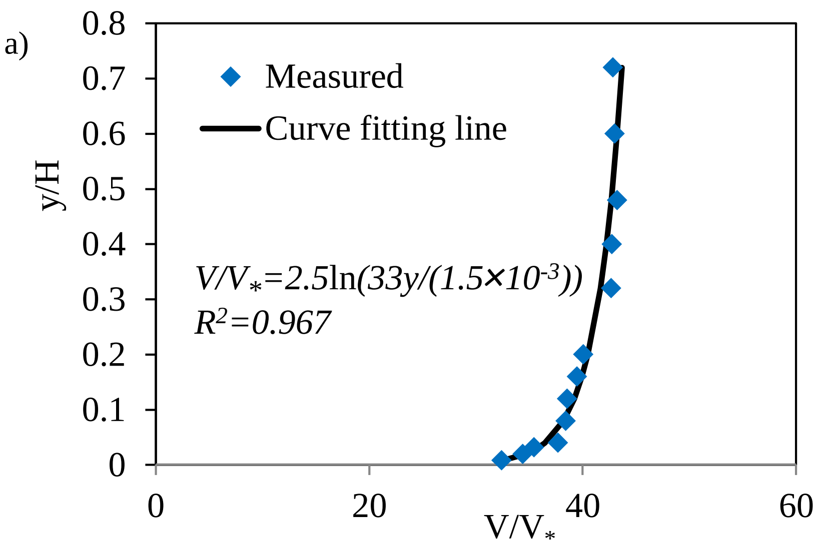


Figure 4-13 Weir setup on Path 1



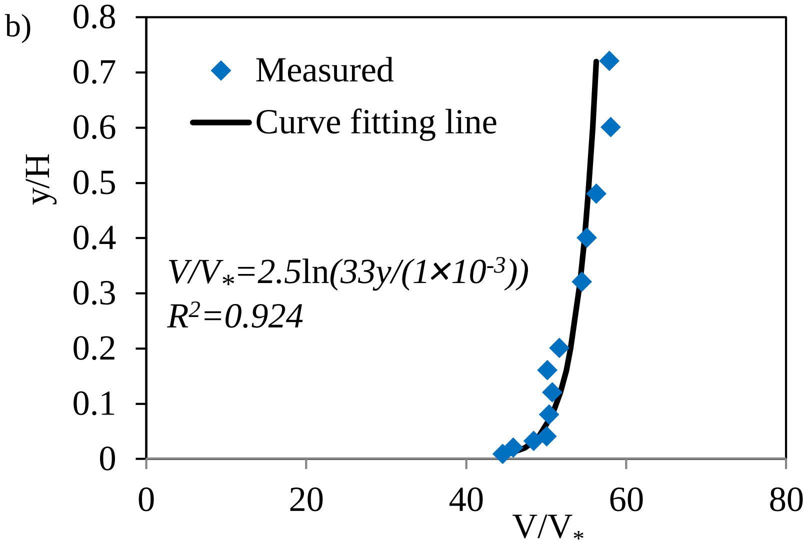


Figure 4-14 Velocity profile over a discharge of (a) 8.80l/s, (b) 12.30l/s

4.4 Experimental data processing and analysis

4.4.1 Residence Time Distribution Curves

Normalised Residence Time Distribution Curves (RTDs) were calculated from the measured tracer concentration vs. time (C vs. t) results, which allowed for comparisons between different experimental conditions. The processing of tracer test results was undertaken following Levenspiel (1999) and Metcalf & Eddy (2003). The normalised concentration, E(θ), is defined such that:

$$\int_0^\infty E(\theta)d\theta = 1 \quad (4\text{-}2)$$

where dθ is the normalised time step, with the normalised time obtained as θ = t/ $\overline{T}$, where $\overline{T}$ is the mean residence time. Quantitative approaches for computing mean residence time from tracer data have been developed through previous experimental studies (Kirchner et al., 2000) whist many studies to date have been designed to compute mean residence time of stream water at a particular site. In this study, $\overline{T}$ was calculated based on the raw tracer data from the concentration-time RTD curves according to the approximated method of Levenspiel (1999) given by:

$$\overline{T} = \frac{\sum C_i t_i \Delta t_i}{\sum C_i \Delta t_i} \tag{4-3}$$

where the summation index "i" is the concentration sample number. C_i is the measured tracer concentration at time t_i.

Mean residence time estimation is increasingly used as simple summary descriptor of the hydrodynamic processes involving storage and mixing of water within catchment systems. Aquatic scientists often estimate residence time and compare it to time scales of inputs or biogeochemical processes to calculate mass balances or understand the dynamics of populations and chemical properties. The theoretical residence time T is calculated simply by dividing the effective operating hydraulic volume (V) by the volumetric flow rate (Q). To describe the water mixing behaviour in the system, the effective volume ratio V_e is used as a measure of hydraulic efficiency based on the following equation (Metcalf & Eddy, 2003):

$$V_e = \frac{\overline{T}}{T} \tag{4-4}$$

Correspondingly, the fraction of dead space volume V_d can be calculated to provide information about the magnitude of dead volume within the system through the equation (Metcalf & Eddy, 2003):

$$V_d = 1 - \frac{\overline{T}}{T} \tag{4-5}$$

4.4.2 Tracer Mass Accuracy Index

Although tracer degradation should be avoided, some loss of tracer is inevitable. One of the most important tests of a tracer's reliability is the tracer mass accuracy index and it should always be reported in the results of a tracer study.

Consequently, in this study, percentage errors in mass balance can be calculated for laboratory datasets since the true total mass in the experimental system was generally a known quantity. Therefore, in the published literature, the mass accuracy index is commonly used as the quantitative indicator of system error. The experimental mass accuracy index E in solute mass balance is evaluated as:

$$E = \frac{(\mathrm{M} - \mathrm{Q}\int \mathrm{C(t)dt})}{\mathrm{M}} \tag{4-6}$$

where M is the injected tracer mass and Q is the discharge. $C(t)$ is the measured tracer concentration at time t. Kadlec & Wallace (2009) referred to an accuracy index of ±20% as an indicator of successful hydraulic tracer studies. An accuracy index below 0 could be related to imperfections in water flow measurements, although these are generally not addressed as problematic in hydraulic tracer studies.

4.4.3 Experimental Data Analysis

The RTD curves obtained in Test 2 were plotted in Figure 4-15a. It can be seen from these results that the RTD curves for Paths 1 and 3 were much closer to the theoretical plug flow (PF) curve than to the complete mixing (CM) curve, which indicated that there was little stagnant fluid and the overall short circuiting and mixing levels were generally low in these paths. This was generally expected for the river reaches. In the corresponding result for Path 2, however, the elongated tail of the RTD curve suggested the existence of dead zones in the lake, which is typical of flow patterns with a relatively high level of mixing. Some parameters determined from the experiments are listed in Table 4-6. For Path1 and Path 3, the hydraulic efficiency were both almost 1, thus indicating the well water mixing in the two paths. In contrast, for Path 2, the value of V_e is only 0.34, demonstrating that a significant portion of the tracer had exited Path 2 before the theoretical residence time. This also indicated hydraulic short-circuiting and the existence of dead volume within Path 2. The quality of the tracer experiments was verified with the help of the accuracy index. As shown in Table 4-6, the mass accuracy index was computed for each test and the percentage values for Paths 1, 2 and 3 were 5%, 14% and 6% respectively, thus meaning that the tracer curves obtained in this study were adequate for describing hydraulics and tracer transport in the experiment.

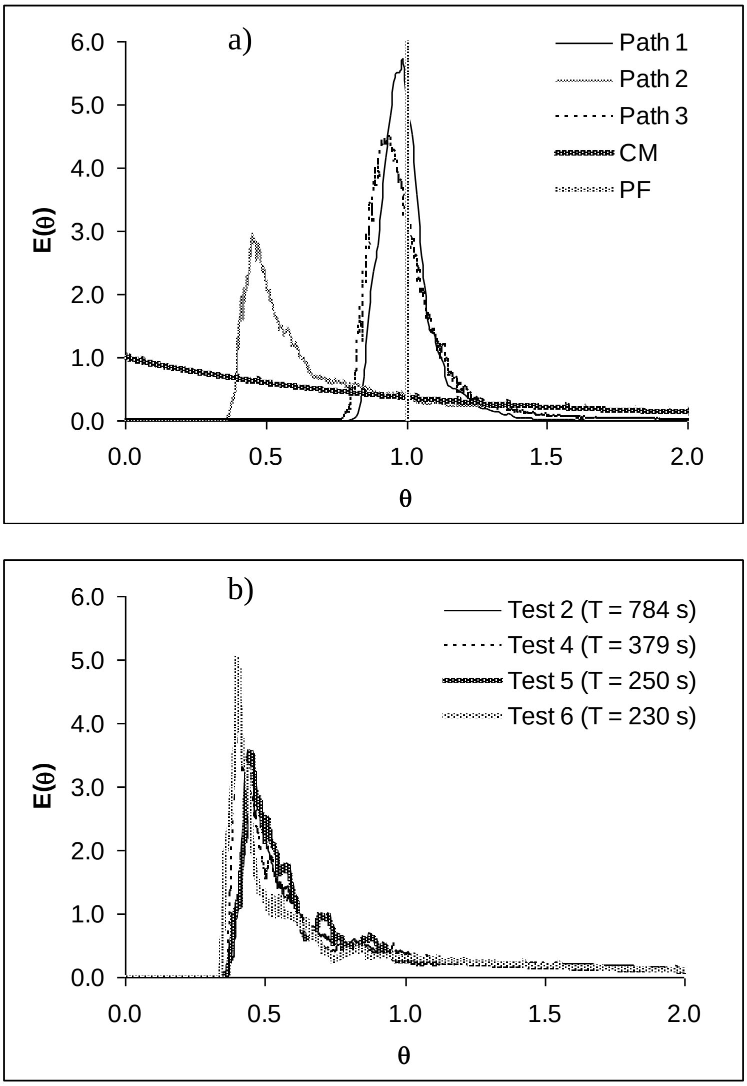


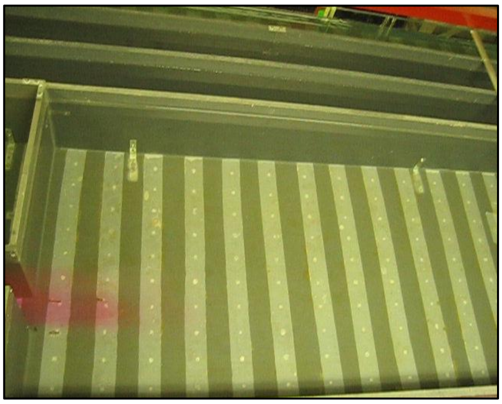
Figure 4-15 Normalised RTD curves obtained for a) Tests 2 and corresponding results for the plug flow (PF) and complete mixing (CM) flow patterns; and b) Tests 2, 4, 5 and 6 and the corresponding mean detention times in the lake

The impact of water diversion and flow augmentation on the hydraulic performance of the lake was assessed by comparing the mean RTD curves and detention times obtained before and after diversion for 5.5 l/s, and due to increasing the experimental discharge to 8.8 l/s and 12.3 l/s after diversion. Thus, the obtained results are illustrated in Figure 4-15b. It can be noted from the results of Figure 4-15b that the normalised RTD curves for different discharges were practically coincidental, with relatively small discrepancies in the peak concentration levels. This suggested that the discharge variation did not have a significant impact on the overall mixing levels in the lake, as assessed in terms of the spread of the RTD curves. The key impact of flow augmentation was the reduction of the mean detention time, as the basin volume was kept constant through the downstream control of water depth in the system. Generally speaking, such an impact would have a positive influence on lake water quality, as far as flushing is concerned. However, relatively long detention times might still be observed in regions subjected to very low flow speeds and with little exchange with the prevailing currents.

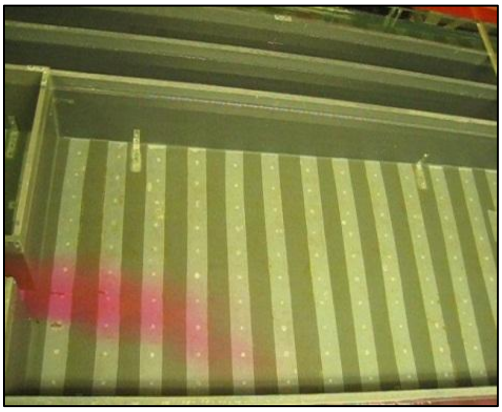
Table 4-6 Parameters of tracer passage curves from Test 2

Parameters	Path 1	Path 2	Path 3
Q/Q_t	0.65	0.17	0.18
Tracer mass(mg)	0.8	8	8
$\bar{T}$ (s)	72	254	391
T (s)	70	784	385
V_e	≈1	0.34	≈1
V_d	0	0.66	0
Accuracy index	6%	14%	5%

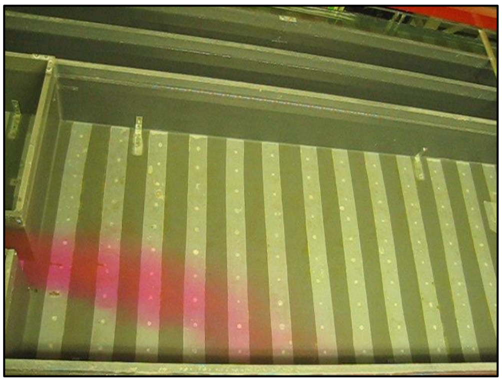
Figure 4-16 shows certain images from the experiments in the lake. These images distinctly indicate that there was an uneven flow pattern in the lake. Due to the increasing area of the traced water cloud and the limited observation angle, it was not possible to observe the complete lake once. As the preferential flow path from the inlet to the outlet sections is a straight line, the current thus generates a recirculation zone on the other side of the simulated lake, and other smaller dead zones. Low fresh water flushing in the recirculation and dead zones has resulted in water quality decline, thus reducing the ecological integrity and economic value of lacustrine ecosystems worldwide over the past few decades. Hypoxia is perceived as one of the most deleterious consequences of low fresh water flushing due to the fact that it can reduce or eliminate the ability of organisms dependent on aerobic respiration to use affected habitat (Levin et al., 2009).



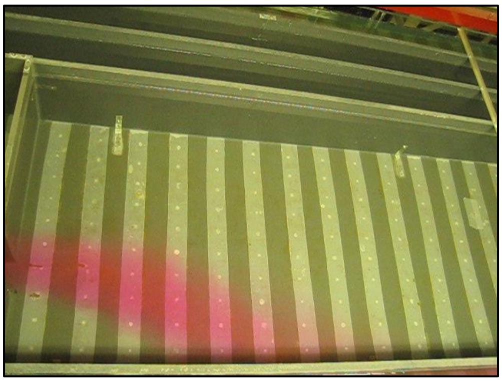
1s



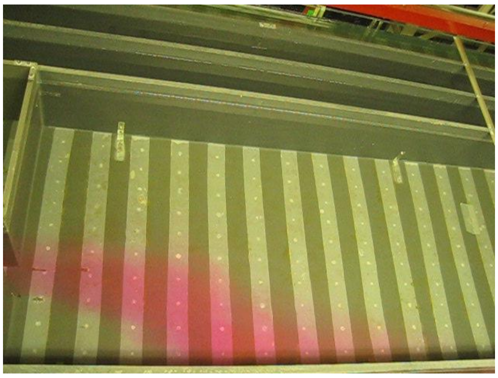
3s



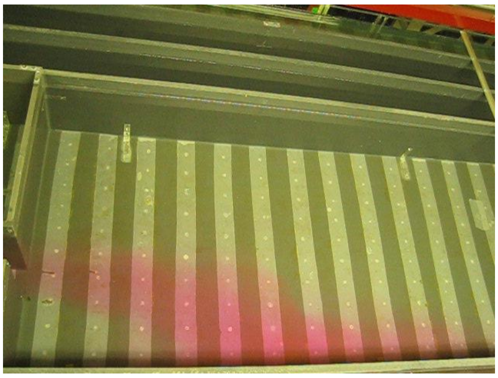
5s



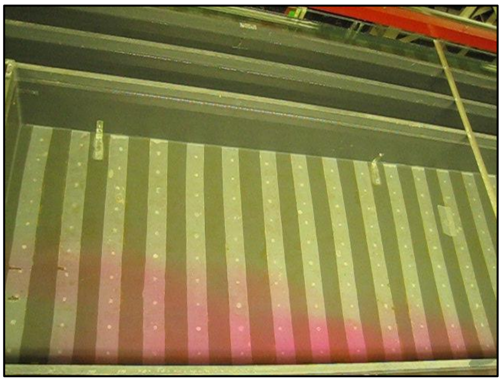
7s



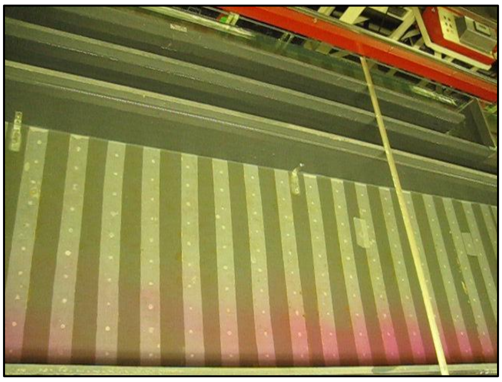
9s



11s



13s



18s

Figure 4-16 Time series of tracer distribution over the discharge of 12.30l/s

4.5 Summary

In this chapter, an idealised river-lake system similar to that of the Eco-City site was assessed by means of hydraulics laboratory experiments before and after the implementation of water diversion and flow augmentation. The details of the experimental procedure and the general tracer data observed are presented. The results analysis shows that water diversion and flow augmentation improved flushing, as seen from the perspective of reducing the mean residence time. However, recharge alone had little impact on the overall mixing level in the lake waters due to poor cross-sectional flow distribution. The persistent low water mixing in the lake makes its ecosystem prone to the development of hypoxic conditions even at low levels of nutrient input. This could lead to localised water quality problems in stagnant and circulating flow regions, despite the interventions. This could in turn impact, for instance, water supply and fisheries productivity.

Therefore, in order to control and prevent lake water problems and keep the water quality at a certain level, additional measures to water diversion may need to be considered to achieve the goal of enhancing water quality in a recharged lake. A flow deflector near the lake inlet combined with flow augmentation could be assessed to mitigate the occurrence of dead zones. This aspect will be assessed in more detail based on numerical modelling predictions made for the lake model in Chapter 5.

CHAPTER 5 EXPERIMENTAL AND NUMERICAL MODELLING RESULTS

5.1 Introduction

With the objective of better understanding the hydraulic and water quality processes affecting the river-lake system, a 2D hydrodynamic and water quality model was used in this study. The use of numerical modelling for conducting flow characteristics and solute transport has become widespread for rivers, lakes, estuaries and coastal studies. In this chapter, experimental data are used to evaluate the performance of the numerical model in predicting solute transport and various flow features. A longitudinal dispersion coefficient is determined for input data to the numerical simulations. Measured and simulated results are then compared and discussed. In addition, lake flushing scenarios with flow deflectors will be carried out using the validated numerical model.

5.2 2D Hydrodynamic and Water Quality Model

A 2D hydrodynamic and water quality model, DIVAST (Depth Integrated Velocities and Solute Transport), developed by Falconer (1986) was used in this study. DIVAST is a 2D, depth integrated, hydrodynamic model, which has been developed for estuarine and coastal modeling. It is suitable for water bodies which are dominated by horizontal, unsteady flow and do not display significant vertical stratification. The model simulates two-dimensional distributions of currents, water surface elevations and various water quality parameters within the modelling domain as functions of time, taking into account the hydraulic characteristics governed by the bed topography and boundary conditions.

DIVAST has been developed in order to simulate the hydrodynamic, solute and sediment transport processes in rivers, estuaries and coastal waters. The hydrodynamic module solves the depth integrated Navier–Stokes equations. The solute module is used to simulate solute transport, including salinity, BOD, DO, the nitrogen and phosphorous cycles and algal growth. The model has been refined and validated in light of data from more than 30 years against many laboratory and field studies (Falconer et al., 2003; Gao et al., 2011).

5.2.1 Governing Equations for Hydrodynamic Processes

The governing equations for the hydrodynamic processes include a continuity equation:

$$\frac{\partial \eta}{\partial t}+\frac{\partial p}{\partial x}+\frac{\partial q}{\partial y}=q_m \tag{5-1}$$

and two momentum equations:

$$\frac{\partial p}{\partial t}+\frac{\partial \beta pU}{\partial x}+\frac{\partial \beta pV}{\partial y}= fq-gH\frac{\partial \eta}{\partial x}+\frac{\rho_a}{\rho}C_w W_x\sqrt{W_x^2+W_y^2}-\frac{gp\sqrt{p^2+q^2}}{H^2C^2} +\varepsilon\left[2\frac{\partial^2 p}{\partial x^2}+\frac{\partial^2 p}{\partial y^2}+\frac{\partial^2 q}{\partial x\partial y}\right] \tag{5-2-a}$$

$$\frac{\partial q}{\partial t}+\frac{\partial \beta qU}{\partial x}+\frac{\partial \beta qV}{\partial y}=-fp-gH\frac{\partial \eta}{\partial y}+\frac{\rho_a}{\rho}C_w W_y\sqrt{W_x^2+W_y^2}-\frac{gq\sqrt{p^2+q^2}}{H^2C^2} +\varepsilon\left[2\frac{\partial^2 q}{\partial x^2}+\frac{\partial^2 q}{\partial y^2}+\frac{\partial^2 p}{\partial x\partial y}\right] \tag{5-2-b}$$

where η=water surface elevation above datum; q_m =source discharge per unit horizontal area ($m^3/s/m^2$); p, q =discharges per unit width in the x and y directions respectively; p=UH, q=VH; U, V = depth averaged velocity components in the x, y directions: H=total water depth and $H=\eta+h$; h=water depth between bed level and datum; β=momentum correction factor for a non-uniform vertical velocity profile; f=coriolis parameter; ρ_a=air density ($\cong 1.292 kg/m^3$); ρ=density of fluid (=1000 kg/m^3); W_x, W_y= ind velocity in the x and y directions respectively; C=chezy roughness coefficient; C_w=air/fluid resistance coefficient (assumed to be 2.6×10^{-3} Falconer & Chen (1991)); ε=depth averaged turbulent eddy viscosity.

