A NOVEL STANDARDIZED ASSESSMENT FOR THE NEW END USES OF RECYCLED WATER SCHEMES

By

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CERTIFICATE OF ORIGINAL AUTHORSHIP

I certify that the work in this thesis has not previously been submitted for a degree nor has it been submitted as part of requirements for a degree except as fully acknowledged within the text.

I also certify that the thesis has been written by me. Any help that I have received in my research work and the preparation of the thesis itself has been acknowledged. In addition, I certify that all information sources and literature used are indicated in the thesis.

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NOMENCLATURE

AAS	atomic adsorption spectrophotometer
ABS	absorbents
AC	activated carbon
ADWG	Australia drinking water guideline
AOP	advanced oxidation process
AP	acidification potential
BC	breakthrough capacity
BOD	biochemical oxygen demand
BN	Bayesian network
BV/h	bed volumes per hour
CAS	conventional activated sludge
CBD	central business district
COD	chemical oxygen demand
COEF	coefficient
CSF	cancer slope factor
CW	constructed wetland
CWW	city west water
DAF	dissolved air flotation
DALY	disability adjusted life years
DBP	disinfection by-products
DMF	dual media filtration
DPR	direct potable reuses
ECOSAR	ecological structure activity relationship
EDCs	endocrine disrupting compounds
EDS	energy disperses X-ray spectroscopy
EIO	economic input-output
ELECTRE	elimination and choice expressing reality
EP	eutrophication potential
EPA	environmental protection agency
ERA	environmental risk assessment
ETP	ecotoxicity potential
FC	faecal coliform

GAC	granular activated carbon
GC	gas chromatography
GHG	greenhouse gas
GL	gigalitres
GL/d	gigalitres per day
GL/yr	gigalitres per year
GWP	global warming potential
GWR	groundwater replenishment
HACCP	hazard analysis critical control point
HQ	hazard quotient
HRA	health risk assessment
HRT	hydraulic retention time
ICP	inductively coupled plasma optical emission spectroscopy
ION	ion exchange
IPR	indirect potable reuses
kWh/d	kilowatt-hour per day
L	litre
LC	liquid chromatography
LCA	life cycle analysis
LCI	life cycle inventory
LCIA	life cycle impact assessment
LOAEL	lowest dose at which adverse effects are observed
L/p/d	litres per capita per day
MAUT	multi-attribute utility theory
MBR	membrane bioreactor
MCA	multi-criteria analysis
MF	microfiltration
MFA	material flow analysis
mg/g	milligram per gram
mg/L	milligram per litre
MIET	missing inventory estimation tool
ML	megalitre
ML/d	megalitres per day
ML/yr	megalitre per year

MOC	maximum operation capacity
MRA	microbial risk assessment
MS	mass spectrometry
MVC	mechanical vapour compression
NF	nanofiltration
NOAEL	highest dose at which no adverse effects are observed
NSW	New South Wales
OCWD	Orange County water district
ODP	ozone depletion potential
OR	odds ratio
ORG	organoclay
ORP	oxidation reduction potential
ORWARE	organic waste research model
PAC	powdered activated carbon
PEC	predicted environmental concentration
PHO	photochemical oxidation
PMHC	Port Macquarie-Hastings council
PNEC	predicted no effect concentration
PROMETHEE	preference ranking organization method for enrichment evaluation
QCRA	quantitative chemical risk assessment
QLD	Queensland
QMRA	quantitative microbial risk assessment
RA	risk assessment
RBC	rotating biological reactor
RC	residual chlorine
RfD	safe risk level
RHDA	Rouse Hill development area
RHWRS	Rouse Hill water recycling scheme
RIRA	recycled water irrigation risk analysis
RO	reverse osmosis
ROWG	rank order weight generation
RQ	risk quotients
RW	recycled water
RWAlterDW	recycled water is an alternative to drinking water

PhACs	pharmaceutical active compounds
SA	South Australia
SAT	soil aquifer treatment
SBR	sequencing batch reactor
SE	standard error
SEM	scanning electron microscope
SP	salinisation potential
SPSS	statistical package for the social sciences
STP	sewage treatment plant
SWC	Sydney Water corporation
TC	total coliform
TDS	total dissolved solids
TN	total nitrogen
TOC	total organic carbon
TP	total phosphorus
TSS	total suspended solids
UF	ultrafiltration
UV	ultraviolet
VIC	Victoria
VOC	volatile organic compounds
WCRWP	Western Corridor recycled water project
WHO	world health organization
WM	washing machines
WRAMS	water reclamation and management scheme
WRP	water reclamation plant
WSP	waste stabilization pond
WWTP	wastewater treatment plant

RESEARCH OUTCOMES

(9 journal papers, 4 conference papers and 7 research awards)

Journal Articles

1. **Chen, Z.**, Ngo, H. H., Guo, W. S., Pham, T. T. N., Lim, R., Wang, X. C., et al. (2014). A new optional recycled water pre-treatment system prior to use in the household laundry. *Science of the Total Environment*, 476, 513-521.

2. **Chen, Z.**, Ngo, H. H., Guo, W. S., Lim, R., Wang, X. C., O’ Halloran, K., et al. (2014). A comprehensive framework for the assessment of new end uses in recycled water schemes. *Science of the Total Environment*, 470-471, 44-52.

3. **Chen, Z.**, Ngo, H. H., Guo, W. S., Wang, X. C., Miechel, C., Corby, N., et al. (2013). Analysis of social attitude to the new end use of recycled water for household laundry by the regression models. *Journal of Environmental Management*, 126, 79-84.

4. **Chen, Z.**, Ngo, H. H. and Guo, W. S. (2013). Risk control in recycled water schemes. *Critical reviews in Environmental Science and Technology*, 43(22), 2439-2510.

5. **Chen, Z.**, Ngo, H. H. and Guo, W. S. (2013). A critical review on the end uses of recycled water. *Critical reviews in Environmental Science and Technology*, 43(14), 1446-1516.

6. **Chen, Z.**, Ngo, H. H., Guo, W. S. and Wang, X. C. (2013). Analysis of Sydney’s recycled water schemes. *Frontiers of Environmental Science and Engineering,* 7(4), 608-615.

7. **Chen Z.**, Ngo H. H., Guo W. S., Listowski, A., O’Halloran, K., Thompson, M., et al. (2012). Multi-criteria analysis towards the new end use of recycled water for

household laundry: A case study in Sydney. *Science of the Total Environment*, 438(1), 59-65.

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9. **Chen, Z.**, Ngo, H. H., Guo, W. S., Wang, X. C. and Luo, L. (2011). Probabilistic risk assessment of recycled water schemes in Australia using MATLAB toolbox. *Journal of Water Sustainability*, 1(3), 75-86.

Conference Papers

1. **Chen, Z.**, Ngo, H. H., Guo, W. S. (2013). Conceptual principle for development of new end uses in recycled water schemes. Proceedings of the 4th International Symposium "Re-Water Braunschweig", November 6-7, 2013, Braunschweig, Germany. p. 27-34.

2. **Chen, Z.**, Ngo, H. H., Guo, W. S., Listowski, A., O' Halloran, K., Thompson, M. and Muthukaruppan, M. (2012). Multi-criteria analysis of Sydney's recycled water schemes towards the new end use for washing machines, Poster presentation, *IWA World Water Congress and Exhibition*, Busan, Korea, 16-21 September, 2012.

3. **Chen Z.**, Ngo H. H., Guo W. S. and Wang X. C. (2011). Analysis of Sydney's recycled water schemes. Oral presentation at the IWA Conference-*Cities of the Future Xi'an: Technologies for integrated urban water management*, China, 15-19 September, 2011.

4. **Chen, Z.**, Ngo, H. H., Guo, W. S., Wang, X. C. and Luo, L. (2011). Probabilistic risk assessment of recycled water schemes in Australia using MATLAB toolbox. Oral presentation at the *International Conference on Challenges in Environmental Science and Engineering (CESE)*, Tainan, Taiwan, 25-30 September, 2011.

Research awards

1. Excellence in Professional Development Program in Civil & Environmental Engineering Research, CTWW, UTS 07/2011−12/2013

2. Finalist at the 2013 UTS final 3 Minutes Thesis Competition 08/2013

3. Best oral presentation award at the UTS Faculty of Engineering and Information Technology (FEIT) 3 Minutes Thesis Competition 08/2013

4. Best oral presentation award at the UTS FEIT Research Showcase Contest 06/2013

5. Best poster presentation award at the International Water Association (IWA) World Water Congress and Exhibition, Busan, Korea, 16-21 September, 2012 09/2012

6. Best student oral presentation award at the International Conference on Challenges in Environment Science & Engineering, Taiwan, 25-30 September, 2011 09/2011

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ABSTRACT

Nowadays, recycled water has provided sufficient flexibility to satisfy short-term freshwater needs and increase the reliability of long-term water supplies in many water scarce areas. It becomes an essential component of integrated water resources management. However, the current applications of recycled water are still quite limited with non-potable purposes such as irrigation, industrial uses, toilet flushing, car washing and environmental flows. There is a potential to exploit and develop new end uses of recycled water in both urban and rural areas. This can contribute largely to freshwater savings, wastewater reduction and water sustainability.

This thesis put forwards a conceptual decision making framework for the systematic feasibility assessment of sustainable water management strategies in related to new end uses of recycled water's planning, establishment and implementation. Due to the transparency, objectivity and comprehensiveness, the analytic framework can facilitate the optional management strategy selection process within a larger context of the community, processes, and models in recycled water decision-making. Based on that, a simplified quantitative Multi-criteria Analysis (MCA) was conducted in Rouse Hill Development Area (RHDA), Sydney, Australia, using the Multi-attribute Utility Theory (MAUT) technique. The results indicated that recycled water for a household laundry was the optimum solution which best satisfied the overall evaluation criteria. Another two management options can be excluded from further consideration in initial stages, namely the implementation of Level 1 water restriction on the use of recycled water and recycled water for swimming pools.

With the identified strengths of recycled water use in washing machines, five relevant management alternatives were proposed according to different recycled water treatment technologies such as microfiltration (MF), granular activated carbon (GAC) or reverse osmosis (RO), and types of washing machines (WMs). Accordingly, a comprehensive quantitative assessment on the trade-off among a variety of issues (e.g., technical, risk, social, environmental and economic aspects) was performed over the alternatives. Overall, the MF treated recycled water coupled with new washing machines and the MF-GAC treated recycled water coupled with existing washing machines were shown to be preferred options. The results could provide a powerful guidance for sustainable water reuse in the long term. However, more detailed field trials and investigations are still needed to understand, predict and manage the impact of selected recycled water new end use alternatives effectively.

Notably, public acceptability becomes important to ensure the successful development of recycled water new application in household laundries. This thesis addresses social issues by extensive social attitude surveys conducted in three locations of Australia, namely Port Macquarie, Melbourne and Sydney. Based on responses from Port Macquarie and Melbourne, the regression models provide conclusions about which characteristics are more likely to lead to the acceptance of recycled water from society. Three attitudinal variables (i.e., recycled water is an alternative to drinking water, attitude and cost) and three psychological variables (i.e., odour, reading and a small treatment unit) were found to be the key driving forces behind domestic water reuse behaviour. Comparatively, survey results in Sydney indicated slightly different aspects of concern. Due to experience in current use on dual pipe systems, Sydney residents interviewed have established good cognitions on the appearance and cost of recycled water. They were more concerned about the colour of clothes and potential damage to washing machines. The overall findings could drive future research to achieve a better public perception of the new end uses of recycled water.

Moreover, the thesis also demonstrates the feasibility and cost-effectiveness of applying a zeolite filtration column as an effective ion-exchange resin for recycled water softening prior to use in washing machines. At the laboratory scale, the column service life for a typical washing machine was approximately one month without material regeneration on the basis of an optimal contact time (i.e., 5 minutes) and the calculated breakthrough capacity (i.e., 14 milligram hardness ions per gram of zeolites). It is believed that with a full application at households, this unit is likely to play a positive role in guaranteeing the recycled water quality as well as changing the public perception on the safe use of recycled water.

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CHAPTER 1

Introduction

1.1 OVERVIEW

Water is vital to human health and wellbeing while supporting ecosystems, agricultural and industrial development and the environment. It is also crucial in underpinning in cultural and social values (ATSE, 2012). As a result of climate change (e.g., flooding, prolonged drought and severe cold), increasing population, rapid urbanisation and deteriorating water quality, water scarcity is considered as one of the most critical threats for the society and a constraint for sustainable development. Due to continuous economic and population growth, water may become the most strategic resource in many areas of the world within the next decades, especially in arid and semi-arid regions (Asano et al., 2007; Gohari et al., 2013).

In light of potential water shortages, many countries, regions and cities have increasingly recognised the significance of water conservation and water demand management as a long-term water supply strategy. However, as water conservation is still unable to close the water supply-demand gap effectively in some cases, technological innovations in driving green growth for the existing water market should be taken into account. These ways can promote water consumption toward enhanced efficiency, productivity and environmental outcomes. Moreover, sustainable management solutions should also be considered to balance the technical, environmental, economic and social issues. The corresponding strategies may involve the exploitation and development of alternative water resources such as rainwater, stormwater, recycled water and desalinated water (Smith, 2011; ATSE, 2012).

Compared to other water resources, recycled water can contribute to a considerable wastewater reduction through reduced effluent discharges to the aquatic environment. It can also provide a relatively constant water supply all through the year, and offer a large number of benefits through various end uses (e.g., irrigation, industrial, residential, recreational, indirect and direct potable reuse applications). To some extent, excessive costs on water infrastructure and energy consumption could be avoided as well (Anderson et al., 2001; Huertas et al., 2008). In addition to implementing the current end uses, the establishment of new recycled water applications can promote environmental sustainability and reduce the ecological footprint. The proposed new end uses include recycled water for household laundry, livestock drinking and servicing, and swimming pools (Chen et al., 2014).

The emphasis and appraisal of the recycled water schemes have traditionally been on solving technological challenges, such as advanced wastewater treatment approaches, infrastructure requirements, human health safety factors and maximum economic returns. The performance of both individual and combined treatment technologies have been evaluated widely using single or integrated assessment models such as life cycle assessment, material flow analysis and environmental/health risk assessment (Tangsubkul, 2005b; Agnes et al., 2007; Urkiaga et al., 2008). Some economic evaluation models that focus on capital and maintenance costs and the price of recycled water have also been constructed (Godfrey et al., 2009). Moreover, substantial guidelines and regulations towards specific recycled water end uses as well as considerable national and local reports on water quality and risk control are being increasingly established. These actions will standardise the treatment level and improve the reliability of recycled water significantly.

Although technical feasibility and economic affordability of producing recycled water of virtual drinking water quality have already been largely achieved, some of the recycled water schemes have been put aside. The main reasons for project failures might be strong public misgivings and the lack of appropriate management including a systematic decision-making framework, rational and structured principles, demand-supply and cost-effectiveness analysis, and holistic quantitative assessment. It is worth noting that sustainable water management requires thorough understanding, comprehension and application of cross-disciplinary approaches and scientific credibility that might influence various technical and non-technical issues.

Consequently, with ambitious water recycling targets formulated by the local government and water authorities, long-term sustainability of the recycled water schemes becomes crucial for future project expansion and new end use exploitation and development (Chen et al., 2012b). To enhance the environmental performance and public acceptability further, a comprehensive decision-making system should be built into the recycled water field, which aims to facilitate the quantitative analysis of the trade-offs among a variety of competing issues (e.g., risk, technical, environmental, social and economic aspects) simply, efficiently and effectively. Meanwhile, case studies, social surveys and experimental investigations are also essential to validate and improve upon the decision-making of recycled water schemes continuously.

1.2 RESEARCH SCOPE AND OBJECTIVES

This thesis focuses on the analysis of recycled water new end use(s) sustainability and public acceptability improvement through a series of decision-making system build-up, qualitative and quantitative multi-factorial evaluations in terms of technical, environmental, risk, social and economic aspects, as well as social surveys and experimental designs. The research work has the potential for application beyond recycled water new end uses. The concepts could be applied to the complex assessment and/or the construction of sustainable development scenarios for urban water resources. The developed models could also be incorporated into integrated water resources planning and management. This study describes a statistical analysis that has direct significance for recycled water-related policy making and strategy creation. This can guarantee the implementation of residential recycled water supply in a smooth and cost-effective way. With further improvement, the pre-treatment unit introduced for recycled water purification in a laundry could be put into practice in households. The designs could also be adapted to other recycled water end uses such as livestock and/or pet drinking and commercial laundry purposes.

The research objectives of this study were formulated in a logical sequence and include the followings:

- To review previous literature critically regarding the status of recycled water use and the adopted models for the holistic water and environmental sustainability evaluation;
- To identify the relevant research gaps and opportunities;
- To develop the assessment framework, principles and methodologies for systematic decision making;

- To conduct a qualitative feasibility assessment for proposed new end uses of recycled water;
- To define specific recycled water management alternatives, to determine the fit-for-purpose assessment criteria and to perform a detailed quantitative assessment;
- To complete a specific decision-making plan for the smooth implementation of the preferred management option(s) and the establishment of risk communication, monitoring and review;
- To investigate the public attitude, knowledge and concerns in different locations of Australia regarding the actual recycled water application in households, especially in the laundry;
- To identify the key areas where efforts can be put to improve further the public acceptability of new end uses of recycled water; and
- To evaluate the feasibility of a small optional pre-treatment unit for additional purification of recycled water for household laundry.

1.3 THESIS STRUCTURE

The thesis comprises eight chapters and is structured as follows: Chapters 2 − 7 address the research objectives specifically. Chapter 8 draws conclusions in relation to management strategies for recycled water new end uses, and systematic analysis and assessment of sustainable recycled water schemes more generally.

Chapter 1 gives a brief overview of the existing circumstances and issues associated with water supply, recycling and reuse. A description of the research objectives and scope is presented afterwards.

Chapter 2 demonstrates the importance, definition and sources of recycled water. It also presents a critical review of the current recycled water end uses and existing environmental assessment models on the evaluation of recycled water schemes. The advantages and weaknesses of each type of model on water reuse are discussed in detail.

Chapter 3 describes the adaptive research methodologies on the analysis of new recycled water applications in three parts: (i) a comprehensive Multi-criteria Analysis (MCA), including the framework, principles and quantitative evaluation algorithms; (ii) a social survey analysis with the chi-square test and regression model; and (iii) a new optional recycled water purification system including experimental materials, apparatus and procedures.

Chapter 4 identifies the potentials for the development of three recycled water new end uses and performs a conceptual decision-making analysis to validate the strengths of these applications.

Chapter 5 investigates the feasibility of implementing sustainable water management strategies in an existing project in Sydney using a simplified quantitative MCA. Furthermore, a complex MCA with advanced weighting and outranking techniques is conducted for the new recycled water use in a household laundry.

Chapter 6 presents the social surveys conducted in three locations of Australia. The respondents' attitude, knowledge and psychological concerns and their final acceptance on recycled water use in a household laundry are studied.

Chapter 7 develops a zeolite filtration column to reduce the total hardness level of recycled water further before water use in a washing machine. The performance of the column is discussed in terms of optimal contact time, maximum operation capacity, breakthrough capacity, column service life and regeneration capability.

Chapter 8 provides summaries, statements and conclusions from this study and offers recommendations for future research.

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CHAPTER 2

Literature Review

2.1 INTRODUCTION

This chapter provides a summary of recent literature related to global water supply situations and recycled water opportunities as well as recycled water sources, end uses and assessment models. This review could facilitate the further understanding of the key factors that affect the exploitation, implementation and expansion of the existing or new recycled water schemes. The findings can also contribute to the development of sustainable recycled water market and the establishment of sound management strategies significantly.

2.2 IMPORTANCE OF RECYCLED WATER

With the social development and population increase, water consumption has increased beyond sustainable levels in many parts of the world, including North America, Australia, the Middle East, the Mediterranean, Asia and Africa (Dolničar and Schäfer, 2009). Climate change, uneven distributed water resources, severe droughts, groundwater depletion and water quality deterioration make the current water supply situation even worse, forcing water authorities and local councils to increasingly consider recycled water as a supplementary water supply (Asano, 2001). This resource can help to alleviate the pressure on existing water supplies, protect remaining water bodies from being polluted and on the other hand provide a more constant volume of water than rainfall-dependent sources (Huertas et al., 2008). In the U.S., recycled water reuse accounts for 15% of the total water consumption, which is tantamount to save approximately 6.4 Gigalitres per day (GL/d) of fresh water. At the same time, environmental loads exerting by effluent discharge can be mitigated to some extent. This strength is fairly distinct as many studies have already demonstrated massive adverse effects on aquatic sensitive ecosystems from wastewater effluent in terms of nutrients pollution, temperature disturbance and salinity increase (USEPA, 2004).

Additionally, Pasqualino et al. (2011) pointed out that replacing potable and desalinated water by recycled water for non-potable purposes (e.g., irrigation, industry, urban cleaning and fire fighting) could result in lower environmental impacts in terms of acidification potential, global warming potential and eutrophication potential. Apart from environmental benefits, recycled water can also introduce some economic benefits to local government or private sectors. For instance, irrigating vineyards at McLaren Vale with recycled water which contains some amount of nutrients has already brought $120 million to the South Australia government (DENR, 2010).

The modern birth of recycled water application was in the mid-19th century along with the prosperity of wastewater treatment technologies. Before 1990s, 70% of reused wastewater was processed to a secondary treatment level by Conventional Activated Sludge (CAS) methods and the effluent was only suitable for agricultural uses in less developed areas. With the rapid development of advanced wastewater treatment technologies such as membrane filtration in the last 10 to 15 years, the application of recycled water has been broadened from non-potable uses (e.g., irrigation, industry, environmental flow and residential uses) to indirect and direct potable reuses. Currently, thousands of water recycling schemes and pilot studies are being carried out worldwide with many more in the planning and construction stages (Pearce, 2008; Rodriguez et al., 2009; Chen et al., 2013a).

2.3 DEFINITION AND SOURCES OF RECYCLED WATER

In some previous literature, water recycling is defined as reclamation of the effluent generated by a given user for on-site use by the same user, such as industry where the recycling system is a closed loop (Asano and Levine, 1996). However, in recent years, there are other more general definitions. Asano and Bahri (2011) stated that water reclamation is the treatment or processing of wastewater to make it reusable while water recycling and reuse are to use treated wastewater in a variety of beneficial ways such as agricultural, industrial or residential purposes. In Australia, the term ‘water recycling’ has been regarded as the preferred term for generic water reclamation and reuse. Sources of recycled water are wastewater effluents coming from previous uses, including greywater, blackwater, municipal wastewater or industry effluents. The stream of recycled water may be comprised of any or all of these waters (ATSE, 2004).

2.3.1 Greywater

Greywater refers to urban wastewater that includes water from household kitchen sinks, dishwashers, showers, baths, hand basins and laundry machines but excludes any input from toilets (Li et al., 2009a). Another definition by Al-Jayyousi (2003) excludes the steam from kitchen wastewater. The quality of greywater varies depending upon the size and behaviour of the residents as well as the volume of water and the chemicals used. Generally, it is less polluted and low in contaminating pathogens, nitrogen, suspended solids and turbidity compared with municipal and industrial wastewaters. However, in some cases, high Biochemical Oxygen Demand (BOD) and Chemical Oxygen Demand (COD) concentrations might be observed, which are attributed to chemical and pharmaceutical pollutants from soaps, detergents and personal care products as well as food wastes in kitchen sinks (Morel and Diener, 2006). With respect to the treatment, physical (e.g., coarse sand and soil filtration, and ultrafiltration) and chemical (e.g., coagulation, photo-catalytic oxidation, ion exchange and granular activated carbon) approaches are suitable to treat low strength greywater (e.g., laundry and showering wastewaters) for either restricted or unrestricted non-potable uses under safe conditions. These treatment technologies are widely used at small-scale residences, which are able to reduce 30-35% of freshwater consumption. Comparatively, for medium and high strength greywater (e.g., kitchen wastewater), additional biological treatment processes such as Sequencing Batch Reactor (SBC), Constructed Wetlands (CW) and Membrane Bioreactor (MBR) are often used to remove biodegradable organic substances (Diaper et al., 2001; Li et al., 2009a). As the involved treatment technologies are relatively simple, easily to be conducted and less costly, its reuse is receiving more and more attention in Australia, Japan, North America, UK, Germany and Sweden. The applications include toilet flushing, garden irrigation, recreational impoundments watering and clothes washing (Pidou et al., 2008).

2.3.2 Blackwater

Blackwater refers to wastewater coming from toilets. It is highly polluted and contains high concentrations of organic pollutants, nutrients and a large variety of micro-organisms such as enteric pathogens. The applications are quite limited due to treatment complexity and strong public objections. Nevertheless, some efforts have still been paid to black water recycling and reuse in regions where severe water crisis occurs. In Australia, one trial conducted in South East Queensland was to reuse black water in sewered areas (DERM, 2009). Other applications were reported sporadically as well, including toilet flushing, agricultural irrigation and outdoor hose tap washing (AWS, 2010). Notably, the nutrient recovery rate of some advanced blackwater stream separation devices, especially for nitrogen and phosphorus, can be as high as 85% (van Voorthuizen et al., 2008). These massive nutrients can be sent back to agriculture to replace industrial fertilizers. Because of the advantages, Sweden and Germany have practiced on advanced dual flush and vacuum urine-separating toilets with more than 3,000 installations (ATSE, 2004).

2.3.3 Municipal wastewater

Municipal wastewater is the largest and most significant resource for water reuse around the world. As separate sewage collection pipelines are often inaccessible and unaffordable in many countries, different waste streams are all discharged into municipal sewage systems. Hence, municipal wastewater normally contains a broad spectrum of contaminants (e.g., organic matters, pathogens and inorganic particles) which can be potential risks to human health and the environment. Particularly, some inorganic chemical pollutants (e.g., sodium, potassium, calcium, chloride, bromide and trace heavy metals) are of particular concern in agricultural and landscape irrigation as highly saline irrigation water can severely degrade the soil quality. Besides, the accumulation of heavy metals in the soil is likely to pose threats to the food chain. Furthermore, when considering the recycled water for IPR and DPR schemes, the trace organic pollutants such as Pharmaceutical Active Compounds (PhACs) and Endocrine Disrupting Compounds (EDCs) are important parameters which can cause adverse effects to health at part per trillion concentrations (Weber et al., 2006). Owing to high hydrophilicity and low adsorption ability, they are poorly removed by CAS.

From microbiological aspects, the main pollution groups in municipal wastewater are excreted organisms and pathogens from human and animal origins. The enteric viruses and protozoan pathogens are significantly more infectious than other bacterial pathogens. To determine the presence of pathogens in recycled water samples, Ecoli, total coliform, Enterococci, Giardia, Campylobacter and Cryptosporidium are commonly used as indicators (Khan and Roser, 2007). Regarding the municipal wastewater treatment, membrane filtration processes perform well in treating total suspended solids (TSS), COD, BOD, microbial pollutants and inorganic compounds such as heavy metals (Table 2.1). Sipma et al. (2010) indicated that MBR is superior over CAS in filtering hydrophobic and low biodegradable compounds such as PhACs and EDCs. Depending on the pore size of the semi-permeable membrane, membrane types include Microfiltration (MF), Ultrafiltration (UF), Nanofiltration (NF) and Reverse Osmosis (RO). MF (0.05-2 μm) membranes typically reject suspended particles, colloids, and bacteria. UF (<0.1 μm) and NF (2 nm) membranes have smaller pores, which can remove natural organic matter/soluble macromolecules and dissociated acids/ pharmaceuticals/sugars/divalent ions, respectively. RO membranes (0.1 nm) are effectively non-porous and retain even many low molar mass solutes as water permeates through the membrane (ATSM, 2010). This advanced technology has received considerable attention in Australia, China, Singapore, the U.S., Canada, Europe and Middle East countries.

2.3.4 Industrial wastewater

Industrial wastewaters are defined as effluents that result from human activities which are associated with raw material processing and manufacturing. The composition of industrial wastewater varies considerably owing to different industrial activities. Even within a single type of industry, specific processes and chemicals that are adopted to produce similar products can differ, which would lead to significant changes in wastewater characteristics over time. Table 2.1 illustrates typical wastewater compositions in several industrial categories including the food, paper and tannery industries. Generally, wastewaters from food processing industries (e.g., potato, olive oil and meat processing) are contaminated with high levels of BOD, COD, oil and grease, TSS, nitrogen and phosphorous. Apart from high COD concentrations, industrial processing wastewaters (e.g., chemical and pharmaceutical producing, paper, textile, tannery, and metal working and refinery wastewaters) might be rich in heavy metals (e.g., Cd, Cr, Cu, Ni, As, Pb and Zn) and other toxic substances. These hazards can potentially pose risks to human health and the environment in terms of waterborne diseases, eutrophication and ecosystem deterioration.

Besides, heavy metals can cause serious health effects, including reduced growth and development, cancer, organ damage, the nervous system damage and even irreversible brain damage (Barakat, 2011; Jern, 2006). To classify these toxic compounds, some toxicity scores or indexes regarding industrial effluents have been developed, which can provide suggestions to wastewater recycling and reuse. Tonkes et al. (1999) developed a four-toxicity-class system which was based on a percentage effect wastewater volume (w/v) ranking. They considered the effective concentration of the organism towards the strongest response at 50% (EC_{50}) value as endpoint (<1% w/v = very acutely toxic; 1-10% w/v = moderately acutely toxic; 10-100% w/v = minor acutely toxic; and >100% = not acutely toxic). Similarly, Persoone et al. (2003) and Libralato et al. (2010) established other toxicity classification approaches in wastewater based on various weighting methods. Toxicity outcomes can facilitate the implementation of best available technologies for wastewater treatment. When toxicity is absent, wastewater might be safely reused. Otherwise, some actions must be undertaken to improve the effluent quality.

According to Table 2.1, MBR is shown to be an effective treatment method, especially in removing low biodegradable pharmaceutical compounds whereas CW can be considered as a relatively low cost option but requires large space for treatment. To treat the heavy metal-contaminated wastewater, Barakat (2011) reported several methods and indicated that new adsorbents and membrane filtration have been the most frequently studied and widely applied in industrial effluent treatment. Specially, the use of biological material (e.g., bacteria, algae, yeasts, fungi or natural agricultural by-products) as biosorbent has received a great deal of interest because of the higher removal efficiency and relatively lower cost compared with conventional methods such as precipitation, ion exchange, etc. (Das et al., 2008; Wang and Chen, 2009). Igwe et al. (2005) demonstrated that the adsorption capacity of maize cope and husk for Pb^{2+}, Cd^{2+} and Zn^{2+} were 456, 493.7 and 495.9 mg/g respectively. Similarly, bacillus could adsorb 467 mg/g of Pb^{2+}, 85.3 mg/g of Cd^{2+}, 418 mg/g of Zn^{2+}, 381 mg/g of $Cu^{2+}$and 39.9 mg/g of Cr^{6+}, respectively (Ahluwalia and Goyal, 2007). Another promising method in the near future would be the photocatalytic approach which consumes cheap photons from the UV-near visible region (Barakat, 2011). After going through sufficient barriers, the treated effluent can be reused as cooling water, boiler feed water or industrial process water in closed industrial processing systems. Alternatively, it might be discharged to centralized municipal treatment plants for external integrated uses (Mohsen and Jaber, 2003).

Table 2.1 The characteristics of major wastewaters and associated treatment methods

Wastewater type	pH	TSS (mg/L)	BOD_5 (mg/L)	COD (mg/L)	TKN (mg N/L)	Total P (mg/L)	Salt (g/L)
Municipal Wastewater							
Municipal wastewater	6-8	6-8	110-400	250-1000	20-85	4-15	<0.5
Treatment process:							
Secondary-UF-RO	–	100%	96%	98%	80.5%	93.5%	–
Secondary- MF		96.3%	42.6%	30%	68.9%	13.7%	
Secondary-Ozonation-MF	–	60%	–	60%	32%	100%	–
Tertiary-SAT	–	100%	99.8%	99%	99.9%	99.1%	–
MBR	–	99%	>97%	89-98%	36-80%	62-97%	–
Industrial Wastewater							
Brewery	3.3-7.6	500-3000	1400-2000	815-12500	14-171	16-124	–
Winery	3.9-5.5	170-1400	210-8000	320-27200	21-64	16-66	0.1-1
Dairy milk-cheese plants	5.2-11	350-1082	709-10000	189-20000	14-450	37-78	0.5
Treatment process: MBR	–	98.9%	97%	88%	10%	–	–
Pulp and paper mill	6.6-10	21-1120	77-1150	100-3500	1-3	1-3	0.05
Treatment process: MBR	–	99.1%	98%	86%	90%	–	–
Tannery industry	8-11	2070-4320	1000-7200	3500-13500	250-1000	4-107	6-40
Treatment process: CW (HRT 7 days)	–	88%	77%	83%	48%	38%	–

Note: Modified from Wang et al., (2004); Arlosoroff, (2006); Melin et al., (2006); Galil and Levinsky, (2007); Oron et al., (2008); Van Houtte and Verbauwhede, (2008); Bielefeldt, (2009); Calheiros et al., (2009).

Abbreviation: % = percentage removal; MBR = Membrane Bioreactor; CW = Constructed Wetlands; HRT = Hydraulic Retention Time; UF = Ultrafiltration; RO = Reverse Osmosis; MF = Microfiltration; SAT = Soil Aquifer Treatment.

2.4 END USES OF RECYCLED WATER

In developed countries, wastewater collection, treatment and reuse have been a common practice. The U.S. and Saudi Arabia are highest-ranked countries associated with the total treated wastewater reuse, while Qatar, Israel and Kuwait are the most noteworthy countries considering the per capita water reuse (Jimenez and Asano, 2008). Comparatively, in many low and middle income countries, irrigation practices often involve the direct use of untreated wastewater. For instance, in Kumasi, Ghana, a population of 2.5 million in 2010, up to 70% of the irrigation water comes from polluted wastewater with the faecal coliform ranges from 104 to 108 CFU/100 ml (Keraita et al., 2003).

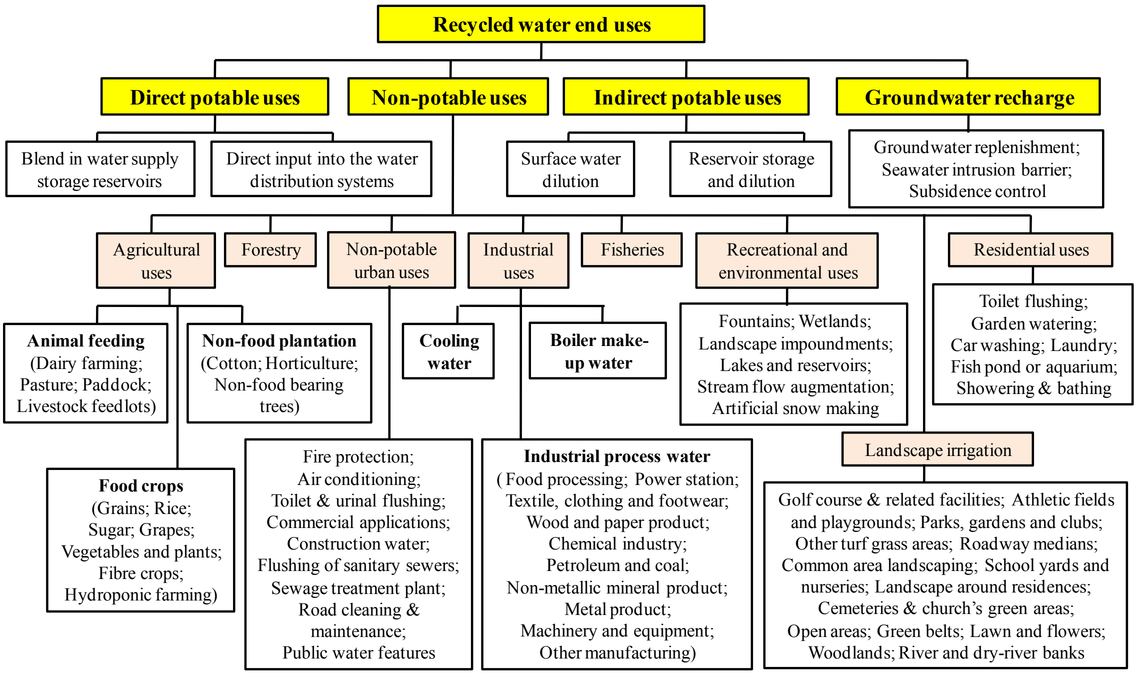


Figure 2.1 Recycled water end uses (modified from Dolničar and Schäfer, 2009; Asano, 2001; Bitton, 2011).

Although some developing countries have begun to conduct wastewater treatment, the treated effluent still fails to fulfill the reuse requirements in some cases. Hence, it can be seen that water reuse situations vary greatly in different countries. The application of recycled water depends heavily on treatment levels, water supply status, environmental conditions, public perceptions and the stringency of waste discharge requirements. According to what degree it might contact with people, the end uses can be generally divided into three categories: non-potable uses, indirect potable uses and direct potable uses. Figure 2.1 illustrates specific end uses in different reuse categories.

2.4.1 Agricultural uses

Wastewater reuse in agricultural irrigation has the longest history that has lasted for 5000 years. As far as 3000-1000 BC, wastewater was used back for irrigation in ancient Greece and the Minoan civilisation (Asano and Levine, 1996). In more recent history, some of the earliest recycling projects for irrigation purposes were implemented in the West U.S. in the late 1920s, along with the publishing of initial water reuse standards in California. At that time, most wastewater effluents only suffered from primary or even no pre-treatment before being utilized in agriculture, triggering health risks and environmental pollution issues potentially. Till the 1980s, primary effluents were still provided for irrigating fodder, fibre and seed crops while secondary treatment was the minimum criteria for food crops and pastures' irrigation in California and France. The guidelines were sketchy and controversial but have allowed a real development of wastewater reuse (Bahri, 1999). In the 1990s, water reuse on agriculture had rapid growth around the world. Despite the technical feasibility of advanced treatment has been achieved, high quality effluents were seldom applied to agriculture due to cost and nutrient lost issues. Accordingly, elaborate water quality guidelines were published over time which became more stringent than earlier ones and mostly regarded secondary and disinfection processes as the minimum requirements (Table 2.2).