5.2.2 Governing Equations for Solute Transport Processes

The governing equation for solute transport processes can be written as:

$$\frac{\partial HS}{\partial t}+\frac{\partial HUS}{\partial t}+\frac{\partial HVS}{\partial t}=\frac{\partial}{\partial x}\left[D_{xx}H\frac{\partial S}{\partial x}+D_{xy}H\frac{\partial S}{\partial y}\right]+\frac{\partial}{\partial y}\left[D_{yx}H\frac{\partial S}{\partial x}+D_{yy}H\frac{\partial S}{\partial y}\right]+\Phi S \qquad (5\text{-}3)$$

where, S= depth averaged solute concentration (unit/volume)； $D_{xx}, D_{xy}, D_{xy}, D_{yy}$ = depth averaged dispersion-diffusion coefficients in the x and y directions respectively (m^2/s)； Φ_S summarises all other sources and sinks of solute apart from advective and dispersive transport. Sources and sinks include discharge from outfalls and rivers as well as chemical and biological transformations.

In DIVAST, all solutes, even algae, enter the model domain via the outfalls. An initial concentration of each pollutant in the cell containing the outfall is calculated by assuming that the discharged material is immediately and homogeneously distributed in that particular cell.

5.3 Parameters and Coefficients Estimation

The hydraulic processes tend to be of a more deterministic nature than water quality processes and variables, which often show significant variability. Similarly, hydraulic data tend to be more deterministic than some of the water quality data. For example, water depth recordings will exhibit less variability than bacterial counts. It is common practice to change the water quality parameters until the “best fit by eye" or the minimum sum of squared deviations is obtained. A complete study requires a sequence of independently calibrated models dealing with hydrodynamics and transport of dissolved pollutants. A single parameter value is retained, which is then used for subsequent analysis and prediction. Some critical parameters in DIVAST were selected and are discussed in detail below.

5.3.1 Eddy Viscosity Coefficient

The turbulence model in DIVAST relates to Boussinesq's approximation for the mean shear stress τ_e in turbulent flow:

$$\tau_e = \varepsilon \frac{dv}{dy} \tag{5-1}$$

where ε is the eddy viscosity, which is dependent on the turbulence characteristics of the flow and may be thousands of times larger than the molecular viscosity. If the turbulent shear stress is dominated by bottom friction, a relationship between the Chezy coefficient and the eddy viscosity exists. In DIVAST, the depth-integrated eddy viscosity is calculated (Fischer, 1979) from:

$$\varepsilon = Ce\frac{H}{C}\sqrt{g\left(U^2+V^2\right)} \tag{5-2}$$

where Ce =eddy viscosity coefficient.

In DIVAST, values of the depth averaged turbulent eddy viscosity, ε, can be estimated using a logarithmic velocity profile, such as (Elder, 1959):

$$\varepsilon = \frac{\kappa}{6}u_* H = 0.0667 u_* H \tag{5-3}$$

where u_* is the bed shear velocity defined as:

$$u_* = \sqrt{\frac{\tau_b}{\rho}} = \frac{\sqrt{g(U^2+V^2)}}{C} \tag{5-4}$$

where τ_b is the bed shear stress

With this said however, Fischer (1973) found that field data is greater than that given by Elder (1959), and discovered the value from laboratory data to be:

$$\varepsilon = 0.15u_*H = 0.15\frac{H}{C}\sqrt{g\left(U^2 + V^2\right)} \tag{5-5}$$

Fischer's (1979) suggestion of $Ce \approx 0.15$ was based upon laboratory data. Bates et al. (1998) reported the range [0.1, 2.0] m^2/s for their large scale 2D river flood simulations.

5.3.2 Semi-slip Boundary Condition Coefficient

Generally speaking, boundary conditions for closed walls are either slip or no-slip. Slip conditions assume a negligible wall shear, whereas no-slip conditions specify a zero velocity at the wall boundary. However, due to the finite grid size used in numerical models, particularly for small scale simulation, the use of a no-slip boundary condition may result in the transmission of a much larger shear stress from the wall to the fluid, thus resulting in an under-prediction of the flow velocity. Previous work conducted by Li & Falconer (1995) had indicated that for a square harbour the semi-slip boundary condition gave accurate results.

Thus, in this study, for a rectangular river lake system, a semi-slip boundary condition was included to cater for velocity gradient at closed boundary. Wall shear stress was given as:

$$\tau_w = \frac{\rho g}{c^2} U_t^{\,2} \tag{5-6}$$

where U_t =undistributed velocity, assuming that the side wall shear stress can be represented by Newton's Law of viscosity, then the normal velocity gradient can be obtained to give (Li & Falconer ,1995):

$$\frac{\partial U_t}{\partial n} = \frac{\sqrt{g}}{C}\frac{U_t}{\sigma H} \tag{5-7}$$

By using a backward difference representation at an upper boundary, and similarly a forward difference representation at a lower boundary, the virtual velocity at the wall to give the correct shear stress was given as:

$$U_w = U_t(1 - \frac{\sqrt{g}}{C}\frac{\Delta n}{\sigma H}) \tag{5-8}$$

where Δn = grid size normal to the wall. For the subsequent model simulations; σ =eddy viscosity constant. In the DIVAST model, a coefficient for velocity gradient at closed boundary α was introduced which was given as:

$$\alpha = \frac{U_w}{U_t} \tag{5-9}$$

5.3.3 Longitudinal Dispersion Coefficient

Longitudinal dispersion coefficient is a critical quantity used to predict solute transport processes, such as the propagation of an accidental spill of a quantity of pollutant. It is also used to compute the downstream concentration of on output from a sewage treatment plant. The longitudinal dispersion coefficient is conventionally estimated using analytical formulations in conjunction with semi-empirical estimates for certain parameters and a set of bulk flow stream data (Seo & Baek, 2004).

The longitudinal dispersion coefficient can be estimated based on concentration measurements collected from laboratory tracer experiments since the empirical formulas are generally of limited accuracy. The equation to obtain longitudinal dispersion coefficients was given as (Fischer, 1979):

$$D = \frac{1}{2}\overline{U}^2 \frac{{\sigma_2}^2 - {\sigma_1}^2}{{t_2}^2 - {t_1}^2} \tag{5-10}$$

where the subscripts 1 and 2 refer to two cross sections, and $\bar{t}$ is the time of passage of the centroid of the cloud at the station. σ^2 is the variance of the time-concentration curve measured at a fixed station. Therefore, the longitudinal dispersion coefficients adopted in the model could be calculated from the experimental data using Equation 5-10.

5.4 Model Calibration

Model calibration is an essential component of all numerical studies. The typical procedure for calibrating a hydrodynamic model involves systematic alteration of the model inputs until the outputs closely match the field or laboratory observations. In this study, the data obtained from Test 2 (see Chapter 4, Section 4.3.2) were used. In this section, the experimental data was compared against model predictions and the results were used to calibrate the performance of the DIVAST model.

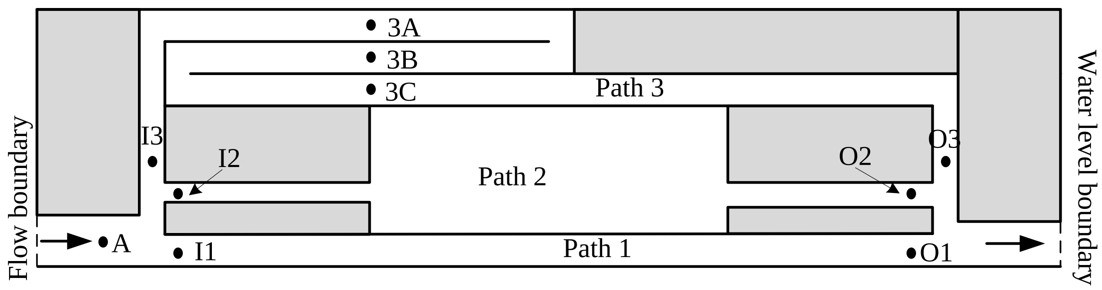


Figure 5-1 Computation domain and boundary conditions

The numerical description of experimental setup involved a rectangular domain of 1000 cm $\times$ 120 cm, as shown in Figure 5-1. A calibration procedure was conducted to evaluate the hydrodynamic and water quality model accuracy in reproducing the time and space variability of velocity and concentration within the domain. Comparisons between the simulation results and the experimental data were carried out. A selected set of model parameters as well as initial and boundary conditions were calibrated for the whole domain. The numerical solutions were obtained using a grid size of $\Delta x = \Delta y$ = 1cm, with a time step of 0.02 s.

Table 5-1 Calibrated values for certain main parameters

Parameter	**Unit**	**Calibrated**	**Harbour**
Roughness length	mm/km	3.0	50
Kinematic viscosity fluid	mm^2/s	1.31	1.31
Minimum Reynolds number	-	1000	1000
Momentum correction factor	-	1.016	1.016
Eddy viscosity coefficient	-	0.15	1.2
Longitudinal dispersion coefficient	m^2/s	0.5	13
Lateral turbulent diffusion coefficient	m^2/s	0.02	0.02
Partial slip boundary condition coefficient	-	0.9	0.95

In Table 5-1, the calibrated values for certain main parameters are listed. Most of the coefficients used in this study were default values which were derived from previous work concerning harbour application of DIVAST (Hakimzadeh & Falconer, 1997), with tests also being undertaken to study the sensitivity of the model results to the coefficients. For the hydrodynamic model a coefficient for a partial slip boundary condition of 0.9 was used. The theoretical value of eddy viscosity coefficient for a logarithmic velocity profile used was 0.07. However, this value has been found to be relatively small for practical open channel flow studies (Fisher, 1978) and a more realistic value is typically 0.15 (Hakimzadeh & Falconer, 1997). An eddy viscosity coefficient of 0.15 was finally adopted in this study for both the lake and river areas by matching the measured velocity with the value computed with DIVAST. The bed roughness length was assumed to be 3 mm for all model simulations. The calibration process gave specific optimal values for longitudinal dispersion, as described in detail later. The initial and calibrated values for each parameter are reported in Table 5-1.

The model was calibrated against water velocity and tracer concentration data. In order to test the ability of the numerical model to predicting the hydrodynamic parameters, comparisons were first made between the numerically predicted and the laboratory-measured velocities. The model accuracy was established by calculating the Relative Error (RE) and the Coefficient of determination (R^2) between simulation results and experimental data time series for water velocity and tracer concentration. Statistics for the calibration of water velocity are presented in Table 5-2. The RE were calculated and listed in the table, which demonstrated that the numerical model predictions agreed closely with the measured flow velocity for Paths 1 and 2, with the RE totalling 2.67% and 6.89% respectively. In contrast, the calibration result for Path 3, with the RE at 9.30%, was not as satisfactory as Paths 1 and 2.

Table 5-2 Comparison of the predicted and observed velocity

Position	Predicted Velocity(cm/s)	Observed Velocity(cm/s)	Error (cm/s)	Relative Error (RE)
A	9.60	10.00	0.4	4.16%
O1	11.2	10.90	-0.3	2.67%
O2	5.80	6.20	0.4	6.89%
O3	4.30	4.70	0.4	9.30%

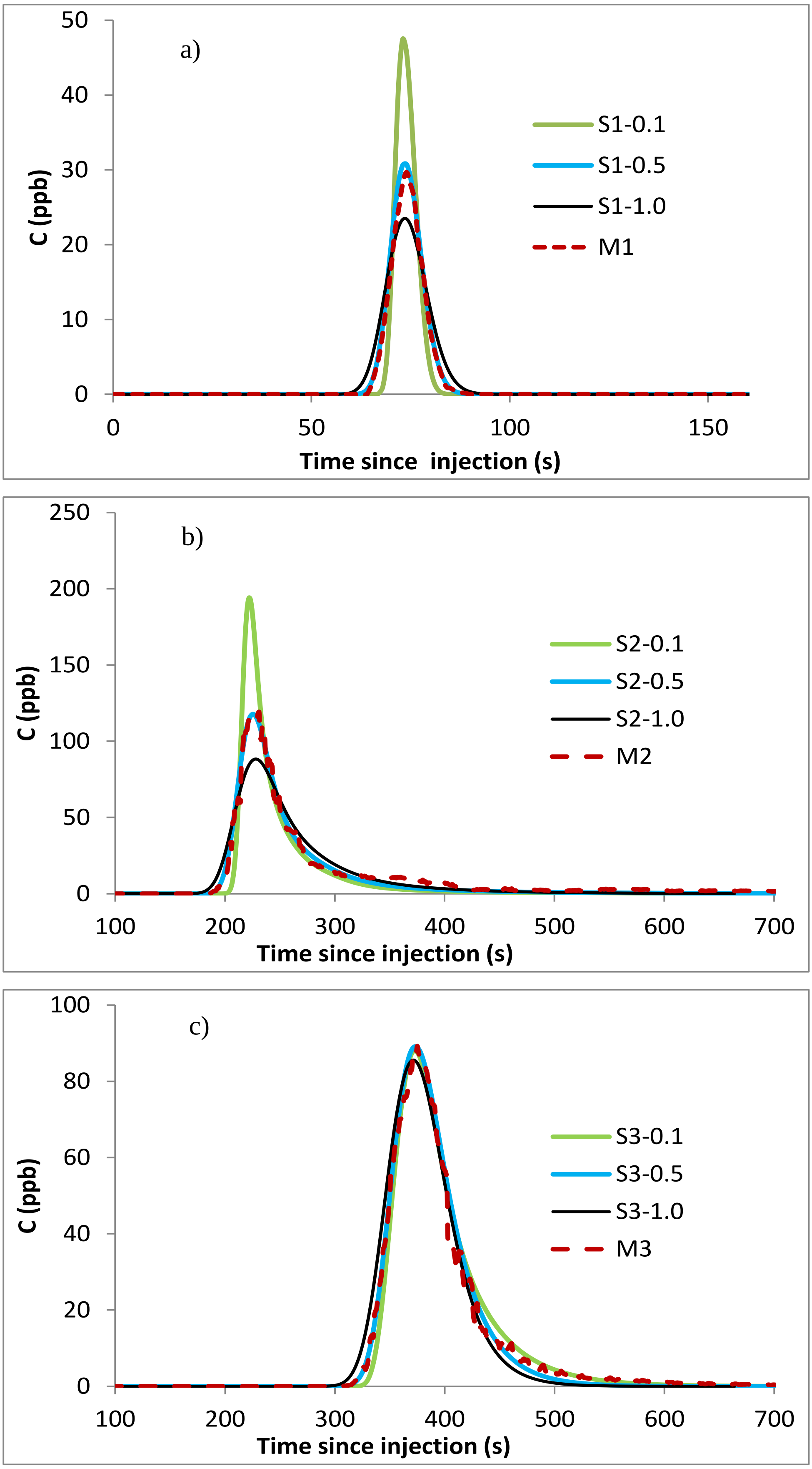
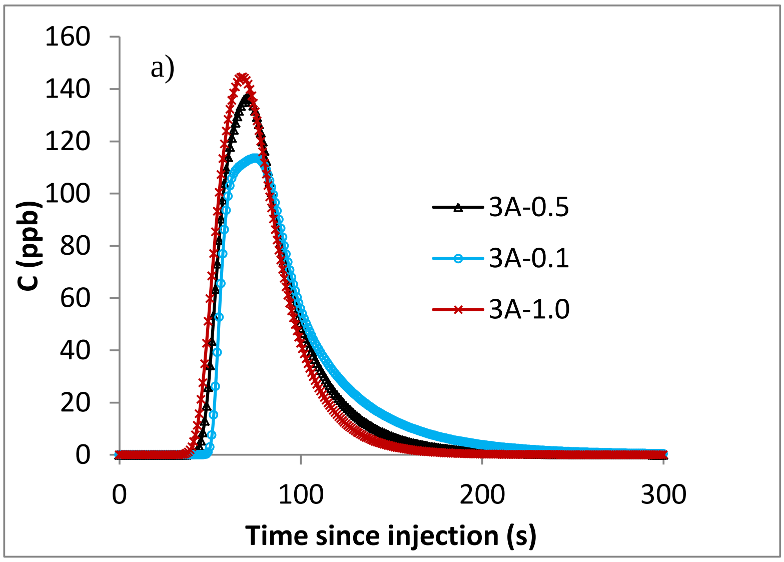
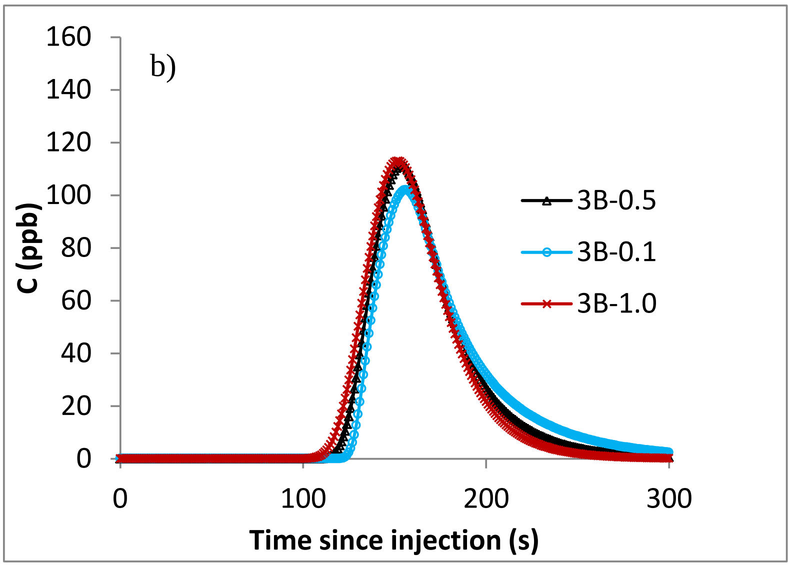


Figure 5-2 Comparison of Observed and Predicted tracer curves with different longitudinal dispersion coefficient at (a) O1 (b) O2(c) O3

The numerical model was also calibrated using the tracer concentration data obtained from Test 2. Three values of the dispersion coefficient, namely 0.1, 0.5, and 1.0 $m^2 s^{-1}$ were used in different runs, as a critical parameter for predicting solute transport. The best model performance was achieved with a value of 0.5 $m^2 s^{-1}$, with close agreement being obtained between measured and predicted results in this condition. Figure 5-2 shows the results of the tracer concentration comparisons at O1, O2 and O3, with the dashed line showing the measured velocities and the solid lines the model predictions. The coefficient of determination (R^2) was calculated between simulation results and experimental data time series for the tracer concentration. It was found that the best model performance was achieved for the longitudinal dispersion coefficient =0.5 with R^2=98%, 96% and 98% for O1, O2 and O3 respectively. The comparison of predicted and observed concentration data for O1, O2 and O3 is reported in Figure 5-2. A sensitivity analysis performed with DIVAST using a Roughness Coefficient of 10mm and 1mm showed no degradation of the quality of the numerical fitting shown in Figure 5-2. The simulated results suggested that changes in dispersion coefficients should be taken into account in DIVAST. The accuracy of the model prediction can be considered as satisfactory.





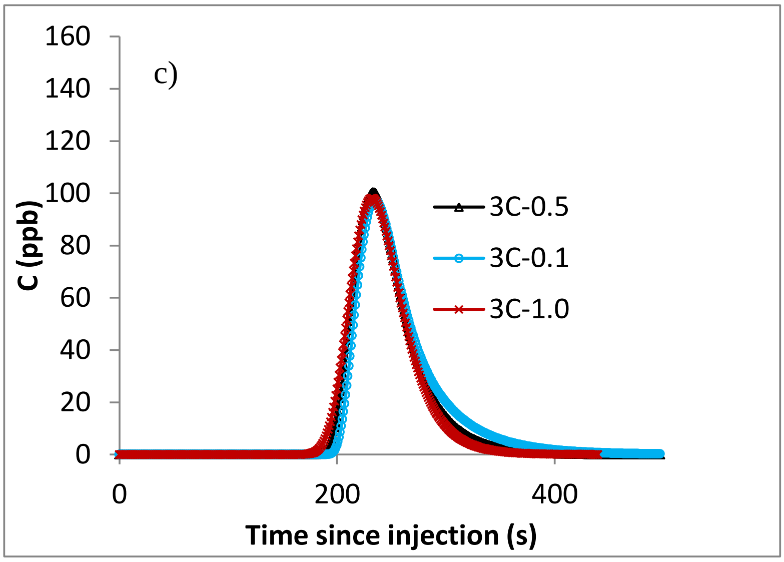


Figure 5-3 Tracer passage curves for points (a) 3A (b) 3B (c) 3C with longitudinal dispersion coefficient variation in Path 3

Figure 5-2 (c) shows that the peak values of the tracer did not vary with the longitudinal dispersion coefficient as was the case with Figure 5-2 (a) and (b). A set of simulations were performed to investigate the effect of variation of longitudinal dispersion coefficient on the tracer transport in Path 3. The concentrations of three selected points, namely 3A (298cm, 116cm), 3B (298cm, 103cm) and 3C (298cm, 90cm) in Path 3 (see Figure 5-1) are presented as a function of time in Figure 5-3. It can be seen that the peak values varied with the longitudinal dispersion coefficient before reaching the first turning on Path 3 (see Figure5-3(a)). In contrast, Figure 5-3 (b) and (c) illustrates that the tracer dispersion might be strengthened by the turnings, therefore meaning that the tracer mixed thoroughly compared with Paths 1 and 2.