Currently, agricultural irrigation still represents the largest use of recycled water throughout the least developed regions (e.g., Middle East, South America and North Africa). Comparatively, in most developed regions (e.g., Australia, Japan, the U.S. and Europe), the number of urban reuse schemes is as high or much higher than the number of agricultural irrigation schemes (Brissaud, 2010). For example, in Australia, the fraction of recycled water used in agriculture decreased from 66% to 29% over the period 2004 and 2009. So far, there are about 270 different agricultural irrigation schemes across the country, using 106 Gigalitres per year (GL/yr) of recycled water. Nonetheless, considering the annual total water consumption in agriculture (7300 GL in 2008-09), the contribution of recycled water was small, which only accounted for 2%

(ABS, 2010). The proportion is being improved and correspondingly, Australian government has published its national recycling guidelines with overall 4 classes of water quality. Likewise, most of the states have their local guidelines that are slightly different from others. Normally, for raw human food crops and vegetables, Class A treatment comprising of tertiary and disinfection is required. While for processed or cooked crops, pastures and fodder used for dairy animals, and non food crops, lower effluent quality (secondary treatment as a minimum) is permitted. There are several large-scale irrigation schemes being successfully implemented in Australia, including the Virginia Pipeline Scheme in Adelaide (18 GL/yr) and the Eastern Irrigation Scheme in Melbourne (11 GL/yr). Remarkably, the Shoalhaven Water's Reclaimed Water Management Scheme (4 GL/yr) in New South Wales has converted the region from dry land to a dairy farm without introducing extra charge and environmental problems (ATSE, 2004).

Table 2.2 Historical recycled water use restrictions and guidelines in agriculture

Time period	Water quality guideline	Minimum treatment required		Water quality criteria
In the 1920s	CSBH[1], 1918	Restricted crops	30 days settlement	–
In the 1980s	WHO[2], 1989	Very restricted crops	Sedimentation & pre-treatment	TC[3] /100 mL <1000
		Restricted crops	8-10 days in WSP[4]	Helminths eggs <1
		Without restrictions	Series of WSP	
In the 1990s	US EPA[5], 1992	Food crops eaten raw	Secondary, filtration and disinfection	BOD <10 mg/L; SS[6] <5 mg/L; FC[7] /100 mL – ND[8] Cl_2 after 30 min >1 mg/L
		Restricted access areas and processed food crops	Secondary and disinfection	BOD <30 mg/L; SS <30 mg/L; FC/100 mL <200 Cl_2 after 30 min >1 mg/L
	Cyprus, 1997	Tertiary (filtration and disinfection)		BOD <10 mg/L; SS <10 mg/L; FC/100 mL <50 Helminths eggs/100 cm^3 – ND[8]

Note: Modified from Kretschmer et al., (2004); Asano et al., (2007); Bitton, (2011).
[1]California State Board of Health; [2]World health organization; [3]Total coliform; [4]Waste stabilization ponds; [5]Environmental Protection Agency; [6]Suspended solids; [7]Faecal coliform; [8]Non-detectable.

In Europe, the wastewater reuse projects for agricultural irrigation in France and Italy cover more than 3,000 and 4,000 ha of land respectively. Moreover, in Spain, the volume of recycled water use in agriculture has amounted to 780 Megalitres per day (ML/d) by the year 2002, accounting for 82% of the total water reuse (Jimenez and Asano, 2008). Presently, one of the largest reuse schemes in Northern Spain is in the City of Vitoria, which supplies 35 ML/d of recycled water for the 9500-hectare spray irrigation. The initial commitment of the project which is to produce high quality recycled water for unrestricted irrigation has benefited from its wide acceptance among current and potential users (Asano and Bahri, 2011). Likewise, in Greece, agricultural irrigation is the main interest for reuse where 20 ML/d of treated wastewater irrigates olive trees, cotton, forest and landscape at regular. Among the total reused water, 3.5 and 0.4 ML/d of effluent from Levadia and Amfisa Wastewater Treatment Plant (WWTP) is used for cotton and olive tree irrigation respectively (EWA, 2007). When it comes to the Mediterranean and Middle East areas, most of the countries have progressively used recycled water for irrigation, especially Israel, Tunisia, Cyprus and Jordan (Angelakis et al., 2003). In Israel, 75% of recycled water is used for agriculture with irrigation of 19,000 ha (Shanahan, 2010). The three largest recycling systems are located in Kuwait, Israel and Saudi Arabia, which reuse 375, 310 and 595 ML/d tertiary treated recycled water in agricultural irrigation respectively (Jimenez and Asano, 2008).

Unlike developed countries which are seeking various end uses of recycled water continuously, wastewater in developing countries is reused predominantly in agriculture. In Asian countries such as India and Vietnam, over 73,000 ha and 9,000 ha of land were found to be irrigated by wastewater respectively. In Jordan, almost 100% of the treated effluent is utilized for irrigation with an area of 13,300 ha either directly at the outlet of the WWTP or after being discharged into reservoirs (Mekala et al., 2008). In Egypt, about 42,000 ha are irrigated with treated wastewater or blended water, where the irrigation area is estimated to reach 210,000 ha by the year 2020. However, IWMI has pointed out that about 46 developing countries are using polluted water for irrigation purposes, at least 3.5 million ha were irrigated globally with untreated, partly treated, diluted or treated wastewater until 2006 (IWMI, 2006; Qadir et al., 2010). In these countries, unplanned and uncontrolled wastewater reuse projects were conducted regardless of health and environmental issues because of limited treatment conditions, socio-economic situations and public recognitions (IWMI, 2010). For example, in Asian countries, this situation is common in Pakistan where nearly 80% of the crop was irrigated by raw sewage, which resulted in enteric diseases and gastrointestinal illnesses. While in Syria, it was reported that in Damascus, some untreated wastewater was discharged to agricultural lands directly, leading to the degradation of surface water and groundwater, especially in the Barada River and Aleppo southern plains. Similarly, the Mezquital Valley, Mexico, also used approximately 3.9-25.9 GL/d of raw wastewater to irrigate over 85,000 ha of crops in the Valley of Mexico and surrounding areas, where the disease spreading was observed as well (Jimenez and Asano, 2008).

2.4.2 Landscape irrigation uses

Recycled water use in landscape irrigation has been practiced around the world for more than 50 years. Significant development has occurred in the last 20 years as a result of growing water demand, increasing costs of acquiring additional water in urban areas and stringent wastewater discharge requirements (Asano et al., 2007; Lazarova and Bahri, 2004). Currently, landscape irrigation has become the second largest user of recycled water in the world. However, the particular water demand for different countries and regions varies greatly by geographical location, season, plants and soil properties.

In Australia, there are around 240 out of total 600 recycled water schemes being applied to urban environmental irrigation. Many of them have been operating for more than 20 years without negative impact on human health or the environment (Stevens et al., 2008). In the U.S., it even represents the largest use of recycled water in Florida and the Irvine Ranch Water District, California, as the state governments recognise that the landscape irrigation schemes are easy to be implemented, especially wherever potable water is used in urban areas. In regard to landscape irrigation applications, the end uses listed in Figure 2.1 can be further categorized into unrestricted access areas and limited or restricted access areas (Tables 2.3), in which different water reuse quality guidelines will be established (Lazarova and Bahri, 2004).

Table 2.3 Landscape irrigation categories

Unrestricted access areas	Limited or restricted access areas
Public parks	Cemeteries
Playgrounds, school yards and athletic fields	Highway medians and shoulders
Public and commercial facilities	Landscaping within industrial areas
Individual and multifamily residences	Green belts
Golf courses associated with residential properties	Golf courses not associated with a residential community

Note: Adapted from Asano et al., (2007).

Table 2.4 Water reuse guidelines for wastewater reuse around the world

Country	Application	Parameters					
		BOD[1]	TSS[2]	Turbidity (NTU)	FC[3]	TC[4]	RC[5]
Victoria, Australia	Unrestricted	<10	<5	<2 (24 hr median); <5 (max)	*E.coli* <10	–	>1 after 30 min
	Restricted	<20	<30	–	*E.coli* <1000	–	–
New South Wales, Australia	Surface irrigation	90%<20 max 30	90%<30 max 45	–	–	90%<30; max 100	>0.2; <2
	Toilet	90%<10 max 20	90%<10 max 20			90%<10; max 30	>0.5; <2
Canada	Unrestricted	<10	<5	<2	2.2	–	–
China	Irrigation	<20	<1000	<20	3	–	>1 after 30 min
	Toilet	<10	<1500	<5	3	–	>1 after 30 min
	Washing	<10	<1000	<5	3	–	>0.2 at point of use
Japan	Irrigation	–	–	<2	ND[6]	<1000	free > 0.1, combined >0.4
	Toilet	–	–	<2	–	ND[6]	
Germany	Unrestricted	<20	<30	<1-2	<100	<500	–
Spain	Unrestricted	–	<10	<2	ND[6]	Helminth Egg/10 L <1	–

Note: Modified from Lazarova and Bahri, (2004); Pidou, (2006); Asano et al., (2007).
[1]BOD (mg/L); [2]TSS (mg/L); [3]Faecal Coliforms per 100 mL; [4]Total Coliforms per 100 mL; [5]Residual Chlorine (Cl_2, mg/L); [6]Non-detectable.
Abbreviation: Unrestricted = Unrestricted urban irrigation; Restricted = Restricted area irrigation; Toilet = Toilet flushing.

As can be seen in Table 2.4, the control of important parameters on each guideline over the unrestricted access areas is so critical that tertiary treatment including filtration and disinfection is usually required as these places are mostly located in urban areas and have frequent contact with people. Generally, as unrestricted access areas are widely distributed everywhere, there are more reuse schemes (e.g., parks, golf courses, gardens, ovals and play fields) related to these areas. However, restricted access areas have less exposure to people and the risk control can be more easily conducted, thus secondary effluent is acceptable.

2.4.2.1 Golf course uses

Nearly half of landscape irrigation schemes are related to golf courses. One of the successful examples is the Darwin Golf Course in Australia, where 450 Megalitres per year (ML/yr) of recycled water provided by the Darwin Golf Course Sewage Treatment Plant (STP) has well connected with the golf course irrigation. In addition to course irrigation, part of the water that sent to the golf course ponds is further utilised in sport fields such as Marrara Sports Complex, saving a considerable amount of fresh water (ATSE, 2004). In the U.S., the average annual water consumption in golf course is 190-230 ML in the East Coast and 300-380 ML in the southwest. In the city of North Las Vegas, a 113 ML/d reuse plant with MBR system was commissioned in May 2011 and treats wastewater for golf course irrigation. Apart from this initial reuse option, the city also plans to provide recycled water for commercial laundries in hotels and concrete mixing plants where recycled water can be used as cooling water in natural gas co-generation facilities as well as dust control water on construction sites (McCann, 2010).

When it comes to Europe, Spain is a representative country where 4 golf courses in Costa Brava, the northeast Spain, have used recycled water as the sole source for irrigation since 2004 (Sala and Millet, 2004). The 2010 scenario of the Spanish National Water Plan has even specified the compulsory use of recycled water for golf course irrigation in many water basins (Candela et al., 2007). Furthermore, the largest project of its kind in the world is the Jumeirah Golf Estates (220 ML/d) in Dubai, the Middle East, which equips an advanced wastewater collection, treatment and tertiary effluent reuse system. In Tunisia, using recycled water in golf course even becomes an important component of the tourism development, where at least 8 golf courses are irrigated with secondary treated effluent (GWA, 2008).

2.4.2.2 Public parks, schools and playgrounds uses

Irrigating public areas with recycled water is also widely conducted. However, concerns have been raised owing to the high potential risk of accidental recycled water ingestion, especially when children fall to or touch the grass and then have hand-to-mouth contact. These can be solved by applying risk control approaches. For example, in Australia, by adopting limiting daytime watering hours and locking the entrance gate when irrigating, the landscape irrigation scheme (580 ML/yr) in the Alice Springs, Northern Territory, is able to minimize the risk exposure of recycled water to people (ATSE, 2004). With respect to China, the Qinghe Water Reclamation Plant (WRP) in Beijing has successfully provided UF treated effluent for the 2008 Beijing Olympic Games. Among the total capacity of 80 ML/d, 60 ML/d has been used as water supply for landscaping the Olympic Forest Park. The remaining 20 ML/d has been used for road washing, toilet flushing and other purposes. The second phase of the plant with a daily average capacity of 320 ML/d and peak capacity of 450 ML/d has been commissioned by the end of 2010, which has become China’s single largest water reuse scheme (GreenTech, 2005). Besides, the Chongqing University, located in southwest of China has conducted a Chongqing greywater demonstration project at the new Huxi campus in 2005. The greywater from a 21 high rise teaching building is collected and treated onsite by CWs. The treated effluent is blended with rainwater and used for landscape irrigation and scenic lake replenishment. This project is capable of reducing the annual potable water consumption by 150 ML (SWITCH, 2008). Moreover, in the Middle East, to ensure the health and environmental safety, the city of Abu Dhabi has treated tertiary recycled water (200 ML/d) with supplement sand filtration and chlorine disinfection before irrigation, which has allowed the city to be a garden city despite high temperature and low rainfall (Jimenez and Asano, 2008).

2.4.2.3 Residential landscape uses

Residential landscape irrigation schemes mainly use the recycled water that comes from municipal wastewater and greywater sources. They are sometimes coupled with other residential end uses such as toilet flushing, car washing and clothes washing. When the water is delivered from area-wide centralized distribution systems, care must be taken to prevent the cross connections of dual pipelines. Generally, the recycled water quality is complied with guideline values for unrestricted areas (Table 2.4). In Australia, the Ipswich Water's Carole Park STP, Brisbane, supplied 1.2 GL/yr of recycled water from a tertiary disinfection reservoir to Springfield with 18,000 homes. This water was used for irrigating residential areas including road verges, grass areas and median strips as well as the Bob Gibbs Park (ATSE, 2004). Besides, the U.S. has another pilot study in California named the El Dorado Hills residential irrigation project. From 2005, the Serrano community in this area has been using recycled water from the Deer Creek WWTP and the El Dorado Hills WWTP to irrigate all front yards of 6100 residences by dual pipe systems. Meanwhile, more than 60 million American people are using decentralised recycling systems which operated onsite individually for their front yard and back yard irrigation (Asano et al., 2007). Owing to broad public acceptance and less stringent water quality requirement compared with potable uses, water reuse in landscape irrigation will possibly experience high development in the future.

2.4.3 Industrial uses

Recycled water has been successfully applied to industry in Japan, the U.S., Canada and Germany since the Second World War for more than 70 years. Recently, industrial use is the third biggest contributor to recycled water consumption. In Australia, because of the severe drought conditions and mandatory water restrictions, industrial recycling schemes have been expanded to about 80 together with the acceleration of the reuse rate by 25% in most industrial sectors (Stevens et al., 2008). In Asia, Japan is one of the world's leading countries in this kind which had achieved a 76.3% of water recovery rate within industrial sectors by 1992 (Schmidt, 2008). Comparatively, the U.S. has the longest history of water reuse in industry. During the 1940s, the prologue of industrial application has been unfolding gradually in the U.S. with the start of using chlorinated wastewater effluent for steel processing (Asano and Levine, 1996). Till the 1990s, the concept of zero discharge which means total reuse without any wastewater being released into the environment was also put forward in the U.S. and Germany. Besides, industrial use occupied 33% and 55% of the total recycled water in northern Europe and Sweden, respectively (Bixio et al., 2006). The major industrial categories associated with substantial water consumption include cooling water, boiler feed water and industrial process water (Chiou et al., 2007).

2.4.3.1 Cooling water

Cooling water creates the single largest industrial demand for water (more than 50%) and becomes one of the predominant areas for water saving and reuse in industry (Asano, 2001). Equipment or processes in refineries, steel mills, petrochemical manufacturing plants, electric power stations, wood and paper mills and food processing all require efficient temperature control to ensure the safety and efficiency. In electric power generation plants, cooling water accounts for nearly 100% of water use. While in other industries, the proportion can range from 10% in textile mills to 95% in beet-sugar refineries. Generally, in the US, more than 90% of water consumed by industries is attributed to cooling purposes in comparison with 70% of that in Japan (Schmidt, 2008). Regarding to cooling systems, non-evaporative ones have been gradually replaced by recirculating evaporative systems which uses water to absorb process heat and transfer the heat by evaporation either in cooling tower or spray ponds. As the evaporative systems are recirculating continuously, recycled water is mainly used as makeup water to recover the evaporation loss, which brings large benefits in terms of thermal pollution mitigation and water conservation. However, it is required to be of high quality due to corrosion, biological growth, scaling, fouling and salinity concerns (USEPA, 2004). In this case, additional processes such as coagulation, precipitation and ion exchange to removal total dissolved solids (TDS) are usually specified in guidelines.

For example, in Asia, Wang et al. (2006) conducted a pilot study at the North China Pharmacy Limited Company. They indicated that the product water from both sand filtration/MF/RO and sand filtration/UF/RO systems could fulfill the cooling water quality requirements. Accordingly, the company used 400 ML/d of treated effluent from the Gaobeidian WWTP as industrial cooling water (Jiang, 2004). You et al. (2001) studied the water reuse in a semiconductor factory in Taiwan. The RO devices generated ultra pure deionized water from tap water to rinse the integrated circuit crystal chips while the RO reject (230 kilolitres per day) was reused as cooling water. They demonstrated that the pre-treatment of the reject water was uneconomical. Increasing the cycles of concentration and reducing the quantity of make-up in the cooling water system would be preferable in the plant. Additionally, the thermal power generation plants of MahaGenco Company at Koradi and Khaparkheda, India, reuse 110 ML/d of treated water for cooling purposes predominantly. The company is going to use treated water constantly for the next 30 years, which will directly benefit 1 million people due to a significant amount of freshwater savings (USAID, 2009). Nevertheless, a clear water quality standard should be specified later as there are still no guidelines associated with recycled water reuse in industries in India (Jamwal and Mittal, 2010). Similarly, in the US, there are numerous petroleum refineries and power stations in California that have used 100% of recycled water for their cooling systems since 1998 (USEPA, 2004). Vourch et al. (2005) conducted studies in 10 French industrial dairy plants and concluded that both total organic carbon (TOC) and conductivity of water treated by a single RO or an NF/RO operation were satisfied for reusing as cooling water. Furthermore, public objection towards recycled water in industrial cooling is as low as 3%, compared with 16% and 53% in agriculture and drinking respectively (Dolničar and Saunders, 2006).

2.4.3.2 Boiler feed water

As boiler feed water plays an important role for the operation of steam generators in many industrial types, the recycled water should be of very high quality. High quality water helps to avoid boiler corrosion, deposits, sludge formation, scaling, fouling and foaming, especially when the boiler is operated under high pressure. Thus, advanced treatments such as UF, RO, demineralization or ion exchange are often required. Mann and Liu (1999) listed the feed water quality requirements for low, medium and high pressure boilers. Other international or local guidelines are also published. Successful examples in Australia include the BP Amoco Company, Brisbane, where 10.6 to 14 ML/d of recycled water from Luggage Point STP with MF and RO treatment steps, is used as boiler feedwater at the refinery (ATSE, 2004). The Eraring Power Station, Lake Macquarie, also uses 1.2 GL/yr of purified recycled water as boiler feed water from Dora Creek STP with MF, RO and demineralisation approaches, to provide steam for driving turbines (Anderson, 2003a). In Asia, the Gaojing Power Plant in Beijing, China, adopts UF/RO membranes to treat the blow-down from its cooling towers and reuses the treated effluent as boiler feed water. The integrated UF/RO treatment system is able to overcome the problems associated with high hardness, alkalinity, silicon dioxide and sulphate which are typically found in cooling water blow-down. Since 2003, around 70% of water in cooling tower has been reused (DCC, 2009). Additionally, the Dagang Oilfield Reclaimed Water Plant in Tianjin, China, commissioned in 2009, uses a submerged MBR (30 ML/d) to treat a 50/50 combination of oil industrial wastewater and local municipal secondary effluent. The treated effluent is sent to a nearby power plant, polypropylene plant and coke calcination plant for cooling and boiler feed water supply purposes (Mo and Chen, 2009; Zheng, 2010).

In the U.S., several refineries in California have also used recycled water as a primary source of boiler water since 2000. The Californian West Basin Municipal Water District guidelines on recycled water prescribed that pure RO is necessary for low pressure boiler feed in refineries while ultra-pure RO is essential for high pressure boiler feed in refineries (USEPA, 2004). Furthermore, in the Middle East, the world's largest produced water reuse project is the Mukhaizna Water Treatment Facility (47.7 ML/d) in Oman which has been operated since late 2008. The plant uses 7 mechanical vapour compression (MVC) brine concentrator trains to treat produced water from oil and gas extraction and then reuses high purity distillate as boiler feed water for steam generation. This project has attracted widely public attention from 2009 because of its scale and the first time to adopt a novel and integrated MVC treatment technology in water reuse sector in the Middle East. Currently, the water reuse rate is as high as 90% and the plant is planning a zero liquid discharge configuration at a later stage (GWA, 2009a). More recently, a remarkable project at an oil recovery plant in the partitioned neutral zone between Saudi Arabia and Kuwait has become the first successful large-scale produced boiler water system for steam generation in an enhanced oil recovery application in the Middle East. The plant has de-oiling and de-gasification pre-treatment facilities and recycles untreated oily sour produced water originating from a carbonate oil reservoir, producing up to 35,000 barrels per day of high-purity distillate for high-pressure boilers. It is also an energy saving plant which only uses 5% of the energy normally required for single-effect steam evaporation (GWA, 2010).

2.4.3.3 Industrial process water

In industry, lots of processes (e.g., dust, pollution and fire control and suppression, acid and alkali dilution, plant and equipment rinse, raw material and product washing, friction reduction and lubrication, etc.) involve using substantial amounts of water (Huertas et al., 2008; VU, 2008). The required recycled water quality depends on particular end uses. Generally, low quality water is acceptable for the tanning industry; medium quality water is suitable for pulp and paper, textile and metallurgical industries while only high quality water can be adopted in electronics, food processing, chemical and pharmaceutical industries (USEPA, 2004). Wastewater reuse in textile, paper and metallurgical industries has been studied for several years. Thus, many recycling schemes have been successfully conducted and much higher water recycling and reuse rates have been reported.

(1) Pulp and paper mill industry

The pulp and paper making industry is highly water intensive, which ranked third in the world after primary metals and chemical industries. In terms of water quality, reusing the effluent within the pulp and paper mills may increase the concentration of organic and inorganic pollutants, which can affect paper formation, increase bacterial loading or cause corrosion and odour. The efficiency of chemicals may also be affected by the quality of preparation water. Besides, the wires must be kept clean to achieve an optimum paper sheet and drainage (Asghar et al., 2008). Hence, to ensure good paper quality and high recycling rate, the water introduced in the paper machine must meet high quality requirements. Nowadays, general water quality guidelines have already set tertiary treatment with colour removal as the minimum level (VU, 2008). In Asia, the Anand Tissues Ltd., located in Fitkari, Uttar Pradesh, India produces unbleached kraft paper and adsorbent paper, and uses recycled water in paper producing sectors. About 20% of the final effluent from activated sludge treatment is recycled to the pulp digester while wastewater generated from the pulp mill and the paper machine is reused for pulp washing. The company also recycles water discharged from the paper machine, the pulp washing stream and the retentate from raw water RO plant (Tewari et al., 2009). In the U.S., water reuse in the paper industry started in the 1950s, during which, freshwater consumption has been reduced by 23%, from approximately 568 ML per ton at the beginning to 133 ML per ton. Between 1955 and 1972, water consumption has been further reduced to 102 ML per ton. Currently, many modern mills have already achieved 100% recycling rate, using only 61 ML or less of freshwater per ton (USEPA, 2004).

Meanwhile, in Europe, Ordóñez et al. (2011) studied the different recycled water treatment systems in HOLMEN Paper Madrid in Spain. The results showed that both the MF/RO/UV and UF/RO/UV systems achieve constant permeate quality with the salt rejection rate above 99%, the coliform concentration below 1 CFU/100 mL and final COD concentration below 5 mg/L. Hence, the company successfully produces 100% recycled paper using 100% recycled water. Mänttäri et al. (2006) conducted a study at Stora Enso Kotka mill in Finland and indicated that the pulp and paper mill effluents treated by activated sludge processes could only be reused for the production of packaging paper. They also found that when the monovalent ion content was low, recycled water by biological pre-treatment plus NF was suitable to be used in the manufacturing processes. In high strength wastewater, low-pressure RO membranes were required to remove monovalent anions and dissolved inorganic carbon. Moreover, Koyuncu et al. (1999) used a UF/RO system to treat pulp and paper mill effluents in Turkey. The overall removal efficiencies of COD, colour, conductivity, NH_3-N were found to be 90-95%, 95-97%, 85-90% and 80-90% respectively together with 85-90% recovery rates after integrated membranes. As the effluent was of very high quality, it could be reused as process water internally. Furthermore, the Mondi Paper Company in Durban, South Africa, uses 47.5 ML/d of recycled water from the Durban Water Recycling Plant, suffering tertiary, ozonation and activated carbon treatment. As a result, considerable water savings in Mondi have been achieved and the water tariff has been reduced by 44% (Holtzhausen, 2002).

(2) Metallurgical industry

Metallurgical industry is the largest water consumption sector among all industrial types in some countries where sinter plant, blast furnace, cold rolling and other processes have high potentials to recycle 80-90% of wastewater (Johnson, 2003). Generally, secondary or tertiary treated recycled water may be suitable for most applications in this category while for sensitive processes such as hot rolling, electroplating and finishing, MF/RO processes may be required (VU, 2008). There are many water reuse schemes regarding metallurgical industry around the world. For example, in Australia, the Port Kembla Steelworks used 20 ML/d of recycled water from the Wollongong STP, saving 130 ML/year of freshwater (Bluescope Steel, 2006). The recycled water was under

MF/RO treatment and used in a wide range of processes including cooling metal, cooling tower makeup, process water for cleaning and rinsing strip, steam generation for heating purposes, dust suppression and washing. Till 2006, the recycled water quality in Port Kembla Steelworks was superior to guideline quality in Sydney (Table 2.5). Besides, the company has also conducted interdepartmental water reuse schemes and installed a 300 KL/d onsite WWTP to provide secondary effluent for internal quench basins. The project was planned to be expanded to 35 ML/d and possibly 50 ML/d (Hird, 2006).

Table 2.5 Comparison of recycled water quality in Kembla Steelworks and guidelines

Important parameters	Industrial water quality	Required recycled water quality in Sydney
Chloride (mg/L)	14.6	20
Hardness	9.5	<20
pH	5.8-6.7	6.5-8.5
Parasites (per 50 mL)	Non-detectable	<1
Viruses (per 50 mL)	Non-detectable	<2
Coliform (per 50 mL)	Non-detectable	<5

Note: Modified from Hird, (2006).

Similar to Port Kembla Steelworks, Port Kembla Coal Terminal also receives recycled water from the Wollongong STP and has been using it for dust suppression since 2009, reducing the fresh water consumption by 70%. Moreover, a new technology using filtration, de-ionisation and UV treatment to process wastewater from the electroplating has been introduced at Astor Metal Finishes Villawood factory in Sydney, which is capable of recovering most of the wastewater (NSWOW, 2010). Besides, the steel industry in China also benefits from recycled water use. The Taiyuan Steel Plant, Shanxi and the Handan Steel Plant, Hebei are both using submerged membrane/RO system for treating combined industrial and local secondary effluents. They provide 50 and 48 ML/d of treated water for internal industrial process uses, respectively (Zheng, 2010).

(3) Food processing industry

Although the use of food processing wastewater for irrigation purposes has been widely reported, it could be more efficient to reuse these effluents within the same industry. Hiddink et al. (1999) pointed out that a great potential for water recycling and reuse in the food industry seemed possible to reduce the water consumption by 20-50%. Till now, most food processing industries have recycled partial wastewater effluents for non-food and plant cleaning, washing or cooling processes. However, seldom of them reused the water for food preparation and processing. Some of the currently acceptable direct reuses are initial washing of vegetables, fluming of unprepared products and scalding water of meat and poultry (Rajkowski et al., 1996). As the quality of food products obtained through recycling or reusing wastewater should be at least equal to that of the food product obtained using tap water, the wastewater treatment system is required to remove undesirable physical, chemical and microbiological components, especially the pathogenic and spoilage-causing organisms. With respect to case studies, in Australia, the Mars Food Water Recycling Project in New South Wales uses UF, RO and UV disinfection to treat both wastewater streams from the food manufacturing process and stormwater onsite and reuse them for non-product utility purposes, saving 355 ML/yr of water (GWA, 2010). Based on a poultry processing plant in Brazil, Matsumura and Mierzwa (2008) found that pre-chiller effluent could be reused during chilling processes or for other non-potable applications after UF and water from gizzard machine could be reused as cascade water in inedible viscera flume without pre-treatment. After filtration, wastewater from thawing and filter wash process might also be reused. By adopting water reuse programs, water consumption was reportedly reduced by 21.9%.

Furthermore, Blöcher et al. (2002) conducted a one-year study on water reuse at a fruit juice production plant in Germany. The plant used MBR plus two-stage NF treatment system. In the MBR, high COD removal (>95%) was achieved. After the two-stage NF filtration, the chemical and bacteriological parameters of the treated water met the limits of the German Drinking Water Act with a water recovery of 81%. Therefore, the treated water can be reused for various purposes (e.g., boiler make-up water, cooling water, pasteurisation or bottle pre-washing). Besides, after investigating the use of several NF and RO membranes in 10 French industrial dairy plants to produce water for reuse, Vourch et al. (2005) concluded that both the single RO and NF/RO treated waters are capable of reusing as cooling water in terms of TOC concentration and conductivity. Hafez et al. (2007) reported the reuse of treated water effluent of the EL-Nile Company for the food industry in Egypt. The wastewater samples were generated from fruit juice and milk products lines and processed by MF/UF/NF/RO system. The WWTP treated 1.2 ML/d of wastewater, in which only 0.9 ML/d of water was processed through RO that can be reused in high pressure boilers. The water resulted from NF (0.3 ML/d) can be reused in industrial processes and low pressure boilers. Nevertheless, there are also many limitations in the implementation of water reuse in the food industry due to high water quality requirements and strong public objections. The city of Toowoomba in Queensland, Australia, could be a good demonstration. As the critical water situation has occurred and level 5 water restrictions have been employed, the water recycling project in Toowoomba food industry was initially supposed to achieve a significant freshwater saving. Despite that the six star water quality has far exceeded the drinking water quality specified in Australia Drinking Water Guideline (ADWG), strong public objections have lead to its failure (Hurlimann and Dolnicar, 2010a). Overall, although water recycling and reuse have been widely conducted in many industries for years, there is still a potential to improve the recycling rate in many processes and sectors further. For example, in Coke making and Plate mill industries, water reuse rates only account for 0-30% (Johnson, 2003). Water reuse in pharmaceutical industries is also stagnant because of psychological issues. These situations are waiting to be improved in the future.

2.4.4 Environmental and recreational Uses

Releasing high quality water has benefited many environmental wetlands and water bodies by providing sufficient water for ambient wildlife and regional water cycle. Recycled water in turn can be further purified by wetlands before discharging to receiving water bodies or permeating into groundwater aquifers. In Australia, Queensland was a leading state with nine experimental wetlands constructed in the north part in 1992-94 to further treat secondary effluent and another two projects in south-east part in 1995 (Greenway, 2005). After detention in wetland, the effluent quality can be largely improved so that the treated water can either be used for wildlife habitation or reused in other fields. Likewise, in the U.S., recycled water from Iron Bridge Plant was supplied to a wetland and then finally discharged into the St. Johns River in Orlando, Florida, breeding hundreds of aquatic animals and plants. House et al. (1999) also confirmed the feasibility of constructing wetlands to treat and recycle 4.5 ML/d of domestic effluent for toilet flushing in North Carolina. In Europe, wetlands have been studied for more than 30 years together with over 100 CWs in Czech Republic (Vymazal, 2002).

While the main objective of recycled water for environmental uses is to protect the ecosystem and public health, human health concern is the primary issue for recreational uses. Depending on the likelihood of human exposure to recycled water, recreational uses can be further categorized into unrestricted and restricted access uses. Unrestricted recreational use includes wading and swimming while restricted use consists of fishing, boating and other non-body contact activities (USEPA, 2004). Generally, this category requires frequent water quality monitoring in terms of pathogen and nutrient concentration, colour, odour and temperature due to potential high exposure to the public. Class A treatment with tertiary and pathogen reduction is also essential (EPA Victoria, 2003). Successful illustrations include the Santee Recreational Lakes project in San Diego, the U.S., where 4 ML/d of Class A recycled water was supplied to supplement evaporation water loss for ensuring fishing, boating and view watching activities without any significant change in water quality. Apart from this, recycled water is also extensively applied for stream flow augmentation in the San Luis Obispo Creek in California and San Antonio River in Texas (Asano et al., 2007).

2.4.5 Non-potable urban and residential Uses

Many developed countries, including Australia, Japan, the U.S., the UK and Germany have recognized the opportunity of substantial water saving from urban areas by constructing dual-reticulation pipe systems (Figure 2.2). These systems can supply recycled water for applications including fire fighting, toilet flushing, garden irrigation and car washing. As a result of high risk exposure to customers and the confirmed community reservations, recycled water is subject to class A or even higher standard. In Australia, one representative example is the Water Reclamation and Management

Scheme (WRAMS) in Sydney Olympic Park. It has extended the urban water recycling concepts to integrated water management by incorporating both stormwater and recycled water in recycled water delivery systems. The novel stormwater reservoir design enabled stormwater from the Olympic Park and excess secondary effluent from STP to be stored and regulated so that the subsequent WRP can be operated at any rate to cope with large events. Up to 7 ML/d of recycled water under MF, UV and super-chlorination was used for toilet flushing and open space area irrigation at sporting venues in Olympic Park, saving 850 ML/yr of Sydney's freshwater supply. The additional recycled water also served 2000 residential houses in Newington in terms of toilet flushing and garden watering. Recently, the end uses have been expanded to over eleven types, including swimming pool filter backwash and ornamental fountains (Chapman, 2006). The Beijing Capital International Airport wastewater reuse project in China, Asia, is another showcase which serves 20,000 visitors per day. The UF/RO facility (10 ML/d) supplies treated water for toilet flushing in airport office buildings and the Airport Hotel. The excess water is used for vehicles washing, plant irrigation, roads cleaning and cooling (DCC, 2008).

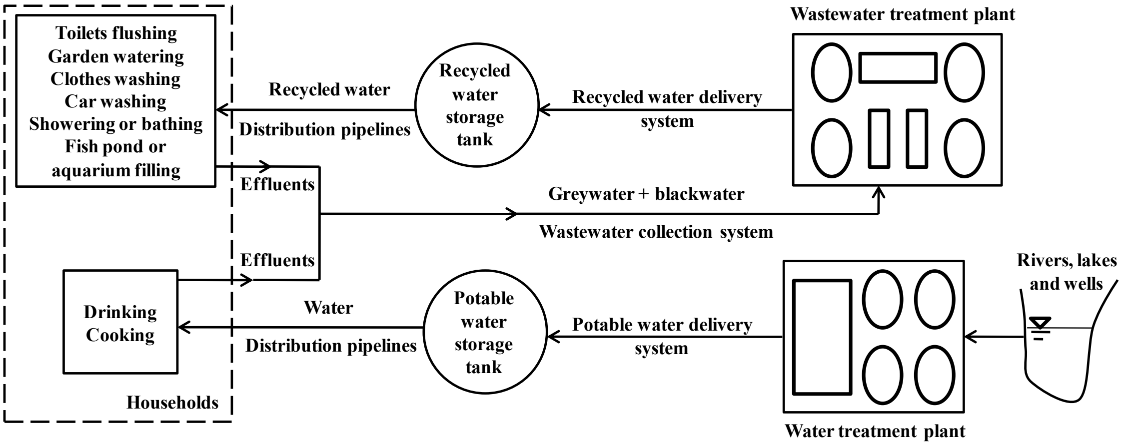


Figure 2.2 Simplified centralised dual pipe system in residential areas

Apart from centralised systems, onsite water recycling schemes are widely operated as well, especially in rural areas. For instance, a two unit family dwelling called Toronto Healthy House was built in Canada in 1997. The house had its own water recycling system (120 L/d) with 4 levels of treatment, including anaerobic, bio-filtration, sand and carbon filtration and UV processes and the water was usually recycled up to 5 times. Treated water was then used for toilet flushing, laundry, bathing and garden irrigation.

This house has also collected rainwater for drinking purposes and used solar energy for electricity consumption (Paloheimo, 1996). Recently, more advanced treatment technologies such as UF, MBR, RO or NF are being applied to onsite systems. However, Friedler and Hadari (2006) found that the Rotating Biological Reactor (RBC)-based system is economical when the building reaches seven storeys (28 flats) while MBR-based system becomes economically feasible only if the building size exceeds 40 storeys. In the U.S., the first large-scale onsite water recycling system was conducted at Solaire building (293-unit) in New York City. The wastewater treatment system, located in the basement, uses MBR and UV to treat more than 95 ML/d of wastewater, of which 34 ML/d is for toilet flushing, 43.5 ML/d is used as makeup water for the building's cooling towers and 22.7 ML/d is for landscape irrigation. The treated water is of high quality with BOD <2 mg/L, TP <2 mg/L and TN <3 mg/L. The system has reduced the freshwater and energy consumption by 75% and 35% respectively (AWMG, 2010).

2.4.6 Groundwater recharges

Groundwater recharge with recycled water can reduce the decline of groundwater levels, dilute, filtrate and store recycled water, partially prevent saltwater intrusion and mitigate subsidence (Asano and Cotruvo, 2004). Asano et al. (2007) listed other 10 advantages over surface storage of recycled water. Currently, it has become the fourth largest application for water reuse either via surface spreading or direct aquifer injection with over 100 projects in the U.S. and countless schemes worldwide. Regarding the recharge methods, surface spreading is simple and widely applied which provides benefits of additional treatment by soil. Besides, direct aquifer injection is particularly effective in creating hydraulic barrier in coastal aquifers. Seepage trench method is also practiced in Glendale, Arizona, the U.S., but biological clogging problem has been observed (Blair and Turner, 2004). Thus, more investigations of aquifer locations and properties are indispensable (Asano, 2001; Asano and Cotruvo, 2004). Moreover, the required wastewater quality for groundwater recharge depends on intended reuses. For instance, in Australia, as the treated water withdrawn from confined aquifers was planned to be used for agricultural applications in Adelaide, South Australia, tertiary treatment and nutrient reduction in wastewater were required, which complied with national recycling guidelines. Comparatively, Israel used spreading basins for wastewater infiltration. Since the treated effluent was often reused for agricultural irrigations after 50 days' retention in groundwater aquifer, only the secondary treatment was required (Guttman et al., 2002).