A parametric study conducted using DIVAST illustrated that, for the test cases considered, variation of effective longitudinal dispersion coefficient has an important effect on tracer transport. During the calibration process, different longitudinal dispersion coefficients were assumed. The best fitting case was chosen by calculating the R^2 at a range of 96% to 98%. Additional simulations showed that the sharp turnings in Path 3 had a strong effect on the tracer transport and flow velocity. The peak values of tracer passage curves did not vary with the longitudinal dispersion coefficients whilst the RE was nearly 10%. Thus, the model simulations of tracer passage curves for Path 3 are not included in the following sections.

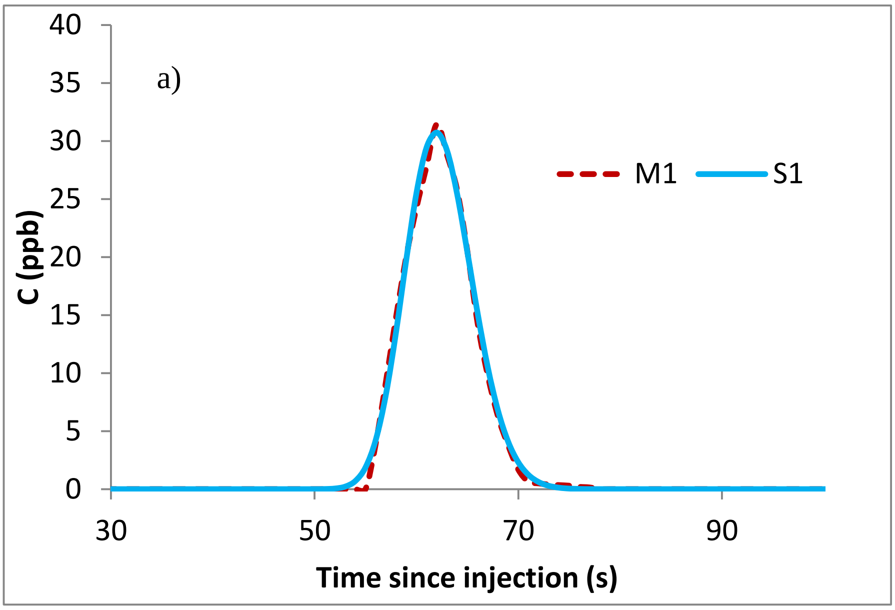
5.5 Numerical Modelling of River Lake System

5.5.1 Model Validation

The following section presents a series of validation checks for DIVAST which demonstrate the accuracy of the model for solute transport in the river lake system, Test 3 was selected to validate the model's predicted and measured flow velocity and dye concentrations. Table 5-3 compares the predicted and measured flow velocity. It can be seen from the table that REs for Paths 1 and 2 were 2.26% and 1.21% respectively, thus demonstrating reasonable hydrodynamic agreement. Figure 5-4 presents the predicted and measured dye concentrations for Test 3. Reasonable agreement between the measured dye concentration data and simulation results was achieved with R^2=98% for Path 1. The coefficient of determination of 86% was achieved for Path 2. It should be noted that the simulations were less effective when it came to describing the tail of tracer curves, as has been found previously (Massei et al., 2006). The validation results indicated that the calibrated DIVAST model was able to closely predict the flow field and the tracer transport in the river lake system, with the exception of Path 3. The model simulations of tracer passage curves for Path 3 are not included in the following sections.

Table 5-3 Comparison of the predicted and observed velocities in Test3

Position	Predicted Velocity(cm/s)	Observed Velocity(cm/s)	Error (cm/s)	Relative Error (ER)
A	12.0	12.1	0.1	0.8%
O1	13.3	13.0	-0.3	2.26%
O2	8.20	8.10	-0.1	1.21%
O3	5.50	6.00	0.5	9.10%



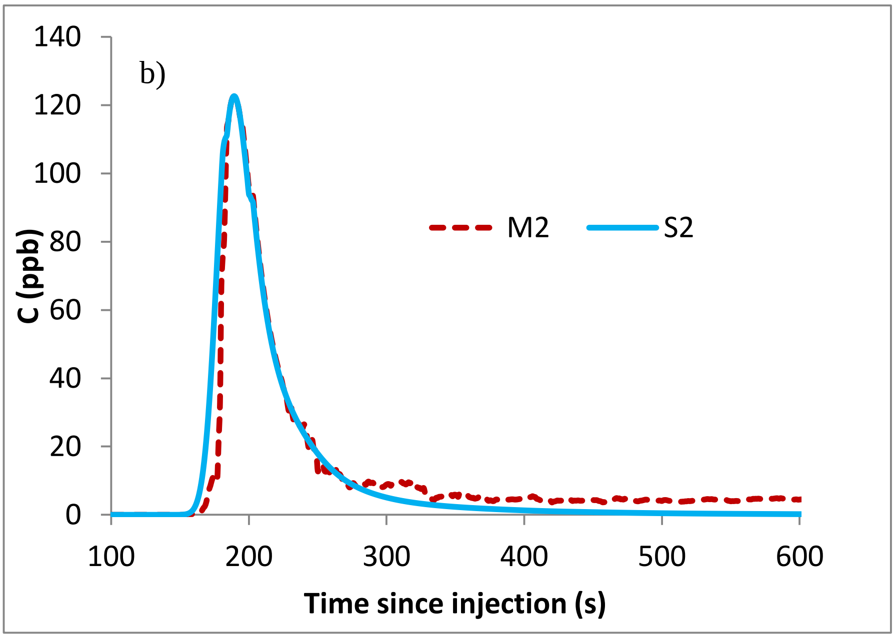


Figure 5-4 Comparisons of the predicted and measured concentrations for Test 3 at a) O1; b) O2

5.5.2 Water Diversion Modelling

Validation of DIVAST showed reasonable agreement with experimental and numerical solutions for tracer transport in the river lake system. Thus, an eddy viscosity coefficient constant value of 0.15, a longitudinal dispersion coefficient value of 0.5 m^2 s^{-1} and partial slip boundary condition coefficient value of 0.9 were used for the numerical simulations. All other parameters were at the default values set within the DIVAST model (see Table 5-1).

Water diversion modelling was carried out using the validated numerical model to assess the impacts of water diversion on the water distribution in the river lake system and the flushing processes within the lake. A weir structure was introduced to divert water from Path 1 to Path 2 and 3, thus meaning that the numerical model was slightly modified and simulated with the effect of the weir placed near the inlet section of Path 1. Several runs were carried out to study flow path distributions by adjusting the height of the weir on Path 1 with the total discharge totalling 5.50l/s. The predicted tracer concentration curves for Paths 1 and 2 over different weir heights were plotted as a function of time in Figure 5-5. For Path1, the higher the weir, the longer the residence time was. Conversely, the water residence time for Path 2 decreased as the weir height increased. The water residence time was reduced from 219s to 132s for Path 2. The simulation results suggested that the use of water diversion to enhance the water flushing in Path 2 was an effective solution. For lakes with longer residence times, long-term average pollutant loadings become more important to overall lake water quality.

The predicted results showed that the effect of water diversion was negligible when the weir height in Path 1 was below 10cm. The water diversion ratios over different weir heights (above 10cm) were calculated and listed in Table 5-4. It can be seen that, with the values of 20cm, the water diversion ratios, namely 35%, 32% and 33% for Paths 1, 2 and 3 respectively, could be obtained from the numerical model.

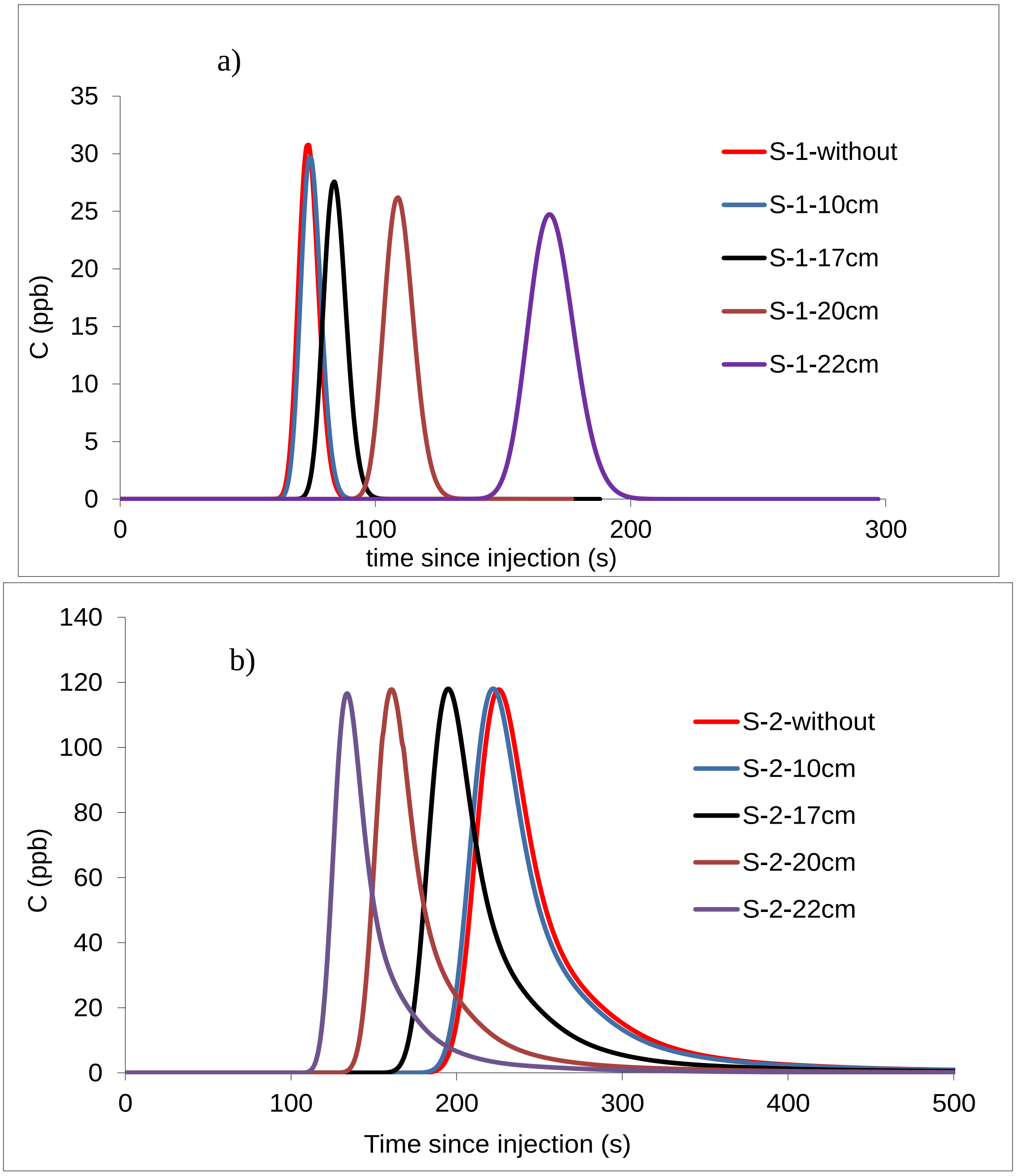


Figure 5-5 Predicted tracer passage curves over a range of weir height on point a) O1; b) O2.

Table 5-4 Water diversion ratios over different weir heights

Weir	**17cm**			**20cm**			**22cm**		
Path	1	2	3	1	2	3	1	2	3
Q (l)	2.68	1.35	1.58	1.95	1.75	1.80	1.32	1.95	2.23
Ratio (%)	48	24.1	28.1	35	32	33	24	35.5	40.5

5.5.3 Flow Augmentation Modelling

Although water diversion is an effective tool with which to enhance the water flushing in Path 2, during the dry periods, the low water flow would reduce the ecological function of the river lake system. As the tracer experiments presented, flow augmentation is of vital importance when it comes to meeting the flow requirements in the river lake system. To accomplish this goal, the corresponding numerical modelling was carried out in the river lake system under controlled flow rates. In this study, flow augmentation modelling was run over a range of flow rates, i.e. from 5.50 l/s to 12.3 l/s. A sketch of the flow field in the river lake system is shown in Figure 5-6, as depicted from the corresponding hydrodynamic model predictions.

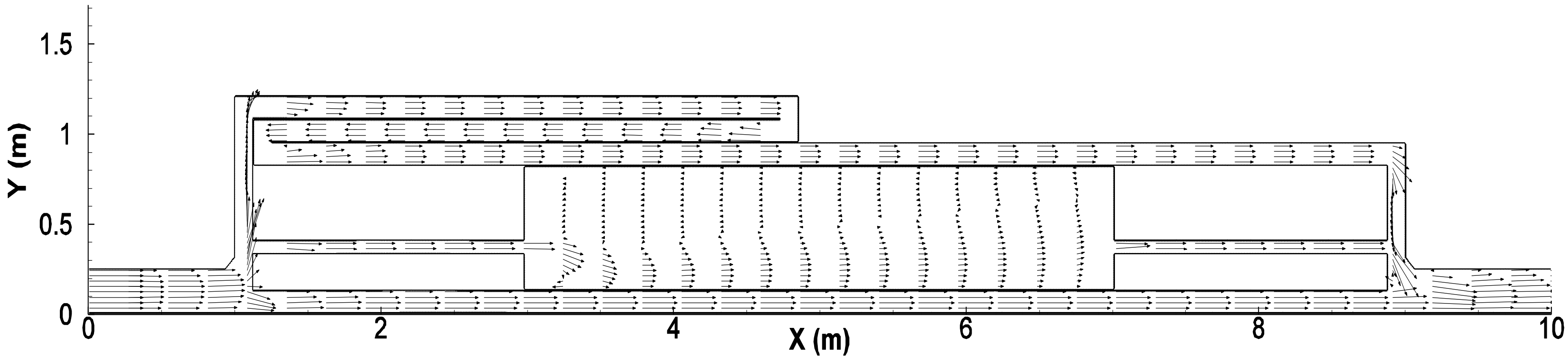


Figure 5-6 A sketch of the flow field in the river lake system

Two simulations with discharges of 8.80l/s and 12.30l/s were used to investigate the effect of flow augmentation on water mixing improvement of the river lake system. In the model, the location of the outfall was point A (70cm, 12cm). Figure 5-7 shows the predicted tracer passage curves flow augmentation. It can be seen that under the flow rate of 12.30l/s the water residence time for Paths 1 and 2 was reduced to 43s and 89s respectively.

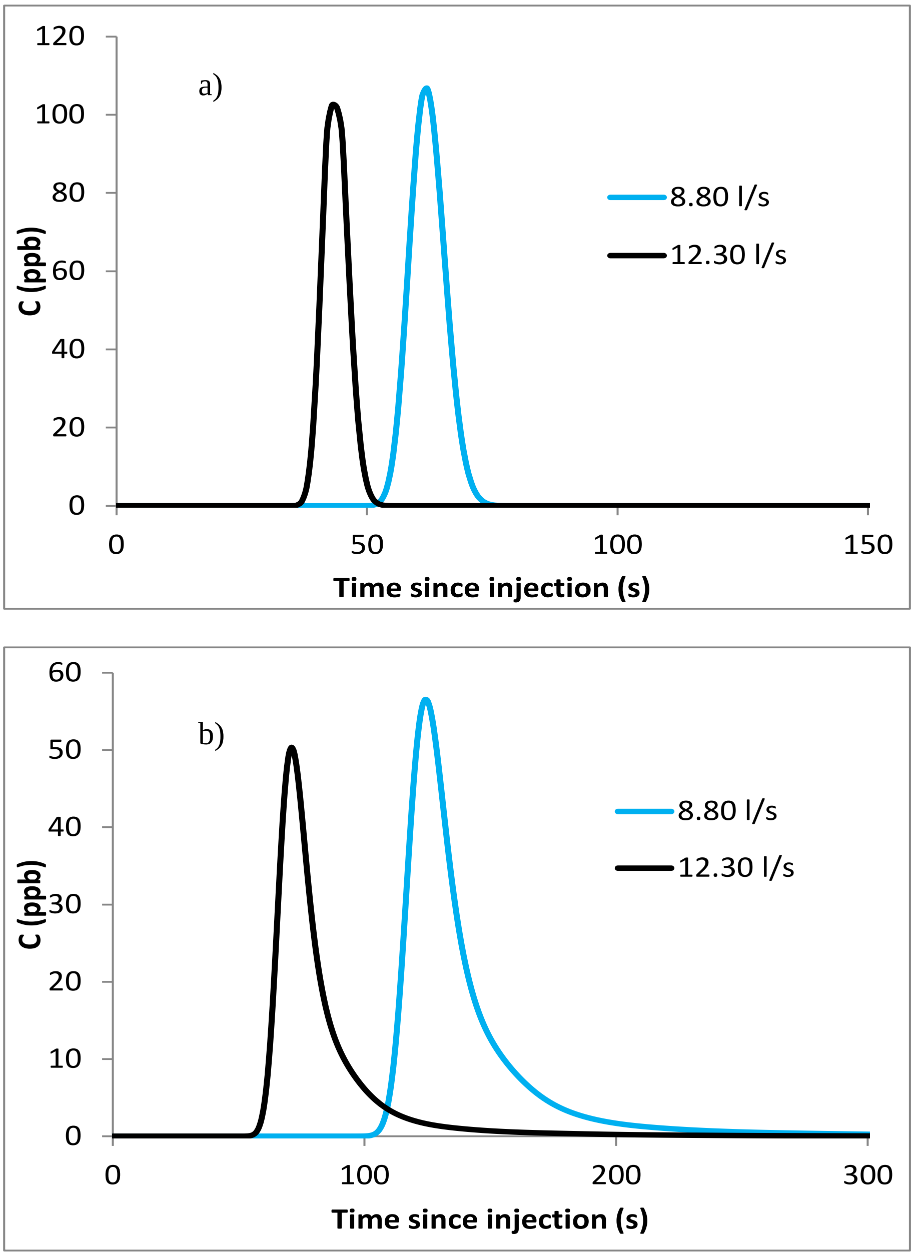


Figure 5-7 Predicted tracer concentrations under flow augmentation at a) O1; b) O2

Flow augmentation from the upstream enhanced the water flushing in the river lake system, particularly for the lake. To evaluate the effect of flow augmentation on the hydraulic efficiency in Path 2, the mean residence time and theoretical residence time were calculated and analysed based on the predicted tracer curves (see Table 5-5). The residence time is an important parameter which measures the effectiveness of water flushing in removing a pollutant from the system, meaning that the pollutants will be prevented from exerting their adverse effects. A short residence time is beneficial to pollutant removal. Results shown in Table 5-5 indicate that the flow rate has a significant effect on the residence time, which ranges from 180s over a discharge of 5.50l/s to 89s over a discharge of 12.30l/s for Path 2. However, the hydraulic efficiency, V_e, has only just increased from 0.42 to 0.44. This numerical modelling result coincides with the experimental results analysis based on Figure 4-15b in Chapter 4. This suggested that the flow augmentation did not have a significant impact on the overall water mixing levels in the lake.

Table 5-5 Parameters of tracer passage curves for Path 2 with flow augmentation

Q_t (l/s)	**5.50**	**8.80**	**12.30**
Q_2(l/s)	1.75	2.64	3.69
$\bar{T}$(s)	180	120	89
T (s)	427	277	202
V_e	0.42	0.43	0.44
V_d	0.58	0.57	0.56

5.6 Numerical Modelling Results for Lake Area

In order to thoroughly investigate the water mixing, numerical simulations in the lake area were undertaken to determine the flow pattern and transport of the water within the lake during flow augmentation. Figure 5-8 shows flow field in the lake under the flow rate of 12.30l/s. In order to more effectively analyse the flow pattern in the lake, three cross sections (C, D and E) were selected where the mean velocities were measured in the physical model with the same hydraulic conditions. Each cross section consistes of 5 vertical profiles whilst 18 points were measured on each vertical, with the near bed points more closely spaced to each other. The lowest point was located at 2 mm above the channel bed; the highest point, 70 mm below the water surface. Results from numerical simulations were compared with the measured mean velocity results and plotted in Figure 5-9. It can be seen that the velocities computed in cross sections C, D and E were in close agreement with those measured.