In the U.S., the state of California has over 40 aquifer recharge projects where the largest one is the Groundwater Replenishment (GWR) in Orange County Water District (OCWD). GWR is designed to purify highly treated sewer water with MF, RO, UV disinfection and hydrogen peroxide technologies. Half of the repurified water is injected into OCWD's seawater intrusion barrier wells along the Pacific coastline, the other half is provided to groundwater spreading basins in Anaheim. The project has 3 stages with the production rate of 265, 321 and 474 ML/d in 2008, 2010 and 2020, respectively (Wild et al., 2010). By 2020, the GWR will be capable of supplying approximately 22% of water needed in OCWD. Other environmental benefits include the reduced water discharge to ocean, improved groundwater quality by decreased mineral levels, and more cost-effective and energy-efficient compared to water importation from northern California (OCWD, 2008).

In Europe, Berlin, Germany, has adopted bank filtration and subsequently pond infiltration since the 1870s which is regarded as the earliest groundwater recharge case in the world. The wastewater after tertiary treatment (160 GL/yr) is discharged into an unconfined and alluvial aquifer. After one year's retention, the water pumped from the aquifer is supplied to drinking water supplies, satisfying 20-70% of the city's total drinking water demand. In the Middle East, the largest water recycling scheme in Israel was the Dan Region Project (270 ML/d). The secondary wastewater effluent was recharged to groundwater by spreading basins and then purified by Soil Aquifer Treatment (SAT), serving about 1.3 million people. With 20 years' operation, the recycled water after SAT in the aquifer has been regarded as suitable for a variety of non-potable uses such as unrestricted agricultural, industrial, commercial, residential and recreational uses (Kanarek and Michail, 1996; Asano and Bahri, 2011). Regarding Africa, the Atlantis Groundwater Recharge scheme in South Africa discharges the separately treated domestic and industrial wastewaters into different portions of the aquifer. About 3 GL/yr of tertiary treated domestic wastewater is recharged to unconfined sand aquifer. After six months' retention time, it is transported to water pipelines, contributing to 25-40% of drinking water supply. Lower quality industrial wastewater (1 GL/yr) is infiltrated through coastal basins and used as saltwater barrier. Notably, cares must be taken to prevent aquifer leakage problems when recharging less treated wastewater (Jimenez and Asano, 2008).

2.4.7 Indirect potable reuses (IPR) and direct potable reuses (DPR)

The IPR refers to the water after discharged from STP is diluted with natural surface water or groundwater and further used as drinking water. The DPR is to convey the highly treated recycled water directly to the drinking water supply system (ATSE, 2004). It is reported that more than 15 planned IPR schemes are running worldwide, some of which has been successfully functioning for more than 20 years. In Australia, despite the failure of some IPR projects due to strong public opposition, major IPR schemes have been partially developed owing to severe water shortages and unforeseen drought conditions. The typical projects include the Western Corridor Recycled Water Project (WCRWP) in South East Queensland (232 ML/d) and the three-year trial of the Leederville aquifer replenishment in Western Australia (25-35 GL/yr). Nevertheless, their full implementation is yet to be realised (Khan, 2011).

In Asia, Singapore is one of the leading countries in IPR practice. After a 2-year study, the produced water through MF, RO and UV facilities at NEWater Factory is demonstrated to be cleaner than raw fresh water drawn from river sources in terms of colour, organic substances and bacteria count. In 2003, about 13.5 ML/d of NEWater was mixed with raw water in the water reservoir and then subject to conventional treatment, which contributed to 1% of total drinking water supply. By 2011, the IPR is projected to meet 3.5% of potable water supply in Singapore (Kelly and Stevens, 2005). In Europe, the Torreele IPR project in Wulpen, Belgium, has been implemented since 2002. The recycled water processed by MF/RO/UV is discharged to the unconfined St-Andre dune aquifer with a minimum retention time of 40 days. The extracted groundwater is further treated with aeration and rapid sand filtration and UV treatment prior to distribution. The full scale project produces 40-50% of the drinking water demand, serving more than 60,000 people (Van Houtte and Verbauwhede, 2008; Rodriguez et al., 2009).

Faced with technological feasibility, health risk concerns on residual pharmaceutical particles and public objections, DPR is often regarded as a last resort in many countries. It will be considered only if the current potable water supply in that area is under severe conditions and other potable water alternatives are inaccessible and/or unaffordable. Till now, Windhoek in Namibia is the only community that has a DPR project which serves approximately 250,000 people and has been applied for domestic supply for about 40 years (Dominguez-Chicas and Scrimshaw, 2010). Windhoek constructed the world's first potable water treatment plant named Goreangab WRT in 1969 with an initial capacity of 4.3 ML/d, which treated blended water from the Goreangab Dam as well as the Gammans STP. The origins of wastewater were domestic and business sources predominantly. The plant was upgraded several times and the last upgrade was undertaken in 1997 with a capacity up to 7.5 ML/d. During the decades, recycled water contributed to 4 and 31% of the total supply in normal and drought periods.

In 2002, a new Goreangab WRT was built next to the old plant with a capacity of 21 ML/d. The main treatment processes in two WRTs are outlined in Table 2.6. In the absence of specific water quality guideline for DPR, Windhoek has compiled a specification for treated water based on Namibian, WHO, USEPA and EU guidelines. The specified value of turbidity, dissolved organic carbon, COD and total heavy metal in the effluent are 0.1 NTU, 5 mg/L, 20 mg/L and 20 μg/L respectively (Lahnsteiner and Lempert, 2007). Fortunately, the public in Windhoek is accustomed to using recycled water as potable supply due to effective education campaigns and extensive media coverage. To date, the DPR schemes run successfully with no adverse effect being detected (Huertas et al., 2008). Noticeably, recycled water is found to be superior over desalinated water in terms of infrastructure cost, treatment energy consumption, green house gas emission and aquatic environmental considerations (Dolničar and Schäfer, 2009). Consequently, DPR would be a viable option for many severe water shortage regions in Africa and the Middle East, where desalinated water is currently being used as an alternative drinking water resource.

Table 2.6 Comparison of treatment processes in old and new Goreangab WRT

Treatment	Old Goreangab WRT	New Goreangab WRT	
Influent	Reservoir water Secondary effluent Q = 4.3 ML/d in 1969, Q = 7.5 ML/d in 1997	Reservoir water (50%) Secondary effluent (50%) Q = 21 ML/d in 2002	Secondary effluent (100%) Q = 24 ML/d in 2007
Purification	Coagulation and flocculation Dissolved air flotation Rapid sand filtration Granular activated carbon filtration/ adsorption	Pre-ozonation Coagulation and flocculation Dissolved air flotation Rapid sand filtration Main ozonation Biological and granular activated carbon filtration/adsorption Ultrafiltration	
Disinfection	Chlorination and stabilisation	Chlorination and stabilisation	
Effluent	Blending and Distribution	Blending and Distribution	

Note: Modified from du Pisani, (2006).

2.5 ASSESSMENT MODELS ON RECYCLED WATER SCHEMES

With the water recycling targets being more aggressive, long-term sustainability of the recycled water scheme becomes critical for further project expansion and new end use exploitation. The current environmental assessment models are playing vital important roles in fast and reliable evaluation of existing or future recycling schemes from the perspective of environment-related considerations. Several studies have applied Material Flow Analysis (MFA) to calculate the systematic material flow of pollutants and nutrients in environmental sanitation systems over a given period of time. Others have used Life Cycle Analysis (LCA) to identify environment-related issues of different wastewater treatment technologies or water resources on the ecosystem and natural resources in the life cycle. Since the risk is one of the determinative factors to the success of recycling schemes, the Risk Assessment (RA) studies have also been conducted. The main purposes are to analyse the potential health or environmental risks (e.g., excessive pharmaceuticals and xenobiotic compounds on the soil, surface water and groundwater) resulted from recycled water projects (Ahmed, 2007; Urkiaga et al., 2008). While most environmental studies have been carried out using a single environmental tool, the integrated models have been increasingly developed to compensate the weaknesses of individual ones. However, investigations on the selection and implementation of appropriate integrated models for particular recycling schemes are still limited and not well documented.

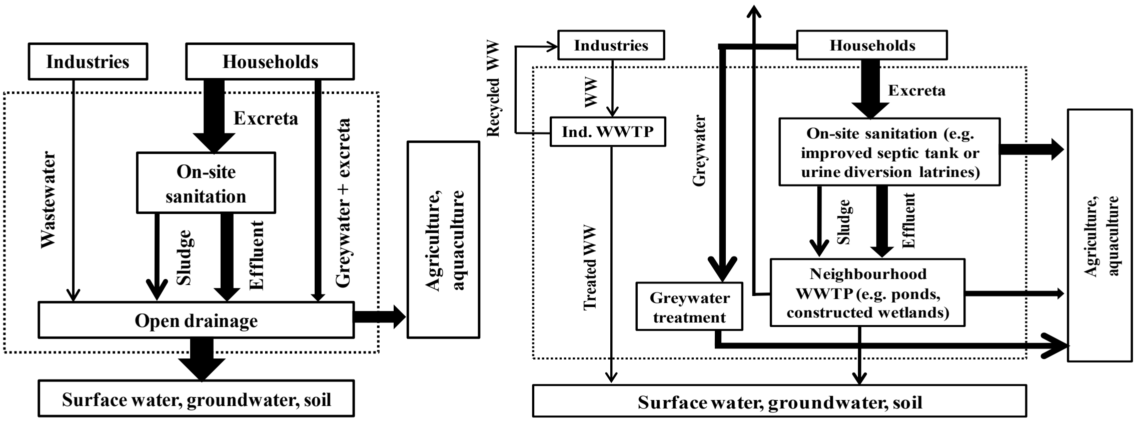
2.5.1 Material Flow Analysis (MFA) models

As MFA examines the material flows and their transformation in regional environmental systems over a given period of time, it addresses the importance of water recycling by linking adverse environmental impacts with possible resource recovery and reuse solutions (Brunner and Baccini, 1992; Jeppsson and Hellstrom, 2002). Generally, MFA consists of four steps: (1) definition of a system which composes of material flows, stocks and processes; (2) measurement of mass fluxes and element concentrations of all goods; (3) calculation of the element fluxes; (4) schematic presentation and interpretation of the results (Sinsupan, 2004). Based on the law of the conservation of matter, environmental impacts of a particular flow can be calculated by a simple mass balance of all associated inputs, outputs and storage. The results can then be interpreted against environmental standards or can be linked to other assessment tools for further analysis.

Regarding to the types of MFA model, qualitative ones are simple and can be quickly performed. They can facilitate decision makers in understanding the metabolism of their region and can also provide early warning signals for future environmental issues. Nonetheless, qualitative approach can only be regarded as an initial assessment since numerical material flow data are not available (Schneider et al., 2002; Agnes et al., 2007). Comparatively, quantitative MFA models normally employ mathematical equations to quantify the processes and flows of transformation, production and consumption as well as the mass and/or water balance within the system. This offers more specific and reliable information in decision making. Depending on the variance of the flows over time, they can be further classified into static and dynamic forms. The static model, where the flows are assumed to be invariable, is suitable to estimate the flows with no primary data and calculate the effectiveness of adopting different policy scenarios in sustainability improvement. When the system is found to be unsustainable, the model is unable to tell when it became unsustainable due to its high uncertainties. On the other hand, the dynamic approach accounts for time dependence and analyses the flows of materials or any accumulation in stocks over a period of time based on mathematical probabilistic distributions (Tangsubkul et al., 2005a; Park et al., 2011).

2.5.1.1 Application of MFA models on environmental sanitation improvement

MFA models have been increasingly applied to environmental sanitation planning in several developing countries such as Columbia, Ghana and Vietnam (Belevi, 2002; Neset et al., 2006). With the water supply, sanitation, solid waste management and urban agriculture are being considered in an integrated way, they are able to identify the key flows or processes associated with huge water consumption and serious environmental pollution. For instance, Agnes et al. (2007) evaluated the current environmental sanitation system in Vietnam as well as the effects of new sanitation concepts or measures. Figure 2.3 depicts two conceptual MFA models where the thickness of arrows indicated the relative importance of the flows. The small scale decentralised wastewater treatment facilities (e.g., CW and waste stabilisation pond) for greywater treatment were shown to be effective in water sustainability improvement as treated greywater can be reused in agricultural irrigation and aquaculture and the amount of open drainage can be greatly reduced. The study also suggested treating industrial wastewater separately from domestic sources and then reusing it internally.



(a) (b)

Figure 2.3 Simplified systems representing nitrogen flows in the current (a) and improved (b) sanitation system in urban areas in Vietnam (adapted from Agnes et al., 2007).

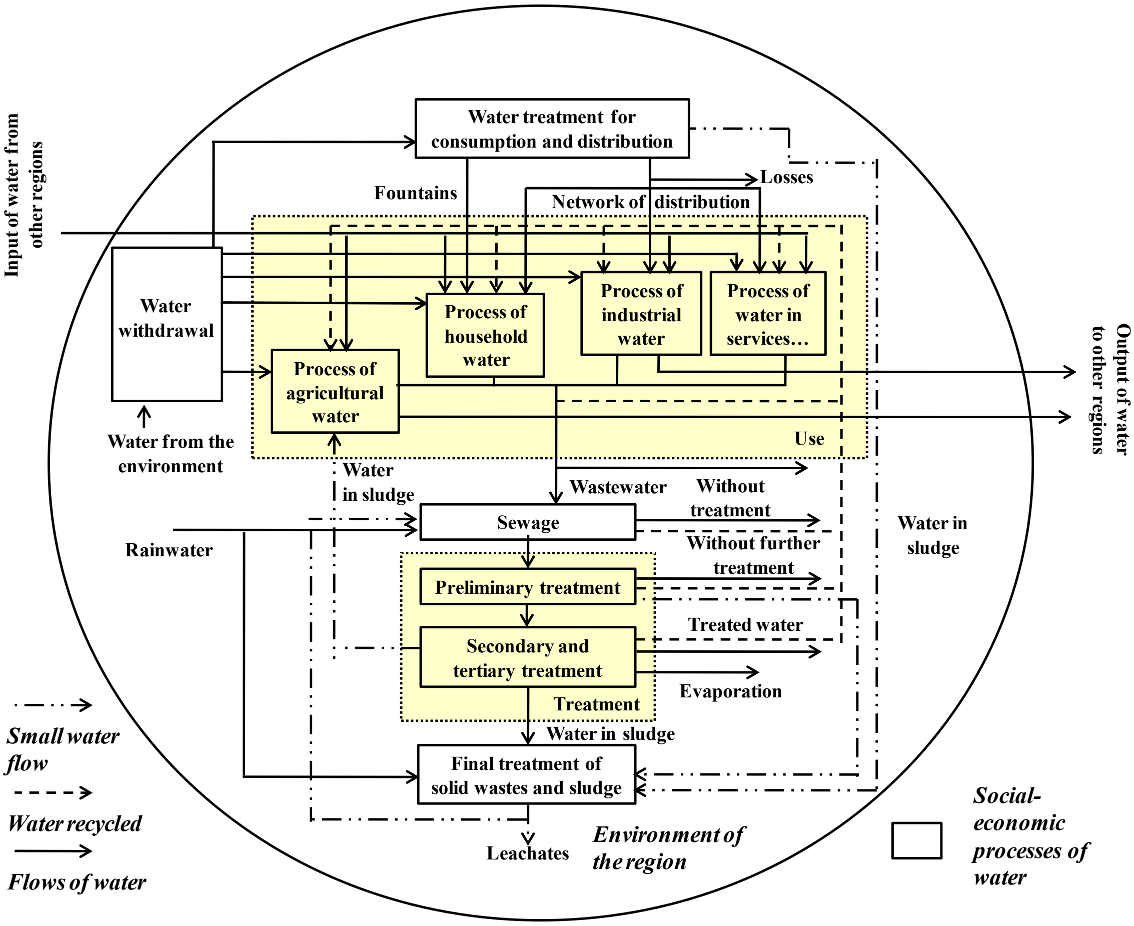


Figure 2.4 Metabolism of water in the socio-economy of a region (adapted from Schneider et al., 2002).

Likewise, Schneider et al. (2002) investigated the metabolism of water within a region of Portugal. As can be seen from Figure 2.4, significant improvements in water sustainability can be achieved from less water consumption, increased internal or external water recycling and reuse, reduced wastewater discharge, lower variability, etc. Although several recycled water end uses (e.g., agriculture, household, industry and services) have been proposed in this water flow analysis, future more detailed quantitative assessments are required.

Further, Tangsubkul et al. (2005a) analysed the phosphorus (P) and water reuse management strategies in the Sydney region for the year 2000. The results demonstrated that the combination of greywater recycling, composting toilet and human behaviour change (e.g., using P-free detergent and adopting a vegetarian diet) was the most effective solution since around 3600 tonnes/year of P can be prevented from entering the wastewater system. Besides, nearly all of the P in wastewater could be recovered in this combined system. Despite data gaps in socio-cultural, economic and health issues, the study has identified the importance of conducting wastewater treatment and reuse in environmental sustainability. However, when temporal changes of the flows are of interest, the dynamic ways should be adopted rather than static ones.

Montangero and Belevi (2008) reported three important mathematical equations in dynamic MFA approach. The most essential one is the mass balance equation:

$$\frac{dM_i^{(j)}}{dt} = \sum_r A_{i,r-j} - \sum_s A_{i,j-s} \qquad \text{(Eq. 2.1)}$$

where *i* is the indicator substance, *j* is the process number, $M_i^{(j)}$ is the stock of substance *i* in process *j*, *t* is the time, *r* is the source process, *s* is the destination process, $A_{i,r-j}$ is the input flow of substance *i* from the source process *r* to the destination process *s*. The left side of the Equation 2.1 represents the stock change rate of substance *i* within the process *j* while the right side expresses the difference between input and output flows. $A_{i,r-j}$ can be derived as follows:

$$A_{i,r-j} = f(p_1, p_2 \dots p_n) \qquad \text{(Eq. 2.2)}$$

where, p_1, $p_2 \dots p_n$ refer to parameters based on scientific and expert knowledge. Additionally, transfer coefficient is also commonly used in modelling material flows, which describes the partitioning of a substance in a process and provides the fraction of the total input of a substance transferred to a specific output good (Equation 2.3).

$$k_{i,g}^{(j)} = A_{i,g}^{(j)} / \sum_r A_{i,r-j} \qquad \text{(Eq. 2.3)}$$

The study also addressed the importance of expressing the model inputs as probability distribution when limited data are given.

Based on these equations, Montangero et al. (2007) carried out a case study to evaluate water and nutrient management strategies (e.g., household consumption patterns, type of sanitation infrastructure and wastewater reuse practices) in Hanoi, Vietnam. The model indicated that reusing a fraction of greywater for toilet flushing would reduce the water consumption from 140 to 113 Litres per capita per day (L/p/d) by 2015, which was tantamount to a 16% decrease in groundwater abstraction. Nevertheless, some important factors (e.g., the fate of organic matter, toxic substances, economic and social conditions) were not considered (Montangero et al., 2006a; 2006b). Moreover, Cencic and Rechberger (2008) introduced a user-friendly software named STAN which supports performing MFA according to the Austrian standard ÖNORM S 2096 under consideration of data uncertainties. Predefined elements such as processes, flows, system boundaries and text fields can be imported from Microsoft Excel or input manually whereas uncertain data are assumed to be normally distributed. With these input data, the graphical MFA model can be automatically translated into a mathematic model using several pre-defined conversion equations. As a result, STAN expresses the mass flows as Sankey arrows which are proportional to their mass flow values. In addition to performing dynamic MFA, STAN is also capable of evaluating the economic, resource and environmental value of the materials.

2.5.1.2 Characteristics and weaknesses of MFA models on water reuse

With respect to the scope, MFA models are not only restricted to flows within the region but also trace the flows beyond the boundary as far as they are relevant to regional activities, thereby enabling the detection of unexpected side-effects within the region to other regions or other time periods. The dynamic approaches are more complex than conventional methods as the temporal changes, transfer coefficient and economic conditions would be involved. With quantitative MFA models becoming more accurate and advanced, they are likely to give more realistic pictures of the regional environmental statuses (e.g., nutrient flow, recycled water consumption and wastewater discharge). Yet the manipulation of sophisticated MFA models remains a big challenge to decision makers. Noticeably, MFA can only deal with one substance and the related environmental interventions at a particular time in one area. The side-effects to other substance chains are beyond the study scope (Hendriks et al., 2000; Brunner and Rechberger, 2004). As such, there is a risk that a critical problem might be overlooked if a wrong judgment is made on the goods/substance selection (Tangsubkul et al., 2005a). While most of the recent MFA studies have recognized the importance of water recycling and reuse in environmental sanitation management, the downstream assessments and discussions on the feasibility and suitability of particular recycled water schemes are still essential. More work on the different fractions of water sources in the STP, subsequent treatment technologies, effluent quality, possible end uses and potential risks to human health and the environment should be done in the following analyses.

2.5.2 Life Cycle Assessment (LCA) models

When dealing with water recycling issues, LCA mainly focuses on the energy and material requirements throughout an entire life cycle of the treatment process as well as the quality of treated effluent associated with fit-for-purpose end uses (Muñoz et al., 2009b). Some major impact categories or indicators in LCA include acidification potential (AP), global warming potential (GWP), eutrophication potential (EP), photochemical oxidation (PHO), ozone depletion potential (ODP) and ecotoxicity potential (ETP). These are able to give an overall picture regarding recycling system performance or contribution to decision makers (Ahmed, 2007; Pasqualino et al., 2011).

There are generally three types of LCA models at present, namely process-based LCA, economic input-output (EIO) LCA and hybrid LCA. The initial and simplest approach is the process-based LCA which is usually carried out in four steps: (1) goal and scope definitions, (2) life cycle inventory analysis, (3) life cycle impact assessment (LCIA) and (4) life cycle improvement analysis and interpretation (Tangsubkul et al., 2005b; Stokes and Horvath, 2006). Particularly, in Step 3, several LCIA methods (e.g., CML 2001, Eco-indicator 99 and EDIP 97) are normally used to quantify the above-mentioned environmental indicators. Dreyer et al. (2003) demonstrated that EDIP 97 and CML 2001 are both midpoint approaches, which showed only minor differences for the most impact categories except for the ones that describe toxicity to humans and the ecosystems. The results of Eco-indicator 99 and EDIP 97 reached opposite directions for some inventories as the former one is an end point method, where the patterns of most important contributors to the impact scores are quite different from EDIP 97. Both Dreyer et al. (2003) and Renou et al. (2008) concluded that more work was required on toxicity indicators as LCIA methods did not converge toward similar results.

Currently, by applying commercial LCA softwares such as Gabi and SimaPro, one can easily choose one or several different LCIA approaches for assessment. Comparatively, EIO-LCA is an economic-based technique which is to capture all economic transactions, resource requirements and transportation to produce recycled water of a certain quality and then calculate the associated environmental emissions and wastes (e.g., energy use, toxic air emissions and hazardous waste) in terms of economic expenditures. To overcome the disadvantages of conventional process-based and EIO-LCA models, hybrid LCA has been developed over the years, which combines the accuracy of process-based LCA and completeness of EIO-LCA (Mattila et al., 2010). Having recognised the advantages of recycled water, some studies have employed LCA models to select optimal wastewater treatment technologies or stages for recycled water planning in the agriculture, industry and urban landscape irrigation, or to evaluate environmental profiles of existing STP (Vlasopoulos et al., 2006; Pasqualino et al., 2009). However, the full application of LCA in holistic recycled water scheme assessment is still quite limited.

2.5.2.1 Agricultural uses

With respect to agricultural irrigation schemes, Ortiz et al. (2007) adopted a processed-based LCA to compare four treatment scenarios (CAS, CAS-UF, external and immersed MBR). The airborne emissions associated with the construction, operation and dismantlement of WWTP were of prime concern. The results indicated an overall lower impact in CAS, followed by immersed MBR and external MBR. On the other hand, systems expect for CAS produced high effluent quality, which not only allowed the water to be safely reused in irrigation but also enabled other applications such as groundwater recharge, household and industrial uses. Thus, considering both environmental impact and water quality, immersed MBR coupled with the renewable energy consumption pattern was optimal. Nevertheless, this study did not take into account the environmental impacts on soil and water nor consider the toxic and health effects. To investigate the toxicity-related impacts of recycled water on agriculture, Muñoz et al. (2009b) evaluated four treatment scenarios, including no reuse, reuse without tertiary treatment, reuse after ozonation and reuse after ozonation and hydrogen peroxide. Two LCIA approaches (i.e., USES-LCA and EDIP 97) have been applied for evaluation, which showed that water reuse scenarios with tertiary treatment were preferred and the ozonation one arrived at lower toxicity score compared with the ozone-peroxide one. It is worth noting that the uncertainties were relatively high, especially in USES-LCA, where the deviations from 1.5 to 6 orders of magnitude were observed. However, Meneses et al. (2010) found that chlorination-UV disinfection was superior over ozonation and ozonation- hydrogen peroxide systems in terms of AP, GWP, EP, PHO and ODP, when considering tertiary treated water for agricultural and urban applications. They also indicated that winter climate conditions contributed to reduced environmental impacts due to variances in population habits, water quality and use pattern. Apart from processed-based LCA models, Tangsubkul et al. (2005b) initially used EIO-LCA to evaluate three treatment technologies, including the CMF (Ozonation-MF-disinfection), MBR-RO and Waste Stabilisation Pond (WSP) systems for irrigation purposes at the Rouse Hill residential area in Sydney, Australia (Figure 2.5).

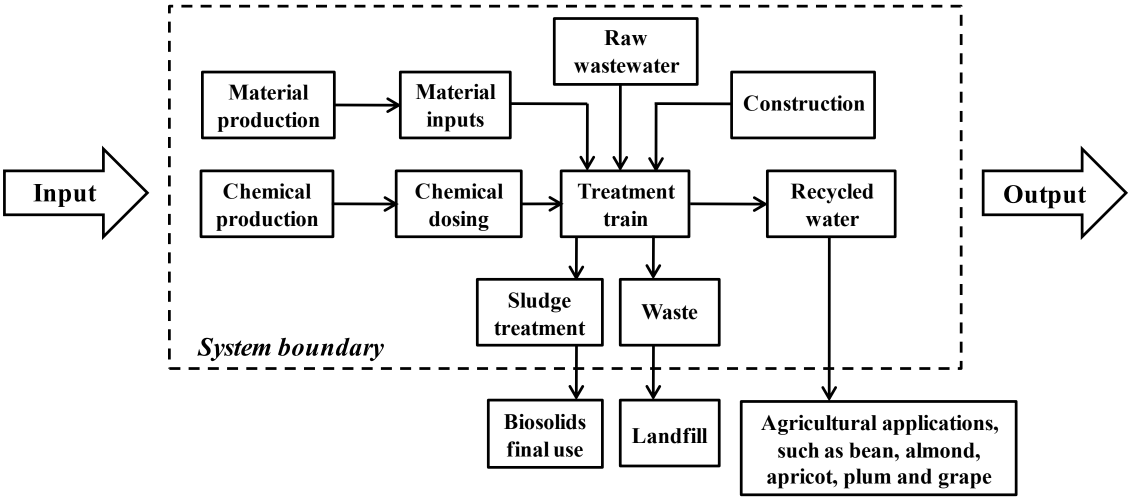


Figure 2.5 System boundary of LCA study on Rouse Hill water recycling scheme (modified from Tangsubkul et al., 2005b).

With the assistance of Missing Inventory Estimation Tool (MIET) 2.0 and GaBi3 softwares, all flows associated with the construction activities of treatment options were converted into a monetary value per functional unit. The environmental impact categories in consideration were GWP, EP, ETP and salinisation potential (SP). The results revealed that MBR-RO system could result in a low SP but was likely to trigger soil dispersion. The WSP system was regarded as the most suitable option for irrigation applications if reductions to SP impact were made. Although MIET was illustrated to be an acceptable means of estimating the impacts caused by the construction phase of wastewater treatment, a major constraint is that the results might be inappropriate when being applied to other countries due to inter-industry cost structure differences.

2.5.2.2 Industrial uses

In regard to cooling and boiler feed purposes, Vlasopoulos et al. (2006) investigated over 600 different treatment technology combinations on treating petroleum wastewaters for industrial and agricultural reuses. The study took into account the environmental impacts of different treatment options in terms of GWP, AP, EP and PHO during the construction and operation phase, using CML 2000 method. The optimal treatment technology combinations associated with four end uses are shown in Table 2.7.

Table 2.7 Optimal treatment technologies associated with different end use categories

End uses	Water quality requirement	Best technology combinations	Descriptions and comments
(1) Barley-wheat	4-staged treatment (53[a])	MF-ORG-RO (in EP) DAF-CW-DMF-RO (except EP)	In stage 2, although CW reached higher environmental impact than ABS, it resulted in smaller design and lower energy consumption in subsequent treatments.
(2) Citrus	4-staged treatment (53[a])	MF-ORG-RO-ION (in EP) DAF-CW-DMF-RO-ION (except EP)	ION was to achieve additional boron removal by 0.5 mg/L. ION only contributed to 1% of the overall environmental impact.
(3) Alfalfa-sorghum-cotton-rhodes-boiler feed	4-staged treatment (104[a])	DAF-CWL-DMF-RO and MF-ORF-RO	Although this end use required less stringent water quality, quaternary treatment was still needed to reduce the sodium concentration.
(4) Cooling system feed	3-staged treatment (139[a])	DAF-ABS-ORG	Although SSF and DMF had better environmental performance than ORG in stage 3, they did not satisfy the required cooling water quality.

Note: Modified from Vlasopoulos et al. (2006).
[a] Number of technology combinations that can meet the requirement.
Abbreviations: DAF = Dissolved Air Flotation; CW = Constructed Wetlands; ABS = Absorbents; DMF = Dual Media Filtration; MF = Microfiltration; ORG = Organoclay; RO = Reverse Osmosis; ION = Ion Exchange.

As can be seen, compared with two agricultural end uses (category 1 and 2) where the crop products can be eaten raw, cooling and boiler feed water required lower water quality (category 3 and 4). Specially, the cooling system allowed the use of three-staged treatment with more technology combinations due to lower water quality requirement for sodium. Remarkably, to select the optimal treatment technology for each treatment stage, one should not only consider the associated environmental impacts but also recycled water quality as well as indirect downstream benefits to subsequent treatment stages.

Jørgensen et al. (2004) studied six alternatives for water recycling and residual handling at an industrial laundry company in Denmark. It was concluded that onsite wastewater reuse scenarios using UF or a bio-filter led to lower environmental impacts compared with current no reuse practice. As laundry process wastewater carried some pollutants such as heavy metals, UF coupled with sludge vitrification were considered to be the optimal technology combination in terms of lowest toxicity impact to the environment. The UF permeate could be safely reused in the washing process. The results demonstrated that LCA was able to identify the best treatment technologies as well as long-term environmental benefits of water reuse in laundries.

Moreover, Zhang et al. (2010) have adopted a hybrid model to measure the life cycle benefit of treated water reuse in industrial and domestic applications. The corresponding life cycle energy consumption in the construction, operation and demolishment phases of the STP in Xi’an, China was also investigated. The study employed the Eco-indicator 99 method in quantifying the environmental impacts of different treatment stages as equivalent energy consumption. Unlike processed-based LCA, the system boundary in this model was confined to the secondary and tertiary treatment units without any sub-boundary between them, which is capable of reducing the possible difficulties in conventional approaches. The study indicated that energy consumption in operating the tertiary treatment (2065.28×10^9 KJ) can be significantly compensated by life cycle benefit of water reuse in terms of reduced wastewater discharge (74.2×10^9 KJ) and freshwater saving (1598.4×10^9 KJ). Although this study successfully linked the life cycle energy consumption with direct benefits of recycled water reuse (e.g., wastewater reduction and freshwater saving), other indirect benefits such as ecosystem protection and water cycle improvement were not considered.

2.5.2.3 Characteristics and weaknesses of LCA models

Among three types of LCA models, process-based LCA is still the most widely used approach. It is noticed that one of the most difficulties is the choice of an appropriate system boundary. Some researches excluded the insignificant contributions within the system as considerable materials or processes can easily lead to an overwhelming number of inputs and outputs whereas others pointed out that such a narrow focus might ignore important effects and cause incorrect decisions (Hendrickson et al., 2011; Matthews, 2011). Moreover, this approach is time intensive and costly, especially in more complicated assessment. When a large variety of impact categories requires to be considered, apart from emission classification and characterization, much time will also be spent on normalization and weighting processes to make the indicators dimensionless thereby enabling comparison and achieving an overall score. The high prices of commercial softwares also limit the model popularization to some extent. Additionally, it would be difficult to apply process-based LCA to newly developed treatment technologies when the relevant material and energy consumption are unavailable in Life Cycle Inventory (LCI) database.

Comparatively, EIO-LCA approach overcomes the system boundary problem. Although the boundary is very broad and inclusive, the transactions and emissions of all processes among all phases are included. It is also relatively quick to be performed and has modest data requirements. This approach is ideal for comprehensive assessments and systems-level comparisons since all environmental flows have been converted into monetary values with an assumption of a proportional relationship between them. Unlike the site-specific results from process-based LCA, which sometimes involve a certain degree of confidentiality, the EIO-LCA results could be reproducible and publicly available as they are economy-wide. However, as EIO-LCA must link physical units with monetary values, it mainly captures environmental impacts associated with raw material acquisition and construction stages of recycling facilities rather than downstream impacts (e.g., water reuse, waste recycling and end-of-life options). With considerable input data requirements, data deficiency sometimes hinders the complete assessment of environmental effects. It is also difficult to be applied to an open economy with substantial non-comparable imports. Besides, the uncertainties are likely to be high through indicator aggregation, monetary transactions between currencies and times, and the possible use of outdated data on interactions and emissions (Hendrickson et al., 2006; Stokes and Horvath, 2006).

Hybrid LCA is considered to be a state-of-the-art method, which involves the integration of more reliable bottom-up process-based LCA data into the comprehensive top-down EIO-LCA methods (Mattila et al., 2010). Although the hybrid approach can take advantage of both methods, the model tends to be sophisticated and hard to be understood or manipulated by decision makers within a short time. Another significant issue in hybrid LCA is to avoid double counting. Simply adding the results of a process-based LCA and an EIO-LCA of the same system will erroneously cause the system components modelled twice. In addition, other practical limitations also exist such as model structure variation, methodology uncertainties, data completeness and software deficiency (Rowley et al., 2009).

2.5.3 Risk Assessment (RA) models

Risk control and assessment are vital important in guaranteeing the safety, acceptability and reliability of recycled water. The excessive chemical and/or microbial hazards in source waters are highly variable over time, which potentially generate a certain degree of risk to human health and the environment (Khan and McDonald, 2010). Particularly, pathogenic microorganisms in recycled water become the prime concerns in health risk assessment (HRA) whereas ecosystems receiving chemical pollutants such as pharmaceuticals and xenobiotic compounds are the first potential targets in environmental risk assessment (ERA). Once the potential hazards, their sources and exposure characteristics have been identified, RA models are able to identify the potential adverse effects associated with each recycling activity either from a qualitative or quantitative approach (Soller, 2006; Toze, 2006; Kamizoulis, 2008). Based on the results from RA models, risk control, management and communication can be established afterwards together with the modifications of existing recycled water guidelines or regulations. The accumulated risk information can also assist in choosing more suitable, reliable and cost-effective treatment processes, and making future project planning (Huertas et al., 2008; NRMMC-EPHC-AHMC, 2008).

2.5.3.1 Qualitative RA models

Qualitative risk can be estimated on the basis of past records, practices, experiences, relevant literature, experiments and/or expert judgements. When numerical data or resources are inadequate under certain circumstances, the risk may be judged from individual's or group's degree of belief, triggering inaccuracy or some errors. This qualitative approach can only be an initial screening for risk assessment and is usually conducted by combining consequences and their likelihood of potential hazards in recycled water (Storey and Kaucner, 2009; Khan, 2010). As shown in Table 2.8, a qualitative estimation of risk can be made using the risk matrix. The likelihood refers to the frequency of adverse effects related to the reuse project (e.g., inadequate or variable water quality, failure of achieving the technical or financial requirements for the correct functioning of the system, acute or chronic effects to public health and the environment). On the other hand, the consequence is the description of the severities of these adverse effects to human health and the environment. Although some scenarios are almost certain or have moderate consequences, they can generate low risks when the likelihood is balanced against consequences (Roser et al., 2006).

Table 2.8 Qualitative risk matrix

Likelihood	Consequences				
	Insignificant[1]	Minor[2]	Moderate[3]	Major[4]	Catastrophic[5]
Rare (once in 100 years)	Low	Low	Low	High	High
Unlikely (within 20 years)	Low	Low	Moderate	High	Very high
Possible (within 5 to 10 years)	Low	Moderate	High	Very high	Very high
Likely (within 1 to 5 years)	Low	Moderate	High	Very high	Very high
Almost certain (once a year)	Low	Moderate	High	Very high	Very high

Note: Adapted from NRMMC-EPHC-AHMC, (2008).
[1] Insignificant: Insignificant impact or not detectable;
[2] Minor: Minor impact for small population;
[3] Moderate: Minor impact for large population;
[4] Major: Major impact for small population;
[5] Catastrophic: Major impact for large population.

Based on Table 2.8, the government of Western Australia has generally specified level of risks towards several end uses in Table 2.9 (GWA, 2009b). Derry et al. (2006) have conducted a rapid health-risk assessment on recycled water reuse at the University of Western Sydney for agricultural and landscape irrigation. The risks together with uncertainty factors were estimated roughly on a scale of 1-100, due to lack of sufficient numerical data (Table 2.10). As can be seen from two tables, when recycled water has frequent contact with people or the injection volume of recycled water is high each time, the risk is likely to be high. Hence, more attention should be paid to these high-risk water reuse categories with risk control actions to the greatest extent.

Table 2.9 Exposure risk levels

End uses	Risk level
Residential dual pipe	High
Internal reuse and external surface irrigation in multi-unit dwellings	
Agricultural irrigation for unprocessed food crops (salad etc.)	
Urban surface irrigation with unrestricted access and application	
Commercial uses; toilet flushing and dedicated cold water taps	
Urban surface irrigation with some restricted access and application	Moderate
Fountains and water features	
Industrial use with potential human exposure	
Urban irrigation with enhanced restricted access and application	Low
Sub-surface irrigation for fruit trees	
Agricultural irrigation for non-edible crops	
Woodlots (forestry) and sub-surface irrigation (non-food crops)	Extra low

Note: Adapted from GWA, (2009b).

Moreover, Roser et al. (2006) investigated the MF/RO treated tertiary effluent discharging into the Hawkesbury-Nepean River, at two locations (i.e., Penrith and North Richmond) of New South Wales, Australia. The risks associated with different water reuse scenarios are listed in Table 2.11. As the Hawkesbury-Nepean River receives around 160 ML/d of treated wastewater, direct drinking of untreated river water on a continuous basis is seen as the worst case but a very unlikely one. However, scenarios associated with consumption of large volumes of water during large scale/extended duration breakdown in the MF/RO system are of high concern. Thus, it could be good ways to collect complete information on MF/RO failure modes and develop critical limits on MF/RO performance so as to ensure a low risk for downstream water users.

Table 2.10 Rapid risk assessment on recycled water

End uses	Exposed population	Exposure routine	Risk value (1-100)[1]	Uncertainty value (1-100)[2]
Landscape irrigation	Mentally challenged or immuno-compromised participants	Ingestion	49	60
	Workers	Ingestion or dermal contact	42	55
	Publics playing on sports fields	Ingestion of aerosols	35	70
Agricultural irrigation	Students, campus staff and work-opportunity participants	Ingestion of fruit, nuts and some vegetables	45	45
	Consumers	Dairy animals	40	50
	Children	Ingestion of fruit	40	60

Note: Adapted from Derry et al., (2006).
[1] Risk value (1-100): 1–Lowest risk; 100–Highest risk;
Higher values indicate the capacity to accommodate more serious hazards.
[2] Uncertainty value (1-100): 1–Lowest uncertainty; 100–Highest uncertainty;
The uncertainty values exceeding 50 indicate a need for further data collection or research in many cases.

Furthermore, Dominguez-Chicas and Scrimshaw (2010) discussed the chemical risks of an IPR scheme where the treatment system consists of pre-screening, MF, RO and an advanced oxidation process (AOP) utilising UV radiation and hydrogen peroxide. Despite high removal efficiency, residual hazards or potential hazardous events at each treatment barrier presented challenges to the treatment processes or resulted in operational problems within the water supply chain. According to the analytical results on the removal rates of 223 potential hazards and the corresponding quality of the final treated effluent, the estimated risks were displayed in a risk heat map (Figure 2.6), which allow for the prioritisation of hazards in IPR to a practical level.

Table 2.11 Qualitative microbial risk assessment for water reuse scenarios

Site	End uses	Scenario	Exposure frequency (year^{-1})	Potential infectious pathogens	Consequences	Likelihood	Risk
Penrith	IPR (direct drinking of untreated river water)	Low flow MF breakdown	5.3 days	*Campylobacter*	Insignificant	Unlikely	Low
		Low flow RO breakdown	5.3 days	*Rotavirus*	Minor	Unlikely	Low
		Low flow RO+MF failure	5.3 days	*Rotavirus*	Major	Rare	High
	Recreational reuse	High flow	26 days	*Rotavirus*	Insignificant	Likely	Low
		Median flow	26 days	*Rotavirus*	Insignificant	Likely	Low
		Low flow	26 days	*Rotavirus*	Insignificant	Likely	Low
		Low flow RO+MF failure	26 days	*Rotavirus*	Moderate	Rare	Low
	Direct consumption of mussels	Low flow	26 meals (1 meal per day)	*Campylobacter*	Insignificant	Unlikely-Possible	Low
	Direct consumption of irrigated lettuce	Low flow	365 days	*Campylobacter*	Insignificant	Possible	Low
North Richmond	IPR (direct drinking of untreated river water)	Low flow	-	*Cryptosporidium*	Insignificant	Almost certain	Low
		High flow	-	*Rotavirus*	Insignificant	Almost certain	Low

Note: Modified from Roser et al., (2006).

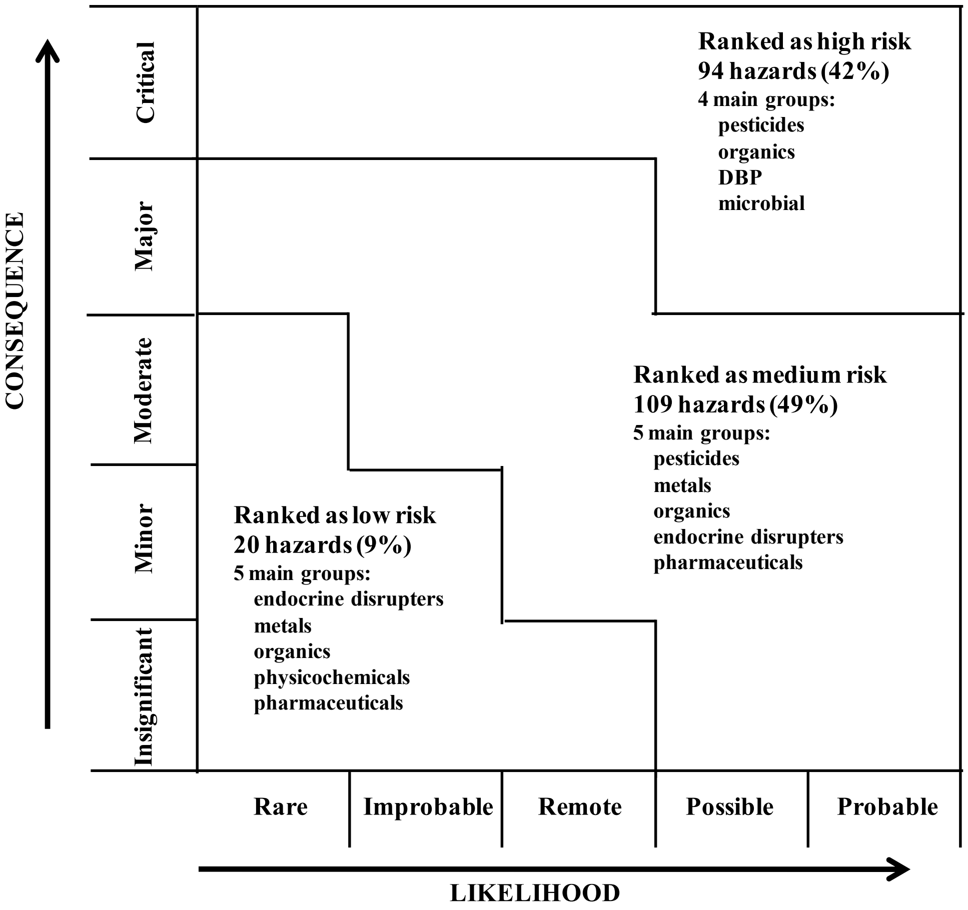


Figure 2.6 Risk assessment matrix of IPR schemes (adapted from Dominguez-Chicas and Scrimshaw, 2010).

The results showed that microbiological hazards and other three chemical groups, although small in the total number, were ranked as high risk, attributing to high consequences. Yet the likelihood data reflecting their occurrence were still insufficient. Whenever possible, more data should be collected throughout the supply chain to revise the outcomes of the risk characterization.

2.5.3.2 Quantitative HRA models

The quantitative approach has been used initially to assess human health effects associated with exposure to chemicals in 1970 and can be analysed based on sufficient numerical data collected from statistical, experimental and other sources for both the likelihood and possible health consequences of exposure in particular circumstances (Hammond and Coppick, 1990; Asano and Cotruvo, 2004). Generally, quantitative assessment involves four steps: hazard identification, dose-response assessment, exposure assessment and risk characterization (Figure 2.7). Each step is essential in identifying risks of recycling schemes and the outputs can feed into risk management and risk communication processes (Huertas et al., 2008; Khan, 2010).

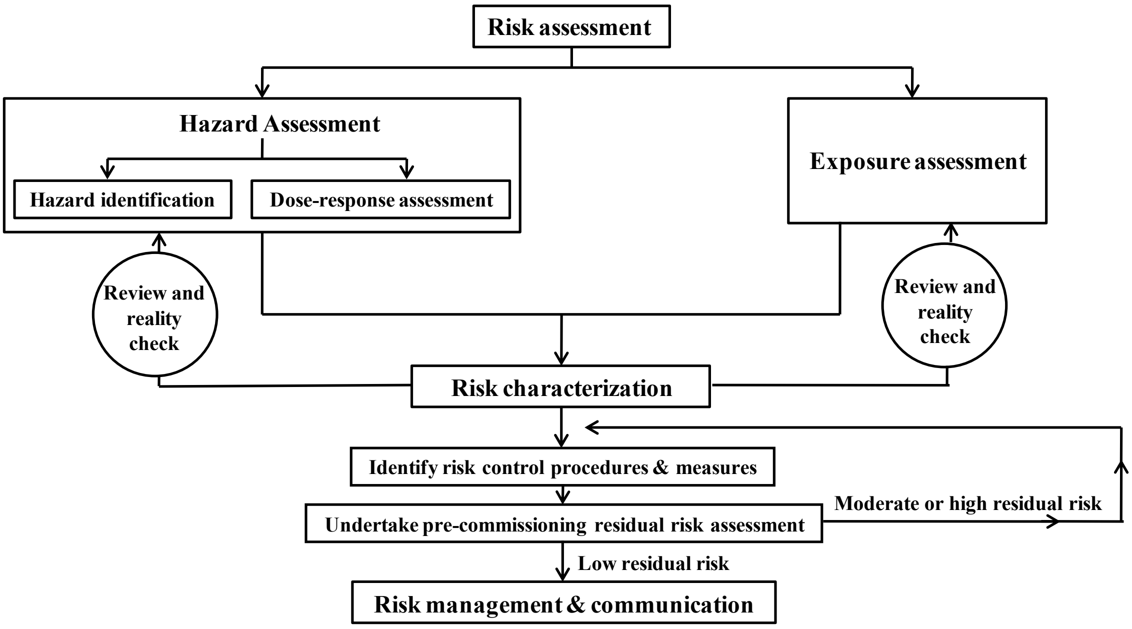


Figure 2.7 Quantitative risk assessment process

(1) Quantitative Chemical Risk Assessment (QCRA)

Chemical risks of recycled water on human health have been largely overlooked or inadequately considered. Guidelines pertaining to chemical contaminants are typically limited to bulk parameters such as pH, TSS, COD and BOD. Although these parameters can be good indicators for the likely presence of chemical species of concern in many situations, the sensitivity is limited for close contact end uses such as IPR and DPR, where an accurate assurance of specific chemical concentrations (e.g., heavy metals, mineral oils, pesticides, EDCs and PhACs) is important. Hence, to provide the most meaningful tools for many water reuse applications, QCRA should be increasingly considered (Weber et al., 2006). Firstly, hazard identification is an initial hazard screening process in QCRA which can eliminate chemicals that do not present significant (or determinant) health risks. This can minimize the unnecessary cost and allow prioritised identification of particular hazards (NRMMC-EPHC-AHMC, 2008). The identification step can be achieved by comparing hazard concentrations in recycled water with corresponding national or local guidelines values.

The second step, dose-response assessment is beneficial for quantitative risk characterization. It normally employs a dose-response curve (Figure 2.8) to characterise the relationship between the exposure dose and the incidence of identified health impacts (Khan, 2010). A clear dose-response curve reflects that the probability of response increases proportionately over a certain dose change. To determine the curve, it is indispensable to collect and analyse relevant data of human health end-points (e.g., acceptable daily intakes and acute reference doses) for the specific hazards (Roser et al., 2006). For non-carcinogenic chemicals, there are threshold doses (Curve A in Figure 2.8), below which no toxic effects are observed (Ritter et al., 2007). In this case, the highest dose at which no adverse effects are observed (NOAEL) or the lowest dose at which adverse effects are observed (LOAEL) can be determined from animal experiments and/or epidemiological data.

Combining NOAEL and LOAEL with uncertainty factors, the safe risk level (RfD) can be derived as follows:

$$\mathrm{RfD} = \frac{\mathrm{NOAEL\ or\ LOAEL}}{(\mathrm{UF}_1 \times \mathrm{UF}_2 \ldots) \times \mathrm{MF}} \qquad \text{(Eq. 2.4)}$$

where UF_1, UF_2… are uncertainty factors, MF are modifying factors. Uncertainty factors may arise from differences in the sensitivity of humans and the test animals, variability in sensitivity between humans, extrapolation of subchronic experiments to chronic exposure, the use of a LOAEL rather than a NOAEL and/or gaps in the available toxicological data. The value of each uncertain factor is assumed to be 3 or 10 with a maximum uncertainty value of 3000 (Khan, 2010). Modifying factors represent the confidence in the study which can be achieved through professional assessments (Asano et al., 2007). As RfD values are designed to protect potentially exposed people, including sensitive sub-populations such as children and the elderly, they tend to be conservative. Some guidelines such as US EPA, WHO, and ADWG have specified RfD values as benchmarks for particular non-carcinogenic chemicals (Rodriguez et al., 2007). Beyond the RfD level, adverse response is likely to increase dramatically. Comparatively, it is assumed that there is no threshold dose for carcinogenic chemicals so that the dose response relationships are straight lines (Curve B in Figure 2.8). The carcinogenic potential of a chemical is normally expressed quantitatively as a cancer slope factor (CSF) which is the gradient of Curve B.

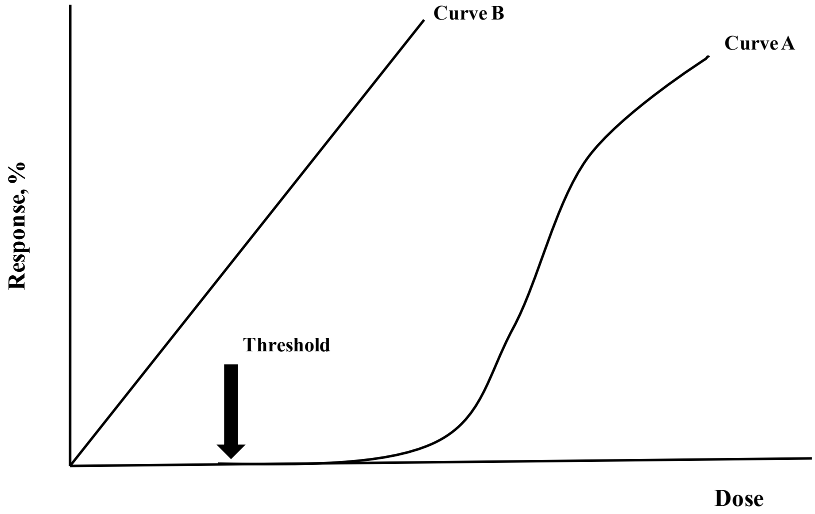


Figure 2.8 Dose-response curve (modified from Asano et al., 2007).

With respect to risk characterization, after identifying the hazards, the corresponding dose-response relationships and the RfD values in a particular exposure scenario, the risk for non-carcinogenic chemicals can be measured by hazard quotient (HQ). HQ is the ratio of an actual exposure to the RfD (Equation 2.5). To demonstrate an acceptable risk to human health or the environment, exposure dose should be less than the RfD. In other words, HQ should be less than 1 (Weber et al., 2006).

$$\text{Hazard Quotient (HQ)} = \frac{\text{Exposure dose } (\text{mg} \cdot \text{kg}^{-1} \cdot \text{day}^{-1})}{\text{RfD } (\text{mg} \cdot \text{kg}^{-1} \cdot \text{day}^{-1})} \qquad \text{(Eq. 2.5)}$$

Additionally, in some guidelines (e.g., WHO and ADWG), the amount and frequency of exposure (e.g., water consumption per person per day) have been added to modify Equation 2.4 so as to derive a maximum safe drinking water level. The adjusted RfD can be written as:

$$\text{Safe drinking water concentration (mg/L)} = \frac{\text{POD (NOAEL or LOAEL)} \times \text{BW} \times \text{PF}}{\text{IR} \times \text{UF}} \qquad \text{(Eq. 2.6)}$$

where BW is the average body weight of an adult (commonly 70 kilograms), PF is a proportionality factor which accounts for the proportion of exposure that may be derived from drinking water (typically 1 or 0.1), IR is the estimated maximum drinking water ingestion rate by an adult (2 L/day), and UF is the uncertainty factor.

For carcinogenic chemicals, as there is no threshold dose, risks can be calculated as follows:

$$\text{Risk (R)} = \text{CSF} \ (\text{mg} \cdot \text{kg}^{-1} \cdot \text{day}^{-1})^{-1} \times \text{Exposure dose} \ (\text{mg} \cdot \text{kg}^{-1} \cdot \text{day}^{-1}) \qquad \text{(Eq. 2.7)}$$

Taking into account of the exposure amount and frequency, the adjusted risk of exposing to carcinogenic chemicals can be written as:

$$\text{Safe drinking water concentration (mg/L)} = \frac{\text{Risk level} \times \text{BW} \times \text{PF}}{\text{CSF} \times \text{IR}} \qquad \text{(Eq. 2.8)}$$

where risk level is the tolerable risk level (usually 10^{-4}, 10^{-5} or 10^{-6}, specified by some international agencies), CSF is the cancer slope factor (Khan, 2010).

Based on these equations, Rodriguez et al. (2007) conducted a screening risk assessment to determine whether the concentration of micropollutants after MF/RO pose any potential health risk for an IPR scheme in Perth, Western Australia. Equation 2.5 was used, in which the detected concentration of each chemical was compared to a benchmark value (non-effect concentration). A total of 134 analytes including volatile organic compounds (VOCs), disinfection by-products (DBPs), metals, pesticides, hormones and pharmaceuticals were sampled at four locations (e.g., WRP inlet, MF permeate, RO permeate and storage dam) and then tested in the laboratory. At the same time, benchmark values were calculated for 3 tiers chemicals. For example, the maximum contaminant level in drinking water from guidelines was used for regulated chemicals, the slope factors or risk specific doses for unregulated toxic chemicals and the threshold of toxicological concern concept for unregulated non-toxic chemicals. The results exhibited that the HQ of final effluent was 10 to 100,000 times below 1 for all VOCs and all pharmaceuticals, except cyclophosphamide (HQ=0.5), while the metals with higher HQ values were arsenic, beryllium, cobalt, lithium and mercury. As all values were well below 1, no increased risk would be posed by recycled water. Notably, additional treatment barriers after RO (e.g., UV light and/or hydrogen peroxide, dilution and retention in the aquifer) can further contribute to a safe drinking water supply.

Moreover, Page et al. (2008) have investigated the risks of three chemicals– diuron, simazine and chlorpyrifos in recycled water for groundwater recharge and IPR schemes. This study used analytical tools for detecting the initial concentration of the chemicals in stormwater and also took the chemical degradation fates into account, where residence time in wetlands and the aquifer, aerobic and anaerobic half life were incorporated in the @Risk Industrial v4.5 software. For each hazard, 10,000 Monte Carlo simulations were performed so that the risk outcomes were statistical distributions and represented the inherent variability as well as uncertainties in each degradation process. Since the initial assumptions used in the risk assessment were extremely conservative, all the predicted concentrations were greater than the guideline values, which indicated that all chemicals posed significant risks. Consequently, it was concluded that the aquifer could not be an effective and reliable barrier and further research would be needed to validate the treatment capacity.

Apart from using instrumental method which is relatively costly to measure the concentration of chemicals, other studies have used the level III fugacity model (Equations 2.9 and 2.10) to predict the transmission fates of chemicals (e.g. steady-state, non-equilibrium concentrations and distributions) from entering into the environment to running out of the WWTP.

$$\text{Concentration (C)} = Z \times f \qquad \text{(Eq. 2.9)}$$

where Z is the fugacity capacity which depends on temperature, properties of chemicals and the nature of the environment into which the chemical is dispersed. f is the fugacity which means the escaping or fleeing tendency of molecules. In level III fugacity model, f can be calculated as follows:

$$f_i \sum D_i = E_i + \sum D_{ji} f_i \qquad \text{(Eq. 2.10)}$$

where E is the chemical discharging rate, D is analogous to the first order rate constant, representing individual process removing the chemical, such as chemical reactions, advective transport, and diffusive exchange between phases. The left part of Equation 2.10 is the rate of transport and transformation that removes chemicals from each compartment, while the right part is emissions and transfers from other compartments (Weber et al., 2006; Cao et al., 2010).

Specifically, Cao et al. (2010) employed the fugacity based model to simulate the distribution of three EDCs (estrone, 17β-estradiol and 17α-ethynylestradiol) in recycled water for an IPR scheme in Southeast Queensland, Australia. This study not only took human as research object but also included fish as comparison. The degradation fates of chemicals in recycled water treated by screening, MF/RO or UF/RO, advanced oxidation (UV/H_2O_2) and chlorination were carefully modelled. Concerning the RfD, the level of plasma vitellogenin was employed as a biomarker of indicated adverse effects for fish, whereas regulation values reported in the Queensland Public Health Regulation were used as benchmarks for humans. The study showed that the majority of EDCs were removed by degradation. The highest HQ was found in 17α-ethynylestradiol with 4×10^{-3} and 2×10^{-4} for fish and humans, respectively. As all simulated concentrations were below fish exposure threshold values and human public health standards, health risks to human are negligible. Overall, fugacity models can be regarded as an effective approach in RA since costly and time-consuming instrumental detection methods can be avoided. They can also trace the chemical degradation fate via wastewater treatment processes, so that it is easy to figure out the removal efficiency of each process. Yet data insufficiency could hinder the determination of HQs and cause a high degree of uncertainty in chemical degradation models.

(2) Quantitative Microbial Risk Assessment (QMRA)

Using QMRA to characterize human health risks associated with exposure to pathogenic microorganisms was first proposed in the 1970s and has been gaining favour since the 1980s (Hamilton et al., 2006; Soller and Eisenberg, 2008). When being applied to recycled water field, QMRA is able to estimate order-of-magnitude risks within a community following exposure to pathogens under specific scenarios (Mena et al., 2008; Toze et al., 2010). Besides, QMRA results are useful in interpreting risk data, preparing further analysis and developing rational objective remediation plans (Ashbolt et al., 2010). As hazard identification and exposure assessment processes in QMRA are quite similar to those in QCRA, they will not be discussed in detail. With respect to dose-response relations, based on historical data (e.g., clinical experiments, epidemiological investigations, animal studies, and/or toxicity assays on mammalian or bacterial cells), dose response relationships for specific species can be established and used to quantify the probability of infection (Soller, 2006). Sigmoidal equations were found to be the best tool to describe the relationship (Fane et al., 2002). Particularly, the dose-response relation for many protozoans and viruses tend to follow the exponential model (Equation 2.11), while beta-Poisson model (Equation 2.12) is more suitable for many bacteria and some viruses (McBride et al., 2002).

$$P_i = 1 - \exp(-rd) \qquad \text{(Eq. 2.11)}$$

$$P_i = 1 - (1 + \frac{d}{\beta})^{-\alpha} \qquad \text{(Eq. 2.12)}$$

where P_i is the daily probability of infection, d refers to the mean ingested dose, r, α, β are empirical parameters which are assumed to be constant for any given host and pathogen. Table 2.12 gives some empirical values for these parameters.

Table 2.12 Dose-response models from various enteric pathogen ingestion studies

Model	Exponential	Beta-Poisson	
Constituent	r	α	β
Virus			
Adenovirus	0.417	–	–
Echovirus 12	–	0.37	186.69
Norovirus	–	0.04	0.055
Rotavirus	–	0.253	0.427
Poliovirus 1	0.0091	0.110	1524
Poliovirus 3	–	0.409	0.788
Bacteria			
Salmonella	0.0075	0.313	2360
Shigella	–	0.2	2000
E.coli	–	0.171	1.61×10^6
E.coli O157:H7	–	0.4	45.9
Campylobacter	–	0.145	7.589
Vibrio cholerae	–	0.097	13,020
Protozoa			
Cryptosporidium	0.09	–	–
Giardia	0.02	–	–

Note: Modified from Asano et al., (2007); Soller et al., (2010b).

Accordingly, annual probability of risk can be calculated as:

$$P_a = 1 - (1 - P_i)^n \qquad \text{(Eq. 2.13)}$$

where n is the number of days. Noticeably, only some amount of the infected person developed clinical disease. The risk of becoming diseased or ill can be written as:

$$P_D = P_{D:i} \times P_i \qquad \text{(Eq. 2.14)}$$

where $P_{D:i}$ is the probability of an infected person developing clinical disease. Additionally, other empirical models (e.g., Weibull-Gamma, Log-logistic and Log-profit models) can be used for some pathogens under certain conditions (Haas et al., 1999; Holcomb et al., 1999).

As above-mentioned equations are only suitable for acute effects in most cases, Disability Adjusted Life Years (DALYs) is an alternative way to quantify the probability of infection. It accounts for not only acute health effects but also delayed and chronic health effects including morbidity and mortality. Overall, DALYs attempts to measure the health of a population with regard to the time lost because of disability or death from a specific disease or risk factor. This facilitates the comparisons of health outcomes (Campos, 2008). Equations 2.15 and 2.16 are commonly applied for the estimation of DALYs:

$$\frac{\text{DALYs}}{\text{year}} = P_{\text{ill/y}} \times \text{DALYs}_{\text{per case}} \times S_{\text{fraction}} \qquad \text{(Eq. 2.15)}$$

$$P_{\text{ill/y}} = P_{\text{ann inf}} \times (\text{ill : inf}) \qquad \text{(Eq. 2.16)}$$

where $P_{\text{ill/y}}$ is the annual probability of illness; ill : inf is the ratio of illness to infection for the specific pathogen; $\text{DALYs}_{\text{per case}}$ is a function of years of life lost due to the disease and years lived with a disability; S_{fraction} is the proportion of the population susceptible to developing the disease following infection. The values of ill:inf, $\text{DALYs}_{\text{per case}}$ and S_{fraction} for specific pathogens can be determined from epidemiological studies (Hamilton et al., 2007). Moreover, new predictive Bayesian methods for dose-response assessment have been proposed in some studies (Englehardt, 2004; Englehardt and Swartout, 2004). These studies concluded that the Bayesian models could handle limited subjective and numeric information, prioritize expenditures for environmental protection and terrorist threats as well as assess health effects of new and existing chemicals and pathogens. Besides, they have other strengths such as flexibility, less data requirement and higher data incorporation than empirical models.

Regarding the types of QMRA models, the most generic ones are static microbial risk assessment (MRA) models and dynamic MRA models. Static models assume that the number of individuals which are susceptible to infection is not time varying. They normally focus on estimating the probability of infection to an individual as a result of a single exposure event. Thus, risk is characterized at an individual level. It is also assumed that the population may be categorized into two epidemiological states: a susceptible state and an infected or diseased state. The susceptible individuals are exposed to the pathogen of interest from a specific pathway under consideration. They move into the infected or diseased state with a probability that is governed by the dose and infectivity of the pathogen (Soller, 2006).

Case studies using static MRA models on recycled water reuse applications have been reported widely. For example, Westrell et al. (2004) investigated the risks of several important pathogen indicators (e.g., *Escherichia coli*, *Salmonella*, *Giardia*, *Cryptosporidium*, *Rotavirus* and *Adenovirus*) in 8 recycled water exposure scenarios using @Risk software. The dose of pathogens for each exposure was estimated from the concentrations in raw sewage and STP based on literature data and previous study at the plant. The corresponding dose-response models in Table 2.12 together with the Monte Carlo technique with 10,000 simulations were adopted in the software for risk characterization. Table 2.13 summarizes the estimated risks of each pathogen associated with four important scenarios. As can be seen, the highest individual risk per single exposure was achieved through exposure to droplets and aerosols for workers at the treatment plant whereas the lowest risk arose from swimming in the lake. Regarding pathogens, viruses gave the highest risk due to high influent concentrations, low infectious doses and high resistances. This study found that the @Risk software was able to compare different reuse scenarios in a relatively short time but did not consider the secondary transmission. Besides, the worst case scenarios such as flooding, a major failure in the wastewater treatment or sudden peaks based on treatment variability were not discussed either, which might be crucial for comprehensive analyses of large-scale recycling schemes.

Additionally, other static MRA studies are summarised in Table 2.14, in which some improvements have been observed. For instance, some studies took the pathogen decay rates into account while others combined the Monte Carlo technique and local hydrological data in the model to reflect the reality better. However, because of difficulties in obtaining sufficient data, MRA models may involve many assumptions. For instance, when considering pathogen decay, some studies assumed the constant decay rates regardless of other dynamic die-off reasons (e.g., desiccation, sunlight or predation) due to data and technical restrictions. Although the above-mentioned static models can provide satisfactory risk estimates when the risks associated with direct exposure to potential hazards are low, they could not simulate the effects of secondary transmission and immunity. When the direct risks increasing to a high level, there is a need for more complex model to account for secondary effects (Soller and Eisenberg, 2008).

Table 2.13 Median number of yearly infections resulting from different exposure scenarios

Exposure scenario	Vol.[1]	Freq.[2]	No.[3]	*E.coli*	*Sal.*[4]	*Giardia*	*Cp.*[5]	*RV.*[6]	*Ad.*[7]
WWTP worker at pre-aeration	1	52	2	0.06	0.004	0.14	0.02	1.98	1.99
Child playing at wetland inlet	1	2	30	0[i]	0	0.0006	0	0.13	0.23
Recreational swimming	50	10	300	0	0	0.0005	10^{-4}	0.04	0.18
Consumption of raw vegetables	1	2	500	0.002	0	0.002	0.01	0.21	0.41

Note: Adapted from Westrell et al., (2004).
[1] Volume ingested per person per exposure (mL); [2] Frequency (times per year);
[3] Number of people affected; [4] *Salmonella*; [5] *Cryptosporidium*; [6] *Rotavirus*; [7] *Adenovirus*.
0 is equivalent to <0.0001 infections.

Table 2.14 Static MRA models for different end uses

End uses	Pathogen (model)	Assumptions	Risk assessment results	Characteristics of model	References
Agriculture	*Virus–Hepatitis A* and *cholera* (beta-Poisson)	Any pathogens contained in recycled water remaining on the irrigated vegetables would be counted	•The risk from consuming cucumbers = 10^{-7} to 10^{-8}/year •The risk from consuming lettuce = 10^{-6} to 10^{-8}/year	•The instruments determined the pathogen doses on vegetables •The assumptions on dose of pathogens do not consider the actual field conditions	Shuval et al., (1997)
Agriculture on paddy field	*E. coli* (beta-Poisson)	•Scenario A assumed that farmers and children are exposed for 100 and 30 days respectively •Scenario B assumed exposure for 30 and 10 days respectively	•Annual risks of 1 h and 24 h after irrigation were 10^{-4} -10^{-5} to 10^{-5} -10^{-6} •Scenario A had greater risk and children had greater risk •UV disinfection significantly reduced the risk	•The dose of E.coli was measured by laboratory instruments •Monte Carlo simulation was performed based on 10,000 trials and risk values were used in the 95% confidence region	An et al., (2007)
Landscape irrigation of parks and golf courses	*Cryptosporidium* (exponential)	•All infections cause illness •No degradation of pathogen by desiccation, predation, sunlight or other reasons	The risk of 1 ml exposure to tertiary treated recycled water = 2.34×10^{-7}	•Samples was tested by laboratory instruments •More pathogen data are needed •The results are conservative	Jolis et al., (1999)
Green space irrigation	•*Rotavirus* and *Campylobacter* (beta-Poisson) •*Cryptosporidium* (exponential)	•Pathogen in secondary effluent were infiltrated at a steady rate and no infiltration or adsorption during passage, only decay •No mix of recycled water with native groundwater	•The mean residual risk to human health was *Rotavirus* > *Cryptosporidium* > *Campylobacter* with the range of 10^{-5} to 10^{-8}. •To obtain a mean risk below the WHO, the residence time in the aquifer need to be 150 days	•The model incorporated pathogen decay, hydrological, uncertainty and variability factors to represent the reality •Pathogen decay rate was determined from the slope of regression line fitted by pathogen numbers over time	Toze et al., (2010)

Note: U.S. EPA's acceptable risk benchmark = 10^{-4}; WHO guideline's acceptable risk value = 10^{-6} DALY.

Table 2.14 (continued)

End uses	Pathogen (model)	Assumptions	Risk assessment results	Characteristics of model	References
Landscape irrigation & residential non-potable reuse	*Rotavirus* & *Giardia* (beta-Poisson)	•Pathogens were shed at fixed rate: 200 g/p/d[1] of faeces and 145 l/p/d[2] of wastewater were generated •4.5 & 2.5 log removal of viruses & protozoa in STP for irrigation; 6 & 4 log removal of viruses & protozoa in STP for residential use •Exposure was 1 and 19.4 mL/p/a[3] for irrigation and residential respectively	•*Giardia* is less infective than *Rotavirus* and the risk is higher in irrigation scenario •The risk increases with the increase of size of population •Risk for small exposures is higher than that from a single large volume of exposure	•The model assumes no thresholds •Some issues that could affect a general acceptance were not taken into account	Fane et al., (2002)
Greywater in-house recycling	*Salmonella* (exponential)	•The population number of reported cases of Salmonella is 60,000 •An infected person sheds organisms into the greywater system for 2 days •4.4 people would be exposed to the system in any day	•The probability of infection: $<1.5\times10^{-7}$ (disinfection system is operating correctly) ; $<1.5\times10^{-3}$ (no disinfection) •The anaerobic COD release rate in the system storage tank increases and DO decreases during pump failure	•The model considered the hydraulic characteristics and system failures and Monte Carlo was used to generate exposure data from frequency distributions •Information on the growth kinetics and different pathogens were insufficient	Diaper et al., (2001)
Drinking water-recycled water cross connection	*Salmonella* (beta-Poisson)	•All microorganisms present in the effluent were detected and all were infectious; a drinking water consumption was 1.4 l/p/d[2] •Salmonella concentrations were constant for the entire duration	•Risks of Salmonella infection range from 0.1 after a 1 day exposure to 0.99 for 30 and 90-day exposure durations •Cross-connection would result in higher risks than USEPA	•Risks associated with the multi-day exposure durations may be over-estimated •The dose-response parameters were determined based on healthy volunteers	Mena et al.,(2008)

Note: U.S. EPA's acceptable risk benchmark = 10^{-4}; WHO guideline's acceptable risk value = 10^{-6} DALY.
[1]grams per person per day; [2]litres per person per day; [3]mL per person per year.

Comparatively, Figure 2.9 shows the possible disease transmission routines in dynamic MRA models. Label S, E, C, D and P stand for different states associated with pathogen infection. C1 represents the individuals who are infected but do not have symptoms of disease, whereas C2 represents the individuals who are still infected but no longer exhibit symptoms of disease. Symbols α, β, σ, δ and γ are the rates of movement from one epidemiological state to another and P_{sym} refers to the probability of a symptomatic response (Soller et al., 2004; 2010a).

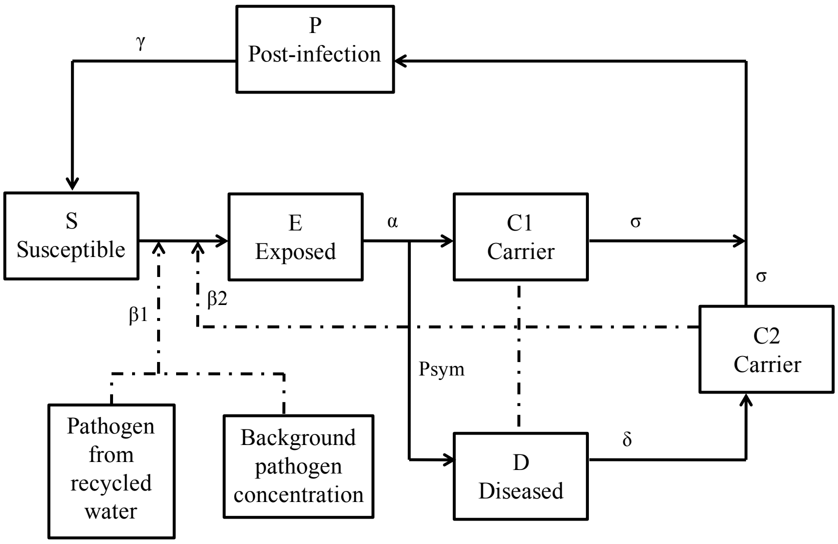


Figure 2.9 Disease transmission model for a dynamic risk assessment (adapted from Soller et al., 2010a).

Compared with static MRA models, dynamic models consider not only the direct exposure to pathogens (S-β1-E-D) but also other indirect factors forming other transmission routines (e.g., S-β2-E-D, C1-P-S-E-D , C2-P-S-E-D, etc.), such as person-to-person transmission, immunity, asymptomatic infection and incubation period. Hence, the dose-response function is an important health component but not critical since factors specific to the transmission of infectious diseases may also be important. Furthermore, as dynamic models also take into account the immunity, exposed individuals may not be susceptible to infection or disease because they may already be infected or may be immune from infection due to prior exposure. If the risk is manifest at the population level, the number of individuals susceptible to infection is time varying. Consequently, dynamic models are undoubtedly more sophisticated (Soller, 2006; Soller and Eisenberg, 2008). The two main forms in dynamic MRA models are deterministic and stochastic, of which the characteristics and applications are shown in Table 2.15.