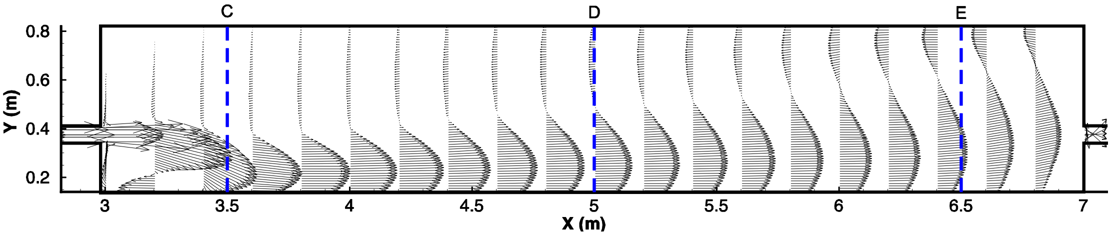


Figure 5-8 Flow field in the lake

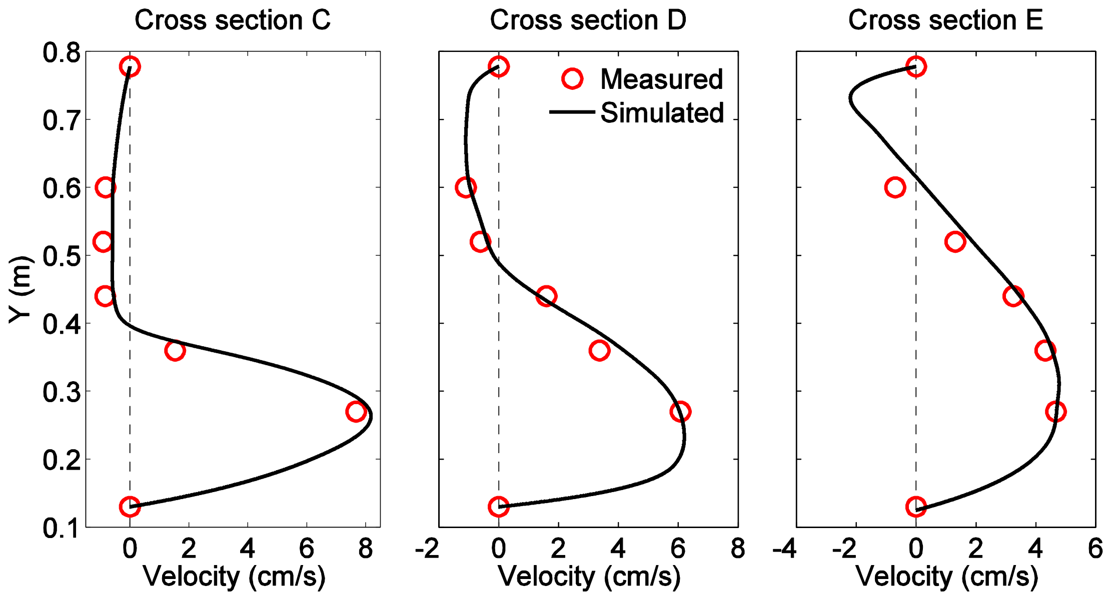
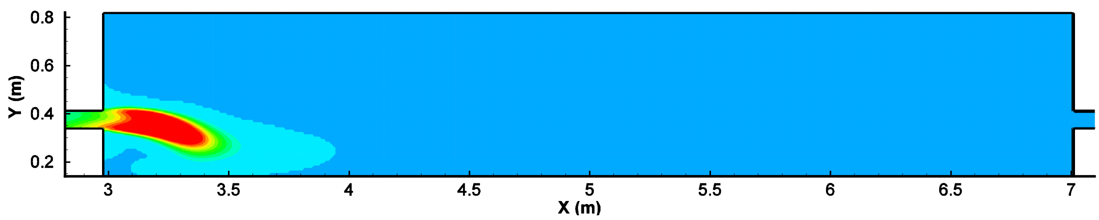


Figure 5-9 Predicted and measured cross-section profiles of the mean flow velocity

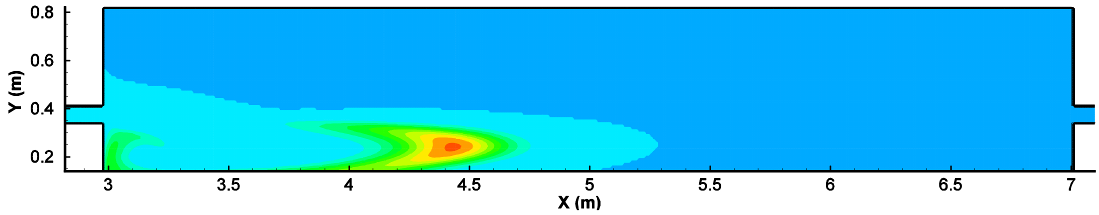
Numerical model simulated results of flow augmentation are displayed in graphical form by displaying concentration values as contours (see Figure 5-10). The tracer distributions at 15s, 30s, 60s, 100s, 160s, 320s, and 440s were presented. It can be seen that a preferential flow path was formed between the inlet and outlet sections, which occupied roughly 40% of the basin space. This region could be associated with the quickest flushing of solutes from the lake. Recirculating flow and dead zones were formed in the remaining space, where overall the flow speed as low and with little exchange of any solutes present therein with the preferential flow path. A sketch of flow and tracer distributions is presented in Figure 5-11 to make the numerical results easily understandable. According to the results discussed in Figure 4-15b in Chapter 4, the relative importance of these two main flow regions, in terms of the solute transport processes in the simulated lake, varied little between tests.

Both the experimental and numerical modelling results indicates that recharge alone had little impact on the overall mixing level in the lake waters due to poor cross-sectional flow distribution. This could lead to localised water quality problems in stagnant and circulating flow regions, despite the interventions, and could impact, for instance, water supply and fisheries productivity. Therefore, additional measures to flow augmentation may need to be considered to achieve the goal of enhancing the overall water mixing in the lake.

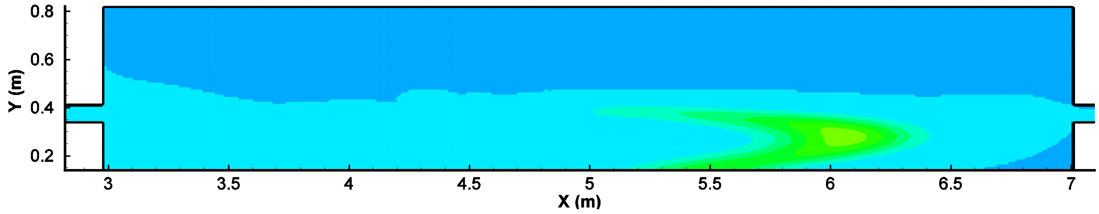
15s



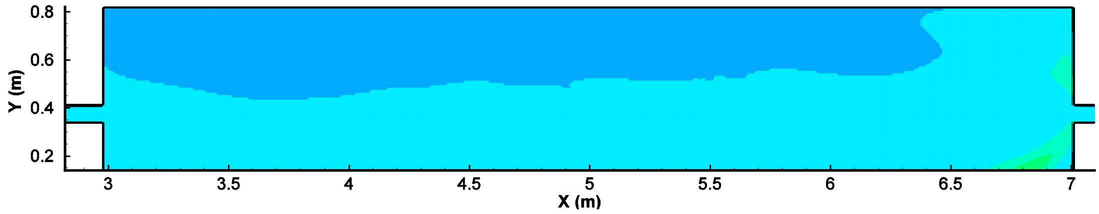
30s



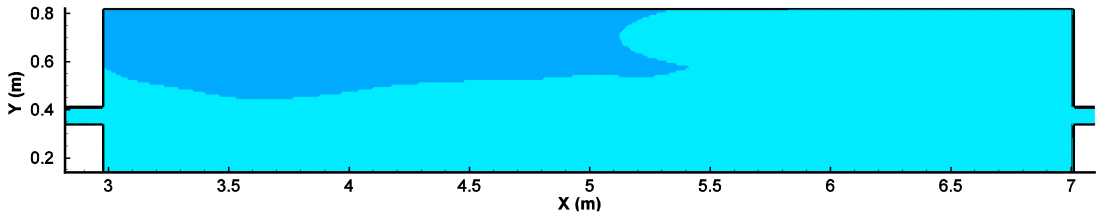
60s



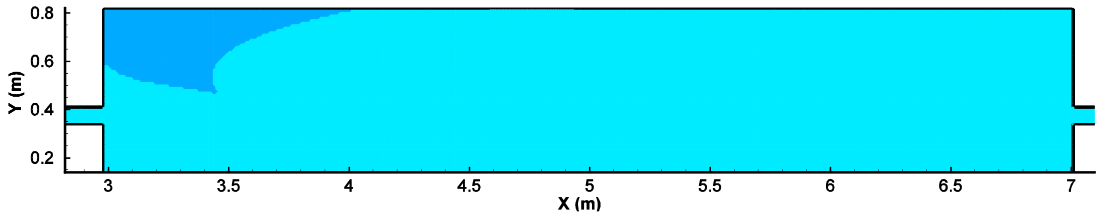
100s



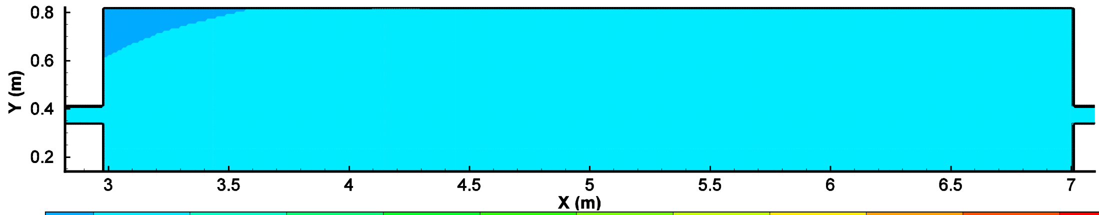
160s



320s



440s



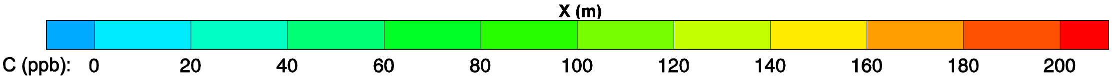


Figure 5-10 Time series of simulated flushing flow over a discharge of 12.30l/s

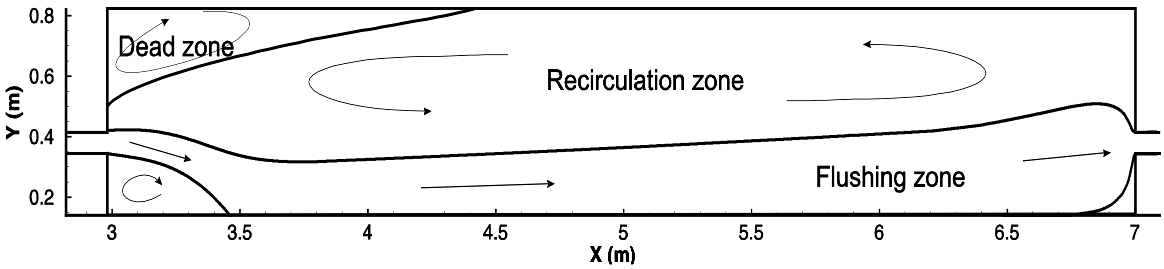


Figure 5-11 Sketch of the flow distribution in the lake

5.7 Lake Flushing Simulation with Flow Deflector

Generally speaking, the flushing of a water body is achieved through transport mechanisms which promote water removal, such as tidal currents, river contributions and the density gradient induced circulation, meteorological events and the topographical configuration (Wang et al., 2004). In this study, due to poor cross-sectional flow distribution, alternative operations of flow structures could be obtained through the application of hypothetical scenarios to improve the water mixing processes affecting lake water quality. The hypothetical scenario simulation with flow deflectors; the aim of which is to deflect incoming flow to low water mixing zones, will be discussed in this section.

5.7.1 Flow Deflector Design

With the objective of deflecting incoming flow to low water mixing zones, two types of deflector were simulated near the lake inlet, as shown in Figure 5-12. Figure 5-12a shows a 38cm long deflector while the other type is combined with 6 small deflectors (see Figure 5-12b). Simulations with these two types of flow deflector were performed using DIVAST. The simulated results at 10s are presented and compared in Figure 5-13. It can be seen that a low water mixing zone was formed in the back of the 38cm deflector while the flow shown in Figure 5-13b was evenly distributed in the vicinity of the lake inlet. It was concluded that, compared with the predicted results for one flow deflector, 6 smaller deflectors positively affected the distribution of flow, by mitigating the occurrence of dead zones.

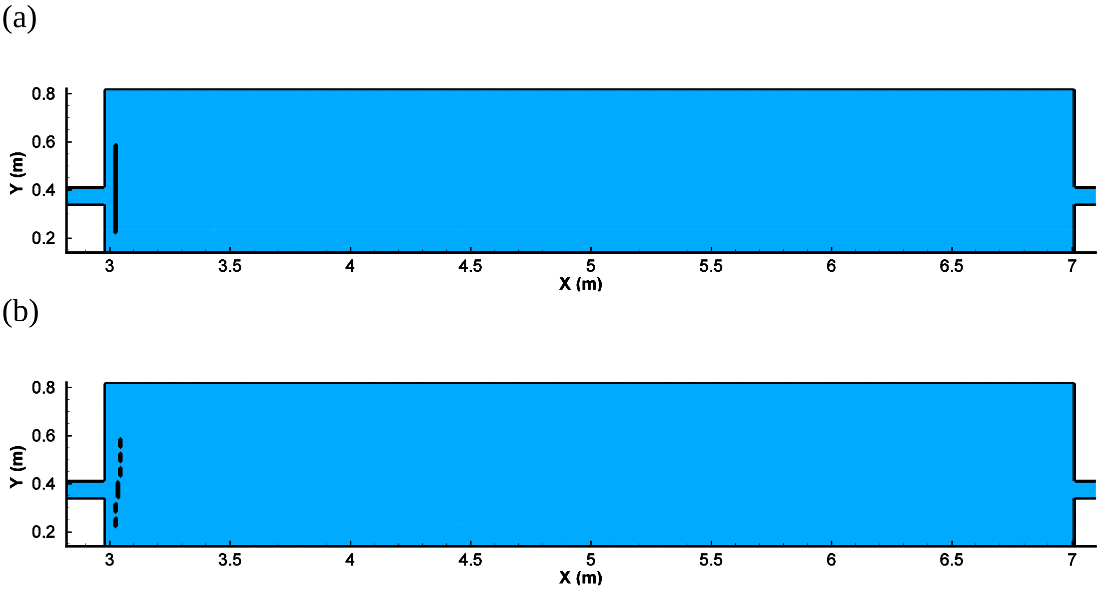


Figure 5-12 Flow deflector design

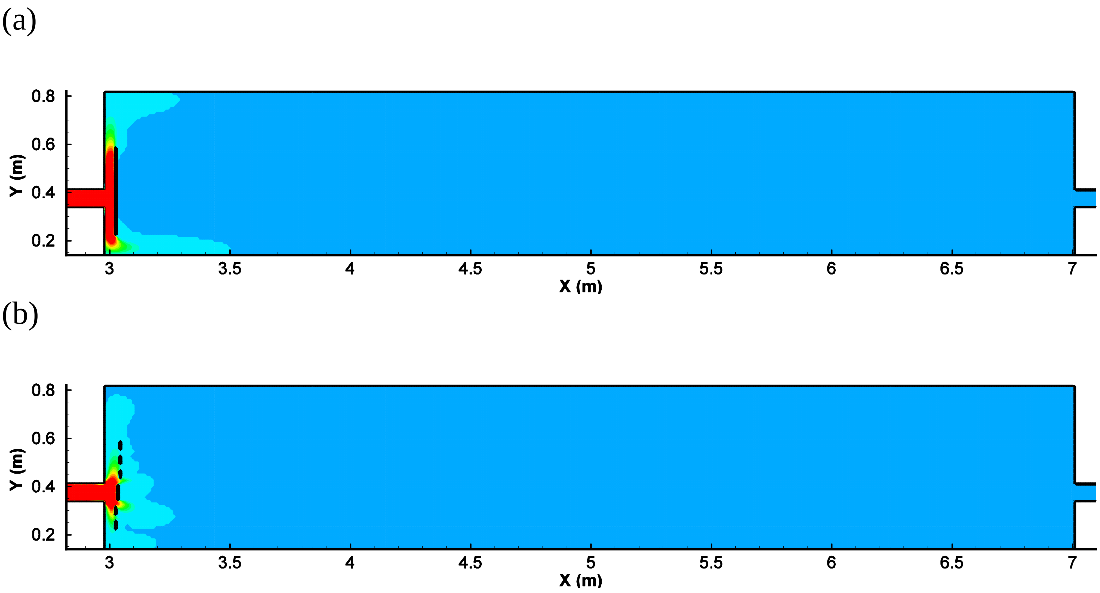


Figure 5-13 Simulated flow distribution at 10s with (a) one deflector (b) 6 small deflectors

5.7.2 Simulation Results with Flow Deflectors

Since the flow deflectors design could positively affect the distribution of flushing flow, simulations for the lake area with flow deflectors design were run. The simulation time for the model was 60mins. The numerical simulated results were displayed in graphical form by displaying the flow fileds as vectors (see

Figure 5-14) and concentration values as contours (see Figure 5-15). Flushing flow distributions at 15s, 30s, 100s, 160s and 240s are presented respectively. It is evident that water in the lake was more effectively mixed with the flow deflector design compared with the simulated result in Figure 5-10.

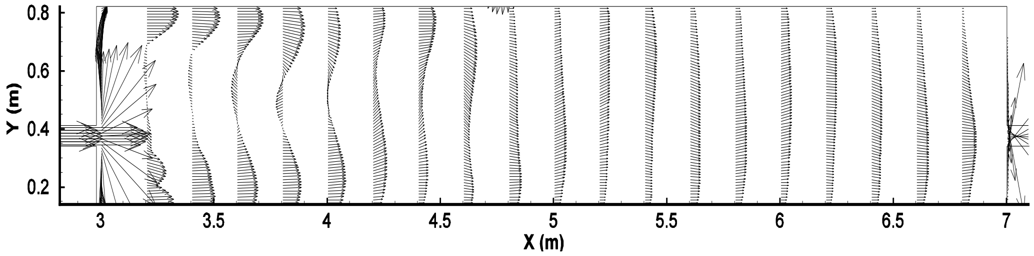
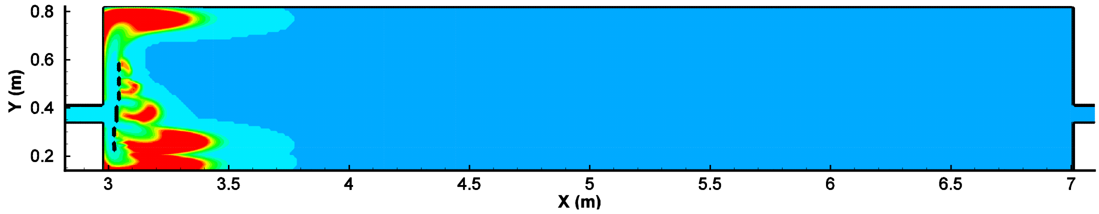
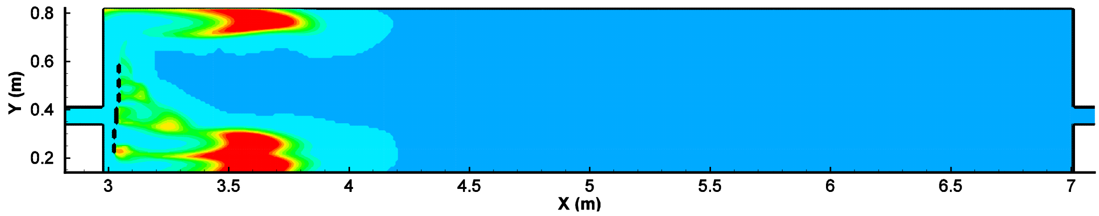


Figure 5-14 Simulation flow filed with flow deflector

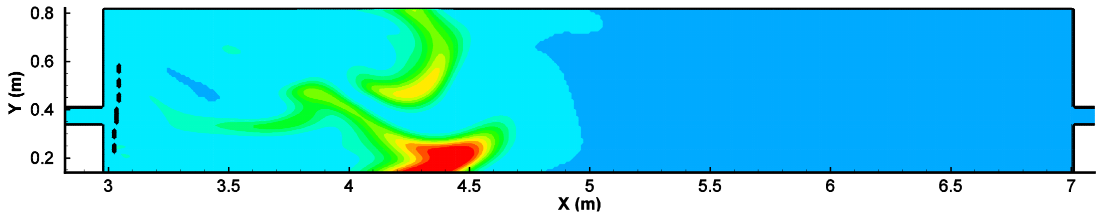
15s



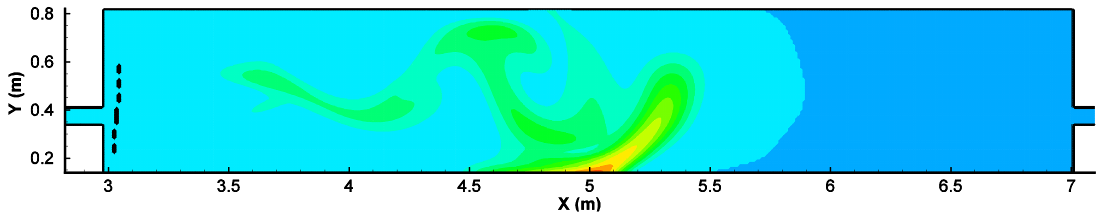
30s



60s



100s



160s

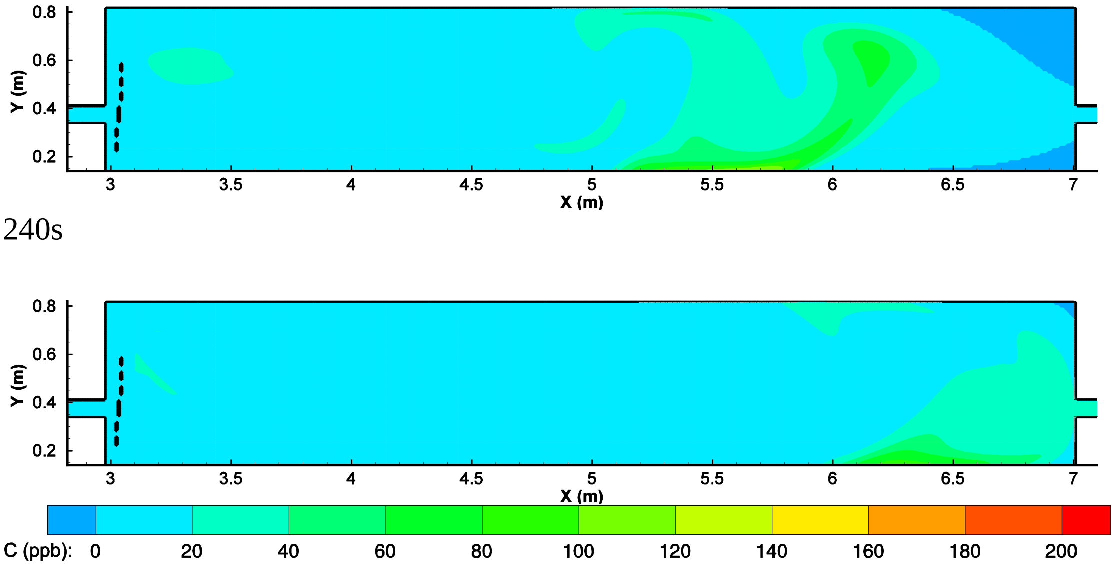


Figure 5-15 Time series of predicted flow distribution with flow deflectors

The analysis of both numerical modelling and experiments for the lake area presents a need to improve the water flushing of low water exchange region in the lake. The flushing refers to the replacement of water in the lake with water from the connected river. However, flow augmentation tests and simulation results showed that recharge alone had little impact on the overall mixing level in the lake waters due to poor cross-sectional flow distribution. To achieve the goal of enhancing water flushing in a recharged lake, the flow deflectors combined with flow augmentation were found to positively affect the distribution of flushing flow, by mitigating the occurrence of dead zones. Overall, water mixing level in the lake was enhanced under the inflow diversion by the flow deflectors and therefore improved the overall water mixing in the lake.