Table 2.15 Characteristics of deterministic and stochastic models

Deterministic model	Stochastic model
• The model is expressed as a set of different equations that have defined parameters and starting conditions	• The model incorporates probabilities at an individual level and is evaluated by an iterative process such as Monte Carlo analysis
• The model does not account for uncertainty and variability associated with model parameters	• The model requires substantially more data to account for population dynamics and protection from infection due to prior exposures
• This model is most suitable for large populations of individuals randomly interacting with one another	• The model accounts for uncertainty and variability to some extent
	• This model is most suitable for small populations with heterogeneous mixing patterns

Note: Modified from Koopman et al., (2002); Soller et al., (2003).

In regard to the applications of dynamic MRA models on recycled water schemes, Hamilton et al. (2007) have introduced a deterministic recycled water irrigation risk analysis (RIRA) tool for Australian irrigation schemes. In RIRA, once pathogen concentration and exposure scenario are inputted together with the chosen dose-response model, the annual risk on health can be obtained immediately. The result is then compared with U.S. EPA's benchmark (10^{-4}) to arrive at the optimal decision. Alternatively, when the DALY metric is selected, the model output is compared to the WHO's tolerable risk level (10^{-6} DALY per year). Overall, the RIRA model is capable of calculating many risk levels in a short period of time with a wide variety of irrigation scenarios, which are convenient and practical for users. The generic and flexible structure of the model also makes it possible to be used in screening level risk assessments for other water reuse scenarios. Besides, the model can investigate the relative merits of different management strategies (e.g., lengthen vs. shorten the time between the last recycled water irrigation event and harvest). Nevertheless, as a deterministic model, RIRA fails to account for the uncertainty associated with the parameters. In future studies, solutions to convert RIRA into a stochastic model might become available.

On the other hand, stochastic models are increasingly being considered, which incorporate uncertainty, variability and a large number of Monte Carlo trials (e.g., 5000 or 10,000 times). These calculations are mostly relied on commercial softwares such as @risk or Crystal Ball. For example, Hamilton et al. (2006) used a stochastic model for QMRA on five different crops (i.e., broccoli, cucumber, Savoy King/Grand Slam cabbage, Winter Head cabbage and lettuce) that were spraying irrigated with secondary effluents. Enteric viruses were chosen as the specific microbial hazard to model as they are highly infective. The daily doses of enteric viruses were calculated from the probability distribution functions according to variation factors. It was shown the constant pathogen decay rate (k=0.69) contributed to a higher risk than the normally distributed decay rate (μ=1.07, σ=0.07 day^{-1}). With respect to crops, consuming lettuce resulted in the highest risk of infection whereas cucumber had the lowest risk potential. The study also evaluated the impact of different duration times in the environment (e.g., 1 day, 7 days and 14 days) on the annual risk. The mean annual risk was demonstrated to decrease with the increase of duration time. Given a 14-day withholding period, the annual probabilities of enteric virus infection derived from consuming vegetables were 10^{-4} to 10^{-7} which were below the U.S. EPA benchmark (10^{-4}). Hence, wastewater can only be safely reused for agricultural irrigation with sufficient decay rate and withholding time.

Table 2.16 illustrates other stochastic models on different water reuse applications. These models were often coupled with other site specific models (e.g., water quality model, hydraulic model and disease transmission model) to reflect local reality. However, as stochastic approach is complicated, combining other models often make the analysis even more difficult to understand and introduce larger uncertainties. Other weaknesses include the inseparability of variability and uncertainty associated with a lack of knowledge.

According to the above-mentioned literature, QMRA models may often be restricted by a paucity of data. They are also hard to determine which process components mainly contribute to disease risk. Even if stochastic form is the most advanced and complicated QMRA model, it is inapplicable when uncertain parameters cannot be expressed as probability distributions (Brouwer and De Blois, 2008). For these reasons, other RA approaches or integrated tools such as fuzzy stochastic modelling and Bayesian network should be increasingly considered (Chen et al., 2010). For example, Donald et al. (2009) introduced a Bayesian Network (BN) model for RA of diarrhea related to recycled water consumption. The model investigated various factors and determined their influence on whether the quality of the water could be classified as acceptable (safe) or unacceptable (unsafe). The various infection factors and pathways were represented by relevant nodes and the values of each node were expressed as probability functions based on an expert opinion. The BN approach on point estimates allowed making various predictions to the risks posed under different scenarios. It was also able to identify the nodes that contribute most to the outcome of gastroenteritis, thereby providing an additional way in modelling recycled water quality.

Table 2.16 Stochastic models for risk assessment on recycled water applications

End uses	Pathogen (model)	Assumptions	Risk assessment result	Characteristics of model	References
IPR of stormwater	•*Rotavirus* and *Campylobacter* (beta-Poisson) •*Cryptosporidium* (exponential)	•The distributions of pathogens have triangular functions •Initial concentration, residence time, aerobic and anaerobic decay rate were specified for pathogens	The risks of infection are 1.5×10^{-3}, 4.6×10^{-3} and 8.4×10^{-3} DALYs for *Cryptosporidium*, *Campylobacter* and *Rotavirus* respectively	•The model outcome is a statistical distribution of risk experienced by the diverse members of the population •The QRA model was developed to facilitate Monte Carlo simulations which can provide a sensitivity analysis of the influence factors	Page et al., (2008); Page et al., (2009)
Recreational use (San Joaquin River recharge)	•*Viral gastroenteritis* (beta-Poisson) •A hydraulic model •A disease transmission model	•The model virus possessed the clinical features of rotavirus •Number of individuals initially in the susceptible state is equal to the total population for the study area •Data below the detection limit are present at that limit	•The risk was calculated under summer •The risk was several orders of magnitude below 8-14 illnesses per 1000 recreation events (less than USEPA) •Winter tertiary treatment would further reduce the risk by 15-50%	•The model consists of 5 state variables, 11 model parameters and 3 intermediate parameters •The risk for winter operation represents a upper bound and the model is not practical to estimate the cumulative risk •The true treatment efficiencies was underestimated and storm events and were not modelled	Soller et al., (2003)
Recreational use (Newport Bay recharge)	•*Rotavirus* (beta-Poisson) •Water quality model •A disease transmission model	•The virus was prevalent and persistent in the environment •The boundary conditions in the water quality modelling were based on the maximum observed concentrations •Data below the detection limit are present at that limit	•The risk estimates for recreation in the Bay were 0.9 illnesses per 1,000 recreation events (less than USEPA) •Control measures reduced pathogen loading by an additional 16% to 50%	• It is not practical to estimate the cumulative risk •A number of other more serious disease outcomes were not modelled •It is not practical to carry out separate assessments for all pathogens	Soller et al., (2006)

Note: U.S. EPA's acceptable risk benchmark = 10^{-4}; WHO guideline's acceptable risk value = 10^{-6} DALY.

2.5.3.3 Quantitative ERA models

The potential environmental risks resulted from recycled water projects include:

- Substantial alteration of land use;
- Conflict with the land use plans or policies regulations;
- Adverse impact on wetlands;
- Affection of endangered species or their habitat;
- Populations displacement or alteration of existing residential areas;
- Antagonistic effects on a flood-plain or important farmlands;
- Adverse effect on parklands, reserves, or other public lands designated to be of scenic, recreational, archaeological, or historical value;
- Significant contradictory impact upon ambient air quality, noise levels, surface or groundwater quality or quantity;
- Substantial adverse impacts on water supply, fish, shellfish, wildlife, and their actual habitats (Urkiaga et al., 2008).

Even if the schemes are conducted far away from human targets or activity zones, they can induce environmental burdens unintentionally when recycled water quality is unacceptable (Tiruta-Barna et al., 2007; Corwin and Bradford, 2008). Due to above concerns, ERA evaluates ecological risk impacts of environmental changes or multiple stressors in the relevant system boundary over long periods. It integrates ecology, environmental chemistry, environmental toxicology, geochemistry and other fundamental sciences to characterise the impacts of natural and man-made disturbances on ecological resources (Bartell, 2008). Figure 2.10 gives general steps in performing ERA where the environmental risk of particular compounds should be carefully identified unless sufficiently low concentrations are observed. Once the requirements from risk managers and decision makers are fulfilled and documented, the processes end (Carlsson et al., 2006; Muñoz et al., 2009a).

In spite of difficulties, numerous ERA models have been increasingly developed, which are considered to be quite useful when empirical measurements of toxic effects are not available, measured values are scarce or the exposure level is being projected into the future (Lee et al., 2007). The simplest ERA approaches normally employ chemical analytical instruments such as atomic adsorption spectrophotometer (AAS), inductively coupled plasma optical emission spectroscopy (ICP), liquid chromatography (LC), gas chromatography (GC) or mass spectrometry (MS) to determine the predicted environmental concentration (PEC) of the pollutant and then compare it with the predicted no effect concentration (PNEC) guideline value so as to obtain the potential risk. Comparatively, instead of using costly detection instruments, some ERA models estimated the PEC using mathematical equations, which are based on initial pollutant concentration, percentage removal rate, dilution factor and the volume of recycled water whereas others even take into account the biodegradation effect of the substance during environmental exposure and tend to be more complicated. Moreover, some studies also adopted empirical models such as the ecological structure activity relationship (ECOSAR), in PNEC calculation when relevant data were not available (Jones et al., 2002).

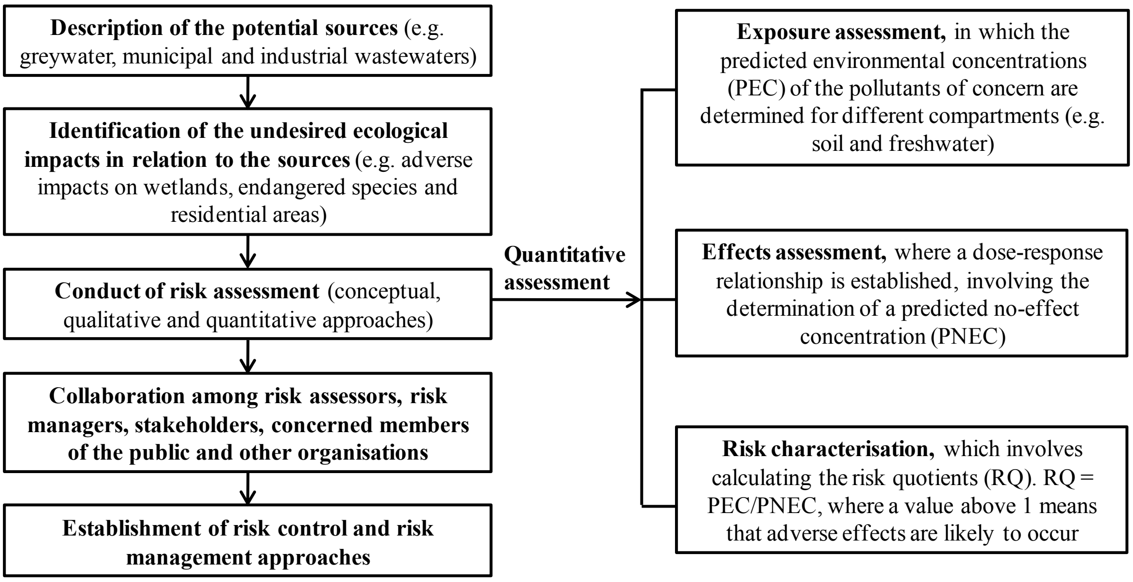


Figure 2.10 Steps in performing ERA (modified from Bartell, 2008).

(1) Agricultural irrigation

To ensure the long term sustainability of agricultural water recycling schemes, ERA models are widely used to evaluate the potential effects of waterborne hazards on soil and surrounding groundwater quality. Regarding inorganic chemicals, Xu et al. (2010) observed the long term (3, 8 and 20 years) recycled water irrigation (56.78 ML/d, processed by primary sedimentation and oxidation pond) on agricultural soils for plots growing trees and feed crops at Palmdale, California, the U.S. Despite nutrient recovery and fresh water savings, the ICP analytical results showed that trace metals including Cr, Cu, Ni and Zn in the upper horizons were accumulated over years, which eventually deteriorated the soil and groundwater quality. A similar study by Li et al. (2009b) using AAS indicated that although irrigating with poorly treated industrial wastewater at Zhangshi irrigation area in Shenyang, China has been ceased since 1992, the Zn, Pb and Cu concentrations were still higher than or close to China's grade A standard due to 30 years' accumulation effects. Thus, the place should be abandoned for cultivated crops and bioremediation or other measures should be carried out. Likewise, Katz et al. (2009) reported the elevation of nitrate, boron and chloride concentrations in groundwater samples from the Sprayfield aquifer, where municipal wastewater under secondary treatment was supplied for agricultural irrigation. Peasey et al. (2000) and Jimenez and Asano (2008) also found a correlation of risk problems with the proximity to farms where recycled water had been applied.

Furthermore, Muñoz et al. (2009a) utilised a more complicated ERA model in the risk characterization of recycled water for agricultural irrigation, where both heavy metals and pharmaceuticals are modelled. The risk exposure to soil organisms (PEC_{soil}) and the second poisoning to top predators via terrestrial food chain ($PEC_{predator}$) were quantified by European Commission Technical Guidance Document on Risk Assessment and level III fugacity model while PNEC values were derived from ECOSAR software. The case studies on 27 pollutants in secondary treated effluents from two Spanish STPs showed that both plants were likely to cause adverse effects on agriculture soil and predators. The Ni concentrations in recycled water from the Alcala de Henares STP which receives a mixture of domestic and industrial wastewater were toxic to both soil and predators whereas pharmaceuticals such as sulfamethoxazole, ciprofloxacin, diclofenac, gemfibrozil and erythromycin in both effluents posed high risks on soil compared with diclofenac in effluents on predators. Thus, additional treatment with membrane filtration, AOP and UV disinfection are recommended.

(2) Environmental and recreational uses

To release reliable environmental and recreational flows and protect the downstream health of the rivers, many ERA studies have been conducted with a focus on pharmaceuticals in recycled water. For instance, Stuer-Lauridsen et al. (2000)

investigated 25 pharmaceuticals in Denmark under worst case scenarios where all sold pharmaceuticals were assumed to be used evenly at temporal and spatial scale in the same year and then released to the sewage system without any attenuation. Accordingly, PECs were calculated as shown in Equation 2.17.

$$\mathrm{PEC_w} = \frac{\mathrm{A} \times (100 - \mathrm{R})}{365 \times \mathrm{P} \times \mathrm{V} \times \mathrm{D} \times 100} \qquad \text{(Eq. 2.17)}$$

where A is the amount used per year (kg/yr), R is the removal in percent (set to zero), P is the number of inhabitants in Denmark (5,200,000 in 1997), V is the volume of wastewater per day per capita (0.2 m^3) and D is the dilution factor in the environment (a default value of 10 is used). The calculated PECs were generally consistent with actual measured pharmaceutical concentrations. On the other hand, as ecotoxicity data were only available for 6 compounds, the corresponding PNECs were derived on the basis of EU draft guideline with a default safety factor of 1000. Finally, high risks were observed on acetylsalicylic acid, paracetamol and ibuprofen while low risks were achieved for estrogen, diazepam and digoxin. Although the mathematical PEC model is easy to perform, the lack of chronic toxicity data for PNECs is the prime obstacle in the study. To solve this problem partially, Jones et al. (2002) used the ECOSAR model in PNEC estimation when assessing the aquatic environmental risks of the top 25 English prescription pharmaceuticals. Four types of pharmaceuticals including mefenamic acid, oxytetracycline, paracetamol and amoxicillin were shown to be of high risk. Nonetheless, the risk quotients (RQ) of the pharmaceuticals might be overestimated in both studies since worst case scenarios were applied in calculating PECs as well as acute ecotoxicity data in quantifying PNEC.

Additionally, Carlsson et al. (2006) have made some improvements in estimating PECs of 27 pharmaceuticals in Sweden. More specifically, PECs were firstly calculated under worst case assumptions using Equation 2.18 and then refined by a Simple Treat 3.1 model to reflect local realistic environmental conditions. In order to trace the steady-state pharmaceutical concentrations in recycled water, sludge or air, several physio-chemical parameters (e.g., molecular weight, K_{ow}, vapour pressure, water solubility, dilution factor, degradation rates and acid-base dissociation constants) were taken into account. Owing to limited biodegradation data, the degradation rate was assumed to be 0.1/h for paracetamol but zero for others. In regard to PNECs, when toxicities of the chemicals were known, they could be derived from the lowest available acute values to organisms (e.g., LC_{50} of fish, EC_{50} of daphnia and IC_{50} of algae) divided by an assessment factor of 1000. Otherwise, ECOSAR model should be applied. Overall, nine substances were considered to be dangerous while only the oestradiol and ethinyloestradiol were likely to cause long term adverse effects to the aquatic compartment. Despite considering local conditions in PEC estimation, most RQ values were still overestimated as chronic toxicity data were only available for 4 substances.

$$\mathrm{PEC}_{\text{surface water}} = \frac{\mathrm{DOSEai} \times \mathrm{Fpen}}{\mathrm{WASTEWinhab} \times \mathrm{DILUTION} \times 100} \qquad \text{(Eq. 2.18)}$$

where DOSEai is the highest recommended daily dose of pharmaceuticals in question, Fpen is the percentage of market penetration. WASTEWinhab is the amount of wastewater used per inhabitant per day and DILUTION is the dilution of sewage water in surface water.

Furthermore, Escher et al. (2011) proposed another ERA approach for pharmaceuticals from hospital wastewater which might be directly discharged to surface waters or infiltrated. The detailed PEC calculations under four scenarios were shown in Table 2.17 and PNECs were estimated from acute toxicity (EC_{50}) values of green algae, water flea and fish divided by 1000. RQs of Top-100 pharmaceuticals in Switzerland were then calculated for the general hospital and psychiatric centre respectively. The results demonstrated that for the chemicals with $RQ_{HWW}>1$, dilution effect significantly decreased the RQ values ($RQ_{\text{WWTP influent}}<1$) and dilution was observed even had a better effect than actual elimination in WWTP. However, it is not the real fact as clotrimazole and ritonavir were found to be highly removed (>80%) in biological treatment. Due to lack of data, no elimination in STP was assumed for 55 and 66 pharmaceuticals in the general hospital and psychiatric centre respectively, which may introduce biases to the final conclusions. Overall, the study allows setting priority risks for further testing and the related equations are also good references for other ERA studies.

Table 2.17 Hospital wastewater treatment scenarios and associated PEC calculations

Option	Descriptions	Equations for PEC calculation
1	Risk potential of the wastewater from the hospital main wing, before discharging to the sewer (full risk potential without any degradation or dilution)	$PEC_{HWW} = \frac{M \cdot f_{excreted}}{V_{HWW}}$ $M = \sum_{i=1}^{n} m_i = \sum_{i=1}^{n} U_i m_{U_i}$
2	Risk potential at inlet of the WWTP (reduction of risk potential through dilution in sewers)	$PEC_{WWTPinfluent} = df \cdot PEC_{HWW}$
3	Risk potential at outlet of the WWTP (reduction of risk potential through degradation and sorption processes with dilution in sewers)	$PEC_{WWTPeffluent} = f_e \cdot PEC_{WWTP\ influent}$
4	Risk potential at the hospital main wing after hypothetical conventional biological treatment (reduction of risk potential through degradation and sorption process without dilution)	$PEC_{WWTPeffluent} = f_e \cdot PEC_{HWW}$

Note: Modified from Escher et al. (2011).

PEC_{HWW} was the concentration of active ingredient expected in hospital wastewater; M is the amount of each active ingredient consumed in the hospital. m_i can be derived from the units consumed for each drug preparation, U_i, and the amount of active ingredient contained in each unit, m_{Ui}. df was the dilution factor in the sewer and assumed to be 0.013. f_e referred to the fraction eliminated in the STP.

Another study by Lee et al. (2008) assessed the environmental risks of most concerned antibiotics at the Gapcheon WWTP in Daejeon, Korea. The exposure doses of the chosen 13 antibiotics were estimated by a Korea ecological risk assessment (KOREOCORisk) model. The model consists of a release rate estimation module, an exposure estimation module and an ERA module. The release rate and biodegradation removal efficiencies of the STPs were taken into account in the release rate estimation module whereas multimedia fate model rather than dilution factor was applied for calculating the site specific regional PECs in exposure estimation module.

Comparatively, PNECs were collected from open literature, review, the ECOTOX database or calculated from ECOSAR model. The model outcomes indicated that RQ of amoxicillin and erythromycin were 151 and 3, respectively, which may chronically degrade the Korean aquatic environment. Lee et al. (2007) also verified the effectiveness of KOREOCORisk in setting management priority among the industrial chemicals in the aquatic environment. Nevertheless, the calculations inside the model are complicated which involves calling sub-models and the uncertainties vary greatly (10 to 10^3).

2.5.3.4 Characteristics and weaknesses of RA models on water reuse

Overall, each RA model has its unique strengths and weaknesses. Some models only address one or a few numerous components of the physical process regarding water treatment and hazard degradation, while others attempt to be more comprehensive. For different recycled water applications, the selection of an appropriate model form and suitable analytical approaches is very important (Soller et al., 2004). Initial efforts aimed towards deterministic models with the assistance of analytical instruments whereas recent studies mainly focus on stochastic or integrated models which account for uncertainty and variability. However, to presume that one model form is most appropriate in different situations is unrealistic (Soller, 2006). Havelaar et al. (2004) explained the steps of converting deterministic approach to stochastic form. Gronewold and Borsuk (2009) also introduced a software tool to translate deterministic model results into stochastic approaches for water quality analysis. Although the complicated methods can better reflect realistic conditions, it should be modified into simpler forms when the variations in modelling outcomes are considerably large.

Despite these efforts, there are still a number of constraints in current RA. For example, in HRA, the dose-response models or curves can often lead to gross overestimates of risk at relatively low pathogen doses. Some accurate models have a maximum risk curve, which limits the upper confidence limit of the dose-response relationship. Similarly, in ERA, most of the PEC values were calculated based on worst case scenarios so that RQs are likely to be overestimated. Hence, more relevant data on chemical metabolism, spreading routes, environmental biodegradation, bioaccumulation potential, partitioning characteristics, human use patterns, wastewater treatment and catchment conditions should be collected. When site specific data are available, several models such as Simple Treat, Pharmaceutical Assessment and Transport Evaluation and Geo-referenced Regional Exposure Assessment Tool for European Rivers can be further applied to refine the PECs towards more realistic. As for PNECs, current ecotoxicity tests rely solely on limited aquatic organisms (e.g., daphnia and algae) which can not represent ecotoxicological responses of the whole ecosystem. Besides, most of the ecotoxicity tests are based on acute responses and are not able to reflect the potential chronic effects following long term exposure to subacute levels. Although the acute to chronic ratios are available for some substances, they are empirical and sometimes the potency to cause chronic ecotoxic effects is not correlated with a potency to cause acute effects. Moreover, since pharmaceuticals mostly exist as complex mixtures, synergistic effects may occur. For example, clofibric acid combined with carbamazepine as well as diclofenac combined with ibuprofen exhibited a much stronger toxic effect than the sum of their individual effects (Cleuvers, 2003). As a consequence, future work should address these difficulties and create a preliminary risk assessment database where the different grades of recycled water versus corresponding health or environmental risks are clarified (Jones et al., 2002; Carlsson et al., 2006; Cooper et al., 2008).

2.5.3.5 Risk control on recycled water

When the chemical or microbial risks of recycled water are predicted to be high to human health and/or the environment according to RA models, risk control should be conducted to ensure the safety and success of the recycling scheme. Risk control approaches include source control, recycled water quality improvement, critical point control and exposure control. While source control can partially prevent trade waste and other hazards from entering the sewage system, recycled water quality improvement through advanced treatment processes is able to reduce the chemical or pathogenic risks to tolerable levels quickly. Besides, safety assurance and monitoring also plays an important role in risk control. This can be achieved by establishing a hazard analysis critical control point (HACCP) system for recycling schemes with a focus on controlling the risk exposure and reducing the hazards through quick and effective treatment (Salgot et al., 2003). Figure 2.11 illustrates the HACCP on a water recycling system in terms of health/sanitation, technical and ecological aspects. The health or sanitation control pays attention to the detection of microbiological quality parameters or indicators (e.g., *legionella spp*, *nematode*, *E. coli*, *enterococci*, *cryptosporidium*, *giardia*, *enterovirus* and organic micro-contaminants). Comparatively, the technical control takes into account of key treatment processes and distribution systems whereas the ecological control focuses on the recycled water quality in the distribution and reuse systems. Derry et al. (2006) pointed out that other biophysical indicators (e.g., thermotolerant coliform, BOD, DO, pH, temperature, conductivity and TSS) are also commonly selected for monitoring at control points. Overall, in addition to risk control, the benefits such as the increase of safety in a recycled water chain, economic cost savings (by reducing the number of inspections), better treated effluent quality, real time information collection can also be achieved when implementing the HACCP (Huertas et al., 2008).

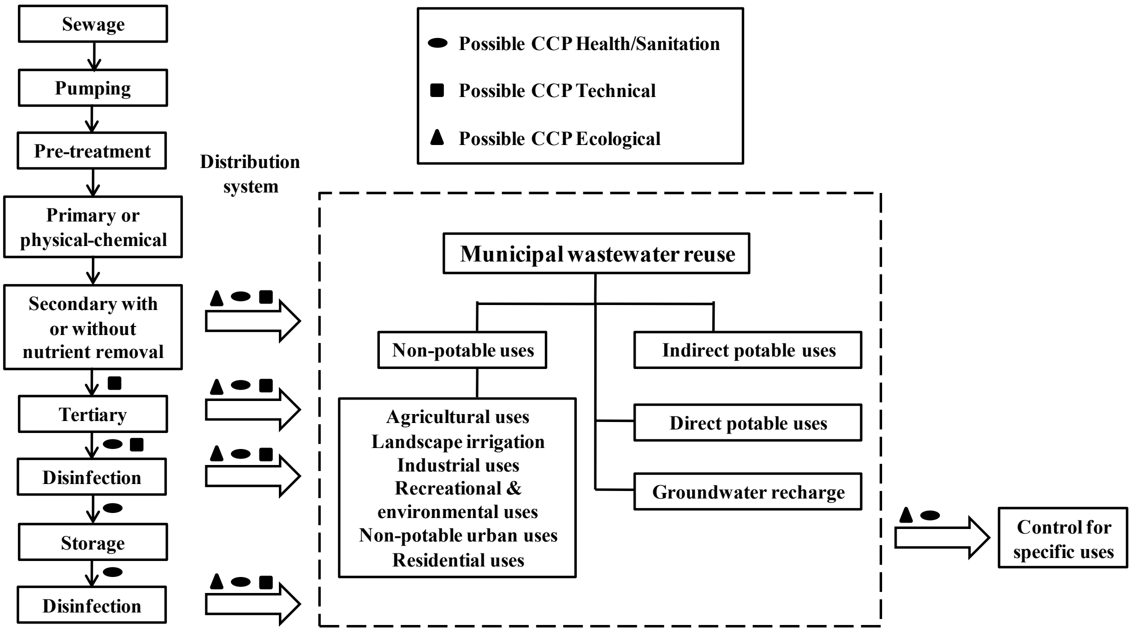


Figure 2.11 Possible critical control points of a water recycling system (modified from Huertas et al., 2008).

Since financial insufficiency hinders the implementation of advanced wastewater treatment and monitoring technologies in many developing countries, exposure control is regarded as a more cost-effective way in risk minimization. Exposure control approaches include public access control, recycled water use restriction, staff access protection, etc. Table 2.18 lists possible pathogen reduction ways in agricultural and urban irrigation schemes (Kamizoulis, 2008; Qadir et al., 2010). Additionally, exposure minimization on potential risk groups during recycled water irrigation can also significantly reduce the risk level (Stevens et al., 2008).

- Workers and crop handlers should wear waterproof and protective coats, boots, gloves and facial masks, cover all wounds during working time, be immunized against Hepatitis A and other diseases that can be transmitted through wastewater use, and wash their hands, arms and legs at the end of each working day.

- Consumers should wash and cook agricultural products before consumption as well as maintain high standards of hygiene (e.g., wash hands with soap and clean water before eating and/or drinking).
- Local residents, golfers and other athletes should be kept fully informed on the use of recycled water by signage and pipe labelling.

Table 2.18 Restrictions and effects on crops and public access

Exposure control methods	Restrictions	Log reduction of pathogens
Crop restriction	Cooking	5-6 logs
	Washing vegetables	2-3 logs
	Peeling	2 log
Crop irrigation management	Drip irrigation of crops	2 log
	Drip irrigation of crops with limited ground contact (e.g., tomatoes, capsicums)	3 log
	Drip irrigation of raised crops with no ground contact (e.g., apples, apricots, grapes)	5 log
	Drip irrigation of plants/shrubs	4 log
	Sub-surface irrigation of plants/shrubs or grass	5-6 logs
	Sub-surface irrigation of above ground crops	4 log
	Spray drift control	1 log
Public access restriction	Withholding periods (1-4 h)	1 log
	No public access during irrigation	2 log
	No public access during irrigation and limited contact following irrigation	3 log
	Buffer zone (50-100 m)	1 log

Note: Modified from Kamizoulis, (2008).

When it comes to residential projects, spray controllers on toilet bowls and washing machines can provide more gentle flows and fewer aerosols. It is also encouraged to apply potable water and soap to wash hands at the end of each reuse activity. Remarkably, other sound solutions should be conducted when targeting recycled waters with high exposure to workers (e.g., industrial uses, road cleaning, fire fighting and car washing). The strategies may include: increasing droplet size if spraying water, notifying and relocating workers when recycled water is in use, training and educating workers regarding hygiene practices, protecting against direct contact with waterproof dressings and gloves and/or providing ready access to adequate hand washing amenities. Since most IPR and DPR projects are successfully operated without detecting any environmental or public health problems, exposure controls on these schemes might not be required.

2.5.4 Integrated assessment models

In most cases, it is difficult to model or analyse all accumulative and interactive health and environmental effects of different activities from a single assessment tool as each approach evaluates recycling projects in different ways. For example, MFA can be seen as an effective tool for the early recognition of environmental sanitation problems as well as the assessment of control measures. It allows decision makers to obtain a first efficient screening of potential environmental effects. This could be vital important in the holistic recycled water sustainability analysis. When the current local environmental sanitation condition is predicted to be sustainable with no changes required, there would be no need to conduct further assessments or actions. This can save lots of time and energy. On the other hand, if the environment situation is shown to be unsustainable, the MFA model can be executed again under different environmental management scenarios and the effectiveness of water recycling and reuse in sanitation improvement could be verified eventually. Once the importance of recycled water has been increasingly noticed by decision makers, more and more recycling schemes together with detailed sustainability assessments are likely to be developed and implemented. In some developed countries with a number of recycling schemes being successfully conducted, the significance of water reuse has already been widely recognised. MFA, in this case, could be regarded as an optional tool in the holistic decision making of existing or future recycled water projects.

Comparatively, LCA is a non-site-specific analysis. Although it is conceived to carry out detailed and complex analyses, it has broader ambitions and gives overall environmental results as both the input and output related interventions are considered, and all releases of different substances (e.g., CO_2 for the whole life cycle) to the environmental media (e.g., air, soil or water bodies) could be summarised. It can be regarded as an important tool in the context of holistic recycled water sustainability analysis since the selection of an appropriate treatment technology or technology combination could not merely benefit the local environment directly but also guide the downstream water quality assessment as well as end use consideration. However, it would be unnecessary to carry out comprehensive LCA assessment in most recycling schemes which normally involves a number of different types of environmental impact categories.

To save time and cost for the whole sustainability decision making process, the assessment categories in LCA should be narrowed down after considering the potential downstream end uses in initial recycled water project planning. For example, when the recycled water is planned for irrigation uses, more concerns should be given to the potential environmental impacts on soil and groundwater (e.g., EP, SP and ETP). Other impact categories related to air quality (e.g., GWP, AP, PHO and ODP) might be more important for industrial applications of recycled water. Even though some insignificant categories have been excluded, difficulties such as the subjectivity in calculation, lack of consideration on alternatives, politicisation of assessment processes and competence of involved authorities, might hamper the quantification of environmental effects in LCA (Asano et al., 2007; Zhang et al., 2010). Besides, other relevant benefits (e.g., water quality improvement, fertiliser consumption reduction and saltwater intrusion prevention) should be further clarified in downstream studies.

Moreover, RA mainly evaluates the health and/or environmental impacts in a site-specific way, which considers the possible releases of a single substance from the different sources and tries to predict the risks of adverse impacts from that particular substance. Notably, HRA models mainly focus on the acute (and possible chronic) chemical and pathogenic risks to human health. ERA models could be adopted to investigate the particular inorganic and trace organic compounds of concern so as to address the output-related environmental interventions caused by different recycled water end uses. This could provide useful information for decision makers to adopt further management strategies in recycling schemes. When these different types of tools are linked together, a broader set of critical issues could be encompassed in study. Thus, the weaknesses of individual tools might be compensated in analysis progress (Wrisberg and Udo de Haes, 2002; Udo de Haes et al., 2006). The integrated approaches have already been reported in several studies.

2.5.4.1 MFA coupled with LCA models

As MFA has been successfully applied in the early recognition of environmental problems, it could be regarded as a prerequisite for the implementation of LCA. Instead of evaluating a wide range of environmental impact categories, LCA, in this circumstance, can focus on the impacts associated with one or several particular elements which are tracked by MFA, saving lots of evaluation time and energy. Thus, the Organic Waste Research model (ORWARE) is developed which is to calculate material flows and energy turnover for various treatment alternatives and transfer the results into environmental effects using LCA methodology. A total of 43 different substances are considered in the model, where the related transformation, transportation, energy and other external resource consumption are able to be simulated. As can be seen in Figure 2.12, the flow data generated by ORWARE is aggregated in a form of effect categories, which indicate the level of environmental damage and resource consumption in terms of emissions to air and water, accumulation in landfill, flow of recycled products, etc. This static model can be further divided into several sub-models (e.g., WWTP, incineration, landfill, compost, anaerobic digestion, truck transports), where WWTP model is one of the most important ones, which is to calculate the emissions and energy from wastewater treatment and reuse. Besides, as ORWARE itself is rather general, other new sub-models can be easily incorporated into the system. Nevertheless, as most sub-models are empirical and do not consider the site-specific conditions (e.g., geographical, industrial and social factors, advanced recalibration is required whenever necessary (Dalemo et al., 1997; Sonesson et al., 1997).

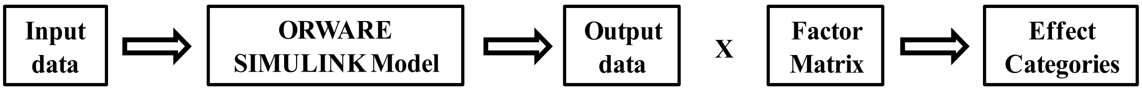


Figure 2.12 Simplified diagram of the ORWARE methodology (adapted from Ramírez et al., 2002).

More specifically, Jeppsson and Hellstrom (2002) used ORWARE model to evaluate two fundamentally different urban water systems in Sweden. One is a centralized system while the other one is the onsite treatment system equipped with source separation of stormwater, greywater, blackwater and urine. To analyse these two systems effectively, several parameters (e.g., COD fractions, exergy, PO_4, particulate P and N, potassium, cadmium, etc.) that are closely related to environmental performance were modelled. Overall, the centralized system was more environmentally friendly in terms of total P and copper to water and cadmium to arable soil whereas the source-separated system was better in regard to total N to water, fresh water consumption and net energy consumption. Similarly, ORWARE has also been applied to urban water systems in Chile by Ramírez et al. (2002). To compare 8 different management scenarios which are different combinations of wastewater treatment (biological vs. chemical), sludge treatment (digestion vs. composting) and disposal (landfill vs. agriculture reuse), five environmental impact categories were considered, including GWP, EP, pathogenic organisms, emissions of heavy metals and toxic organic substances and energy use or production. The results indicated that the biological/composting/agriculture reuse scenario achieved an overall lowest environmental impact except for EP. Regarding pathogen reduction in recycled water, both chemical precipitation and biological treatment were good enough for treated effluent discharging into the sea while additional disinfection was needed in the case of effluent to rivers. The effluent could not be reused in agriculture as concentrations of toxic substances were too high.

2.5.4.2 LCA coupled with ERA models

There has been an effort to incorporate ERA partially or fully within a LCA in some of research areas, such as tin-lead solder in electronics, nanomaterials and mineral waste reuse scenarios (Montangero et al., 2006a; Socolof and Geibig, 2006; Sweet and Strohm, 2006). Overall, by linking LCA and ERA within the same toolbox, the whole of a material's life cycle risk can be considered in an integrated manner, thereby promoting continuous improvement as well as proactive risk reduction and adaptive approaches under current situations. However, Udo de Haes et al. (2006) pointed out that the implementation of a combined approach required a careful study on similarities, differences and synergism between LCA and ERA. Although the full integration was recommended, it was not achieved in practice due to the fundamentally different model structure (i.e., the use of the functional unit concept in LCA versus the use of flows of actual size in ERA). The use of the two tools in a combined form has not been reported in water recycling field. Yet it is possible to use ERA as a more detailed and site-specific analysis after an LCA has been carried out.

2.5.4.3 Comprehensive decision making

Furthermore, to achieve a more comprehensive and holistic decision making for recycled water schemes, the outputs from the MFA, LCA and RA models can be combined with additional economic and/or social perception assessments (Figure 2.13).