5.8 Summary

In this chapter, a 2D hydrodynamic and water quality model, DIVAST, was applied to investigate in more detail the water flushing in the river-lake system. The model calibration and validation of the model DIVAST were initially conducted against water velocity and tracer concentration data obtained from experiments. It was found that an eddy viscosity value of 0.15, longitudinal dispersion coefficient value of $0.5m^2s^{-1}$ and partial slip boundary condition coefficient value of 0.9 gave the closest agreement between tracer passage predictions and measurements. The validated model was then used to perform water diversion modelling and flow augmentation modelling.

As supported by numerical and experimental investigations, a better understanding of the hydraulic and water quality processes in the river-lake system model was developed. For instance, based on the experimentally determined RTD curves, it was found that overall flushing improved with increasing discharges, as indicated by the corresponding reduction of the mean detention time. Having said this, mixing levels varied little in the basin, thus potentially still causing localised and relatively long detention times and water quality issues. The numerical modelling result coincided with the experimental results analysis. Modelling results indicate that a preferential flow path was formed between the inlet and outlet sections whilst recirculating flow and dead zones were formed in the remaining space. Hydraulic efficiency in the lake varied little as the flow increased. Recharge alone had little impact on the overall mixing level in the lake waters due to poor cross-sectional flow distribution. Addressing this issue, the simulations with flow deflector near the lake inlet were carried out. It was concluded that flow deflectors combined with flow augmentation could positively affect the distribution of flushing flow, by mitigating the occurrence of dead zones.

CHAPTER 6 MODELLING HABITAT SUITABILITY FOR FISH IN THE FLUVIAL AND LACUSTRINE REGIONS

6.1 Introduction

Over the past few years, linking physical habitat conditions in rivers to their ecological characteristics has become a fundamental requirement in river management and river restoration. Due to excessive pollutant discharge and inappropriate regulation of water resources, the structure, function and health of many freshwater ecosystems have been degraded to the point that they can no longer support biodiversity (Wang et al, 2012). The use of combined numerical models is becoming a very effective tool with which to study the ecosystem habitat requirement. The most promising aspect of numerical 2D models in ecological flow studies is their potential to accurately and explicitly quantify spatial variations and combinations of flow patterns important to stream habitats (Bovee, 1996). Many researchers have combined hydrodynamic and ecological models to determine the relationship between stream flow and physical habitat suitability for an aquatic species. Habitat suitability curves have been used to characterise aquatic species' habitat preference, availability and quality. For example, Nagaya et al. (2008) used a horizontal 2D numerical model to predict the environmental flow requirement of ayu (Plecoglossus altivelis) with the preference curves of the flow depth and velocity. Yi et al. (2010) developed a mathematical model to predict the minimum in-stream flow and suitable daily discharge during the reproduction season for the carp species in the Yangtze River. In this study the habitat suitability curves were coupled with the mathematical model.

The habitat suitability in a stream varies seasonally and annually according to the river discharge. Hughes & Louw (2010) pointed out that accurate hydrological and hydraulic data, together with a sound understanding of the ecosystem dynamics based on field surveys over extended periods of time would help to generate high confidence related to aquatic species' habitat assessment. It is therefore important to study the habitat suitability in relation to the natural flow pattern and the species bioperiods. Many researchers have emphasised the necessity to account for flow and habitat variability in predicting the distribution and performance of different biota in streams (Poff et al., 1997; Arthington et al., 2006). However, most existing methods do not account for the different bioperiods habitat requirements of the target species and the seasonal variability of the study area. Therefore, the main aims of this chapter are to link the hydrodynamic characteristics of the water system to the fish bioperiods and to determine the levels of habitat suitability in the Eco-City using an integrated eco-hydraulic model. This eco-hydraulic model combines a 2D hydrodynamic and water quality model (Falconer & Lin, 2005) with a HSI model. Habitat suitability curves, as a component of the HSI model, have been developed and used to determine which type of habitat is preferred by a single species. Both water quantity (depth, velocity) and quality (dissolved oxygen) parameters are considered. To reduce the poorly and moderately suitable areas, the influence of different interventions on the improvement of habitat suitability conditions, such as dissolved oxygen (DO) enhancement device and flow augmentation, have also been investigated.

6.2 Target species in the Jiyun River

To create suitable physical habitats for fish in an urban river, target species which are predominant in the river should be reviewed, whilst their ecological characteristics should be carefully examined first (Lee et al., 2010). Grass carp (*Ctenopharyngodon idella*), silver carp (*Hypophthalmichthys molitrix*), bighead carp (*Hypophthalmichthys nobilis*) and black carp (*Mylopharyngodon piceus*) are known as the "Four Domesticated Fish" in China (see Figure 6-1). Indeed, these four fish species were selected as the target species as they represent the major fish population of the Jiyun River. These four fish species belong to the family Cyprinidae with similar life bioperiods and could be collectively labelled as carp. These fish are native to Central Asia, but have been introduced to many regions of the world including Europe, North America, the Middle East, Canada and Australia. The bioperiods of carp, during which the management of river flow and habitat conditions is of particular importance, are determined by evaluating the seasonal needs of the target species. The life pattern of carp could be divided into three stages: spawning stage (March-June), growth stage (July-November), and overwintering stage (December- February).

Spawning activity is associated with high spring flows from March to June (Verigin et al., 1978). Carp produce eggs which are semi-buoyant and require current to keep them from sinking to the bottom. The sticky eggs are deposited onto submerged vegetation and hatch in less than a week. If carp spawn in a river of sufficient length with a current of 0.2 m s^{-1} or faster, hatching of eggs could occur. The optimal velocity for spawning is from 0.4 to 0.8 m s^{-1} and water temperature in the range of 18-30 ℃ (Kolar et al., 2007). In reservoirs, rising water levels may provide access to terrestrial vegetation, which is a good substrate for spawning. In lakes, peak spawning occurs during the 4 or 5 days when water levels fluctuate only slightly (<7 cm) or increase rapidly following a level period.

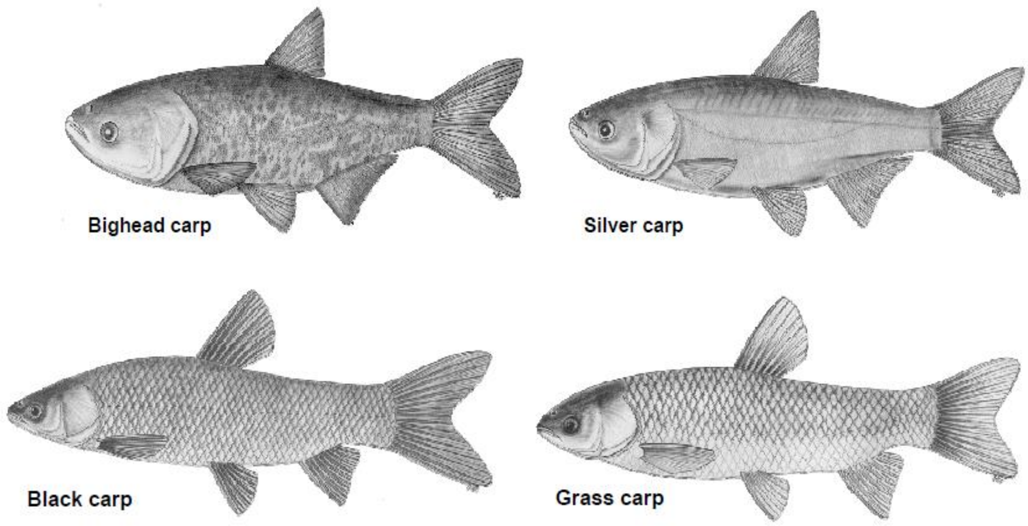


Figure 6-1 Four domesticated fish in Jiyun River

The growth stage for these fish is from July to November. The embryonic development of carp takes approximately 3 days at 20-23℃. Under natural conditions, hatched fry stick to the vegetation. After hatching, the fry remain in shallow (< 2 m), warm, fertile, sluggish waters for 2 to 8 weeks. Around three days after hatching the posterior part of the swim bladder develops, meaning that the larvae swim horizontally and start to consume external food with a maximum size of 150-180 µm. The young carp grow quickly in warm plankton rich water (McCrimmon, 1968). Dense vegetation is also required by fry and juveniles for cover. In stable streams, the water velocity in the plant-free area of a river appears to control the maximum vegetation abundance in the stream. Riis & Biggs (2003) calculated the percentage area of stream cross-sections occupied by vegetation along with the mean velocity in the plant-free area across the channel. Juveniles and adults were found in deeper waters feeding predominantly on aquatic plants, algae and small invertebrates near the bottom. In both riverine and lacustrine habitats, carp prefer enriched, relatively shallow, warm, sluggish, and well-vegetated waters with a mud or silt substrate. Adults spend summer and early autumn in shallow areas of dense vegetation.

As temperature drops from December, the carp move into deeper waters (Jester & Moody, 1969). Winter is commonly regarded as a critical period for fish to survive in rivers. To survive in low temperature and with declining food resources, carp have adapted and evolved varying overwintering strategies, e.g. residency or migration to alternative habitats. Winter survival is mainly influenced by factors other than food abundance .Water depth and DO are considered to be the crucial parameters (Lukowicz & Gerstner, 1998). A weight loss (WL) of 5-10% is considered acceptable for successful overwintering.

6.3 Methodology

6.3.1 Habitat Suitability Index Model

The Habitat Suitability Index (HSI) scoring system was originally developed by the US Fish and Wildlife Service, as a means of evaluating habitat quality and quantity, following which it was then used by Oldham et al. (2000) to evaluate the habitat suitability level of great crested newt. The HSI is a numerical index which varies between zero and one, thus indicating unsuitable to optimal habitat conditions. The overall HSI value is calculated as the geometric mean of the related suitability indices using the following equation:

$$HSI = \left(SI_1 \times SI_2 \ldots SI_n\right)^{1/n} \quad (6\text{-}1)$$

where $SI_1, SI_2,\ SI_n$ are the factors converted to Suitability Index scores, on a scale from 0.01 to 1 (0.01 is used as the lower end of the scale instead of 0). The HSI scoring system shown in Table 6-1 was adopted herein to define the habitat suitability of carp on a categorical scale.

Table 6-1 Categorisation of HSI scores

HSI	Suitability
< 0.5	Poor
0.5 – 0.8	Potential
> 0.8	Ideal

Previous studies of habitat distribution have found that different habitats had different optima. The HSI for fish incorporates several suitability indices, all of which are factors affecting the living conditions of fish, such as velocity, flow depth, pH, cover, water temperature, and DO. In this study, certain important indices were selected. These indices are known to affect carp's habitat based on field data for the River Jiyun and information found in the scientific literature: velocity, depth and DO. As carp have different water requirements during their life stages, which should be taken into account when evaluating the water allocation needs to the river and lake, an HSI system was created based on the literature describing the habitat suitability for three life stages of carp (see Table 6-2). Local expert fish knowledge was also consulted in designing this system so as to account for differences between the site-specific situation and the general conditions found in the literature. The basic form for the expression of suitability is a habitat suitability curve, as shown in Figure 6-2.

Table 6-2 Suitability Indexes for the three life stages

Stage	Index	Note
Spawning (March-June)	Velocity	The minimum velocity to support eggs is 0.2 m s^{-1}. The optimal velocity is 0.4-0.8 m s^{-1} (Liu & He, 1992).
	Depth	Preferred spawning areas are over aquatic or inundated terrestrial vegetation, at depths of 0.5-0.8 m (Edwards &Twomey, 1982).
	DO	Percentage hatching increases with increasing DO content. At 3 mg l^{-1} DO, 40% of the embryos hatched; at 6 mg l^{-1}, 65% hatched; and at 9 mg l^{-1}, 92% hatched (Kaur & Toor, 1978).
Growth (July-Nov)	Velocity	Carp occurred in pools (0.2-0.6 m s^{-1}) and in the main channel borders (0.6-1.2 m s^{-1}), but are most abundant in marshes and backwaters (<0.2 m s^{-1}) (Kallemeyn & Novotny, 1977).
	Depth	The maximum depth for spawning is included because carp primarily frequent shallow waters.
	DO	Optimal DO level for adults is assumed to be ≥ 6 mg·l^{-1} (Edwards & Twomey, 1982)
Over-wintering (Dec-Feb)	Depth	As the temperature drops, carp move into deeper waters. Hence, deep water areas are necessary to maintain carps in the winter, with a minimum depth of 0.4m.
	DO	Optimal DO level for this period is assumed to be ≥ 6 mg l^{-1} (Edwards & Twomey, 1982)

As shown in Figure 6-2(a), the spawning suitability is represented by three indices: velocity, DO and depth. The minimum velocity required to support eggs is 0.2 m s^{-1}, the optimal velocity is 0.4-0.8 m s^{-1} and the suitability index then decreases with further velocity increase. If the velocity is too high then the fish population will reduce, caused by scouring eggs, altering fish habitat and washing adult fish away from their spawning habitat (Naghibi & Lence, 2012). The ranges of velocity, DO and depth which are associated with carp growth are illustrated in Figure 6-2(b). Very little documentation exists on the actual overwintering habitat of carp, although from fish farming it is known that adults prefer to spend the winter in deeper water with optimum depth ranging from 1.0 to1.5 m (Lukowicz & Gerstner, 1998). DO is used as a major indicator of water quality and habitat condition because it affects survival and feeding. Herein, the water depth and DO are considered to be the crucial parameters for carp's overwintering habitat, and the corresponding suitability curves for carp are shown in Figure 6-2(c).

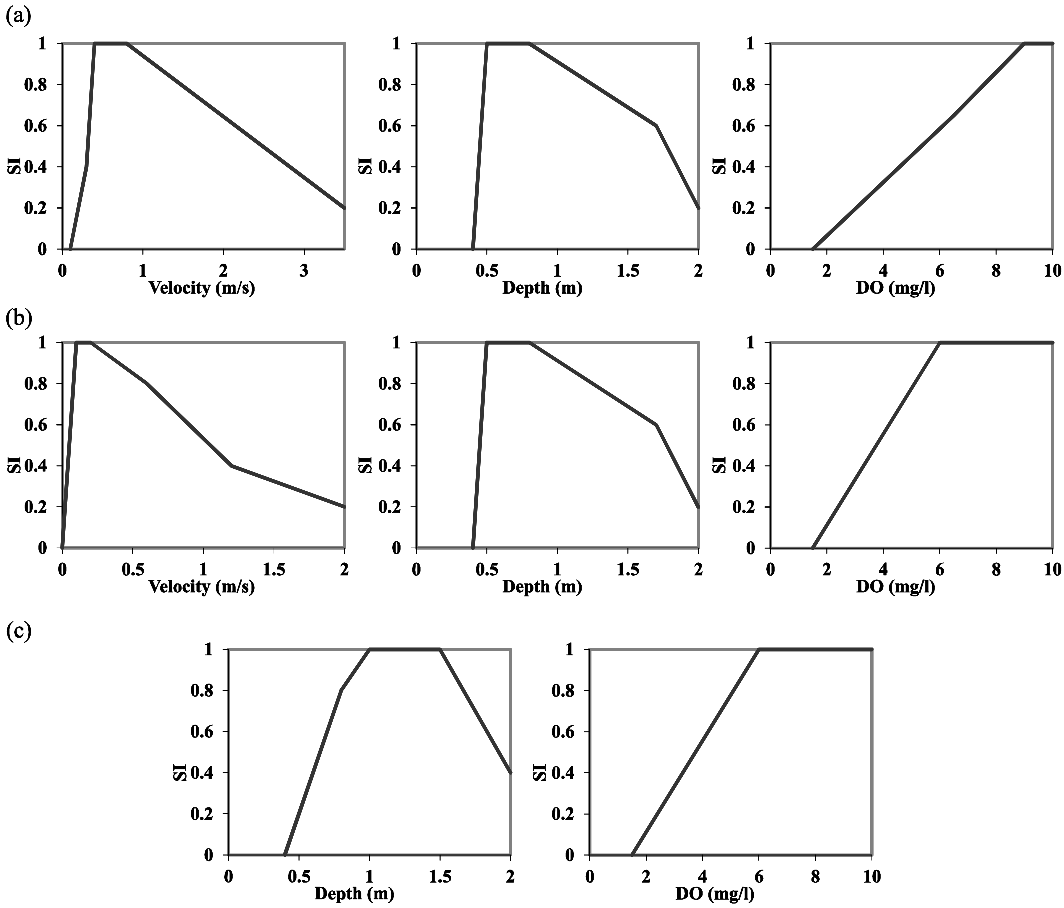


Figure 6-2 Suitability Index (SI) curves for three life stages of carp, (a) spawning; (b) growth; and (c) overwintering

6.3.2 Eco-hydraulic Model

The eco-hydraulic model used in this study combined the 2D hydrodynamic and water quality model discussed in Chapter 5 with the HSI model. Habitat suitability curves, as a component of the HSI model, were developed and used to determine what type of habitat a single species prefers. The governing equations for the hydrodynamic and solute transport processes were discussed in Chapter 5 (Equation 5-1 to 5-3). At the present time, the primary water quality problem that the Jiyun River faces is DO depression in the water column due to excessive pollutant discharge. Oxygen consumption materials, such as aerobic biological organisms and ammonia nitrogen from domestic wastewaters and industrial effluents, are the main sources of river pollution. In the current study, the oxygen dynamics was modelled using the following equation (Stefan & Fang, 1994):

$$\frac{\mathrm{d}C_{DO}}{\mathrm{dt}} = P(t) + k\left(C_S - C_{DO}\right) - S_{BOD} - S_{SOD} - R \qquad (6\text{-}2)$$

The photosynthesis P (t), as a function of time, can be approximated as a half sine wave during daylight hours and zero at night (Chapra & Di Toro, 1991):

$$P(t) = P_{\max}\sin(\frac{\pi t}{f}) \qquad 0 \le \mathrm{t} \le \mathrm{f}$$

$$P(t) = 0 \qquad \mathrm{f} \le \mathrm{t} \le \tau \qquad (6\text{-}3)$$

where C_{DO} = DO concentration; $P_{\max}$ = maximum photosynthesis rate (50 mg l^{-1} day^{-1}) (Portielje & Lijklema 1995); f = the photo-period (12 h), and τ = the diurnal period (24 h); k = reaeration rate constant; C_S = saturated DO concentration. The sources for oxygen are photosynthesis rate $P(t)$ and physical surface reaeration rate $k(C_S - C_{DO})$. The oxygen saturation concentration has a constant component, namely Henry's constant for oxygen (see Chapra, 1997 for details), and a variable component, which is dependent on the pressure of oxygen in the atmosphere, the salinity and the temperature of the water. It decreases with increasing elevation above water level and with increasing temperature and salinity. Reaeration is referred to as the transfer of oxygen between the atmosphere and the oxygen unsaturated water. It is proportional to the saturation deficit. The negative terms S_{BOD} , S_{SOD} , and R in the oxygen model are the losses due to nitrification, which is proportional to biochemical oxygen demand (BOD), the sediment oxygen demand (SOD) and the plant respiration in the water respectively. Based on the water quality report for the Jiyun River in 2006, the initial BOD on the spawning, growth and overwintering stages was set to 67mg l^{-1}, 78 mg l^{-1}, and 60 mg l^{-1} respectively. The SOD is dependent on the water temperature and depth (Brown & Barnwell, 1987). As the aerobic degradation of organic material proceeds, DO is depleted, causing an oxygen deficit in the water.

The calculation domain measures 10,510 m in length by 3,510 m in width and is shown in Figure 6-3. The upstream boundary condition was a measured flow rate, while the downstream boundary condition was given in the form of water elevation. The model was used to predict flow velocities and water depths under different flow rates. The size of the computational grid was uniform and it was 10 m in both x and y directions. The time step was 2 s. The model simulation time was typically 50 hours for each scenario. The 2D hydrodynamic and water quality model was combined with the HSI model. Grid values of each of the indexes were combined with the suitability curve information and thus the overall HSI from the linked eco-hydraulic model was obtained for each scenario.

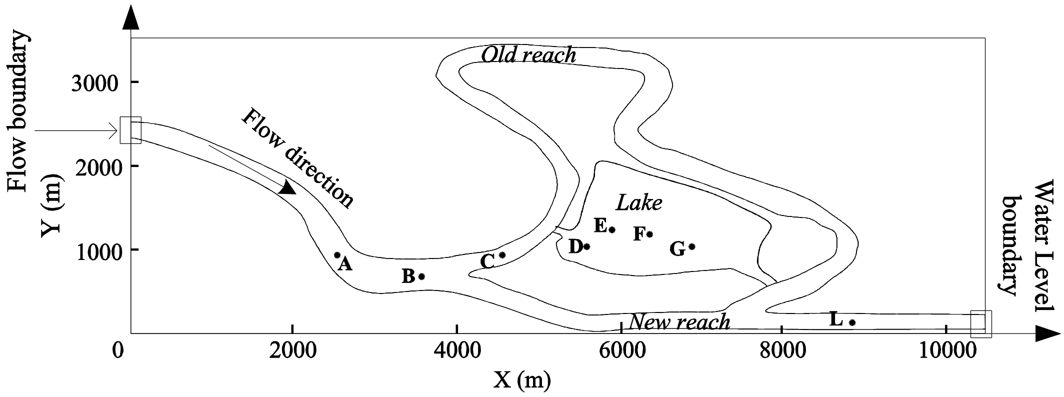


Figure 6-3 Simulation domain and boundary conditions

6.4 Results

6.4.1 Hydrological Data Analysis

The river discharge was found to have a major effect on the inter-annual variation in fish abundances (Carassou et al., 2011). Many water-related problems, including the shortage of water for ecosystems, occur during the annual dry season or, a prolonged period of drought. In this study efforts were made to incorporate the ecological water requirement with the natural flow pattern, which can be derived from existing hydrological data. Mao & Li (2009) investigated the recurrence characteristics of wet and dry years in the Jiyun River network, and found that the number of years with an average precipitation was greater than the numbers of wet and dry years.

In this study, 10 years of hydrologic data (2001-2010) recorded at Fangchaozha gauging station was analysed and the year 2006 was found to be a typical year of average rainfall. Based on the daily averaged flow rates obtained from the gauging station, the annual hydrograph was computed. In addition to flow data, monthly rainfall data were also used, as shown in Figure 6-4. Both the daily discharge and monthly rainfall in Figure 6-4 showed significant variability within a year. In 2006, the annual average flow rate in the Jiyun River was 39.45 m^3 s^{-1}.

The seasonality of precipitation and distribution of rainfall have a major influence on the river discharge, indicated by the high flow events during the months of June to August with the maximum flow rate of 107 m s^{-1}. The river's flow rate during the winter is the lowest, with a minimum flow rate of 2 $m^3 s^{-1}$. The temporal variation of flow regime can be investigated further through the Flow Duration Curve (FDC) analysis, as illustrated in Figure 6-5, which is a cumulative frequency curve showing the percentage of time during which specified discharges were reached or exceeded for a given period. From the FDC, the natural flow rates were evaluated: the low flow (Q90=13.81$m^3 s^{-1}$, 90% exceedance), average flow (Q50=35.11$m^3 s^{-1}$, 50% exceedance) and high flow (Q10=71$m^3 s^{-1}$, 10% exceedance). In the current study they were used to represent the discharges for the three stages of the carp life cycle including overwintering, spawning, and growth.

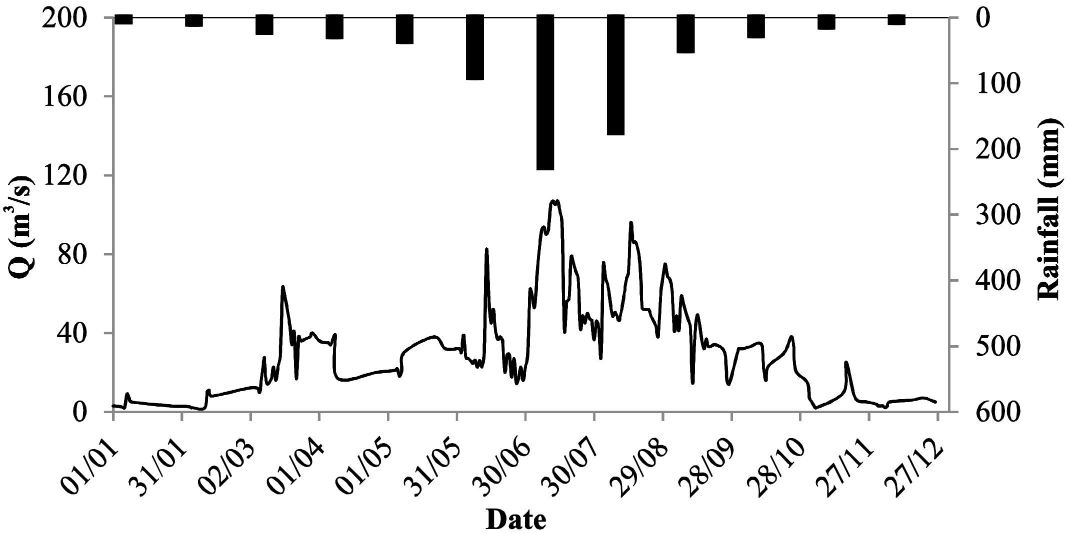


Figure 6-4 Variability of daily discharge and monthly rainfall at Fangchaozha hydrological station in year 2006

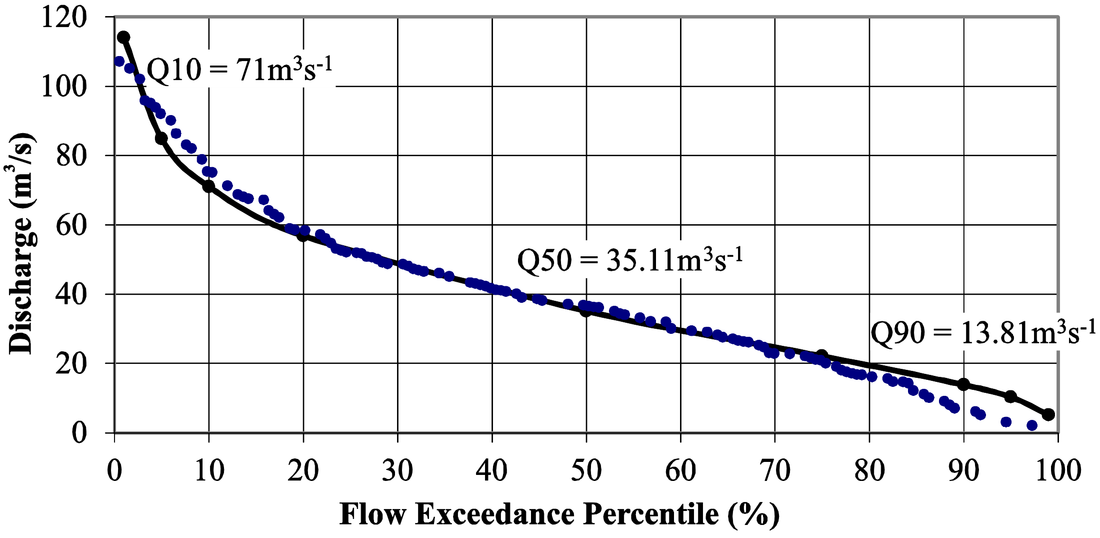


Figure 6-5 Flow duration curve for Jiyun River in 2006

6.4.2 Modelling Results

The HSI distribution at the spawning stage in the study area was determined by combining the suitability index values for the water depth, flow velocity and DO using Equation 1. For the growth and overwintering stages, the velocity was not included in calculating the HSI for the lacustrine region as evidence suggested that it was less important for species abundance in the lacustrine region (Reiser & White, 1983). Carps occur in rivers and estuaries but are most abundant in reservoirs, lakes, and farm ponds with slow moving water (≤0.2 m s^{-1}). Numerical model predicted values of each of the parameters were combined with the preference curves. The suitable areas were then estimated cell by cell to determine the HSI distributions for the whole reach. Figure 6-6 shows the predicted flow field using the integrated model whilst the HSI distributions obtained for the spawning, growth and overwintering stages are plotted in Figure 6-7.

As Figure 6-7 shows, in a large part of the study area the HSI values were less than ideal for carp to settle in the spawning and overwintering stages while nearly half of the area met the ideal conditions for the growth stage. For the spawning stage (Q = 35.11 $m^3 s^{-1}$), only 8.6% of the study area had the ideal condition in the current river state, whilst an additional 14.1% had the potential for fish habitat to develop while the remaining 77.3% was poorly suited. The upstream river reach and the lake inlet area were found to be generally more suitable for the carp life cycle while the downstream of the river and most of the lake were less than ideal, as the HSI value ranged from 0.2 to 0.6 (see Figure 6-7(a)). For the growth stage (Q = 71$m^3 s^{-1}$), 44.5% of the area met the ideal condition, whilst an additional 24.7% had the potential for carp habitat to develop and the remaining 30.8% was poorly suited. The predicted HSI value in the fluvial area was generally above 0.7, thus indicating an ideal condition in the Jiyun River from July to November (see Figure 6-7(b)). As for the overwintering stage (Q = 13.81 $m^3 s^{-1}$), the area with ideal habitat conditions covered 24.4% of the total area, while 20.5% had the potential for development and 55.1% was poorly suited (see Figure 6-7(c)).

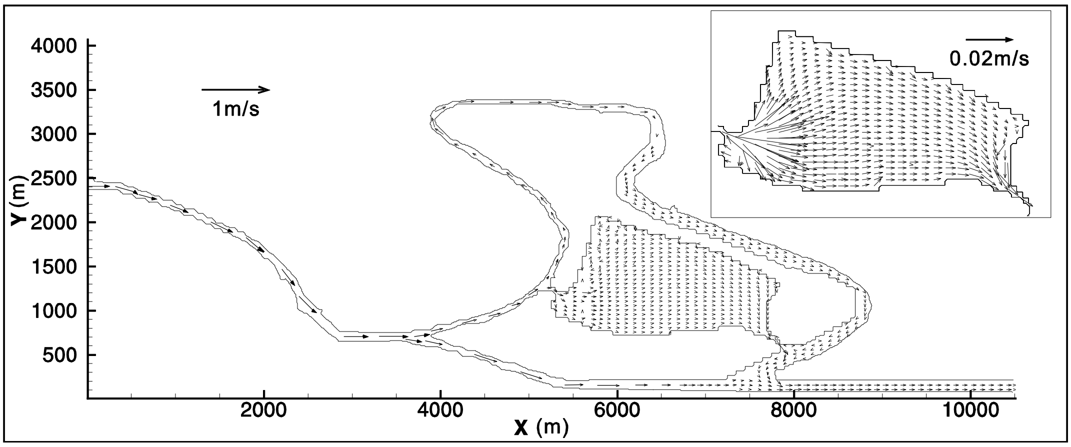


Figure 6-6 Model predicted depth averaged flow field in the study area.

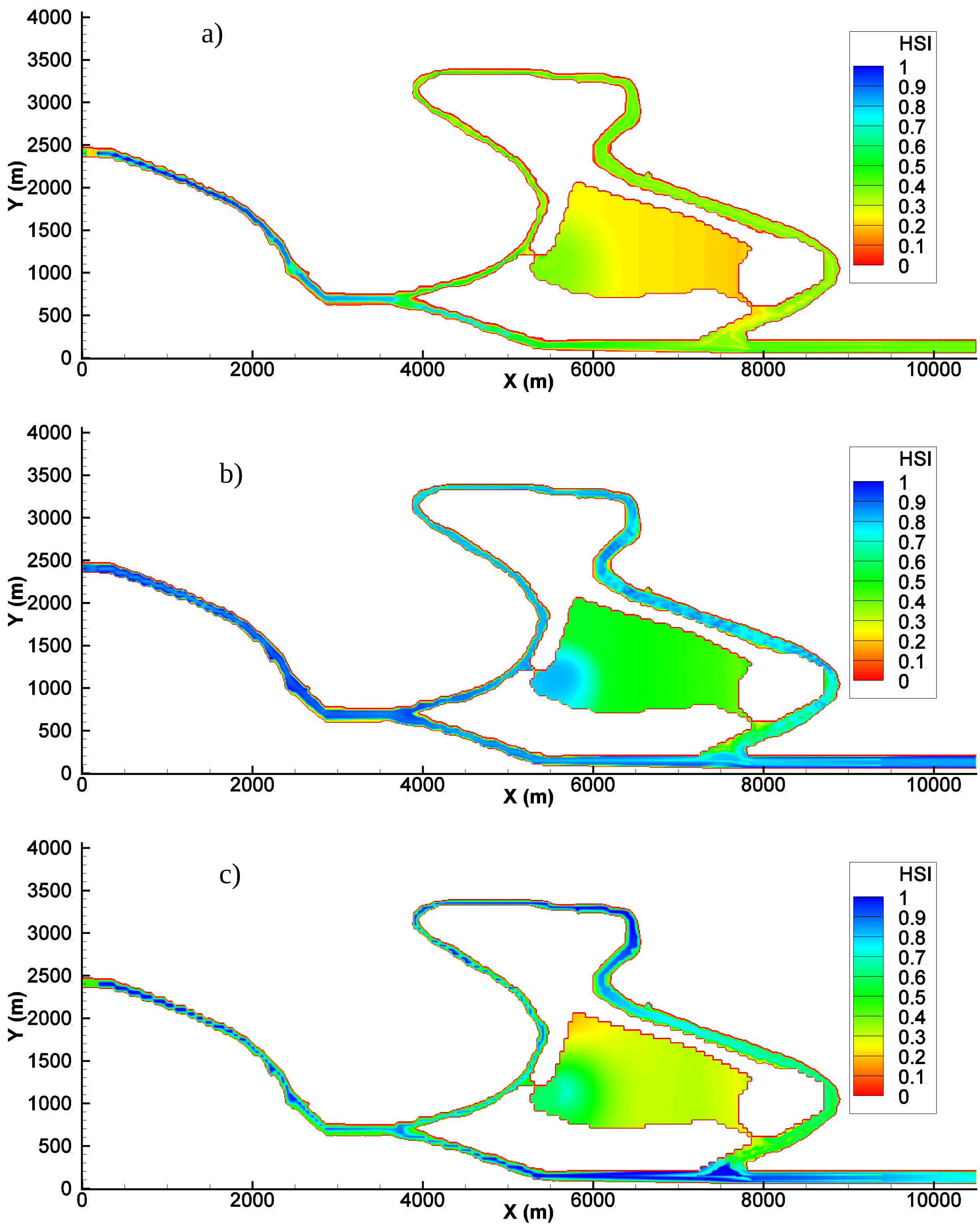


Figure 6-7 Predicted HSI under current flow conditions for the three life cycle stages of (a) Spawning stage Q=35.11 m^3 s^{-1} (b) Growth stage Q=71 m^3 s^{-1} (c) Overwintering stage Q=13.81 m^3 s^{-1}

6.5 Discussion

6.5.1 Habitat Suitability in the Fluvial and Lacustrine Regions

From the simulation results, it was noted that the lacustrine region would be poorly suited for carp to settle, with the spawning period being the most critical. In order to determine the main reasons attributed to the low HSI values in the lacustrine region, a more detailed analysis on each individual suitability index was undertaken.

At the spawning stage, as discussed in previous studies, the water velocity is the main factor for successful spawning since eggs need to be kept in suspension until hatching (Yi et al, 2010; Kolar, 2007). In this study, the predicted mean flow velocity in the lacustrine region was only approximately 0.05 m s^{-1} while the carps will generally not spawn in waters with an average velocity below 0.2 m s^{-1}. The lacustrine region is thus unable to provide carp with adequate spawning conditions. Chinese carp are known to be mainly riverine species and migrate into river habitats for spawning. As carp prefer slow-moving and warm plankton rich water for growth, the young carp swim to the lacustrine region and grow up there from July to November. As a consequence, lake connected rivers have significant benefits for the carp life cycle. The U.S. Geological Survey study identified that the Maumee and other rivers flowing into Lake Erie have the right conditions to serve as spawning sites for Asian carp (Murphy & Jackson, 2013).

For the growth and overwintering stages, the percentage of the predicted ideal water depth is approximately 70% while the HSI value of DO in the lake is generally below 0.5. The DO level, overriding the role of velocity and depth, appears to have a critical impact on the carp behaviour and the habitat suitability level in the lake at the growth and overwintering stages. The DO concentration in the river is more satisfactory because of the amount of photosynthesis and the fact that it is well-mixed. In the old and new reaches of Jiyun River, the DO level is more uniform with values largely ranging from 5 to 8 mg l^{-1}. However, the small flow velocity in the lake limits the potential for reaeration. Particularly in the winter, the predicted DO concentration is very low with the concentration value generally less than 2 mg l^{-1}. Based on the habitat suitability analysis in the fluvial and lacustrine regions, it is noted that,

(1) Carp spawning predominantly occurs in flowing fluvial regions. In the current study, the lake connected rivers would provide suitable spawning sites for carps. However, a combination of the low river discharge and high velocity requirement results in a poor suitability level for carps in the fluvial regions. The flow velocity could not be maintained due to the limited river discharge. The spring droughts in the study area caused by the lack of precipitation from March to June also affect the spawning of carps. Consequently, flow augmentation during the spawning season should be considered in order to improve the carp spawning conditions in the fluvial region.

(2) For the growth and overwintering stages, the habitat suitability conditions in the lacustrine region are less than satisfactory for carps to settle and DO depletion is the main reason attributed to the low HSI in the lacustrine region, particularly during the overwintering stage. The main factor, and the one has the greatest impact on the aquatic ecosystem in the lake is the DO concentration (Stefan & Fang, 1994). During the winter, respiration is likely to occur at a greater rate than photosynthesis. Photosynthesis still occurs, but typically at a much lower rate than other stages. It is, therefore, imperative to look for a way to improve the DO concentration which can be utilised to maintain the healthy ecological water environment in the lacustrine region.

6.5.2 Lacustrine Habitat Suitability Improvement

It is recognised that a low DO concentration can be stressful to aquatic organisms in the lake. Bottom hypoxia occurs when the rates of DO consumption exceed those of the DO supply (Lee & Lwiza, 2008) and lake managers are facing a challenge with regard to determining what to do to improve the DO level. The numerical model results also indicate that a low DO level is the main reason for the poor habitat suitability conditions in the lake, and thus a DO enhancement approach was considered to improve the lacustrine carp suitability. The DO concentration improvement by artificial aeration has been increasingly used in fish farming and even in some recreational waters. Tyler (1946) was the first to report the use of an in stream aerator to enhance the DO level in the Flambeau river in Wisconsin. In 2001, an aeration system and a telemetry control system were used to increase the DO level in Cardiff Bay, UK (ARUP, 2001). Many other studies demonstrated that supplemental aeration is an economic and effective means to improve the stream water quality (Landman & Heuvel, 2003). Alp & Melching (2011) showed that during some periods the DO standards were not met in the Chicago Water System (CWS), which was mainly used for commercial and recreational navigation and urban discharge. The DUFLOW model was then used to evaluate the effectiveness of employing aeration devices to increase the DO level. The results showed that four introduced aerators with oxygen supply capacities ranging from 30 to 80g s^{-1} would achieve 90% compliance with the 5 mg l^{-1} DO standard throughout the Chicago River.

In the current study, 6 hypothetical aerators with an oxygen transfer rate of 20 g s^{-1} were set up in the lake to increase the DO level. Results from the model simulations are presented in Figure 6-8. From this figure it can be seen that the predicted HSI values were significantly improved after the oxygen enhancement. A comparison between the predicted HSI distributions before and after the oxygen enhancement is shown in Table 6-3. The ideally suited area for carps has increased from 24.4% to 67.9% for the overwintering stage and from 44.5% to 77.5% for the growth stage, while the ideally suited area for carps to reproduce was only from 4.5% to 5.5%.

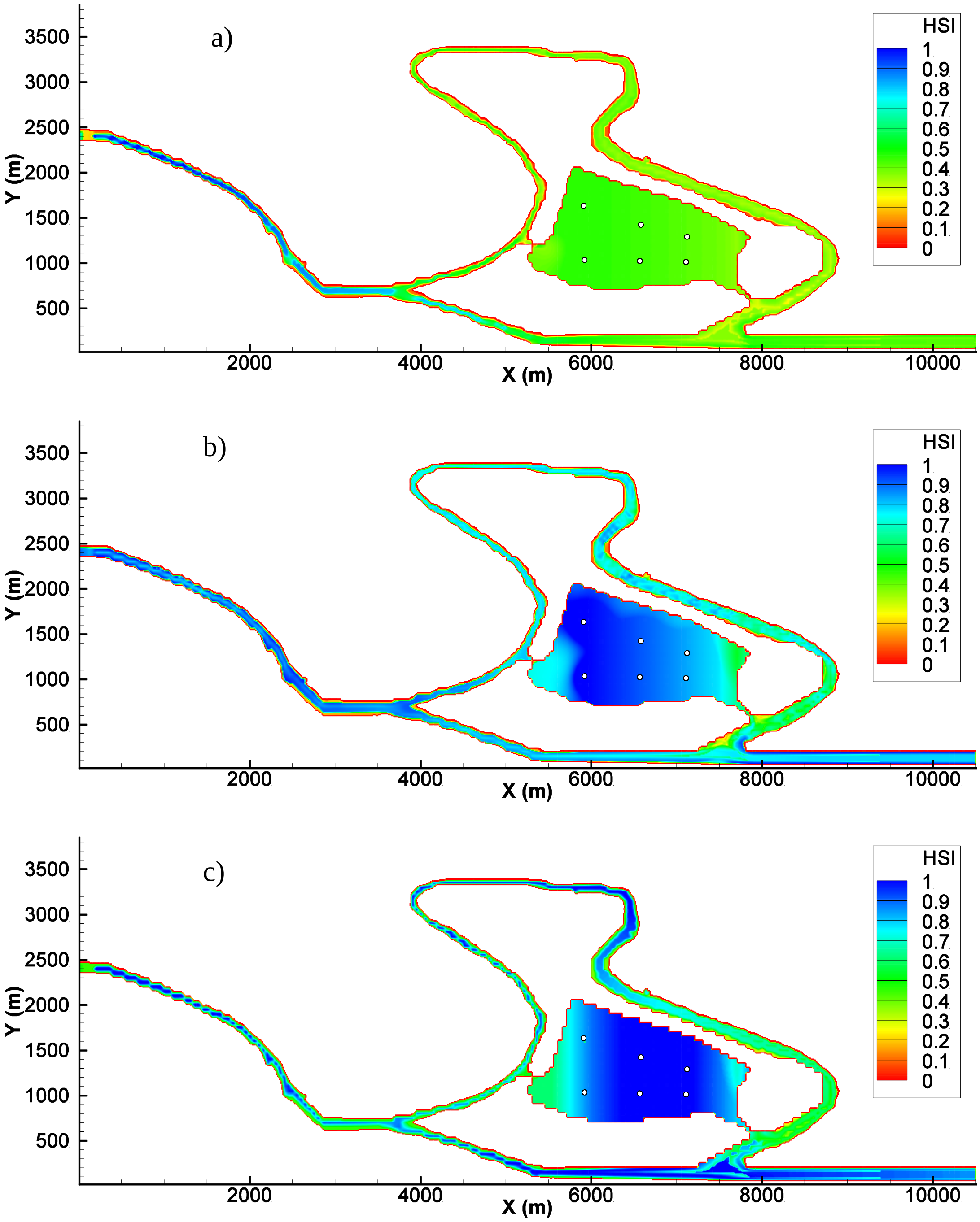


Figure 6-8 Predicted HSI after DO enhancement for the three life cycle stages of (a) Spawning, (b) Growth and (c) Overwintering

The DO concentration values at several locations before and after the oxygen enhancement are plotted in Figure 6-9, in which points A, B, C and L were located in the fluvial region while D, E, F, and G were located in the lacustrine region (see Figure 6-3). It can be seen that the DO concentrations in the fluvial region during the spawning and growth stages were less abundant than in overwintering. The ability of water to maintain oxygen in the dissolved state decreased with increasing temperature. Due to less surface reaeration and limited water inflow, low DO concentrations (<4 mg l^{-1}) were found in the lacustrine region (see Figure 6-9(a)).