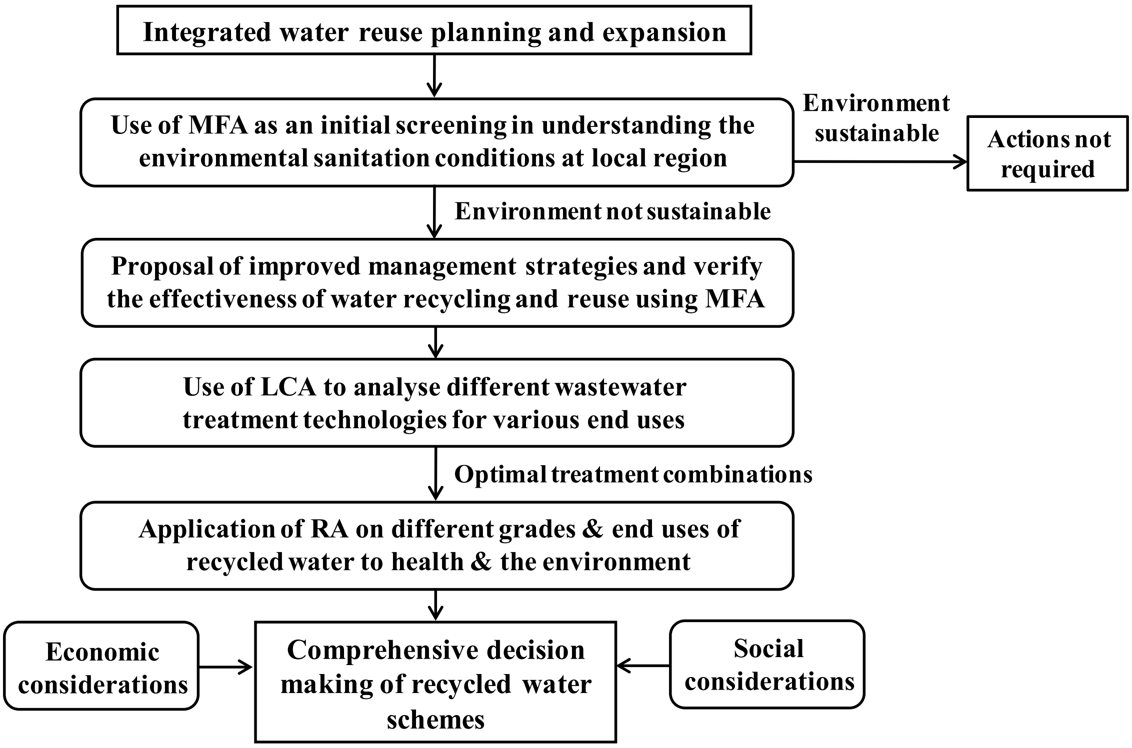


Figure 2.13 Outline of the comprehensive decision making for recycled water schemes

Notably, the economic assessment of recycling schemes is often carried out by a cost-benefit analysis, which should not only include the internal impacts (capital, operating and maintenance costs and the price of recycled water) but also the external impacts of environmental or social nature. The prime objective is to maximise the total benefits, which is the difference between income and costs (Equation 2.19).

$$\text{Max } B_T = B_I + B_E - OC \qquad \text{(Eq. 2.19)}$$

where B_T = total benefit (total income-total costs), B_I = internal benefit (internal income-internal costs), B_E = external benefit (positive externalities- negative externalities), and

OC = opportunity cost. Internal income can be earned by multiplying the selling price of recycled water and the volume obtained. In Australia, to encourage the usage of recycled water, its price is currently set at approximately 75% of the price of drinking water, which is much lower than the real cost of providing recycled water (MacDonald and Dyack, 2004; Hurlimann and McKay, 2007). However, the interaction of willingness to adopt recycled water and pricing strategies has not reached a conclusive result so far (Dolničar and Saunders, 2006). Future pricing strategies should be based on costs as well as include the value of the water itself, its environmental effects and its own opportunity cost. The internal costs consist of:

- *Investment costs*. Investment costs account for 45-75% of the total cost of a recycling project, which include land, civil works, machinery and equipment, distribution facilities and connection works (Hernández et al., 2006).
- *Financial costs*. Financial costs result from financing the investment. Some projects have been state-financed or funded by private initiatives while others have received public participation in the form of investment subsidies, long-term loans or interest rebates.
- *Operating and maintenance costs*. The costs include water treatment, storage systems and pressure maintenance, water quality monitoring and life cycle costs (Urkiaga et al., 2008).
- *Taxes*. Taxes should also be considered if the scheme attracts tax (Godfrey et al., 2009).

Table 2.19 presents the internal costs of several recycled water schemes in Australia. To ensure the internal benefit, Urkiaga et al. (2006) have specified a minimum capacity of the agricultural scheme, which is to serve 10,000-20,000 inhabitant equivalents, or to irrigate a golf course and/or a crop extension of 3,500,000 m^2. They also indicated that two or more different types of treatments and end uses are more economically suitable than a single option. While internal impacts can be calculated directly in terms of monetary units, there are a series of external influences where no explicit market exists. Hence, external benefit is to capture the most tangible and measurable ones and quantify the available aspects based on hypothetical scenarios or patterns observed in related markets. For instance, Godfrey et al. (2009) identified the health benefits of greywater reuse from the reduced number of diarrhoeal cases, work or school absenteeism avoided, etc. Besides, the opportunity cost is normally estimated by an alternative use of the land with certain profitability (Hernández et al., 2006).

Table 2.19 Internal costs of several recycled water schemes in Australia

State	Scheme	Capacity (ML/yr)	Capital Cost ($[4])	Government Fund ($[4])	Unit cost ($[4]/ML)	Completed
VIC[1]	Virginia Irrigation	18,000	36 M[5]	2.03 M[5]	2,000	1999
VIC[1]	Mawson Lake	800	8 M[5]	–	10,000	2005
QLD[2]	Western Corridor Recycling	115	1.7 B[6]	408 M[5]	–	2008
SA[3]	Glenelg Adelaide Parklands	3,800	76 M[5]	–	20,000	2009
SA[3]	Southern Urban Reuse	1,600	63 M[5]	–	39,000	2010

Note: Modified from Radcliffe, (2008); Wang, (2011).
[1]Victoria; [2]Queensland; [3]South Australia; [4]Australian dollar; [5]Million; [6]Billion.

With respect to social aspects, the considerations include aboriginal and heritage, aesthetics, traffic disruption, community recognition, public education opportunities and political impacts (Muthukaruppan et al., 2011). Community perception and acceptance should be highly addressed, which are now recognized as the key elements of success for any recycling project. Dolničar and Saunders (2005) concluded that the acceptance of recycled water was correlated with a high level of education, followed by being in the younger age category, while income and gender appeared significant in only one third of the studies. Hurlimann and McKay (2007) indicated that males had more knowledge about recycled water and were more supportive than females. McKay and Hurlimann (2003) predicted that people aged 50 years and over raised the greatest opposition to water recycling schemes. Po et al. (2003) suggested that there are many other factors influencing the acceptance, including the disgust emotion, perceptions of risk from recycled water, sources of recycled water, specific uses of recycled water, trust of authorities and knowledge, attitudes towards the environment, environmental justice issues and the cost of recycled water.

Hurlimann and McKay (2007) found that higher water quality such as low salt, colourless and odourless contributed to the increased acceptability. Non-potable reuse carries the least public health risk and the public supports for agricultural, golf courses, parks and industries are generally high. Recycled water use inside the home was less preferred, where more than 70% of the respondents agreed to uses it for toilet flushing, gardens and car washing, but only 60% and 13% supported for washing clothes and filling swimming pools respectively (Pham et al., 2011). There are greater concerns on IPR and DPR projects due to health issues. To improve the social acceptance, Pham et al. (2011) suggested informing people about the different benefits and terms of recycled water as well as continuously seeking feedback from the community whereas Hurlimann et al. (2007) believed that authorities should focus on gaining the community's trust. A willingness to use model constructed by Menegaki et al. (2007) indicated that information and education might be useful tools in making people realize the benefits of recycled water, allowing them to pass from negative to positive attitudes.

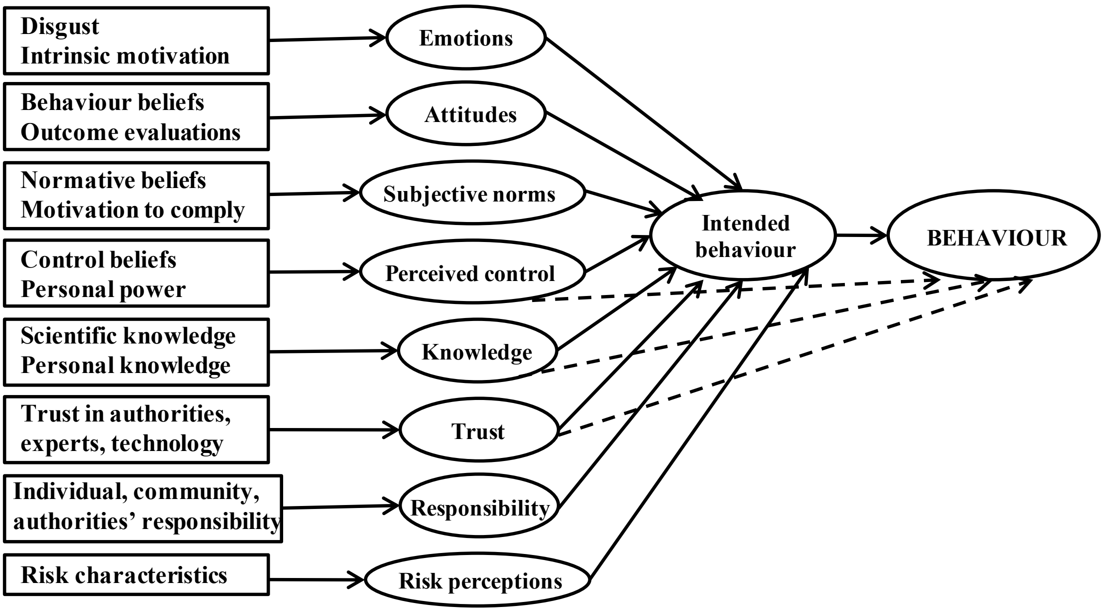


Figure 2.14 The hypothesised model in predicting community behaviour (adapted from Po et al., 2005).

On the contrary, based on Ajzen's theory of planned behaviour, Po et al. (2005) developed a model (Figure 2.14) to examine participants' behavioural responses to recycled water. The model on an IPR case study at the Managed Aquifer Recharge Scheme in Perth, Western Australia, manifested that knowledge and risk perceptions were not dominant in influencing behavioural intentions to drink the recycled water. Overall, the relationships between individual's level of knowledge and perception versus acceptance of recycled water are fairly complicated, broad conclusions on theses factors from a small amount of research are still not enough (Marks, 2003).

When the assessment results from a comprehensive decision making are shown to be unsustainable, several management or control approaches should be established, and the recycling scheme can be re-evaluated. Nevertheless, the model structure and boundary differences of different tools need to be carefully considered, which are likely to introduce misinterpreted conclusions in the end. For instance, when applying LCA to previous MCA results, the extended system boundaries are usually hard to define, which include other parts of the urban infrastructure (e.g., incineration plant, landfill, arable and receiving waters). Besides, the results from the LCA and RA may suggest opposite solutions for the environmental preferred choice as sometimes the wastewater treatment technology with lowest environmental impacts could not arrive at the highest recycled water quality for health and environmental protection. These outcomes, together with results from economic and social assessment should be weighted to ensure the final evaluation results are consistent with the preferences of the decision makers.

2.6 CONCLUSIONS

Recycled water has received great attention over the recent decades in many countries, which can provide an opportunity to supplement fresh water supply partially and alleviate environmental loads and energy consumption substantially. To ensure safe and reliable use of recycled water, membrane technologies such as MBR, MF, UF, NF and RO coupled with real-time monitoring programs are demonstrated to be highly efficient in wastewater hazardous compound removal. It was shown that when low strength greywater is through physical and chemical treatments whereas medium and high strength greywater are treated by additional biological processes, the concentrations of chemicals- and pathogens-of-concern in the effluents can be very low. Similar conclusions are achieved from municipal wastewaters under UF/RO or MBR treatments and industrial wastewaters through MBR or CWs processes. Nevertheless, because of different natural, social and economic conditions, the end uses vary markedly around the world. While agricultural irrigation still represents the largest current use of recycled water on a global scale, other end uses such as industrial and non-potable urban uses and groundwater recharge have made considerable progress in recent years, especially in Australia, Asia, southern and western America, Europe, and the Mediterranean countries. The potential for implementing long term IPR or DPR exists in arid and semi-arid countries and regions, such as in the Middle East and African regions. Along with the historical development, water quality criteria become more stringent, increasingly addressing public health and acceptance issues.

As the sustainability of the recycled water scheme directly influences the introduction of new end uses and the expansion of the current scheme, this chapter has reviewed several assessment models. Each model was shown to link particular technologies and measures to differing conceptions of sustainability and recycled water. MFA was found to be an effective initial screening in understanding the environmental sanitation conditions at local region. LCA has been widely used in selecting the optimal wastewater treatment technologies while RA is mainly used in evaluating the potential health and/or environmental effects of chemical and microbial hazards in recycled water. A key conclusion is that the integrated models are able to address the weaknesses of single approaches and provide more systematic and viable options in decision making. However, the complexities and conflicts in model integration always become troubles, which are waiting to be solved by future simple, sound and manageable techniques.

This chapter also highlights the importance of adopting risk control and management strategies such as source control, wastewater quality improvement, HACCP control and exposure control in recycling schemes. Besides, an integrated approach to plan and manage all available water resources coherently and comprehensively on the local scale is being increasingly emphasized and will be a future tendency in the following years. Establishing uniform water reuse guidelines, building public confidence and getting financial and political support from the government and organizations will contribute to integrated water resource management and sustainable development. From an optimistic view, with focused effort, recycled water can be well managed and be reused in a sustainable way for more end uses, which will benefit both the environment and mankind in the long term.

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CHAPTER 3

Research Methodology

3.1 INTRODUCTION

This chapter describes the adaptive research methodology of a novel and comprehensive assessment for the new end uses of recycled water schemes. It involves a rigorous and consistent evaluation process, including the framework, principles, evaluation criteria and mathematic algorithms, to the analysis of complex information and issues related to the development of new recycled water applications. In addition to a detailed description of the feasibility assessment approach, the chapter also introduces the analytical techniques of social surveys for the identification of key factors on improving public acceptability towards the recycled water application in residential areas of Australia. Furthermore, the experimental materials, treatment unit set-up and analytical procedures associated with a new optional recycled water purification system prior to use in household washing machines are presented as well. The methodology could offer important information regarding how to utilize previous environmental-related knowledge and findings for tracking improvements and for maximizing the potential applications so as to enhance the recycled water sustainability.

3.2 COMPREHENSIVE ASSESSMENT ANALYSIS ON NEW END USES OF RECYCLED WATER

3.2.1 Significance of the research

Although many water recycling schemes and reuse activities have been reported widely in water deficient areas, the inappropriate management of these water systems has pushed the local areas beyond their sustainable limits. For example, some of the schemes sought to achieve great environmental savings through maximized water recycling targets regardless of utility, economic feasibility and geographical conditions whereas other water reuse activities that have political or financial underpinnings might trigger the degradation of surrounding ecological habitats in the long run (Chapagain and Orr, 2009). To ensure the long term regional development, a comprehensive and systematic assessment in water reuse planning is essential which is to investigate the trade-offs among a variety of competing issues (e.g., engineering technical feasibility, energy consumption, ambient ecosystem, risk condition, cost, water pricing and social attitude) under certain analytical techniques and algorithms. Figure 3.1 outlines how to collect the water-related data from different aspects systematically and transform these conceptual principles into a performable and practical regulation system, which can effectively contribute to a quick measurement of the recycled water sustainability (Agnes et al., 2007).

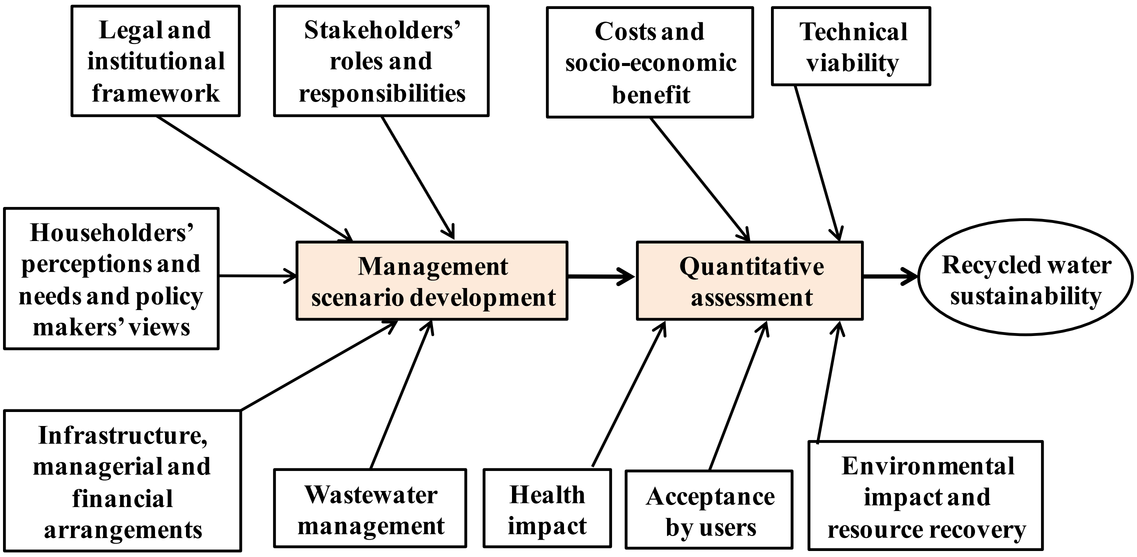


Figure 3.1 Systematic data collection for recycled water planning and sustainability management (modified from Agnes et al., 2007).

3.2.2 Evaluation framework and execution procedure

By reviewing previous literature and information regarding the existing recycled water end uses in current recycling schemes and different environmental evaluation models (e.g., life cycle assessment, material flow analysis and environmental/health risk assessment) for the evaluation of water and environmental sustainability, the relevant research gaps and opportunities have been identified. Accordingly, a systematic framework for the evaluation of new recycled water end uses was established with a full assessment procedure that follows the characteristic Multi-criteria Analysis (MCA) decision-making process in water resource management (Alvarez-Guerra et al., 2010). This assessment framework can provide an opportunity to interpret the complex technical and scientific norms into a more widely understandable format for decision makers and the community.

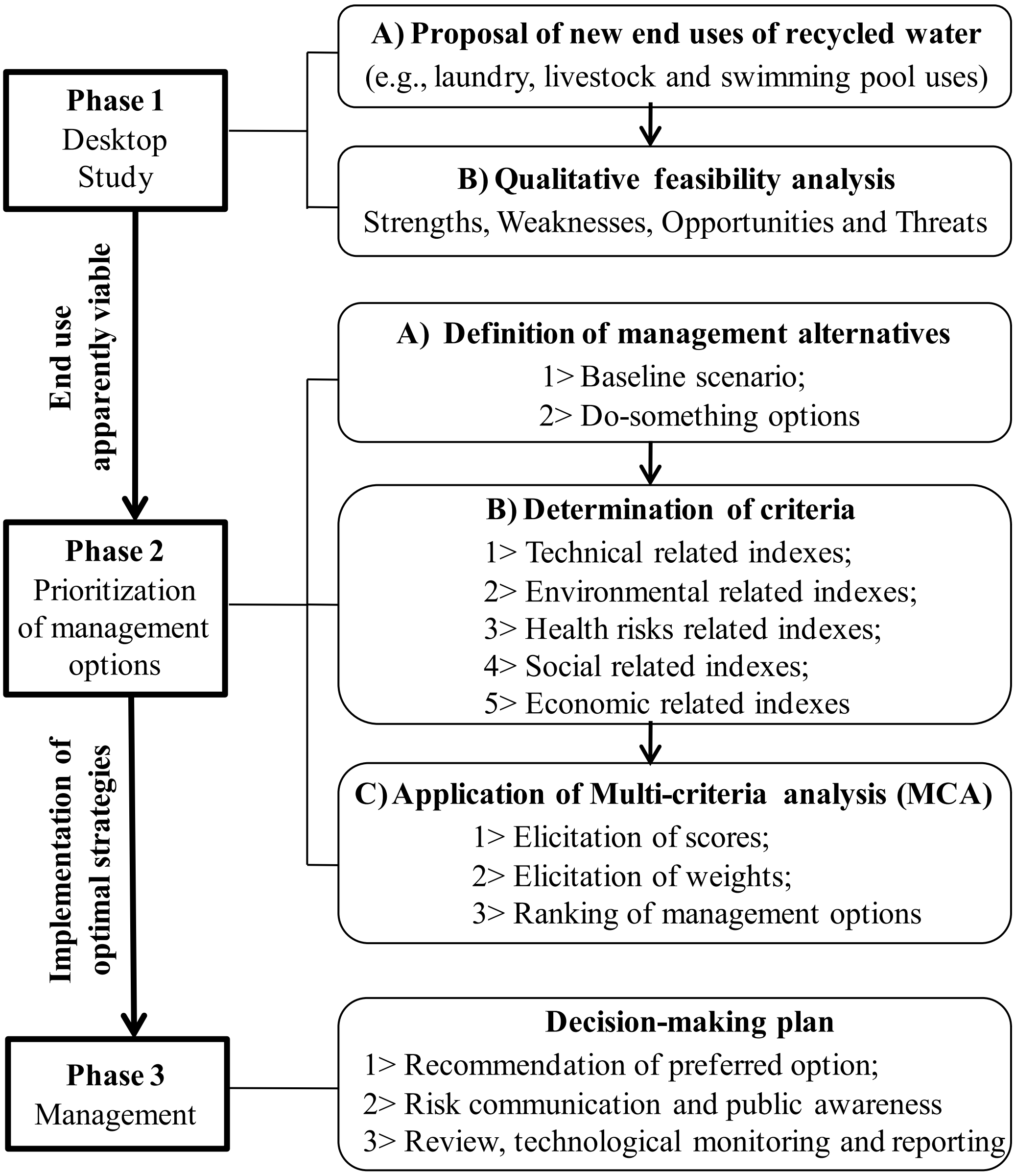


Figure 3.2 Proposed framework for decision making in new end use management

As demonstrated in Figure 3.2, the framework consists of three phases, where phase 1 is the primary screening step to identify the prospects of specific new end uses and verify the viability through initial qualitative approach. Phase 2 starts with the development of several management alternatives which employ different facilities, equipment and/or varied treatment technologies towards one/several end use(s). After selection of particular evaluation criteria from five identified categories, phase 2 involves the application of MCA to further prioritize the management options. Finally, phase 3 is the management step that includes the result discussion, communication, review and reporting.

Based on the framework, a full assessment procedure was developed and it consists of the following main steps (Coutts, 2006; Muthukaruppan et al., 2011):

- Determination of intentions. The first objective is to determine the major purpose of the scheme, whether to provide recycled water for agricultural irrigation, residential use, industrial consumption or Indirect Potable Reuses (IPR). The second is to recognize whether to assess existing schemes, evaluate the viability to upgrade the existing schemes or implement new end uses.

- Prioritization of management options. This step mainly aims to consider different end uses of recycled water, site specific conditions, sustainable water management and regional water cycle planning strategies so as to identify the initial management alternatives. It also requires consultation and communication with water authorities, stakeholders, technical experts and consumers who are potentially involved in water reuse activities to determine the fit-for-purpose evaluation criteria and principles (i.e., technical, environmental, risk, social, and/or economic indexes) and exclude unsuitable management options. The assessment criteria can be selected by referring to similar case studies conducted nearby, to site assessment and to local planning reports as well.

- Evaluation of management alternatives. Once several options are determined, they should be investigated in detail. In this case, MCA can be implemented with a series of assessment criteria, and then the scoring, normalization and weighting techniques can be employed to achieve an overall value for each option.

- Implementation of the preferred option(s). Before implementing the preferred option, viability assessment should be conducted again, especially for newly developed schemes. When undertaking the detailed design, stakeholders should consult with the government, water authorities and local councils, establish the water reuse supply and consumption agreement and prepare an environment improvement plan. If necessary, they should also acquire the approvals from the environmental protection agency or other relevant departments.

- Monitoring and review. After implementing the preferred option, risk communication, monitoring and review are required. The system should also enable comparative analysis against other existing or potential projects.

3.2.3 Multi-criteria analysis (MCA) in decision making

As can be seen in Figure 3.2, the last procedure in Phase 2 involves the application of MCA, which is a decision making tool developed for complex problems. The prime objective of the MCA is to investigate the tradeoffs among the selected evaluation criteria and then obtain rankings of the different management alternatives under certain mathematical algorithms, such as the Multi-attribute Utility Theory (MAUT), compromise programming, analytical hierarchy process, cooperative game theory, Elimination and Choice Expressing Reality (ELECTRE) and the Preference Ranking Organization Method for Enrichment Evaluation (PROMETHEE).

Hajkowicz and Higgins (2008) found that there was a high level of agreement between these different MCA techniques. Consequently, in this research, MAUT has been firstly conducted for the simplified quantitative MCA to identify the potential of implementing three possible recycled water end use management strategies in an existing project. Secondly, the PROMETHEE algorithm along with the Rank Order Weight Generation (ROWG) model, were utilized to further analyse the specific new recycled water end use for a household laundry. The corresponding calculation processes were all achieved by a computerized MATLAB programming technique. From the calculation and simulation results, the least preferred end use(s) or management alternative(s) could be quickly eliminated whereas the superior options can be further investigated.

3.2.3.1 Multi-attribute Utility Theory (MAUT)

The MAUT approach that is comprised of additive utility functions and preference modelling, has been successfully applied to compare different water control plans in the

Missouri River, the U.S. (Prato, 2003), to consider the expansion of water supply to Cape Town, South Africa (Joubert et al., 1997) and to manage the water resource in Oregon, the U.S. (Gregory and Wellman, 2001). MAUT has strengths such as it is easy to compare alternatives; it is transparent and has high public preference. The procedures in the application of this method include: (1) Define a utility function and score each attribute on the scale from best to worst that reflects the range of conditions that might occur; (2) Normalize the attribute scores for a given scenario; (3) Weight the attributes to reflect user preferences about the relative importance of each attribute; (4) Calculate the overall multi-attribute score (OST, 2013).

More specifically, as the evaluation criteria are fundamentally different by their nature, the scoring process is initially to generate a matrix where the elements represent scores of each option against each criterion. While the modelling and monitoring results regarding water quantity, water quality, greenhouse gas emission and energy consumption are usually presented as quantitative estimates, the environmental, risk and social considerations incorporate a higher degree of qualitative judgment by the decision maker or the project team (Kiker et al., 2005). For qualitative data, the state of Victoria has developed a scoring system on a 9-point scale (Table 3.1) whereas the 11-point scale, 5-point scale and 7-point scale have been reportedly used in cases studies in the Australian Capital Territory, and the states of Queensland and South Australia respectively (Coutts, 2006). Since the qualitative information is likely to introduce bias towards or against certain technologies due to personal judgment, it is advisable to use quantitative data wherever possible.

Table 3.1 The qualitative scoring system

Impact	Score	Impact	Score	Impact	Score
Very much better	+4	Little better	+1	Much worse	–3
Much better	+3	No change	0	Moderately worse	–2
Moderately better	+2	Very much worse	–4	Little worse	–1

Note: Adapted from Coutts, (2006).

The quantitative data normally require a normalization or data scaling process to make the final score dimensionless, thereby enabling comparison. There are generally four data scaling methods, including the min-max approach, zero-max approach, range approach and distance-to-target approach. With regard to the min-max and zero-max approach, the normalized score is expressed in Equations 3.1 and 3.2 respectively.

$$\mathrm{CA}_{\mathrm{normalized}} = \frac{CA_i - \min(CA_1, CA_2 ... CA_i)}{\max(CA_1, CA_2 ... CA_i) - \min(CA_1, CA_2 ... CA_i)} \qquad \text{(Eq. 3.1)}$$

$$\mathrm{CA}_{\mathrm{normalized}} = \frac{CA_i}{\max(CA_1, CA_2 ... CA_i)} \qquad \text{(Eq. 3.2)}$$

where $\mathrm{CA}_{i,j\ normalized}$ is the normalized score of option *i* on criterion *j*. min(CA_1, CA_2…CA_i) is the minimum score and max(CA_1, CA_2…CA_i) is the maximum score in regard to one criterion among all selected options. The range approach is similar to min-max approach except in the denominator where the boundary conditions are set on the basis of other information such as the best available technology. Comparatively, as for the distance-to-target approach, it is necessary to define a target and express the normalized score as a ratio of the distance (CA_i) to that target (Hajkowicz and Higgins, 2008; Benedetto and Klemes, 2009).

At the heart of MCA is the aggregation process which must be carefully assessed to ensure the results of the evaluation are consistent with the preferences of decision makers. The MAUT has been regarded as an effective aggregation tool (Equation 3.3).

$$\mathrm{U}(x_1, x_2, ..., x_n) = \sum_{i=1}^{n} k_i u_i(x_i), \quad \text{if} \quad \sum_{i=1}^{n} k_i = 1 \qquad \text{(Eq. 3.3)}$$

where, U and u_i are utility functions normalized from 0 to 1. u_i equals to $\mathrm{CA}_{i,j\ normalized}$ and k_i is the corresponding weighting factor. $0 < k_i < 1$, $i=1,2,…,n$. U_i is the final score of option *i* (Kainuma and Tawara, 2006).

As a result, options can be ranked according to their overall scores. Remarkably, the rank of the options can be sensitive to the set of weights (Linkov et al., 2006). Generally, decision makers assign higher weights to more important criteria and smaller weights to less important criteria. Due to different personal perceptions, the weighting process may be highly variable. To increase the confidence in decision making, the final outcomes can be subject to a sensitivity analysis which involves examining how the ranks might be changed under different scoring or weighting systems so as to refine the validity of the preferred option within specified bounds. Alternatively, to minimize the man-made errors, some computerized weighting models can be employed to narrow down the competing options through statistical weight results (Chen et al., 2013c; 2014).

3.2.3.2 Preference Ranking Organization Method for Enrichment Evaluation (PROMETHEE)

The PROMETHEE approach is an outranking method that outputs the ranking of different alternatives by comparing their criteria values. More specifically, a preference function $P_j(i, i')$ can be defined which measures the decision-maker's preference intensity for alternative i over alternative i' for each criterion j (Equation 3.4). However, for criteria to be minimised, the preference function should be reversed (Equation 3.5). There are several different models on describing the preference intensity function, from fairly simple methods (e.g., usual and level shapes) to more complicated forms (e.g., Gaussian and concave shapes). Four types of the preference function have been introduced in this study (Table 3.2).

$$P_j(d) = f_j[d_j(i,i')] \qquad \text{(Eq. 3.4)}$$

$$P_j(d) = f_j[-d_j(i,i')] \qquad \text{(Eq. 3.5)}$$

where $d_j(i, i')$ is the deviation between values of pairwise alternatives. An aggregated preference function index incorporating the weights is defined as:

$$\pi(i, i') = \sum_{j=1}^{k} w_j P_j(i, i') \qquad \text{(Eq. 3.6)}$$

where w_j (j=1,…,k) are assigned weights associated with the criteria. As each alternative faces (n-1) other alternatives, a positive and negative outranking flow is determined by:

$$\varphi^{+}(i) = \frac{1}{n-1}\sum_{i'=1}^{n}\pi(i, i') \qquad \text{(Eq. 3.7)}$$

$$\varphi^{-}(i) = \frac{1}{n-1}\sum_{i'=1}^{n}\pi(i', i) \qquad \text{(Eq. 3.8)}$$

It is then possible to determine an overall score for each alternative by calculating the net outranking flow:

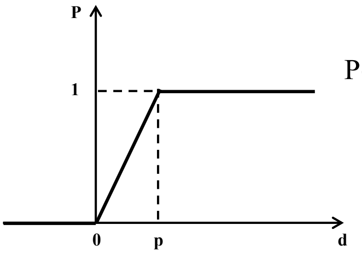
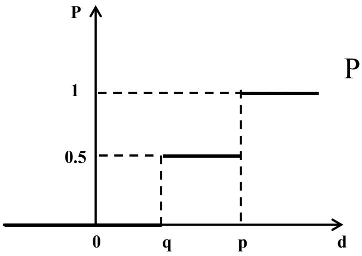
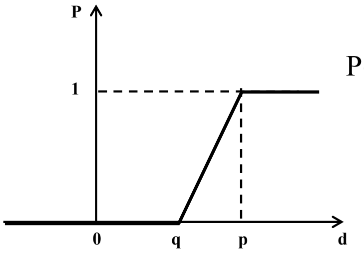
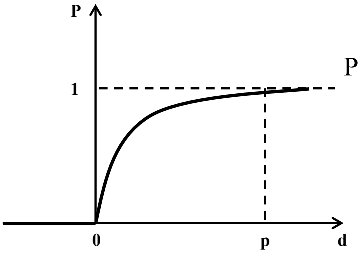
$$u_i = \varphi(i) = \varphi^{+}(i) - \varphi^{-}(i) \qquad \text{(Eq. 3.9)}$$

The alternatives are finally ranked according to their net flows, from highest to lowest (Hajkowicz and Higgins, 2008; Alvarez-Guerra et al., 2010).

When a fixed set of criteria values and a fixed set of weights are available, the whole PROMETHEE MCA processes can be achieved by a computerized MATLAB programming technique. Yet, in some cases, due to a lack of sufficient expertise, decision makers are more confident of criteria importance order than assigning specific weight to each criterion. Thus, to reduce the man-made errors, a Monte Carlo ROWG model can be used which is based on a given precedence order of the selected criteria.

Particularly, in ROWG, the model will firstly assign random numbers (e.g., 10,000 iterations) to each criterion weight from a uniform distribution [0, 1]. The random numbers, together with the number of criteria (n) will be then put into a matrix with n×10,000 values. The values in each column will be sorted in a descending order. Supposing the ranked numbers are: $1 > r_{(n-1)} > \ldots > r_{(2)} > r_{(1)} > 0$, the first differences of these ranked numbers can be obtained as: $k_n = 1 - r_{(n-1)}$, $k_{n-1} = r_{(n-1)} - r_{(n-2)}, \ldots, k_1 = r_{(1)} - 0$. This is to ensure that the set of values $(k_1, k_2, \ldots, k_n)$ in each column will have a sum of 1 and be uniformly distributed (Butler et al., 1997). After the transformation process, the numbers in each column will be sorted again to match with the given precedence order.

Table 3.2 Preference functions used in PROMETHEE

P(x, y)	Graph	Definition	Applicable objects
V-shape		$P(d)=\begin{cases}0 & d\leq 0\\ \frac{d}{p} & 0<d\leq p\\ 1 & d>p\end{cases}$	Quantitative criteria (e.g., RW supply, energy consumption, GHG emission). The indifference threshold (q) is 0; p is a threshold of strict preference.
Level shape		$P(d)=\begin{cases}0 & d\leq q\\ \frac{1}{2} & q<d\leq p\\ 1 & d>p\end{cases}$	Qualitative criteria (e.g., RW operability, ecology, political support, education opportunities).
Linear shape		$P(d)=\begin{cases}0 & d\leq q\\ \frac{d-q}{p-q} & q<d\leq p\\ 1 & d>p\end{cases}$	Quantitative criteria (e.g., capital and operational cost). The indifference threshold (q) is non-zero.
Concave shape		$P(d)=\begin{cases}0 & d\leq 0\\ \sqrt{1-(1-\frac{d}{p})^2} & 0<d\leq p\\ 1 & d>p\end{cases}$	Criteria representing contamination values (e.g., parameters in RW quality). An increase in contamination values leads to a rapid decline in utility scores.

Note: Modified from Brans and Mareschal, (2005).
Abbreviations: RW= Recycled water; GHG= Greenhouse gas.

3.3 SOCIAL SURVEY ANALYSIS ON THE NEW END USES OF RECYCLED WATER

3.3.1 Chi-square test

The geographical differences on household laundry behaviour and community attitude on receiving recycled water were measured by the chi-square test, using software MATLAB R2012b as the analysis tool. The Pearson chi-square test is a statistic analysis that has been commonly used to compare two different data (or frequency distributions), namely the observed and the expected ones, according to a specific hypothesis. This test always examines the null hypothesis which states that there is no significant difference between the expected and observed results (or the frequency distributions are the same). Hence, a rejection of this null hypothesis indicates that the observed data (or frequencies) exhibit significant departures from the expected frequencies (Azen and Walker, 2011). The formula for calculating the Pearson chi-square test is:

$$X^2 = \sum_{\text{all categories}} \frac{(\text{observed value - expected value})^2}{\text{expected value}} = \sum_{i=1}^{c} \frac{(O_i - E_i)^2}{E_i} \qquad \text{(Eq. 3.10)}$$

where O_i represents the observed value in the i^{th} category and E_i represents the expected value in the i^{th} category. This X^2 test statistic follows a χ^2 distribution with c-1 degrees of freedom, where c is the total number of categories.

Additionally, after obtaining the degrees of freedom, it is essential to determine a relative standard as the basis for accepting or rejecting the hypothesis. In this research, a p value higher than 0.05 ($p > 0.05$) is adopted as the relative standard. The p value is the probability that any deviation of the observed value from the expected value is due to chance alone 5% of the time or less. Consequently, if the p value for the calculated χ^2 is $p > 0.05$, the null hypothesis can be accepted. By contrast, if the p value for the calculated χ^2 is $p < 0.05$, the null hypothesis will be rejected, indicating that some factor other than chance is operating for the deviation (Fisher and Yates, 1963; Azen and Walker, 2011).

3.3.2 Regression model

Since the chi-square test could only tell whether the two variables were associated with each other or not rather than measuring the depth of relationship, regression models were further employed to identify the linear relationship between predictor variables (e.g., behavioural, attitudinal and psychological variables) and the response variable (likelihood of using recycled water in a laundry). The regression analyses were performed by the software Statistical Package for the Social Sciences (SPSS) package. The basic form of the regression function is:

$$\text{logit}(\pi) = \ln(\frac{\pi}{1-\pi}) = \alpha + \beta_1 X_1 + \beta_2 X_2 + \ldots + \beta_\text{p} X_p \qquad \text{(Eq. 3.11)}$$

where π represents the probability of being supportive to recycled water use in a laundry, α is the intercept parameter and β is the coefficient associated with the j^{th} predictor variable, X_j (Azen and Walker, 2011; Tutz, 2012).

Particularly, the constant term (α) is the estimated log odds of being supportive on recycled water use in a laundry with whole observations, holding all predictor variables at the reference categories. The negative estimates (β) indicate that an increase in the variable (e.g., the higher the perceived cost) leads to a decrease in acceptance. The regression models also give additional information on the standard error and odds ratio. The standard error indicates the precision of the coefficient and the 95% confidence interval for the coefficient is approximately given by: Coefficient ± 2 Standard Error (Dolnicar et al., 2011), while the odds ratio provides the information about how sensitive the response variable is to each of the factors.