Figure 6-9(b) shows that the predicted DO concentration in the lacustrine region has been significantly improved by the supplementary aeration, with the DO concentration ranging from approximately 4 to 8 mg l^{-1}, which are satisfactory in the lacustrine region for carp to settle. The model results confirmed that using the aerators in the lacustrine region provided 95% and 91% ideally suitable areas for the growth and overwintering stages, respectively. The DO enhancement by aeration devices would be an effective means by which to bring the DO concentration to the target level and therefore improve the water quality in the lacustrine region at the growth and overwintering stages. However, as Figure 8(a) illustrates, the suitable area for carps to spawn in the lacustrine region was not improved even though the DO concentration was increased from approximately 2 to 8 mg l^{-1}. As mentioned above, the main factor initiating spawning is the high flow while spawning activity for carp does not take place with a velocity under 0.2 m s^{-1}. The connected old and new reaches of the Jiyun River could potentially provide suitable conditions for successful spawning.

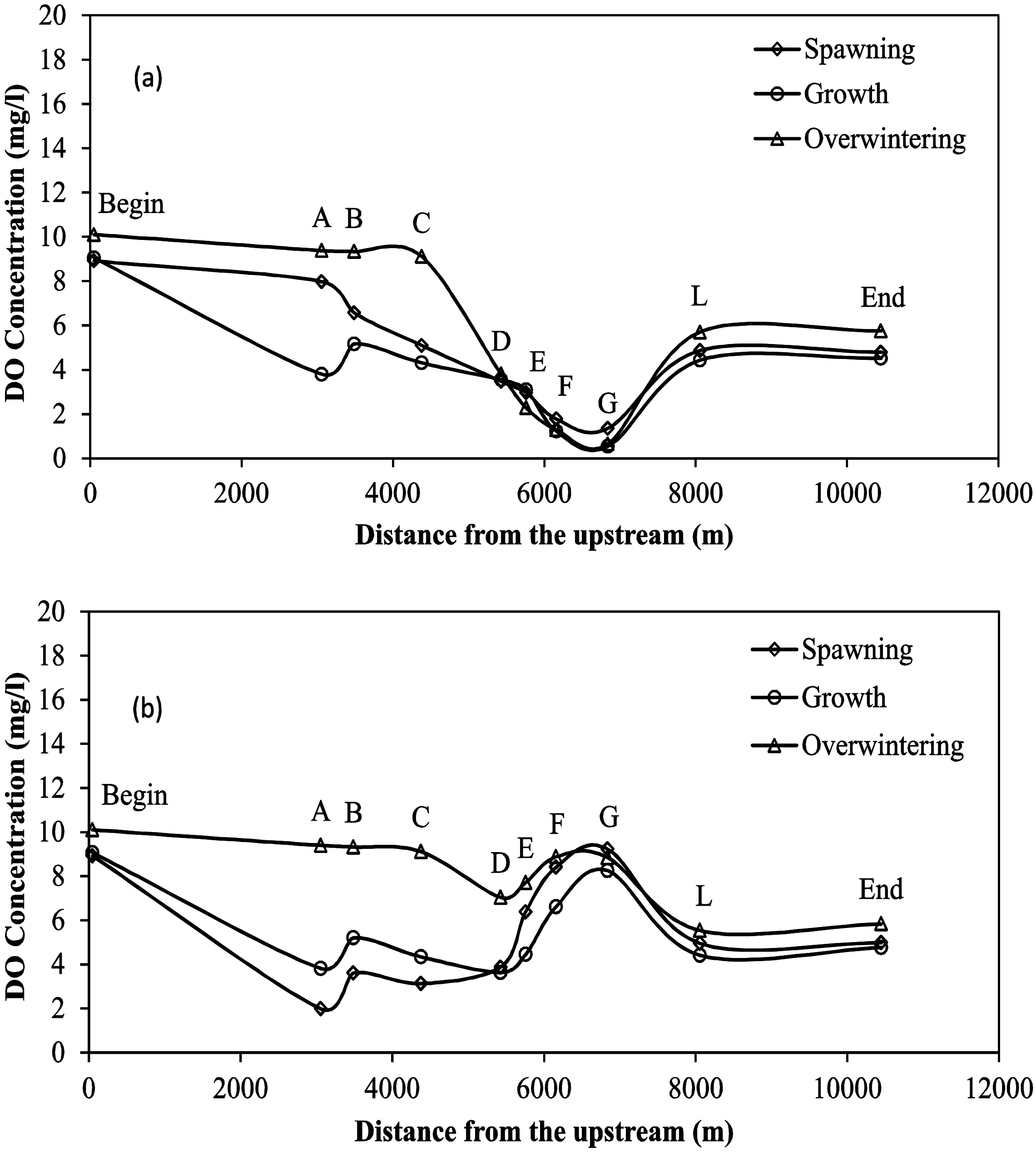


Figure 6-9 DO concentration from upstream to downstream before (a) and after (b) oxygen enhancement

Table 6-3 Predicted HSI distribution area before and after DO enhancement

	Spawning		**Growth**		**Overwintering**	
	Before	After	Before	After	Before	After
Ideal	4.5%	5.5%	44.5%	77.5%	24.4%	67.9%
Potential	10.2%	10.8%	24.7%	11.7%	20.5%	17.7%
Poor	85.3%	83.7%	30.8%	10.8%	55.1%	14.4%

6.5.3 Fluvial spawning habitat suitability improvement

At the spawning stage, the predicted flow velocity in the fluvial region was lower than that required to support successful spawning. To fulfil the spawning requirements during the spring drought, the effect of flow augmentation on the habitat conditions was investigated using the numerical model. The model was run for a variety of inflow discharges, ranging from 35 to 70 $m^3 s^{-1}$, to improve the habitat suitability level at the spawning stage. Since at the current condition more than 60% of water from the upstream main channel flows to the new reach, a weir was introduced to divert water from the new reach to the old one (see Figure 6-10). The modelling results indicate that, as the discharge increased to around 2 times that of the original discharge, the area of ideally suited habitats for carps to reproduce in the fluvial region increased to 78% (see Figures 6-7 and 6-10).

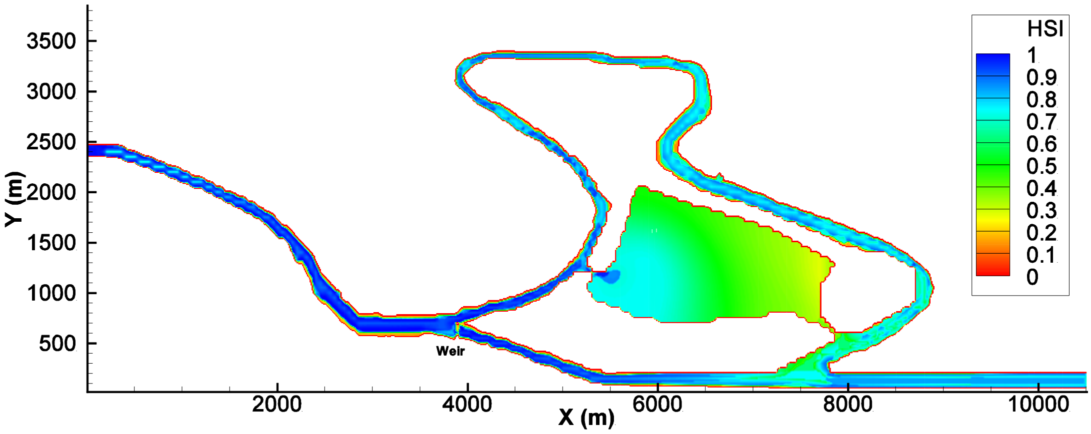


Figure 6-10 Predicted HSI after flow augmentation for the spawning stage

Although flow augmentation is an effective measure to improve the spawning condition, a continuous high flow for the spawning stage is not easily achieved. This is due to the fact that the Jiyun River is located in a low rainfall region and has relied on external water sources for many years. To overcome this constraint and to reduce its reliance on external water sources, additional water supplies from non-traditional sources, such as rainwater, desalinated water and recycled domestic and industrial wastewater, were also considered. As a city designed to be ecologically friendly, non-traditional water would be taken into account for river flow augmentation in order to reduce the use of normal water resources. The potential non-traditional water supply in the Eco-City and the volume of the augmented flow was calculated and plotted as a function of monthly time in Figure 11. The non-traditional water supply in the Eco-City was calculated based on the amount of domestic and industrial wastewater available in the Eco-City multiplied by the recovery factor (Tianjin Water Authority, 2012). As shown in Figure 11, the non-traditional water supply exceeded 4×10^6 m^3 of flow from March to June while approximately 2.5×10^6 m^3 of augmented flow is needed at the spawning stage. The analysis indicated that Tianjin Eco-City has sufficient non-traditional water to meet the high flow requirement at the carp spawning stage.

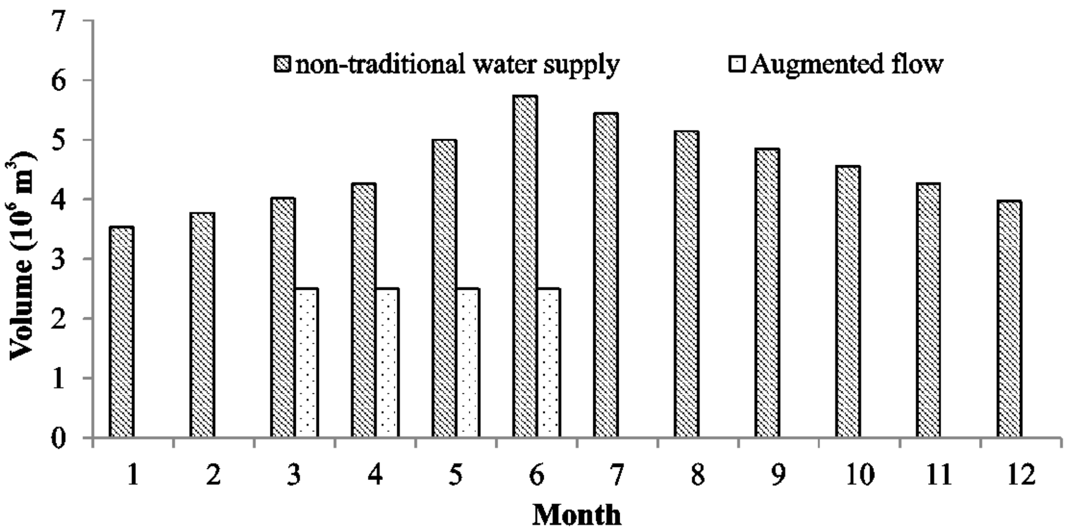


Figure 6-11 Non-traditional water supply and Augmented flow demand in the Eco-City

6.6 Conclusions

In this paper, a 2D hydrodynamic and water quality model has been coupled with a HSI model to investigate the levels of fish habitat suitability in a combined fluvial and lacustrine region in a new Eco-City. Both water quantity and quality parameters were considered in this model. Hydrological information was incorporated into the hydraulic data and the bioperiod needs of carps were evaluated using different suitability indexes. Model simulations were undertaken for three scenarios based on the flow rates for the three representative life stages of carps: spawning, growth and overwintering.

The model results indicated that under the current flow regime the habitat suitability level in the lacustrine area was poorly suitable for carps both at the growth and overwintering stages. The depletion of DO concentration was the main reason for the low habitat suitability level in the lacustrine region. To improve the habitat suitability level in the lacustrine region, the DO enhancement by aeration devices was used. It was found that the habitat suitability conditions were significantly improved by setting up 6 hypothetical aerators, particularly for the overwintering stage with HSI values increasing from 24.4% to 67.9%. Supplemental aeration is an effective means by which to improve the habitat suitability levels during the growth and overwintering stages. Due to the high flow requirements for carps to spawn, the effect of flow augmentation on the fluvial spawning habitat conditions was also studied. The simulation results showed that sufficient flow velocity to support successful spawning was achieved by raising the discharge to 2 times that of the current flow in the study area. The Eco-City is located in a water-lacking region, thus meaning that the augmented flow may not be easily achievable from reservoir storage. The analysis indicated that Tianjin Eco-City has sufficient non-traditional water to meet the high flow requirement for carps to spawn in the fluvial.

CHAPTER 7 CONCLUSIONS AND RECOMMENDATIONS

In this chapter, the main findings and conclusions of the study are summarised first, following which possible recommendations for further work are discussed.

As outlined in Chapter 1, the main objectives of this thesis were to:

- Determine the availability of non-traditional water resources and the potential water use in the Eco-City;
- Estimate the minimum ecological water requirements in order to maintain the environmental water quality at a certain level in the Eco-City;
- Develop an eco-hydraulic model to determine the fish habitat suitability in the river lake system in the Eco-City.

Firstly, in order to determine the availability of non-traditional water resources and the potential water use in the Eco-City, historical annual rainfall records of the Eco-City area were analysed to determine the rainwater harvesting whilst monthly industrial and domestic effluent data were used to estimate the reclaimed wastewater collection. Secondly, to estimate the minimum EWRs of the Eco-City river lake system, an improved wetted perimeter method was used to estimate the minimum EWRs based on the available data for the 8 cross sections of the river system. Finally, to determine the aquatic habitat suitability of the river lake system in the Eco-City, 1) an idealised river-lake system was assessed by using hydraulics laboratory experimentation and 2D numerical modelling. The analysis was based on residence time distribution curves, mean detention times and overall solute distribution patterns obtained for the simulated lake; 2) an eco-hydraulic model has been developed by combing a 2D hydrodynamic and water quality model with a Habitat Suitability Index (HSI) model. This model was used to investigate the levels of fish habitat suitability in a combined fluvial and lacustrine region in a new Eco-City. Both water quantity and quality parameters were considered in this model. The hydrological information was incorporated into the hydraulic data and the bioperiods needs for the three life phases of the target carp species were also evaluated. Model simulations were undertaken for three scenarios based on the flow rates for the three representative life stages of carp, including: spawning, growth and overwintering.

7.1 Conclusions

Non-traditional Water Resource

In Chapter 3, it was found that rainwater had a great potential for domestic use in the Eco-City from June to September. The total harvested rainwater from June to September was $8.66\times10^6 m^3$ while the domestic water consumption was $4.66\times10^6 m^3$. Reclaimed domestic and industrial water was determined to be the major water resource to fulfill the basic ecological function of the Jiyun River from January to March, during which the flow rate is too low to meet the ecological water requirements. The results showed that the flow rate in the new reach was only around 15.24 m^3/s in January. Overall, there were major opportunities to use non-traditional water to reduce demand for surface water in the Eco-City.

Minimum EWRs

The estimated results indicated that current monthly flow conditions are fairly satisfactory except for the flow from January to March. The total Minimum EWRs in January, February and March were 33.2 m^3/s, 33.3m^3/s, and 33.6m^3/s

respectively. In contrast, the measured flows from January to March ranged from 15.24 m^3/s to 30.29 m^3/s. An amount of 82.9×10^6 m^3 of the non-traditional water volume was needed to meet Minimum EWRs in Jiyun River. The reclaimed water calculation in Section 3.2.2 indicated that even though the total potential reclaimed water collected in January, February and March could not meet the EWRs, the ecological function could be retrieved by flow augmentation from the volume stored in November and December.

River Lake System

In Chapters 4 and 5, the results from hydraulics laboratory experimentation and 2D numerical modelling showed that recharge alone had little impact on the overall mixing level in the lake waters due to poor cross-sectional flow distribution. This could lead to localised water quality problems in stagnant and circulating flow regions. The effect of inserting a flow deflector near the lake inlet combined with flow augmentation was then assessed and was found to positively affect the distribution of solutes, by mitigating the occurrence of dead zones. It was concluded that additional measures to water diversion may need to be considered to achieve the goal of enhancing water quality in a recharged lake.

Fish Habitat suitability

In Chapter 6, it was concluded from the simulation result that under the current flow regime the habitat suitability level in the lacustrine area was poorly suited for carp at the growth and overwintering stages. The depletion of DO concentration was the main reason for the low habitat suitability level in the lacustrine region. Supplemental aeration was found to be an effective means by which to improve the habitat suitability levels during the growth and overwintering stages. The habitat suitability conditions were significantly improved by setting up

6 hypothetical aerators, particularly for the overwintering stage with HSI values increasing from 24.4% to 67.9%. Due to the high flow requirements for carp to spawn, the effect of flow augmentation on the fluvial spawning habitat conditions was also studied. The simulation results showed that sufficient flow velocity to support successful spawning was achieved by raising the discharge to 2 times that of the current flow in the study area. The analysis indicated that the non-traditional water supply exceeded 4×10^6 m^3 of flow from March to June while approximately 2.5×10^6 m^3 of augmented flow was needed at the spawning stage. Indeed, it was concluded that Tianjin Eco-City has sufficient non-traditional water to meet the high flow requirement for carps to spawn in the fluvial.

7.2 Recommendations

The findings of this study have significant implications for the planning and management of an Eco-City. However, with the increase of non-traditional water use for different purposes, concerns over the environmental and health implications of this use have also increased. Further research is necessary in order to set up a universal strategy and use systemic indicators for the assessment, monitoring and evaluation of the sustainability of non-traditional water use.

Future flow augmentation and water diversion will be required in the lake as low water mixing will significantly affect the ecology functions in the lake. In this sense, water management practitioners assisted by reliable predictive tools of flow and water quality processes can select the best intervention options. Further research is certainly needed for temporary structures which can be erected to store water for immediate or later diversion.

It should be pointed out that the results obtained from this study are specific to Tianjin Eco-City, although the methods can also be applied to other fluvial or lacustrine environments. The Non-traditional water use for sustainable development will be a key feature for other Eco-Cities. Although the combined model provides a useful tool to assess habitat suitability conditions in the fluvial and lacustrine region, only three important indices (water velocity, depth and DO) were considered in this study. Further studies are still needed to advance our understanding of the habitat suitability conditions in the fluvial and lacustrine regions by taking into account other relevant variables.

REFERENCES

Abdel-Ghaffar, A.S., El-Attar, H.A., Elsokkary, I.H., 1985. Egyptian experience in the treatment and use of sewage and sludge in agriculture, Proceedings of the Regional Seminar, Nicosia, Cyprus.

Acreman, M., Dunbar, M.J., 2004. Defining environmental river flow requirements - a review. Hydrology and Earth Systems Sciences 8, 861-876.

Alexander, R.B., Smith, R.A., Schwarz, G.E., 2000. Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico. Nature 403, 758–761.

Allen, E., Browne, J., Hynes, S., Murphy, J.D., 2013. The potential of algae blooms to produce renewable gaseous fuel. Waste Management 33, P 2425-2433.

Alp, E., Melching, C.S., 2011. Allocation of supplementary aeration stations in the Chicago waterway system for dissolved oxygen improvement. Journal of Environmental Management 92, 1577-1583.

Alsharhan, A.S., Rizk, Z.A., 2001. Hydrogeology of an Arid Region: The Arabian Gulf and Adjoining Areas, 137-146.

Al-Subaie, K.Z., 2007. Precise way to select a desalination technology. Desalination 206, 29-35.

Alves, M.H., 1994. Uma proposta de caudal ecolo ́gico para a barragem de Alqueva, 3rd Congresso da A ́gua VII Silubesa, III, 501–512.

Anderson, J.R., Morison, A.K., 1989. Environmental Flow Studies of the Wimmera River,Victoria. Part D. Fish Populations – Past, Present and Future. Conclusions and Recommendations. Technical Report Series 76, 75 pp.

Annear, T.C., Conder, A.L., 1984. Relative bias of several fisheries instream flow methods. North American Journal of Fisheries management 4, 531-539.

ARMCANZ, ANZECC. 1996. National Principles for the Provision of Water for Ecosystems. (Occasional Paper SRW No. 3): Sustainable Land and Water Resource Committee.

Arthington, A.H., Bunn, S.E., Poff, N.L., Naiman, R.J., 2006. The challenge of providing environmental flow rules to sustain river systems. Ecological Applications 16, 1311-1318.

Arthington, A.H., D.L. Conrick and B.M. Bycroft., 1992a. Environmental Study: Barker–Barambah Creek. Volume 2. Scientific Report: Water Quality, Ecology and Water Allocation Strategy. 457 pp.

ARUP, 2001. Website of ARUP. [online] http://www.arup.com/Projects/Cardiff_Bay_aeration_system.aspx.

Ashley, R.M., Tait, S.J., Styan, E., Cashman, A., Luck, B., Blanksby, J., Saul, A., Sandlands, L., 2007. Sewer system design moving into the 21st century—a UK perspective. Water Science and Technology 55, 273–281.

Bates, P.D., Steward, M.D., Siggers, G.B., Smith, C.N, Hervouet, J.M., Sellin, R. J.,1998. Internal and external validation of a two dimensional finite element model for river flood simulation. Proceedings of the Institution of Civil Engineers Water, Maritime & Energy 130, 127–141.

BBC, 2007. Website of BBC. [online] http://news.bbc.co.uk/1/hi/uk_politics.

Bentivegna, V., Curwell, S., Deakin, M., Lombardi, P., Mitchell, G., & Nijkamp, P., 2002. A vision and methodology for integrated sustainable urban development: BEQUEST. Building Research and Information 30.