3.4 EXPERIMENTAL ANALYSIS OF THE NEW OPTIONAL RECYCLED WATER PURIFICATION SYSTEM

3.4.1 Feed solution

In this research, recycled water from CWW Western Treatment Plant has been employed as the feed solution. When the recycled water was insufficient, synthetic water was used to simulate the composition of CWW recycled water. The synthetic feed solution was prepared by dissolving 300 mg/L of $CaCl_2 \cdot 2H_2O$ in deionised water. Overall, the feed solution used for all the experiments had a total hardness level between 175 to 200 mg/L as $CaCO_3$, indicating relatively high levels of hardness.

3.4.2 Pre-conditioning of natural zeolites

The natural zeolites were provided by Castle Mountain Zeolites (Quirindi, New South Wales, Australia) which are clinoptilolite-rich minerals composed of clinoptilolite (around 85% by weight) and mordenite (around 15% by weight) with trace amounts of quartz.

Table 3.3 Chemical composition of Castle Mountain Zeolites

Mineral content	Percentage by weight (weight %)
SiO_2	71.81
Al_2O_3	12.10
Fe_2O_3	1.14
Na_2O	2.33
K_2O	0.90
CaO	2.60
MgO	0.65
TiO_2	0.22
MnO	0.03
P_2O_5	<0.01
SrO	0.22
Loss on ignition	7.77

Note: Adapted from An et al., (2011); CMZ, (2013).

The nominal mineralogical composition of Castle Mountain Zeolites is listed in Table 3.3 and the Si/Al molar ratio calculated from the composition data is 5.03 (An et al., 2011; CMZ, 2013). The zeolite type FM 16/30 was used in the experiment with a particle size of 0.5-1.5 mm and bulk density of around 1100 kg/m^3 (CMZ, 2013). Regarding the pre-treatment and preparation of filtration materials, natural zeolites have been initially washed and cleaned with deionised water so as to remove impurities. As shown in Equation 3.12, the main mechanism of zeolites in hardness removal is ion exchange, in which hardness ions such as calcium and magnesium are taken up by the zeolite column and an equivalent amount of more desirable ions such as sodium and potassium are released simultaneously (GE, 2012).

$$\begin{bmatrix} \mathrm{Ca} \\ \mathrm{Mg} \end{bmatrix} \cdot \begin{bmatrix} \mathrm{SO_4} \\ \mathrm{2Cl} \\ \mathrm{2HCO_3} \end{bmatrix} + \mathrm{Na_2} \cdot \mathrm{Zeolite} \rightarrow \mathrm{Zeolite} \cdot \begin{bmatrix} \mathrm{Ca} \\ \mathrm{Mg} \end{bmatrix} + \begin{bmatrix} \mathrm{Na_2SO_4} \\ \mathrm{2NaCl} \\ \mathrm{2NaHCO_3} \end{bmatrix} \qquad \text{(Eq. 3.12)}$$

Consequently, to attain a high sodium form of the exchangeable surface ions, the cleaned zeolites were then conditioned with a saturated sodium chloride solution (360 g/L) in flasks under a magnetic stirring thermostatic water bath at 50-60 ˚C for 24 hours. By this means, higher adsorption capacity of modified zeolites can be obtained (Inglezakis, 2005; Sivasankar and Ramachandramoorthy, 2011; Lin et al., 2013). The treated zeolites were finally rewashed three times with deionised water to remove excessive sodium chloride deposited on the particle surface and then dried at 105 ˚C for 2 hours. The dried zeolites (around 1.6 kg) were packed in the column.

3.4.3 Experimental set-up

Figure 3.3 showed the schematic of the zeolite pre-treatment unit. The feed solution in the feeding tank was pumped continuously into the zeolite column reactor, which had a height of 0.55 m, an internal diameter of 0.06 m in the reaction zone and an effective volume of 1.5 L (Figure 3.4). The purified recycled water can then be sent to a washing machine for laundry water supplies. The system could be operated with by-pass controllers under three scenarios.

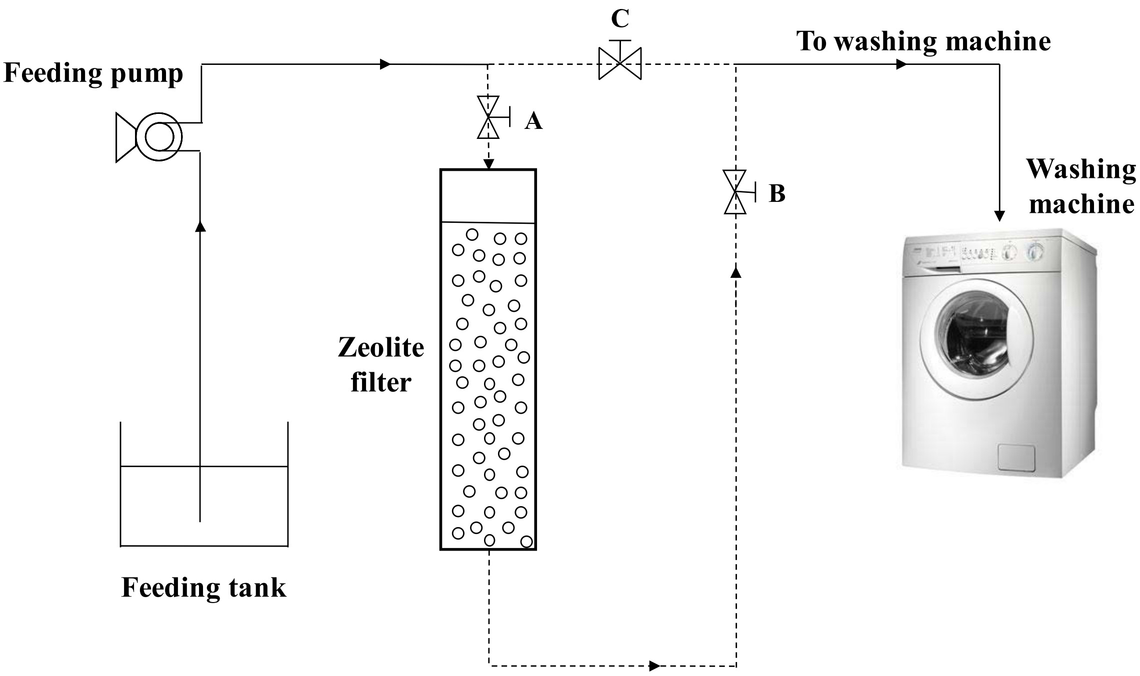
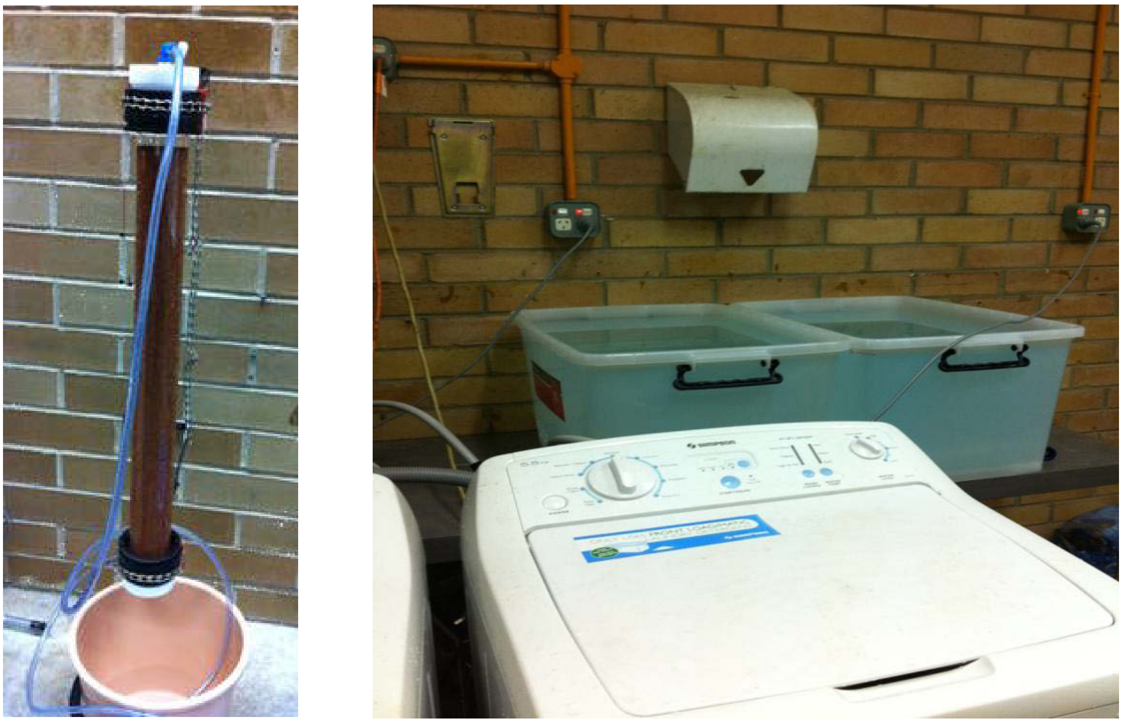


Figure 3.3 Schematic diagram of the pre-treatment unit



(a) Zeolite filter (b) Recycled water connected to the washing machine

Figure 3.4 Zeolite column experiment in the laboratory

For scenario 1, the recycled water was delivered directly to the washing machine without any further treatment (valves A and B closed, valve C open). Under scenario 2, the recycled water fully passed through the zeolite filter column prior to entering the washing machine (valves A and B open, valve C closed). With regard to scenario 3, the recycled water partially went through the zeolite filter column based on a bypass percentage before entering the washing machine (valves A, B and C open partially). To ease the observation of the filtration effectiveness, the experiment was operated under scenario 2 with a continuous influent pumping rate at room temperature (25 ˚C).

3.4.4 Zeolite regeneration with sodium chloride solution

The whole experiment also included two zeolite regeneration phases. High solubility, low cost and safety of sodium chloride makes it suitable as an effective regenerant for removal of hardness ions from the saturated ion exchange resin. Under the regeneration phases, 10-15% of sodium chloride is generally deemed to be a good concentration of salt in the feed solution. At this concentration, good capacity of the regenerated ion exchange reactor can be achieved, without excessive consumption of salt (Wist et al., 2009). Consequently, sodium chloride solution (100 g/L) was adopted as the feed solution and the zeolite column reactor was operated in a fluidized condition at the up flow mode. The regeneration pump worked continuously at 0.3 L/h for 24 hours. The regeneration proceeds according to Equation 3.13 (GE, 2012).

$$\text{Zeolite} \cdot \begin{bmatrix} \text{Ca} \\ \text{Mg} \end{bmatrix} + 2\text{NaCl} \rightarrow \text{Na}_2 \cdot \text{Zeolite} + \begin{bmatrix} \text{Ca} \\ \text{Mg} \end{bmatrix} \cdot \text{Cl}_2 \qquad \text{(Eq. 3.13)}$$

After regeneration, the zeolite column was washed thoroughly with deionised water until the washings showed no more excessive sodium chloride; and this made the column was now ready for the next cycle of water softening. Noticeably, small residual amounts of hardness might remain in the zeolite column.

3.4.5 Analytical techniques

3.4.5.1 pH

The pH was measured using a portable pH meter (HANNA Instruments HI 9125, Australia). It was a pH meter coupled with pH and temperature probes (Figure 3.5).

Figure 3.5 HANNA pH meter, HI 9125 (adapted from HANNA Instruments, 2013).

3.4.5.2 Alkalinity

The alkalinity was determined using a titrator (HANNA Instruments HI 84431, Australia), where titrations were conducted using the low range reagent HI 84431-50 (10 to 500 mg/L as $CaCO_3$). The titrator utilizes an electrometric titration with a pH electrode to determine the total titratable alkalinity in water (Figure 3.6).

Figure 3.6 HANNA total titratable low to high alkalinity titrator, HI 84431 (adapted from HANNA Instruments, 2013).

3.4.5.3 Nutrient analysis

Both nitrogen and phosphorus were measured using a photometric method, the Spectroquant® Cell Test (NOVA 60 Merck, Germany). Total nitrogen (TN) was based on the sum of NH_4-N, NO_2-N and NO_3-N rather than an independent TN test. Sample cell test was used to analyse the ammonium nitrogen (NH_4-N), nitrite (NO_2-N), nitrate (NO_3-N) and phosphate (PO_4-P) concentrations (Figure 3.7).

Figure 3.7 NOVA 60, Merck spectrophotometer

3.4.5.4 Turbidity analysis

The turbidity of recycled water was measured using a portable turbidimeter (HACH 2100Q, USA with a measuring range of 0 to 1000 NTU (Figure 3.8).

Figure 3.8 HACH 2100Q, portable turbidimeter (adapted from CHEM17, 2013).

3.4.5.5 Scanning electron microscope (SEM) analysis

The surface morphology of the zeolite material was observed by Scanning Electron Microscope (SEM, Zeiss EVO® LS 15, Germany). SEM uses electrons instead of light to form an image of a sample by scanning it with a high-energy laser of electrons in a raster scan pattern (Figure 3.9).

In this research, 2-dimentional images were generated over the selected typical areas of the surface of the zeolite samples, which displayed spatial variations in the surface properties with a high magnification of 1000X.

Figure 3.9 Zeiss EVO® LS 15 analytical environmental SEM (adapted from ZEISS, 2013).

3.4.5.6 Energy disperses X-ray spectroscopy (EDS) analysis

Energy disperses X-ray spectroscopy (EDS, Bruker XFlash® Detector 5030, Germany) was used to determine the elemental compositions of zeolite samples. EDS measures the number and energy of the X-rays emitted by the solid specimen. The energies of the characteristic X-rays allow the elements making up the sample to be identified, while the intensities of the X-ray peaks allow the contents of the elements to be quantified (Figure 3.10). The images and chemical characterizations of the natural zeolites under raw material, pre-conditioning and adsorption completion conditions were compared to see the changes.

Figure 3.10 Bruker XFlash® EDS Detector for SEM (adapted from Bruker, 2013).

3.4.5.7 Hardness analysis

Total hardness was measured using a hardness meter (HANNA Instruments HI 93735 Hardness Ion Specific Meter, Australia). It measures the total hardness in three different scales: 0 to 250, 200 to 500 and 400 to 750 mg/L as $CaCO_3$ (Figure 3.11). As this study only needs to measure the hardness levels in low range, the required reagents for measurement include: hardness indicator reagent LR (HI 93735A-0), hardness buffer reagent (HI 93735B-0) and fixing reagent (HI 93735C-0).

Figure 3.11 HANNA Hardness Ion Specific Meter, HI 93735 (adapted from HANNA Instruments, 2013).

All experiments followed the measurement procedures described in the analyser user manuals.

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CHAPTER 4

Conceptual Principle and Assessment for Analysis of New End Uses in Recycled Water Schemes

4.1 INTRODUCTION

This chapter identifies the potentials for the development of three recycled water new end uses, household laundry, livestock feeding and servicing, and swimming pool, in the future water use market. To validate the strengths of these new applications, a conceptual decision-making analysis was performed. This can be able to facilitate the optional management strategy selection process and thereafter provide guidance on the future end use studies within a larger context of the community, processes, and models in decision-making. Moreover, as complex evaluation criteria were selected and taken into account to narrow down the multiple management alternatives, the methodology can successfully add transparency, objectivity and comprehensiveness to the assessment. Meanwhile, the proposed approach could allow flexibility to adapt to particular circumstances of each case under study. Remarkably, the findings from this chapter also provide fundamental information for the subsequent quantitative model construction, validation and modification.

4.2 RESEARCH BACKGROUND

The growing environmental problems, including the diminishing natural water resources, greater water demand triggered by population growth and urbanisation, deteriorated water quality, and highly changing climate, have highlighted the importance of exploiting all other possible water sources before using up limited surface water and groundwater supplies. Recycled water, which is the wastewater being treated to a specified quality in order to be reused again, has been increasingly considered as a supplementary water supply (Lazarova et al., 2003; DWR, 2009). The merits of recycled water use have been demonstrated all over the world. In addition to economic, social and environmental benefits, a distinct benefit of water reuse is the steadiness of water supply for both household and local industries, which is superior to rainfall-dependent water sources (Lazarova et al., 2012).

Moreover, when bringing recycled water and other water resources together into the management, the ecological footprint of water, sewage and drainage system could be potentially reduced by over 25% (Anderson, 2003b). In a broader sense, water management can be further incorporated into the climate change adaptation and environmental sustainable development (Angelakis and Durham, 2008; Asano and Bahri, 2011). However, despite apparent strengths of recycled water, the further adoption of water reuse might be affected by a variety of issues, including water rights, environmental concerns, public acceptance and cost (NRC, 2012).

In developed countries, especially the cities and regions where freshwater resources are approaching the sustainable limit, recycled water would continue to be an important alternative water resource, especially for non-potable purposes (Chen et al., 2013a). More stringent water treatment standard (e.g., tertiary treatment and additional nutrient removal) is expected to be required in most recycled water schemes. As highly advanced technologies are available for producing clean water from wastewater without adverse health effects, the focus of motivating water reuse should shift away from technological issues to environmental, social and economic concerns (Van der Bruggen, 2010).

While agricultural and industrial purposes are the dominant end uses of recycled water presently, urban and residential applications such as landscape irrigation, toilet flushing and car washing, are experiencing rapid development, the amount of which are likely to be as high as or much higher than that of agricultural irrigation schemes (Brissaud, 2010; Wild et al., 2010). High value end uses with potential close human contact (e.g., recycled water for household laundries and swimming pools) will be promising but still somewhat ambiguous due to strong public misgivings.

Comparatively, in less developed countries, owing to technical and economic constraints, a large proportion of water reuse activities still involve secondary wastewater treatment. There would be a tendency in recycled water market towards a higher level of treatment. With respect to end uses, apart from agricultural irrigation that will continue to be the major user of recycled water, other agricultural activities such as livestock consumption, using recycled water, can be beneficial to alleviate freshwater stress and maintain economic development. According to these recent trends in both developed and developing areas (Chen et al., 2013a), current end uses are mostly limited to a few non-potable purposes. To meet aggressive water recycling targets, beyond the implementation of more recycled water schemes, the development of new end uses might be prospective and should be realized accordingly.

4.3 IDENTIFICATION OF POTENTIALS FOR THE DEVELOPMENT OF RECYCLED WATER NEW END USES

4.3.1 Use of recycled water for the household laundry

Generally, household laundry accounts for 15-20% of household water usage and is regarded as the second largest indoor user of water (Babin, 2005; Savewater Alliance, 2012). However, the water consumption for a laundry in different households may vary substantially due to a variety of washing machine types, number of washes, wash temperatures, load sizes, etc. Table 4.1 summarizes the different household behaviours in laundry worldwide. In Europe and Turkey, most of the households employ the state-of-art front loading washing machines with integrated heating rods, using electricity to heat up water internally. Turkish households even use high wash temperatures more frequently, where more than 75% of the clothes are washed at water temperatures higher than 50 ˚C. While in Asia, North America and Australia, top loading washing machines are widespread which use water from external cold and/or warm water taps that is not heated by the washing machine further. Due to the traditional laundry habits and practices, low wash temperatures have been widely adopted in these countries (Pakula and Stamminger, 2010). In Australia, the percentage of cold water used by washing machine was over 70% and varied between 70 and 90% (Bertone and Stewart, 2011). As the choice of washing machine type is the main factor affecting the annual water consumption in laundry and front loaders typically consume less than half as much water per wash as top loaders, European households use significantly less amount of water than Asian and North American households. However, they consume additional electricity and/or energy to heat up water from the cold water tap.

Overall, more than 9.9 kilolitres (kL) of fresh water can be saved per household per year worldwide if recycled water could be reticulated to the cold water input tap to the washing machine. The installation of an additional recycled water tap and the upgrade of recycled water treatment plant due to increased demand would incur extra charges. However, considering the total resource cost and operating/maintenance cost perspectives, the life cycle unit cost of the proposed new laundry use scenario might be financially viable. Moreover, the water authorities will also benefit from this new end use as the treated recycled water can be utilized more efficiently and result in higher revenue rather than being directly discharged to the environment (Bertone and Stewart, 2011). When it comes to water quality, the Class A recycled water which undergoes tertiary treatment has been shown to be suitable for washing clothes (YVW, 2010). Particularly, DOH (2013) prescribed that the microbial contents of the *Escherichina coli*, somatic bacteriophage and *Cryptosporidium* should be less than 1 cfu per 100 mL, 1 pfu per 100 mL and 1 oocyst per 1 L, respectively. In terms of heavy metal concentrations, Mainali et al. (2013) indicated that 1 mg/l of iron (Fe), 1 mg/l of lead (Pb), 10 mg/l of zinc (Zn), 5 mg/l of copper (Cu) and 1 mg/l of manganese (Mn) are the maximum allowable values for recycled water use in the household laundry in terms of tensile and tearing strengths.

Table 4.1 Household laundry behaviour in different locations

Region	Washing machine type	Load size per wash (kg)	Wash temperature (˚C)	Water use per wash (L)	Number of wash (phpy)	Water consumption for laundry (kL phpy)
West Europe	>98% Front	3-4*	40	60	165	9.9
East Europe	>98% Front	3-4*	40	60	173	10.4
Turkey	>90% Front	–	60	60	211	12.7
North America	>98% Top	3-4	15-48	144	289	41.6
Australia	>68% Top	–	20-40	106	260	27.6
China	>90% Top	1.3-2	Cold water	99	100	9.9
South Korea	>90% Top	–	Cold water	140	208	29.1
Japan	>97% Top	~3	Cold water	120	520	62.4

Note: Modified from Pakula and Stamminger, (2010); ABS, (2011a).
Abbreviations: phpy= per household per year; Front= Front loader; Top= Top loader.
*Asterisks indicate 75% of machine capacity.

4.3.2 Use of recycled water for livestock feeding and servicing

While recycled water in a household laundry could be a considerable contributor to freshwater savings, especially in highly populated urban areas if managed properly, there is also a high potential to exploit and implement new end uses in rural and regional areas. For instance, Figure 4.1 illustrates that less than 2% of annual water consumption on livestock farming activities in Australia is sourced from recycled water, compared with 61% and 37% from self-extracted and distributed water sources (e.g., surface water and groundwater), respectively (ABS, 2012). The total volume of water consumed by agricultural industry will grow fast, which is expected to rise 14% in the next 30 years, putting more pressure on dwindling water resources (UN, 2010). As the global water demand is likely to exceed supply by 56% by the year 2025 (WWO, 2010), there would be a significant decrease in agricultural productivity, especially during the time when prolonged drought and continuing unavailability of water happen. Since the water requirement for the livestock industry is high, from feed production to servicing and product supply processes, the related recycled water application should be taken into account to expand the recycled water market in non-metropolitan areas further.

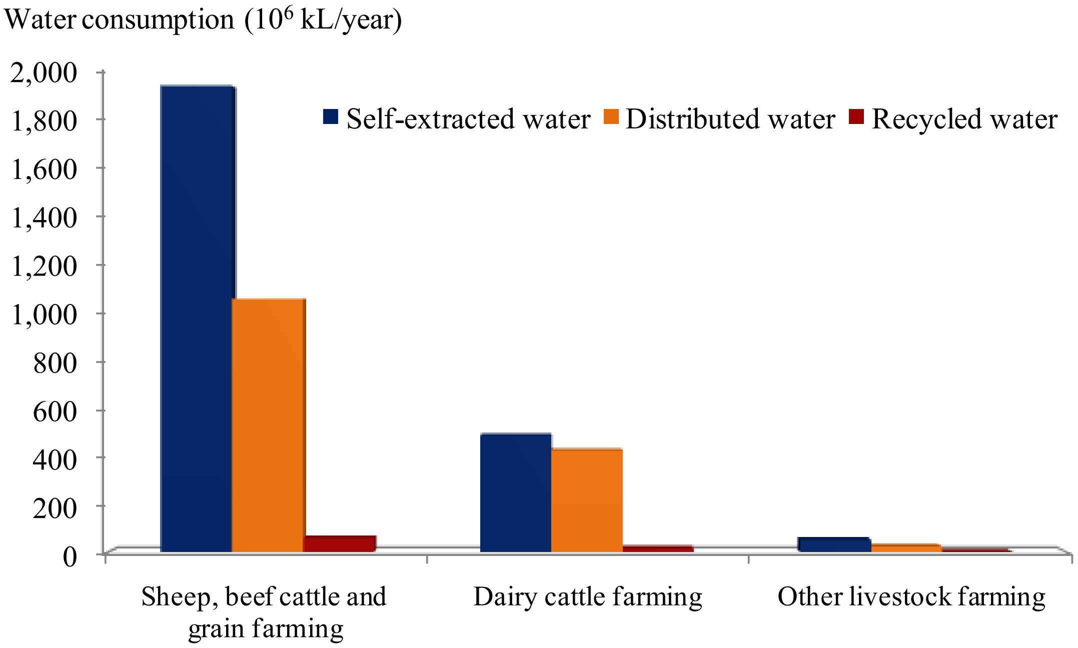

Figure 4.1 Water consumption on livestock farming activities in Australia by different origins of water (modified from ABS, 2012).

Particularly, livestock have to maintain their vital physiological functions with water content higher than 60 to 70% of the body weight. Reduction of water intake can result in lower meat, milk and egg production as well as weight loss. Drinking water is the prime way to meet the daily water requirements despite that livestock are able to ingest water contained in hydrated feedstuffs and/or absorb the metabolic water produced by oxidation of nutrients. Table 4.2 gives the water use information of different stocks. As can be seen, water needs vary because of the discrepancies of the animal species, breed, age, weight, the level of dry matter intake, the physical form of the diet, water availability and quality, temperature of the supply water, ambient temperature and the farming system.

Given that the water demand increases linearly with age and becomes constant after animal reaches adulthood (Chapagain and Hoekstra, 2003), the rough daily drinking water demand of an animal can be estimated in Equations 4.1 and 4.2. In some cases, water requirements can be extremely high for highly productive animals under warm and dry conditions due to increased water losses with high temperature and low humidity (FAO, 2006).

$$\text{For Age} < \text{Age}_{\text{adult}}:\quad q_d[e,a] = \left(\frac{q_{\max}[e,a] - q_{\min}[e,a]}{Age_{adult}}\right) \times Age + q_{\min}[e,a] \qquad \text{(Eq. 4.1)}$$

$$\text{For Age} > \text{Age}_{\text{adult}}:\quad q_d[e,a] = q_{\max}[e,a] \qquad \text{(Eq. 4.2)}$$

where, q_d[*e, a*] is the daily drinking water requirement of animal a in exporting country e. $q_{\max}$[*e, a*] and $q_{\min}$[*e, a*] are the average daily drinking water requirements of an adult and a body animal respectively. $\text{Age}_{\text{adult}}$ is the age of an animal in days when adult (Chapagain and Hoekstra, 2003).

Table 4.2 Drinking water and service water requirements for livestock

Stock type	Sub group	Average weight (kg)	Drinking water requirement (L/head/day)	Service water requirement (L/head/day)	
				Industrial system	Grazing system
Sheep	Weaners	15-20	3.6-5.2	2	0
	Adult dry sleep: Grassland (Saltbush)	40-50	2-6 (4-12)	5	5
	Ewes with lambs	36-45	4.0-6.5	5	5
	Lactating meat ewe	40-130	9.0-10.5	5	5
	Gestating dairy ewe/ram	–	4.4-7.1	5	5
	Lactating dairy ewe	36	9.4-11.4	5	5
Goats	Lactating	27	7.6-11.9	5	5
Beef Cattle	Feedlot cattle: backgrounder	200-680	15-40	2	0
	Feedlot cattle: short keep		27-55	11	5
	Lactating cows: Grassland (Saltbush)		40-100 (70-140)	11	5
	Dry cows, bred heifers & bulls	680	22-54	11	5
Dairy Cattle	Milking cows	680	68-155	22	5
	Dairy calves (1-4 months)	200-250	4.9-13.2	0	0
	Dairy heifers (5-24 months)	350-450	14.4-36.6	11	4
	Dry stock	400-550	35-80	22	5
Swine	Weaner	5-35	1.0-3.2	5	0
	Feeder pig	18-37	3.2-10	50	25
	Gestating sow/boar	–	13.6-17.2	50	25
	Lactating sow	175	18.1-22.7	50	25
Chicken	Broiler summer (winter, fall & spring)	4-5	0.45 (0.28)	0.09	0.09
	Broiler breeders	1-3	0.18-0.32	0.09	0.09
	Laying hens	1-3	0.18-0.32	0.15	0.15
	Pullets	0.5-1	0.03-0.18	0.01	0.01
Horses	Small	225-360	13-20	0	5
	Medium	275-500	26-39	5	5
	Large	450-700	39-59	5	5
Camel	Mid-lactation	350	31.5-52.2	–	–

Note: Modified from Attwood, (1997); Chapagain and Hoekstra, (2003); FAO, (2006); Markwick, (2007); Dennis, (2008).

Additionally, service water are also required to clean the livestock production units, wash animals, cool the facilities, animals and their products as well as discharge the wastes. Table 4.2 shows some indications of different service water requirements. It can be seen that the water consumption in industrial systems is generally higher than that of grazing systems, owing to extra cooling and cleaning purposes of facilities. Specifically, pigs require a large quantity of water when kept in industrial “flushing systems”, where service water requirements for washing the manure down a gutter can be seven times higher than drinking water needs.

Overall, the proportion of livestock production met by specialized and intensive industrial systems is rapidly increasing as these systems react faster to growing demand in production and consumption across the globe. Although the pace of expansion of livestock production may diminish, the growing trend will endure over the next 20 years (Gerber et al., 2005). Hence, if the recycled water can be properly treated to a standard that is appropriate for livestock production, considerable freshwater savings would be achieved, especially in intensive farming systems. While this new end use has not been extensively discussed globally, some areas such as the State of Victoria, Australia, have already formulated guidelines regarding the recycled water quality for use in livestock production. The Class A recycled water with tertiary treatment and pathogen reduction is recommended for general livestock (SGV, 2009).

4.3.3 Use of recycled water for swimming pools

Aquatic centres and swimming pools are major public facilities that provide significant benefits in terms of community development, sport, health and fitness to local residents. They require a large amount of water and energy to operate and to maintain so that a number of public pools have been closed during the drought conditions. If no action were taken to mitigate inevitable water shortages in the future, there would be higher risks of closure for more pools in extreme weather situations, affecting the aquatic and recreational industry. Nevertheless, there is still a lack of information on water saving and reuse strategies in existing public aquatic centres around the world (Sydney Water, 2011a).

The major water consumption categories of a typical aquatic centre are depicted in Figure 4.2. While strategies such as dual flush toilet systems, water saving and flow regulation devices in shower heads, and pool covers to reduce evaporation are commonly reported approaches being successfully implemented in many newly constructed aquatic centres, there will be a great potential in water saving and reuse when adopting measures on treating backwash water for use as pool make-up water.

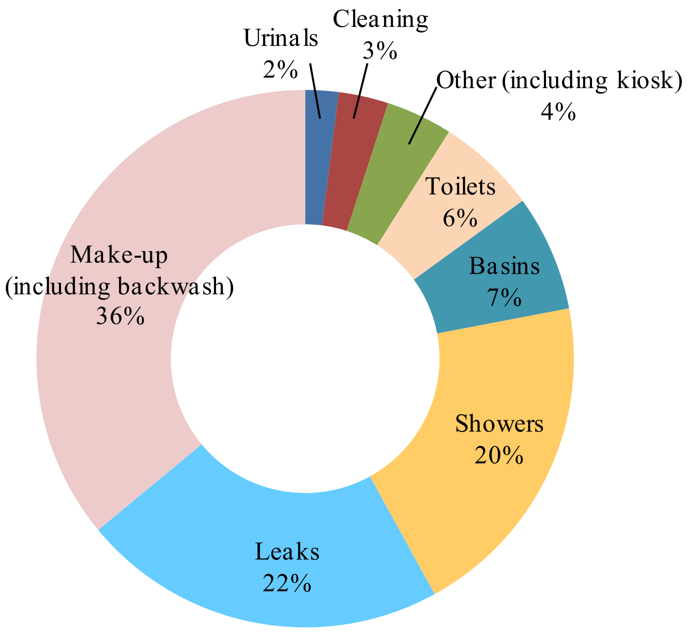

Figure 4.2 Water use breakdown of a typical aquatic centre (modified from Sydney Water, 2011a).

However, the health and environmental risks associated with the use of treated backwash water vary greatly in terms of different water sources, end uses, treatment and management options, etc. (Table 4.3). To control the risks under low levels, it is advisable to use advanced treatment technologies such as Reverse Osmosis (RO), Ultrafiltration (UF) and/or Granular Activated Carbon (GAC), and to conduct frequent monitoring and maintenance. For instance, the state of New South Wales, Australia, stipulates that the quality of recycled backwash water should meet the Australian Drinking Water Guidelines as pool water would likely to be accidentally swallowed during recreational activities (Environmental Health, 2012).

Table 4.3 Risk management for backwash water reuse

Overall risk classification	High risk[1]	Medium risk[2]	Low risk[3]
Source of pool backwash	Shallow pools[4]	Medium depth pools[5]	Deep pools[6]
End use for treated backwash water	Shallow pools[4]	Medium depth pools[5]	Deep pools[6]
Pool operation	Manual control	Automated	Automated including pH and ORP[7] (or chlorine probe) controls and alerts
Pool maintenance	Breakdown maintenance only	Non scheduled preventative maintenance combined with breakdown maintenance	Scheduled preventative maintenance by qualified staff (e.g., pump servicing)
Pool super chlorination	Monthly or less	Fortnightly	Weekly
Backwash frequency	Every month	Every fortnight	As determined by pressure drop across filter or weekly
Backwash water treatment	Settlement or filtration only	Settlement, filtration and disinfection	Advanced filtration and disinfection (RO[8] and ultraviolet light)
Monitoring of backwash treatment process	Monitoring of some treated batches	Periodic monitoring during process; Water quality testing before discharge to pool	Online water quality monitoring (e.g., particle counting/turbidity for MF[9], conductivity for RO[8]; Automatic diversion to wastewater if needed

Note: Adapted from Sydney Water, (2011a).

[1] High risk: if the scheme matches any of the below;

[2] Medium risk: if the scheme matches any of the below and does not match any of the high risks;

[3] Low risk: if the scheme matches all of the below;

[4] Shallow pools: toddler and learn-to-swim pools;

[5] Medium depth pools: family, general purpose and hydrotherapy pools;

[6] Deep pools: Olympic and diving pools;

[7] ORP= Oxidation Reduction Potential;

[8] RO= Reverse Osmosis;

[9] MF= Microfiltration.

Notably, when considering the use of these approaches, a lack of supporting information and understanding may hinder the implementation processes or cause the systems remain dysfunctional for a period of time. Hence, it is essential to ensure that adequate training, information, manuals and some level of feedback have been obtained on how to operate and maintain the strategy efficiently and effectively (Hazell et al., 2006). Besides, the staff members should also pass on water saving and reuse information to patrons and future customers, which could further improve water sustainability. There are successful applications of recycled backwash water in several Australian public aquatic centres, including pools in Penrith and Ryde city councils, New South Wales and centres in the city of Whittlesea, Victoria. Nonetheless, the availability of well documented information is still limited (Hazell et al., 2006; Chen et al., 2013c).

4.4 ASSESSMENT ANALYSIS

4.4.1 Qualitative feasibility analysis

The qualitative feasibility analysis is often used in the preliminary stages of decision making, which acts as a precursor to strategic management planning of recycled water schemes. It can be applied as a tool to identify the critical factors associated with the successful implementation of the schemes, including the project’s internal aspects such as strengths and weaknesses, plus external factors including opportunities and threats (Mainali et al., 2011a).

Table 4.4 summarises the qualitative profiles of three proposed new end uses of recycled water (SGV, 2009; Mainali et al., 2011b; Sydney Water, 2011a). As can be seen, the end uses are apparently viable based on their foreseeable positive aspects (strengths and opportunities) to the community and environment.

Table 4.4 Qualitative feasibility analysis of proposed new end uses of recycled water

Qualitative feasibility analysis	Potential new end uses		
	Household laundry	Livestock using	Swimming pool
Strengths	• Washing clothes− a year round activity • Significant laundry water consumption	• Large water use on drinking and servicing purposes in rural and regional areas	• Little information on water reuse for the option • Large water consumption for pool make-up water
Opportunities	• Expand the dual pipe water supply system • Considerable freshwater saving and reduced effluent discharge • Current recycled water (MF[1] or advanced treatments) can be safely used • Higher possibility of the public acceptance	• Expand the dual pipe water supply system • Considerable freshwater saving, especially in intensive farming • Current recycled water (tertiary and disinfection) can be safely used • Related guidelines in some areas have been formulated	• Considerable freshwater saving and avoidance of sewage discharge • Reduced health risks via improved water quality and system management • Lower risks of closure of some pools in extreme weather conditions
Weaknesses	• Close human contact • Need of extra taps to connect the dual pipe system to laundry • Lack of safety data and relevant guidelines • Lack of comprehensive quantitative assessment	• Health risk concerns • Need of extra taps to connect the dual pipe system to stockyards • Variety in water needs at different stockyards • Lack of comprehensive quantitative assessment	• Close human contact • Strong public objection to the end use • Additional costs on water quality improvement • Frequent water quality monitoring and system maintenance
Threats	• Distrust the quality of water and concerns about health issues • Water hardness forming scum • Public concerns on colour, odour, potential damage to clothes (e.g., iron staining garments) and washing machines, and increased cost	• Colloidal suspensions of oils and greases • Livestock illness and discomfort (e.g., salinity and water hardness) • Suspicion and distrust from farmers • Staff reluctance due to close human contact with recycled water	• Requirement of advanced treatment technologies (e.g., RO[2]) to produce high quality water • Need of adequate staff training •Public concerns on colour, odour and disease transmission •Difficulty in acquiring public acceptance

Note: [1]MF= Microfiltration; [2]RO= Reverse Osmosis.

Future work still needs to find the ways and means to offset the weaknesses by distinct strengths and convert the threats to opportunities. As the descriptive results are unconvincing to some extent, there is a need for a more comprehensive quantitative assessment of new end uses with respect to technical, environmental, risk, social and economic considerations. As such, the optimal decision-making solutions for particular recycling schemes can be demonstrated and highlighted, which provide powerful guidance for sustainable water reuse management in the long term (Chen et al., 2012a, 2013c).