Bernhard, H.S., Ilaria, I., Umberto, S., 2010. Characterizing solute transport with transient storage across a range of flow rates: The evidence of repeated tracer experiments in Austrian and Italian streams. Advances in water resources 33, 1340-1346.

Bockelmann, B.N., Fenrich, E.K., Lin, B.L, Falconer, R.A., 2004. Development of an ecohydraulics model for stream and river restoration. Ecological Engineering 22, 227-235.

Bovee, K.D., 1996. Perspectives on two-dimensional river habitat models: the PHABSIM Experience. Ecohydraulics 2000 Proceedings of Second International Symposium on Habitat Hydraulics B. INRS-Eau & FQSA, IAHR/AIRH, pp. B149–B162.

Brown, L.C., Barnwell, T.O., 1987. The enhanced stream water quality models QUAL2E and QUAL2E-UNCAS: documentation and user manual. Environmental Protection Agency1987, 40p

Caissie, D., El-Jabi, N., Bourgeois, G., 1998. Instream flow evaluation by hydrologically-based and habitat preference (hydrobiological) techniques,

CCICED, China Council for International Cooperation on Environment and Development. 2005. China's Sustainable Urbanization.

Chapra, S.C., 1997. Surface Water Quality Modelling. Mcgraw-Hill Companies, Inc., USA.

Chapra, S.C., Di Toro, D.M., 1991. Delta method for estimating primary production, respiration and reaeration in streams. Journal of Environmental Engineering, ASCE 117, 641-655.

Chatwin, P.C., 1980. Dispersion in natural streams. J. Hydraulic Div., ASCE 97 (HY1), 71–83.

China academy of urban planning and design, 2008. Tianjin Eco-City water system planning. 2008 (2).

CIRIA, 2007. The SUDs manual. Website of CIRIA. [online] www.ciria.org.

Clifford, N.J., Wright, N.G., Harvey, G., Gurbell, A.M., Harmar, O.P., Soar, P.J., 2010. Numerical Modeling of River Flow for Ecohydraulic Applications: Some Experiences with Velocity Characterization in Field and Simulated Data. Journal of Hydraulic Engineering 136,1033-1041.

Collings, M.R. 1974. Generalisation of spawning and rearing discharges for several Pacific salmon species in western Washington. USGS Open File Report. 39 pp.

Crittenden, J.C., Trussel, R.R., Hand, D.W., Howe, K.J., Tchobanoglous, G., 2005. Water treatment: principal and design, second ed. John Wiley & Sons, Inc, New Jersey. Anonymous 1432-1435.

Davies, P.E., Humphries, P., 1995. An Environmental Flow Study of the Meander, Macquarie and South Esk Rivers, Tasmania. 151 pp.

Day, T.J., 1975. Longitudinal dispersion in natural channels. Water Resource Research 11, 909–918.

Desai, A.Y., 2012. Development of a hydraulic sub-model as part of a desktop environmental flow assessment method. PhD TheSis. Rhodes University.

Edwards, E.A., Twomey, K.A., 1982. Habitat suitability index models: Common carp. U.S. Dept. Int. Fish Wildl. Servo FWS/OBS-82/10.12. 27 pp.

Environment Agency., 2010. Harvesting rainwater for domestic uses: an information guide. UK.

Environmental Flow Guidelines., 1999. A Technical Background Paper. ACT government, Australia.

EPA., 1977. Process design manual for land treatment of municipal wastewater Report 625/1-77-008. US Environmental Protection Agency, Cincinnati, Ohio.

Ersson, O., Ersson, M., 1996. Rainwater Harvesting and Purification System.

European Parliament. 2000， EU Water Framework Directive.

Falconer, R.A., 1986. A water quality simulation study of natural harbour. Journal of Waterway, Port, Coastal and Ocean Engineering, ASCE 112, 234–259.

Falconer, R.A., Lin, B.L., 2001. DIVAST user manual. Environmental Water Management Research Centre. Cardiff University. UK.

Falconer, R.A., Lin, B.L., 2003. Hydro-environmental modelling in riverine basins using dynamic rate and partitioning coefficents, International Journal of River Basin Management, IAHR 1 , 81-89.

Falconer, R.A., Lin, B.L., Kashefipour, S.M., 2005. Modelling water quality processes in estuaries, Computational fluid mechanics: Applications in Environmental Hydraulics. Chapter 12 305-328 Bates P D, Lane S, Ferguson R, John Wiley & Sons Ltd ISBN 0-470-84359-4.

Fischer H. B. 1978. On the tensor form of bulk dispersion coefficient in a bounded skewed shear flow. Journal of Geophysical Research. No. C5, 2373–2375.

Fischer, H.B., List, E.J., Imberger, J., Brooks, N.H., 1979. Mixing and Dispersion in Inland and Coastal Waters, Academic Press, Inc., California. Laboratory of Hydraulics and Water Resources Division of Engineering and Applied Science California Institute of Technology. Pasadena, California.

Fischer, H.B.,1966. A note on the one-dimensional dispersion model, Air and Water Pollution, An International Journal, Vol.10, pp. 443-452.

Fisher, S.G., Grimm, N.B., Mart, E., Holmes, R.M., Jones, J.B., 1998. Material spiraling in stream corridors: A telescoping ecosystem model. Ecosystems 1, 19–34

Fogler, H.S., 2006. Element of Chemical Reactor Engineering, 4th ed. Pearson Education Inc./Prentice-Hall, NJ.

Gao, G.H., Falconer, R.A, Lin, B.L., 2011. Numerical modelling of sediment-bacteria interaction processes in surface waters, Water Research 45, 1951-1960.

GB50015-2009, 2009. Water and Drainage Design Standard for Building, China.

Gilvear, D.J., Heal, K.V., Stephen, A., 2002. Hydrology and the ecological quality of Scottish river ecosystems. The Science of the Total Environment 294, 131–159.

Gippel, C. J., Stewardson, M.J., 1998. Use of wetted perimeter in defining minimum

Grismer, M.E., Tausendschoen, M., Shepherd, H.L., 2001. Hydraulic characteristics of a subsurface flow constructed wetland for winery effluent treatment, Water. Environment Research 73, 466–477.

Hakimzadeh, H., Falconer, R.A., 1997. Turbulence modelling of secondary tide-induced circulation in rectangular harbours. Proceedings of 27th IAHR Congress, San Francisco, IAHR 1, 785–790.

Henthorne, L., 2009. The Current State of Desalination. International Desalination Association. Retrieved 2011.

Hiroaki, F., 2008. Rainwater and reclaimed wastewater for sustainable urban water use. Physics and chemistry of the earth 33, 340-346.

Hu, W.P., Zhai, S.J., Zhu, Z.C., Han, H.J., 2008. Impacts of the Yangtze River water transfer on the restoration of Lake Taihu. Ecological Engineering 34, 30–49.

Hughes, D.A, Louw, D., 2010. Integrating hydrology, hydraulics and ecological response into a flexible approach to the determination of environmental water requirements for rivers. Environmental Modelling Software 25, 910-918.

Hughes, D.A., Hannart, P., 2003. A desktop model used to provide an initial estimate of the ecological instream flow requirements of rivers in South Africa. Journal of Hydrology 270, 167–181.

Hughes, D.A., Hannart, P., 2003. A desktop model used to provide an initial estimate. Hydrology 270, 167-181.

ILEC/Lake Biwa Research Institute [Eds]. 1988-1993 Survey of the State of the World's Lakes. Volumes I-IV. International Lake Environment Committee, Otsu and United Nations Environment Programme, Nairobi.

Indelicato, S., Tamburino, V., Barbagallo, S., 1988. Agronomical and health aspects-of municipal wastewater irrigation, 2nd International Conference on Environmental Protection, Sant'Angelo d'Ischia, Naples, Italy, 5-7 October.

Indrani, Gupta., Shivani, D., Chandorkar, A.A., 2004. Numerical modelling for Thane creek. Environmental Modelling & Software 19, 571–579.

Jester, D.B., Moody, T.M., 1969. Statewide fisheries investigations: a study of game fish reproduction and rough fish problems in Elephant Butte Lake. New Mexico Dept. Game Fish, Project F-22-R-9, Job F-1. 73 pp.

Jowett, I.G., 2003. Hydraulic constraints on habitat suitability for benthic invertebrates in gravel-bed rivers. River Research and Applications 19, 495–507.

Junk, W.J., Bayley, P.B., Sparks R E., 1989. The flood pulse concept in river-floodplain systems. Can. J. Fisheries Aquat. Sci 106, 110-127.

Kadlec, R.H., Wallace, S.D., 2009. Treatment wetlands, 2nd ed. CRC Press, Boca Raton, FL.

Kallemeyn, L., Novotny, J.F., 1977. Fish and fish food organisms in various habitats of the Missouri River in South Dakota,Nebraska and Iowa. U.S. Fish and Wildlife Service, FES/OBS-77/25.

Kaur,K, Toor H.S., 1978. Toxicity of pesticide to embryonic stages of cyperinus carpiocommunis. Indian journal of experimental biology, 193-196.

King, J., Brown, C., Sabet, H., 2003. A scenario-based holistic approach to environmental flow assessments for rivers. River research and applications 19, 619-639.

Kirchner, J. W., Feng, X.H., Neal, C., 2000. Fractal stream chemistry and its implications for contaminant transport in catchments, Nature 403, 524– 527.

Kolar, C.S., Chapman, D.C., Courtenay, W.R., Housel, C.M., Williams, J.D., Jennings, D.P., 2007. Bigheaded carps: a biological synopsis and environmental risk assessment. American Fisheries Society, Special Publication 33, Bethesda, Maryland.

Laenen, A., Bencala, K.E., 2001. Transient storage assessments of dye-tracer injections in rivers of the Willamette Basin, Oregon. Journal of the American Water Resources Association 37, 367–377.

Landman, M.J., Heuvel, M.R., 2003. An improved system for the control of dissolved oxygen in freshwater aquaria. Water Research 37, 4337–4342.

Leclerc, M., Boudreault, A., Bechara, J.A., Corfa, G., 1995. Two dimensional hydrodynamic modeling: a neglected tool in the instream flow incremental methodology. Trans. Am. Fisheries Soc 124 , 645–662.

Lee, J.H., Kil, J.T., Jeongc, S., 2010. Evaluation of physical fish habitat quality enhancement designs in urban streams using a 2D hydrodynamic model. Ecological Engineering 36, 1251-1259.

Lee, Y.J., Lwiza, K.M.M., 2008. Characteristics of bottom dissolved oxygen in Long Island Sound, New York. Estuarine, Costal and Shelf Science 76, 187 – 200.

Levenspiel, O., 1999. Chemical Reaction Engineering. John Wiley and Sons, New York.

Levin, L.A., Ekau, W., Gooday, A.J., 2009. Effects of natural and human- induced hypoxia on coastal benthos. Biogeosciences 6, 2063–2098.

Li, C. W., Falconer, R.A., 1995. Depth integrated modeling of tide induced circulation in a square harbor." Journal of Hydraulic Research 33, 321–332.

Liu, J., He, B., 1992. China freshwater fishery cultivation. Beijing, Science Press (in Chinese).

Lukowicz, M., Gerstner, P., 1998. In: Lehrbuch der Teichwirtschaft, 4. neubearbeitete Auflage pp. 495-503. Verlag Paul Parey, Berlin, Germany.

Mall,B., 1999a. Personal message on November fifth, Mall-Beton GmbH, Hufinger Str. 39-45, D-78166 Donaueschingen.

Mao, H.H, Li, J.Z., 2009. Analysis on Wet-Dry and Compensative Characteristics of Regional Rainfall. Journal of Tianjin University 5, China (in Chinese).

Mark Boslet, 2010. Eco City Movement Expands in Sweden. [Online] http://www.greentechmedia.com/articles/read/eco-city-movement-expands-in-sweden.

Marsili-Libelli, S., Giusti, E., Nocita, A., 2013. A new instream flow assessment method based on fuzzy habitat suitability and large scale river modelling. Environmental Modelling & Software 41, 27-38.

Mas, Q., 2009. Eco-city and eco-planning in china: taking an example for caofeidian eco-city. The 4th International Conference of the International Forum on Urbanism, Amsterdam/Delft.

Massei, N., Wang, H.Q., Field, M.S., Dupont, J.P., Bakalowicz, M., Rodet, J., 2006. Interpreting tracer breakthrough tailing in a conduit-dominatedkarstic aquifer. Hydrogeology Journal 14 , 849–858.

Master Plan. Sino-Singapore Tianjin Eco-city. Retrieved 2008-08-05.

McCrimmon, H. R., 1968. Carp in Canada. Fish Res. Board Can. Bull. 165. 93 pp.

Metcalf and Eddy Inc., 2003. Wastewater Engineering: Treatment and Reuse. McGraw-Hill, New York, NY.

Michael, T.K., 1992. Master Thesis. hydrodynamic with a wall-slip boundary condition for a particle moving near a plane wall bounding a semi-infinite viscous fluid

Mielke, J.E., 1999. Desalination R&D: The New Federal Program.

Milhous, R.T., Wegner, D.L., Waddle, T., 1984. User's guide to the physical habitat simulation system. US Fish and Wildlife Service Biological Services Program, FWS: OBS- 81:43.

MISTRA, 1999. Sustainable urban water management. Vatten 1999;55:7–17.

Nagaya, T., Shiraishi, Y., Onitsukab, K., Higashinoc, M., Takamic, T., Otsukad, N., Akiyamab, J., Ozekie, H., 2008. Evaluation of suitable hydraulic conditions for spawning of ayu with horizontal 2D numerical simulation and PHABSIM. Ecological modelling 215, 133–143.

Naghibi, A., Lence, B., 2012. Assessing impacts of high flow events on fish population: Evaluation of risk-based performance measures. Ecological Modelling 240, 16-28.

Nordin, C.F., Troutman, B.M., 1980. Longitudinal dispersion in rivers: the persistence of skewness in observed data. Water Resource Research 16, 123-128.

Odeh, R., Jayyousi,A.I., 2003. Grey water resue:towards sustainable water management. Desalibation156, 181-192

Oldham, R.S., Keeble, J., Swan, M.S, Jeffcote, M., 2000. Evaluating the suitability of habitat for the Great Crested Newt (Triturus cristatus). Herpetological Journal10,143-155.

Papanicolaou, A.N., Elhakeem, M D., Dermisis, D., Young, N., 2011. Evaluation of the Missouri River shallow water habitat using a 2D-hydrodynamic model. River research and application 27,157–167.

Papanicolaou, A.N., Elhakeem, M., Krallis, G., Prakash, S., Edinger, J., 2008. Sediment transport modeling review – Current and future developments. Journal of Hydraulic Engineering 134, 1–14.

Parasiewicz, P. 2008. Application of MesoHABSIM and target fish community approaches for selecting restoration measures of the Quinebaug River, Connecticut and Massachusetts, USA. River Research and Application 24, 459–471.

Parasiewicz, P., 2007. The MesoHABSIM model revisited. River Research and Application 23, 893–903.

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E., Stromberg, J.C., 1997. The natural flow regime. BioScience 47, 769-784.

Portielje, R., Lijklema, L., 1995. The effect of reaeration and benthic algae on the oxygen balance of an artificial ditch. Ecological Modelling 79, 35-48.

Prasad, N., Ranghieri, F., Shah, F., Trohanis , Z., Kessler, E., and Sinha, R., 2009. Climate resilient cities- a Primer on reducing vulnerabilities to disasters. The World Bank, Washington DC, pp 186.

Quality Domains Ltd, 2010. Website of lowenergyhouse. [online] http://www.lowenergyhouse.com/rainwater-harvesting.html.

Queensland Water Commission, 2010. South East Queensland Water Strategy. Summer, G., 1988. Precipitation: Process and Analysis. John Wiley and Son, Chich-ester.

Register, R., 1987. Ecocity Berkeley: building cities for a healthy future. North Atlantic Books.

Reiser, D.W., White, R.G, 1983. Effect of complete red dewatering on salmonid egg-hatching success and development of juveniles. Transactions of the American Fisheries Society 112, 532-540.

Richardson, B.A., 1986. Evaluation of instream flow methodologies for freshwater fish in New South Wales. Stream Protection, the Management of Rivers for Instream Uses. Water Studies Centre, Chisholm Institute of Technology, Caulfield. pp. 143–167.

Riis ,T., Biggs, B., 2003. Stream vegetation and flow regimes. Water & Atmosphere 11.

Roseland, M., 1997. “Dimensions of the eco-city.” Cities 144: 197-202.

Roseland, M., 2001. The eco-city approach to sustainable development in urban areas. In: Devuyst D, Hens L, De Lannoy W (eds) How green is the city? Sustainability assessment and the management of urban environments. Columbia University Press, New York, pp 85–104.

Seo, I., Baek, K., 2004. Estimation of the longitudinal dispersion coefficient using the velocity profiles in natural streams. Journal of Hydraulic Engineering 130, 227-236.

Seyed, M., Kashefipour, Falconer, R.A., 2002. Longitudinal dispersion coefficients in natural channels. Water Research 36, 1596–1608.

Singapore government, 2008. Website of the Sino-Singapore Tianjin Eco-city. [online] http://www.tianjinecocity.gov.sg.

Sino-Singapore Tianjin Eco-City Investment and development Co., Ltd. http://sstec.dashilan.cn.

Soulie, M., Tremea, L.L., 1991. Technolgie pour le traitement.et la reutilization des eaux esees dans le basin mediterraneen, 3eme Rencontre de la Agence Regional pour l'Environnement, Department Haute Cote d'Azur, Marseille, France, 24-26 September.

Stalnaker, C., Lamb, B.L., Henriksen, J., 1995. The Instream Flow Incremental Methodology-A Primer for IFIM: National Biological Service, U.S. Department of the Interior, Biological Report 29, p 45.

State Environmental Protection Administration, General Administration of Quality Supervision, Inspection and Quarantine, 2002. Environmental Quality Standards

Stefan, H.G., Fang, X., 1994. Dissovled oxygen model for regional lake analysis. Ecological modelling 71, 37-68.

Stephane, A, Malcolm, L.,1993. Flushing times for the Providence River based on tracer experiments. Estuaries and coastes 16, 830-839.

Surface Water Quality Standard (GB 3838-2002[S]). Beijing, China (in Chinese).

Sushil, K.S., Beck, M.B.,2003. Dispersion Coefficient of Streams from Tracer Experiment Data. Journal of Environmental Engineering 129, 539.

TEMC, 2008. Master plan of Sino-Singapore Tianjin Eco-City. Tianjin Eco-city Management Committee. Tianjin, China (in Chinese).

Tennant, D.L., 1976. Instream flow regimes for fish, wildlife, recreation and related environmental resources. Fisheries 1, 6-10.

Tianjin Water Authority. 2010. Tianjin water resources bulletin. Tianjin, China (in chinese).

Tyler, R.G., 1946. Polluted streams cleared up by aeration. Civil Engineering 16, 348.

UNEP, 2008. Water Quality for Ecosystems and Human Health. 2nd edition. ERCE, UNESCO.

USACE, River Analysis System HEC-RAS, Release Notes. 2010.

Vannote, R.L., Minshall, G.W., Cummings, K.W., Sedell, J.R., Cushing, C.E., 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37, 130–137

Verigin, B.V., Makeyeva, A.P., Mokhamed, M.I., 1978. Natural spawning of the silver carp (Hypophthalmichthys molitrix), the bighead carp (Aristichthys nobilis), and the grass carp (Ctenopharyngodon idella) in the Syr-Darya River. Journal of Ichthyology 181, 80-92.

Waddle, T.J., 2001. PHABSIM for Windows: User's Manual and Exercises: Fort Collins, CO, U.S. Geological Survey, 288 p.

Wang, C.F., Hsu, M.H., Kuo, A.Y., 2004. Residence time of the Danshuei River Estuary, Taiwan. Estuarine, Coastal and Shelf Science 60, 381-393.

Wang, R., 1996. Thinking about urban interactions: a Chinese approach. Nature Resources 32, 7-8.

Wang. H.Q., Meselhe, .A.E., Waldon, M.G., Harwell, C.M., Chen, C.F., 2012. Compartment-based hydrodynamics and water quality modelling of a Northern Everglades Wetland, Florida, USA. Ecological Modelling 247, 273–285.

Wang. H.Q., Meselhe, A.E., Waldon, M.G., Harwell, C.M., Chen, C.F., 2012. Compartment-based hydrodynamics and water quality modelling of a Northern Everglades Wetland, Florida, USA. Ecological Modelling 247, 273–285.

Washington Department of Fish and Wildlife (WDFW), 2004. U.S. Fish and Wildlife Service, Washington Department of Ecology.

Watts, J., 2011. Can the sea solve China's water crisis?. The Guardian (London). Retrieved 2011-04-19.

World Commission on Environment and Development. Our Common Future, Chapter 2: Towards Sustainable Development. Un-documents.net. Retrieved 2011-09-28.

World Health Organization., 2005. World Summit Outcome Document. USA.

Yi, Y.J., Wang, Z.Y., Yang, Z.F., 2010. Impact of the Gezhouba and Three Gorges Dams on habitat suitability of carps in the Yangtze River. Journal of Hydrology 387, 283–291.

Zhou, N., Williams, C., 2013. An international Review of Eco-City Theory, Indicators, and Case Studies. Report. Berkeley National Laboratory, USA.

PUBLICATION LIST

The following paper has been published, which is based on this research.

[1] Wang, F., Lin, B.L., 2013. Modelling habitat suitability for fish in the fluvial and lacustrine regions of a new Eco-City. *Ecological Modelling* 267, 115-126.

[2] Wang, F, Lin, B.L, Rauen, W.B., 2013. Modelling the impact of water diversion in an idealised river-lake system. 35th IAHR World congress, Chengdu, China.

[3] Wang, F, Lin, B, Rauen, W.B., Ren, Y.H., 2011. Eco-hydraulics modelling of the ecological water requirement in an Eco-City, XIVth IWRA World Water Congress, Pernambuco. Brazil.

[4] Wang, F, Lin, B., 2013. Ecological water requirements supplementation by Non-traditional water in a new Eco-City (Preparation).