4.4.2 Quantitative analysis for prioritization of management options

The procedures regarding the quantitative analysis of the proposed new end uses of recycled water include: i) consideration of specific management alternatives related to each new end use; ii) selection of key criteria that might affect the implementation of new end uses; iii) application of multi-criteria analysis; iv) recommendation of preferred option(s); v) communication, review, monitoring and reporting (Chen et al., 2012a).

4.4.2.1 Management options

1) Baseline scenario. This scenario identifies the "business as usual" projection and can be regarded as a hypothetical reference case. It simply projects the future recycled water end uses based on existing recycling schemes (e.g., toilet flushing, garden watering and car washing). In other words, the baseline for a newly recycled water project reasonably presents the recycled water use activities that would occur in the absence of the proposed new end uses (i.e., laundry, livestock using or swimming pool). Thus, the baseline scenario can be used to compare and determine whether a new end use of recycled water is additional, and the additional savings and benefits achieved by the implementation of new end use activity (CDM, 2008).

2) Do-something options. This step is to identify all plausible alternative scenarios which can deliver outputs or services with comparable quality and properties to the proposed new end use project activity. Specifically, some scenarios include the selection of different equipment and/or facilities. For instance, the washing machines in households possess a number of different characteristics (e.g., loading type and capacity,

water and energy consumption per wash, brand and model name). A scenario regarding the adoption of front loading washing machine may involve with less water, energy and detergent consumption, but higher initial cost and inconvenience of loading and unloading clothes, compared with the use of top loaders (Bansal et al., 2011; Gato-Trinidad et al., 2011; Chen et al., 2012a). Likewise, in livestock feeding and servicing industry, a scenario in which the intensive farming system instead of grazing system is employed, may relate to a larger amount of water requirement, higher capital, maintenance and staff training costs, but greater production efficiencies (Gerber et al., 2005). For swimming pools, the installation of water efficient facilities such as dual flush toilets and filtration systems is likely to minimize water consumption and environmental footprint but induce additional investments simultaneously (Hazell et al., 2006).

Besides, some scenarios are also associated with the use of advanced treatment technologies to achieve different recycled water quality. Currently, although the Class A recycled water which undergoes tertiary treatment such as Microfiltration (MF), ultraviolet disinfection for pathogen reduction, is generally regarded to be protective of the environment, public and animal health and food safety (O’Toole et al., 2009), more advanced techniques are supposed to be discussed to further improve the recycled water reliability and community acceptance. For example, some studies indicated that zeolites are good materials for water purification due to the advantages of low cost, operational simplicity and unique compositions for high level of ion-exchange, adsorption and regeneration. At tertiary treatment stage, the traditional system equipped post-treatment using zeolite column could significantly improve the effluent quality, especially the removal of ammonium in wastewater (Li et al., 2007; Widiastuti et al., 2008).

After usage, zeolites are able to be regenerated for reuse purposes. Since the cost of chemical regeneration could be relatively high, some hybrid biological-ion exchange systems have been developed, where ammonium ions are initially absorbed by zeolites and bacteria attached to zeolite surface can subsequently convert ammonium to nitrite and nitrate nitrogen, contributing to bioregeneration of zeolites without the use of chemical regenerants. It was found that zeolites also have the ability to remove PO_4^{3-} and enhance the sedimentation rate by the stable floc formation (Chung et al., 2000;

Kimochi et al., 2008). Besides, the potential of zeolites to remove bacteria and organic matter has been reported as well (Bowman, 2003).

Comparatively, other studies applied a MF-Activated Carbon (AC) system, either a GAC filter or a Powdered Activated Carbon (PAC) suspension, rather than MF alone as tertiary treatment, which demonstrated better removal efficiency on synthetic organic chemicals and natural organic matters that cause taste, odour and colour. Similar to zeolites, as replacing AC in adsorptive column is relatively cumbersome and expensive, many systems adopt hybrid biological GAC/PAC filter systems where the media is operated essentially under low flux mode to support high heterotrophic and nitrifying biomass. In addition, compared with post-treatment AC units, the pre-treatment and hybrid configurations show possibility to control the AC age and mitigate membrane fouling (Kim et al., 2009; Stoquart et al., 2012). However, the addition of zeolite or AC in wastewater treatment would probably lead to a longer contact time (5 to 20 minutes) and therefore a lower flux rate, and introduce additional installation and usage fees.

Another advanced approach is to use MF-RO treatment system, which is able to produce recycled water of potable water quality. This could be regarded as a much reliable option for swimming pools owing to the potential close contact of treated backwash water with human body. The MF-RO system enables the water to be filtered, and most importantly, the dissolved salts (e.g., sodium and chloride) could be removed from the water during the backwash process, allowing it to be put back into the pool or used for other purposes such as irrigation and toilet flushing. Remarkably, both the installation fees and life cycle cost of RO are relatively high as the energy consumption is expected to be around 1.1 KiloWatt-hour per cubic meter (kWh/m^3) compared with 0.23 kWh/m^3 of MF (Côté et al., 2005). In spite of a long cost-recovery period, when the system is fully functioning, there will be no need of sewage discharge system as all backwash water in the pool is able to be treated and supplemented for pool make up, saving approximately 52 kL of freshwater per year (Hazell et al., 2006). The system also has positive effects on the environment in the long term. However, appropriate operation and maintenance would also be needed to keep long term performance of these advanced treatment technologies.

4.4.2.2 Evaluation criteria

This step identifies relevant evaluation criteria by which management alternatives on end uses would be judged. To ensure comprehensiveness and objectiveness of the assessment, it is advisable to take into account of relevant technical, environmental, health risks, social and economic aspects of alternatives appropriately in the decision making procedure. Technical indexes generally refer to recycled water availability and operability. On the one hand, water availability analyses incorporate detailed calculations on supply-demand relationship, service coverage, continuity and accessibility. Specifically, the supply-demand analysis is to identify the amount of recycled water needed in a certain area (e.g., household, farm and aquatic centre) for basic end use activities on a daily, monthly or annual basis. The service coverage reflects the size of the population/livestock that receives recycled water supply compared to the size of population/livestock without the service in that area while the water flow continuity and accessibility imply that recycled water should be available during working and livestock feeding time or at any time when it is required by households or aquatic centres. On the other hand, water operability analyses include the investigations on the ease of operation and maintenance and system flexibility to upgrade or extend, as well as risk and/or reliability assessment regarding the occurrence probability of treatment system malfunction (Ali, 2010).

Moreover, with respect to environmental aspects, highly treated recycled water is able to mitigate nutrient loads to surface or groundwater and reduce freshwater and chemical fertilizer consumption whereas poorly designed schemes may substantially alter the land use, affect the wetlands and endangered species and trigger adverse effects on surface or groundwater quality. Thus, several environmental indexes (e.g., greenhouse gas emissions, ecology, freshwater savings, energy consumption and recycled water quality) need to be quantified. The major greenhouse gases- carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) can be produced in wastewater treatment and power generation processes, which would lead to global warming and then rapid climate changes. Their greenhouse gas effect is typically weighted by global warming potential (unit: t CO_2e) which is dependent upon the timeframe of consideration, usually 100 years (Listowski et al. 2011; Gupta and Singh, 2012). Additionally, the ecology generally refers to the impacts on land, surface water, groundwater, air, sediment and ecosystem as a result of reduced wastewater discharges. Furthermore, recycled water quality would not only impact the environmental ecosystem, but also be directly related to the risks on human health. Risk assessment can be conducted by either qualitative or quantitative approaches. A qualitative risk level (i.e., low, moderate, high or very high) can be estimated from the severity and expected frequency of the adverse event to human health and the environment while quantitative measurement involves detailed hazard identification, dose-response assessment, exposure assessment and risk characterization using static or dynamic assessment models. Other integrated approaches such as the hybrid fuzzy-stochastic model and Bayesian network model could also be employed as alternative ways (Chen et al., 2013b).

With regard to social indexes, public acceptance, political support and educational opportunities are the main components to be considered for smooth expansion and development of recycled water supply and new end uses in local communities. Hence, research surveys on non-users, perspective users and current users are encouraged to be performed for understanding the holistic public knowledge, behaviour and attitude about water saving and recycled water use, and the impacts as well as measures that people are concerned when implementing the new end uses. Political support is relevant to the rebate, subsidy and policies from water authorities, providers or the government decision makers on the adoption of new water resource strategies. Educational opportunities include the education campaigns, offered information (e.g., leaflets, brochures and articles on newspapers/magazines), personal communications and workshops that could be provided to increase the public comprehension on the importance/advantage of recycled water as an alternative water resource (Chen et al., 2013d).

For economic indexes, it is recommended to consider both the internal factors (e.g., capital, operational and maintenance costs, and recycled water affordability) and external factors (e.g., personal health and financial savings from reduced diseases and work/school absenteeism avoided) whenever possible. As there is typically no explicit market for external influences, the primary target is to quantify the internal factors of different reuse options in terms of monetary units. More precisely, capital cost embodies the initial investment and installation fees on wastewater treatment and supply facilities as well as end user devices while operational and maintenance costs represent the continuous investment over the whole running process (Urkiaga et al., 2008). The affordability index reflects whether the price of recycled water is affordable to householders, farmers, workers and/or consumers, which mainly depends on the annual income and recycled water tariff. Nevertheless, as a result of uncertainties existed in environmental (e.g., climate, geographical and water availability), demographical and economic conditions at different regions and/or time periods, the assessment data collection process would be time consuming and somewhat challengeable, which requires detailed site investigations, recycled water quality monitoring, analyses and reviews as well as extensive public surveys.

4.4.2.3 Multi-criteria analysis (MCA) in decision making

The adoption of MCA methodology as the last procedure in phase 2 of assessment framework (Chapter 3) is to investigate the tradeoffs among these selected multiple conflicting criteria and then obtain rankings of different management alternatives under certain mathematical algorithms. From the computerized MCA simulation, the least preferred options towards one/several end use(s) could be quickly eliminated whereas the superior alternatives can be further discussed. This can provide a powerful guidance for sustainable water recycling and reuse management in the long term as it is possible to suggest how much a successful strategy could benefit the decision maker in exploitation, planning, development and expansion stages of new end uses. With these highly persuasive data, the public acceptability and trust on recycled water applications can also be greatly improved, which in turn further accelerate the booming of potential recycled water markets (Abrishamchi et al., 2005; Chen et al., 2012b).

Initially, the scoring process aims to generate a matrix where elements represent evaluation scores of each option against each criterion. As the criteria are fundamentally different by nature and their values are normally manifested in varied forms (e.g., quantitative estimates or qualitative judgments) with different unit scales (e.g., monetary, volume and concentration units), classification and normalization processes might be required to make the final score dimensionless thereby enabling comparison. For qualitative data, 5-, 7-, 9- and 11-point scale systems have been reportedly used in different locations of Australia, where higher values represent more positive effects

(Coutts, 2006). Since qualitative information is likely to introduce bias towards or against certain facilities or technologies due to inevitable personal perceptions, quantitative data are supposed to be employed to the great extent. Secondly, to embody varying degrees of concern on different evaluation criteria in decision making, weighting process becomes essential in MCA which is to assign higher weights to more important criteria and smaller weights to less important criteria. Although recent works have given more attention to recycled water quality and operational and maintenance cost (Ngo et al., 2009), or highlighted the importance of environmental performances (Chen et al., 2013c), the case-specific context can facilitate the need for differential weighting. Yet in some cases, due to lack of sufficient expertise, decision makers might be less confident in assigning specific weight to each criterion, making the weight values be highly variable. Thus, to reduce man-made errors, some sophisticated models (e.g., random weight model, rank order weight generation model and response distribution weight model) have been developed. For instance, the Rank Order Weight Generation (ROWG) model is capable of evaluating all possible combinations of weights via computerized simulation with a given precedence order in the criteria. The competing options could be then narrowed down through statistical weight results (Chen et al., 2012a).

At the heart of MCA is the aggregation process which must be carefully assessed to ensure the results of the evaluation are consistent with the preferences of the decision makers. There are a number of aggregation algorithms, including the weighted summation, Multi Attribute Utility Theory (MAUT), compromise programming, analytical hierarchy process, Elimination and Choice Expressing Reality (ELECTRE), Preference Ranking Organization Method for Enrichment Evaluation (PROMETHEE) and cooperative game theory. Despite the fact that the aggregation functions are distinct with different levels of calculation complexity and accuracy, a core procedure is to incorporate weight information with the evaluation matrix scores to attain a result. Hajkowicz and Higgins (2008) found strong agreement between different MCA algorithms used for water resource management. Hence, in many applications there is no overwhelming reason to adopt one MCA technique over another approach. The ease of understanding, transparency and preciseness would be the prime concerns.

In circumstances of determining the robustness of the options, it is suggested to use more than one MCA technique (Alvarez-Guerra et al., 2009). The detailed applications of MAUT and PROMETHEE in recycled water new end uses especially the household laundry are shown in Chen et al. (2012a, 2013c). Lastly, with a fixed set of criteria values and a fixed set of weights, the statistical values of management options can be obtained. According to the ranking order, the least preferred options can be eliminated and the optimal option(s) for new end use implementation should be intensively investigated. Additionally, when adopting a single set of weights, sensitivity analyses of weights can be effective to minimize uncertainty in scores and guarantee the reliability and accuracy of rankings and the final decision. Meanwhile, continuous communications and conversations among stakeholders, local authorities, water providers and community members might also be essential.

4.4.3 Decision-making plan

At management stage, this phase involves the clarification of water reuse goals associated with the recommended option(s), conduct of risk communication to increase public awareness, and completion of reporting, technological monitoring and review for organizational entities' approval. Based on MCA results, some targeted water reuse goals on superior alternatives that are expected to accomplish in a short term can be established and verified through committee meetings and discussion as well as external counselling and resources. A detailed assessment report can be then presented to relevant government departments, which should include the major strengths and barriers regarding the new end use strategy implementation and expansion, together with periodic review and evaluation plans in future operational stages.

4.5 CONCLUSIONS

The successful establishment and implementation of new applications in existing or future schemes may depend on a series of issues, comprising technical concerns, environmental impacts, health risks, social attitudes and economic statuses. As a systematic procedure for analysis of multiple constraints is still lacking, this chapter proposed a novel approach for the holistic assessment of three possible new end uses, including household laundry, livestock feeding and servicing, and swimming pool. It was convinced that the methodology and findings would not only offer fundamental information for the subsequent model design and construction but also benefit the decision making with a clear, sound and reliable strategy. Consequently, the whole decision making process for recycled water new end use exploration and implementation would lead to a more robust, efficient and credible solution for prospective water market.

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CHAPTER 5

Multi-Criteria Analysis towards the New End Use of Recycled Water Schemes for a Household Laundry: A Case Study in Sydney

5.1 INTRODUCTION

This chapter presents an overview of the current recycled water schemes in Sydney and identifies the potential of implementing sustainable water management strategies in an existing project using a simplified quantitative Multi Criteria Analysis (MCA). With analytical results on the ways of realizing the feasibility in conducting the new recycled water end use for a household laundry, more specific management alternatives towards the new application are proposed and evaluated. This was achieved by a complex MCA using the Rank Order Weight Generation (ROWG) together with the Preference Ranking Organization Method for Enrichment Evaluation (PROMETHEE) outranking techniques. Particularly, the generated combinations of weights via Monte Carlo simulation were able to reduce the man-made errors of single fixed set of weights significantly because of its objectivity and high efficiency. As a result, the research highlighted the viability of utilizing highly treated recycled water for existing and new washing machines. This could provide powerful guidance and advice for sustainable water reuse governance and management in the long term. However, more detailed field trials and investigations are still required to understand, predict and assess effectively the impact of selected criteria for new management alternatives.

5.2 WATER RECYCLING SITUATIONS IN SYDNEY

Prolonged drought conditions, increased water consumption, deteriorating water quality and highly variable climate in Australia have forced water authorities, consumers and local councils to consider recycled water as a supplementary water supply (Al-Rifai et al., 2007). Recycled water can help to alleviate the pressure on existing water supplies, protect remaining water bodies from being polluted, as well as provide a more constant volume of water than rainfall-dependent sources (Chen et al., 2013c). Sydney, the capital city of the state of New South Wales (NSW) in Australia, has invested heavily in recycled water schemes as water supply dams have been reduced to extreme low water levels (less than 65% of full operating storage) since early 2004 due to extended droughts. A large population of 4.5 million people with its continuing 1.3% growth every year also puts pressures on current water supply strategies.

Presently, Sydney recycles about 33 Gigalitres per year (GL/yr) of wastewater for non-potable uses including agriculture, irrigation, industry, residential indoor and outdoor activities and indirect potable reuses, accounting for 4.3% of the annual total water consumption. Nevertheless, compared with rural areas and some other big cities in Australia, the recycled water use in Sydney is still low. As the city has the highest level of effluent discharge but a lower recycling rate than any other Australian city, there is a large potential to expand the existing recycled water market. To meet the aggressive targets in the NSW Government's Metropolitan Water Plan which is to increase the water reuse to 70 and 100 GL/yr by 2015 and 2030 respectively, more recycled water schemes together with new end uses should be further exploited and developed (NSWOW, 2010; Sydney Water, 2012).

Sydney Water Corporation (SWC), as the biggest government-run water supplier in Australia, has been operating water supply, stormwater management, sewage and large-scale recycled water treatment functions since 1992. Local councils have been left with only small-scale onsite water recycling services and residual drainage functions (Stenekes et al., 2006). Presently, there are about 20 large-scale water recycling schemes and 150 smaller local-scale projects running in greater Sydney, with many more in the planning and construction stages (NSWOW, 2010; Sydney Water, 2011b). Most of these schemes are related to non-potable uses (e.g., irrigation, industry, residential uses and environmental flows) whereas Indirect Potable Reuse (IPR) and Direct Potable Reuse (DPR) schemes are not widely discussed in the literature.

5.2.1 Agricultural and landscape irrigation uses

In the past five years, the greater Sydney region has put considerable effort into expanding and increasing irrigation schemes which use about 4.6 GL/yr of recycled water for irrigating farms, parks, sports fields and golf courses (ABS, 2010). For example, large-scale agricultural irrigation schemes such as the Carlton Farm, the Elizabeth Macarthur Agricultural Institute and the Warwick Farm Racecourse are being successfully implemented, saving considerable fresh water and reducing fertilizer costs. Many local authorities are also actively involved in irrigating parks and sports fields with recycled water, including the Hawkesbury campus of the University of Western

Sydney and the councils of Penrith, Wollongong and Camden. Besides, the Penrith and Ryde city councils also treat and return the backwash water, which is considered as a newly developed end use, to public swimming pools. Additionally, irrigating golf courses with recycled water is widely conducted at many golf clubs (e.g., the Ashlar Golf Club, the Dunheved Golf Club, the Richmond Golf Club and the Kiama Golf Club), which was demonstrated to be beneficial during the severe drought in 2002-03 (ATSE, 2004; Sydney Water, 2011b).

While most of the above-mentioned schemes are centralized ones that import recycled water from the nearby Sewage Treatment Plant (STP), some small scale water recycling schemes are decentralized options with individual onsite wastewater collection, treatment and supply systems. The sewer mining projects include the Kogarah Municipal Council's Beverley Park Water Reclamation Project, the Macquarie University Playing Fields Project, Workplace6, the North Ryde Golf Club Scheme, the Pennant Hills Golf Club Scheme and the Sydney Turf Club Scheme. With respect to the effluent quality, to ensure its safety and sustainability, NSW water recycling guidelines for irrigation continue to be revised toward more stringent requirements so that most of the STPs have upgraded their treatment facilities to meet tertiary or even drinking water quality requirements. For instance, in 2005, SWC carried out extensive transformations on the Richmond STP which used to supply secondary effluent to nearby consumers. Currently, the old trickling filter has been replaced by the intermittently decanted aerated lagoon process coupled with additional tertiary treatment (i.e., sand filtration, chlorination and dechlorination), contributing to the harmless discharge of nitrogen gas to the atmosphere (Aiken et al., 2010). Likewise, the Ryde City Council also introduced new technologies at the Ryde Aquatic Leisure centre where the Ultraviolet (UV) approach has replaced the ozone water treatment system for filtering and recovering backwash water in swimming pools.

5.2.2 Residential uses

Since only 1-4% of residential water consumption is actually used for drinking, SWC has recognized an opportunity for substantial water saving and has constructed dual-reticulation pipe systems in some residential areas, where recycled water has been provided for garden watering, toilet flushing, car washing, etc. For instance, the Rouse Hill Water Recycling Scheme (RHWRS), located in north-western Sydney, is the largest residential water recycling scheme in Australia which started in 2001 and uses up to 2.2 GL/yr of recycled water, reducing drinking water demand by about 40% (NSWOW, 2010). Another representative case is the Water Reclamation and Management Scheme (WRAMS) owned by the Sydney Olympic Park Authority. It has extended the urban water recycling concepts to integrated water management by incorporating both stormwater and recycled water in recycled water delivery systems. In addition to serving 2000 houses in the neighbouring residential suburb of Newington, WRAMS also supplies recycled water to all commercial premises and sporting venues at Sydney Olympic Park. Recently, the end uses have been expanded to over eleven types, which include swimming pool filter backwash and ornamental fountains (Chapman, 2006). At the same time, some decentralized schemes in remote or rural suburbs are carried out where households have installed greywater diversion and treatment systems either in a group or individually. Taking the Mobbs' house in Chippendale as an example, about 100 kilolitres per year of wastewater from the house is processed by three filter beds and a UV radiation unit and then used for toilet flushing, clothes washing and garden watering (ATSE, 2004). The recycled water quality in the above-mentioned schemes has satisfied all mandatory chemical, physical and microbiological performance standards prescribed in the guidelines (Sydney Water, 2011b).

Having noticed the great benefits from these dual reticulation schemes, SWC continues to develop large-scale recycling schemes on newly released residential areas. For example, it has now expanded the RHWRS project to eventually serve 36000 homes. A similar scheme at Hoxton Park, south-western Sydney is planned for completion by 2013; it is expected to supply 2.3 GL/yr of recycled water to about 14000 future homes and surrounding industrial development areas. With respect to the potential recycled water new end uses, as the household water use is the second largest user of water in Australia and approximately 20% of overall Australian household water usage is in the laundry, significant fresh water savings could be achieved if potable-quality water used for clothes washing is replaced by recycled water.

Recognizing the considerable potential benefits, several studies on the use of recycled water for washing machines have been carried out. By reviewing eight original American studies on the willingness of people to adopt certain usage forms of recycled water, Dolničar and Saunders (2005) found the average willingness on using recycled water for a household laundry was 80%. However, in Sydney, Pham et al. (2011) performed similar research surveys on public attitudes towards recycled water and showed that only 60% of the respondents supported this new end use of recycled water. Their research also indicated that the large family with a big washing machine and frequent use would give more support on using recycled water. Ngo et al. (2009) indicated that major concerns by the public over this end use are public health, water clarity (i.e., discoloration, smell and hygiene), cost and machine durability. Additionally, O'Toole et al. (2009) investigated the degree to which pathogens could be transferred from recycled water to hands, sample fabrics, nearby surfaces and the air during a typical household laundry cycle. They concluded that the Class A recycled water, as used in Sydney's current recycled water schemes, was unlikely to pose a health threat when given the inadvertent ingestion volume of 0.01 mL.

However, these studies only focused on a single aspect (e.g., public attitudes, public concerns or health risks) of the new end use. Although a descriptive feasibility study of recycled water use for washing machines in terms of strengths, weaknesses, opportunities and threats has been conducted (Mainali et al., 2011b), the results were somewhat unconvincing due to the lack of quantitative analyses. Taking into account the long-term sustainable development of this new end use, there is a need for a more formalized approach in selecting optimal decision-making solutions for particular recycling schemes (Chen et al., 2012b).

5.2.3 Industrial uses

Compared with irrigation and residential uses, industrial water consumption is relatively small in Sydney, accounting for 12% of the total water demand (Sydney Water, 2011b). Nevertheless, due to the imposition of water restrictions during drought conditions but constant high water demand in specific industrial sectors, many companies attempt to enhance the water use efficiency by introducing advanced facilities along with new recycled water schemes. For instance, SWC's STPs deliver up to 4 and 20 ML/d of recycled water to the Eraring Power Station and BlueScope Steel, respectively, where high-quality recycled water treated by Microfiltration (MF), Reverse Osmosis (RO), UV and demineralization processes are utilized as cooling or boiler feed make-up water (Anderson, 2003a). Besides, BlueScope Steel has conducted interdepartmental water reuse schemes (i.e., wastewater from one sector is reused in another sector) and installed an onsite treatment plant to provide secondary treated recycled water (0.3 ML/d) for internal quench basins (Hird, 2006). Similarly, the Port Kembla Coal Terminal receives recycled water from the Wollongong STP and has been using it for dust suppression since 2009, reducing the fresh water consumption by 70%. Moreover, a new technology using filtration, de-ionisation and UV to process wastewater from the electroplating industry has been introduced at the Astor Metal Finishes in Villawood. It is able to recover most of the wastewater and has become a pioneer in wastewater recycling technology in this industry in Australia.

Apart from existing industrial recycling schemes in mining, refinery, fibre cement, commercial laundry and food processing industries, many more recycled water projects are being constructed or under consideration. Particularly, the Rose Hill-Camellia Recycled Water Scheme, the first one owned by the private sector in NSW, is expected to deliver 4.7 GL/yr of recycled water to six major industrial customers in western Sydney and will become one of Sydney's largest industrial recycling projects (NSWOW, 2010; Sydney Water, 2011b). It is worth noting that stringent water quality guidelines should be applied in line with industrial production requirements and staff safety considerations.

5.2.4 Environmental uses

SWC operates 17 inland STPs that discharge recycled water into the Hawkesbury-Nepean River System where water supply dams and weirs have been built in the upper catchment. To release reliable environmental flows and protect the health of the downstream river, these STPs have been upgraded to advanced tertiary standards since 2004. For example, the new St Marys Water Recycling Plant in the west of city is now in operation as the first of its kind in the world. Tertiary treated wastewaters from the

Penrith, St Marys and Quakers Hill STPs are transferred to this plant and undergo additional UF, RO, decarbonation and disinfection processes. The recycled water is released to the Hawkesbury-Nepean River, providing 18 GL/yr of water for environmental flow regulation. This represents a very large saving of freshwater as this flow would otherwise have been provided by freshwater from the Warragamba Dam.

Until now, due to the high water quality requirement and limited exposure to the public, most of the environmental-related schemes have been successfully implemented and neither adverse environmental impacts nor human health problems have been identified. Although IPR schemes have not been pursued in Sydney, the incidental IPR does occur since major water supply sources – the Warragamba Dam and the Nepean River periodically receive effluents from the Goulburn and Penrith STPs, respectively. Despite the fact that there is a significant dilution of the treated wastewater with the catchment source water which lowers risk profiles in most situations, IPR is a relatively recent topic because the initial potable water recycling plant in Quaker's Hill, north-west of Sydney, was put aside during the 1990s owing to public misgiving (Stenekes et al., 2006).

5.3 MANAGEMENT ALTERNATIVES AND EVALUATION CRITERIA

While the water recycling targets are projected to be more aggressive, several constraints (e.g., water quality, environmental impacts, social or community attitudes, economic, and site-specific conditions) may hamper the progress of the potential water reuse market. Currently, systematic analyses on these issues have already been conducted in some states of Australia, including Victoria, Australian Capital Territory, Queensland and South Australia, but that is not the case in NSW. Consequently, based on the comprehensive assessment framework and MCA methodology presented in Chapter 3 as well as the current recycled water use situations in residential areas of Sydney, this section describes the specific MCA decision-making processes, including the discussion of possible management alternatives and the selection of fit-for-purpose evaluation criteria for existing or future recycling projects.

5.3.1 Possible recycled water end use options

In this study, three possible recycled water management options have been proposed. While the first two options (i.e., recycled water for a household laundry and swimming pools) have been discussed before in Chapter 4, the Level 1 water restriction has also to be considered as a prospective approach since 15% of the recycled water sold in Sydney is actually clean drinking water due to STP failure and maintenance issues. The Level 1 water restriction includes no use of sprinklers or other watering systems (excluding drip irrigation) as well as no hosing of hard surfaces and vehicles at any time.

Table 5.1 gives the general descriptions and characteristics of these three management options. Regarding the assessment criteria, the technical aspects including the water supply and operability, risk related issues, environmental and social considerations and cost benefit aspects can be taken into account for detailed analyses. Each main index category consists of several sub indexes.

Table 5.1 Descriptions of three possible recycled water management options

Options	1) RW for a household laundry	2) RW for swimming pools	3) Level 1 water restriction on the use of RW
Water quality requirement	Current RW can be safely used without any quality improvement	Additional advanced treatment (e.g., membrane technology) is required to recycle backwash water	Current RW can be safely used without any quality improvement
Water quantity	Laundry accounts for 20% of total household water use	36% of freshwater consumed in swimming pools can be saved if RW is used as pool make-up water	Level 1 restriction can result in 12% reduction of water demand
Risk	The risk is even lower than RW used for toilet flushing because of less exposure	Although the exposure to RW is high, the improved quality can sufficiently reduce the risk	Reduced RW consumption can lessen its exposure to human and the environment to some extent
Environmental considerations	Reduced effluent discharge and freshwater use	Reduced effluent discharge and freshwater use	High water use efficiency and low ecological footprint
Community attitudes	60% of respondents in Sydney agree with this option	13% of 116 householders agree with this option	Public acceptability is low as the frequencies of washing hard surfaces and using sprinklers are high
Costs and benefits	Need to add extra taps for RW connection	Need to add extra taps for RW connection. Require additional costs on water quality improvement	Neither additional taps nor costs on water quality improvement are required

Note: Modified from Spaninks, (2000); Cooper, (2003); Pham et al. (2011); Sydney Water, (2011b).
Abbreviations: RW= Recycled water.

5.3.2 Recycled water use in a household laundry

To analyse further the new recycled water end use for a household laundry, more specific management options include the following:

5.3.2.1 Baseline scenario (do nothing scenario)

This scenario identifies the "business as usual" projection. In this scenario, the recycled water in the existing scheme has been supplied for toilet flushing, garden watering and car washing rather than a laundry.

5.3.2.2 Recycled water for existing washing machines

In Australia, the existing installed washing machines in households include top loading machines (80%) and front loading machines (20%) with a number of different characteristics such as loading type and capacity, water and energy consumption per wash, brand and model name (Fisher and Paykel Australia, 2005). If recycled water is used in a laundry, there will be three taps suitable for a washing machine connection: a potable hot water tap, a potable cold water tap and a recycled water tap. Currently, the charge for adding an additional recycled water tap by SWC is $215 (Sydney Water, 2012). Generally, the average volume of water used per wash is approximately 153 litres (L) for top loaders and 78.5 L for front loaders (Gato-Trinidad et al., 2011). The average load of laundry is 4.5 times per household per week (6.4 and 2.0 during summer and winter respectively). Because of the residential makeup differences in different households, the water consumption for clothes washing varies between 23 litres per person per day (L/p/d) (large family with more than four people) and 45.1 L/p/d (single person households). Although the higher occupancy can lead to lower total per capita water use, the higher recycled water quality might be required for big families associated with the addition of more young children (Willis et al., 2013).

In regard to the environmental benefits, the use of recycled water for clothes washing reduces the total daily potable water demand as well as the morning drinking water peak use demand. This can be a significant saving on the size of drinking water supply infrastructure for peak demands (Willis et al., 2011). Additionally, the reduced effluent discharge can also alleviate the waste loadings on the environment, safeguard biodiversity of the lagoons and support sustainable tourism.

On the other hand, considering the recycled water quality, although the use of Class A recycled water which undergoes tertiary treatment such as MF and UV disinfection has been shown to be safe to human health and the environment (O'Toole et al., 2009), to improve further recycled water reliability and public acceptance by elderly people and by households with younger children, more advanced techniques need to be discussed. By using an MF-Granular Activated Carbon (GAC) filter system rather than an MF alone as tertiary treatment, Kim et al. (2009) demonstrated better removal efficiency of synthetic organic chemicals and natural organic matters that cause taste, odour and colour.

Alternatively, a post GAC filtration contactor could be installed after the MF. The addition of GAC in wastewater treatment led to a longer contact time (5 to 20 minutes) and introduced additional installation and usage fees. Besides, the MF-RO treatment system could be an advanced approach to produce recycled water of potable water quality. However, the life cycle cost of RO was relatively high (\$0.28/m^3) due to high energy consumption which was around 1.1 Kilowatt-hour per cubic meter (kWh/m^3) compared with 0.23 kWh/m^3 of MF (Côté et al., 2005). In this chapter, three different recycled water treatment techniques (MF, MF-GAC and MF-RO) coupled with existing washing machines were compared thoroughly in the case study section (Table 4).

5.3.2.3 Recycled water coupled with new washing machines

Normally, the estimated lifespan of a washing machine is approximately 11 years, and the existing washing machine in each household laundry will be replaced by more efficient new washers (Demesne, 2012). Several studies indicated that the front loading machines (4-5 star rating) generally use less water, less energy and they require only half the detergent that top loading machines (1-3 star rating) use (Sustainability Victoria, 2009; Bansal et al., 2011; Gato-Trinidad et al., 2011).

Table 5.2 shows the potential environmental savings by adopting different star rating of new washing machines (WELS, 2012). Research on the “Choice” website showed that the average price of a front loading washing machine was \$1500 (ACA, 2008).

Table 5.2 Washing machine efficiency comparisons

Description	Washing machine efficiency		
Star rating	3 stars	4 stars	5 stars
Water savings (L/load)	50%	65%	75%
Energy saving (kWh/load)	50%	65%	75%
Greenhouse gas (GHG) savings (kg CO_2/ load)	3.31	4.29	4.94

Note: Adapted from WELS, (2012).

Apart from improved machine efficiency, many modern washers are now equipped with sensors and microprocessors that can detect the size, type, and soil level of a load of clothes. This allows the washers to match precisely the wash requirements such as the amount of water and detergent, wash temperature, spin speed and rinsing for different clothes loads. Such advancements in washing machines can lead to a further 20% reduction in electricity and water use.

Moreover, other technologies (e.g., turbidity sensors, bubble-action designs, suds saver and delay-start function) prove to be energy savers as well. However, the long-term reliability is yet to be determined (Olsson et al., 2008). In spite of these strengths, some residents might be reluctant to adopt a new, more efficient washing machine due to the high initial cost, inconvenience of loading and unloading clothes, inability to add clothes once underway and longer cycle times (Bansal et al., 2011). In this case, public education or awareness campaigns should be launched more widely and frequently. Nevertheless, in this study, taking into account the cost issues, only the less costly water treatment approach (e.g., MF) coupled with a new washing machine was considered in the case study section.

With regard to the evaluation criteria by which management alternatives on recycled water for a household laundry will be judged, it is necessary to include each of the relevant technical, environmental, social and economic aspects of alternatives appropriately in the decision making procedure (Table 5.3).

Table 5.3 Descriptions of evaluation criteria

Index	Descriptions
Technical index	
Recycled water supply (I_{R1})	This refers to increased water supply caused by additional demand for recycled water for washing machine
Recycled water operability (I_{R2})	This refers to the ease of operation and system flexibility
Environmental index	
Recycled Water Quality (I_{R3})	This index is closely related to risks on environmental and human health
Water savings (I_{R4})	This refers to freshwater savings via recycled water use
Energy consumption (I_{R5})	This refers to electricity usage on the washing machine and water treatment
GHG emissions (I_{R6})	This relates to energy use on the washing machine and water treatment
Ecology (I_{R7})	The reduced wastewater discharge is beneficial to the local ecosystem, surface water and groundwater
Social index	
Community acceptance (I_{R8})	This relies on public surveys of household residents
Political support (I_{R9})	This refers to the rebate and subsidy from SWC
Education opportunities (I_{R10})	This refers to the community's knowledge on recycled water
Economic index	
Capital Cost (I_{R11})	This reflects the initial investments on tap installation, new washing machines, new treatment facilities, etc.
Operational Cost (I_{R12})	This reflects the investments on treatment unit operation and washing machine maintenance

5.4 CASE STUDY

The proposed MCA decision-making processes were applied to a case study on the Rouse Hill Water Recycling Scheme in the Rouse Hill Development Area (RHDA), Sydney. The recycling scheme started in 2001 which utilizes up to 2.2 GL/yr of recycled water from the Rouse Hill STP for toilet flushing and outdoor irrigation uses, serving over 20,000 homes. Before being pumped to the dual reticular system, the tertiary treated effluent is further treated by chlorination, ozonation and MF and stored in the service reservoirs. A complete explanation of the treatment processes can be found in Cooper (2003). The recycled water (Class A quality) meets the "NSW Guidelines for Urban and Residential Use of Reclaimed Water" and it can be used freely without water restrictions. The project capacity will be doubled to benefit an additional 16,000 households (NSWOW, 2010).

However, the current end uses of recycled water are limited as most households only use the recycled water for toilet flushing, garden watering and car washing (Cooper, 2003). Moreover, because of no restrictions on the recycled water use, households in RHDA use significantly higher amounts of water than households without the connection to the dual reticulation system. Hence, the recycled water use efficiency in RHDA has been kept low (O'Toole et al., 2008). Regarding the recycled water treatment technology, despite the capability of producing high quality effluent, the existing continuous MF method performed relatively poorly in terms of most environmental impact categories due to the high levels of energy and chemical consumption (Tangsubkul et al., 2005b). These constraints are likely to limit the long-term sustainability of RHDA. Since the capacity of this scheme has already been greatly expanded, it is necessary to further develop new end uses as well as improve water use efficiency.

5.4.1 Analysis of three possible recycled water end use options

This study conducted a simplified quantitative MCA using the Multi-attribute Utility Theory (MAUT) technique to investigate three possible recycled water end use management strategies listed in Table 5.1. Due to a lack of data, all criteria are assessed using the scoring system described in Table 3.1 of Chapter 3.

As can be seen in Table 5.4, higher scores are generally assigned to the options with positive impacts whereas negative scores are associated with adverse impacts. For instance, as all options have positive effects on the environment via reduced effluent discharge and/or lower ecological footprint, positive scores were given to them. Yet the varying degree of environmental impacts makes the scores of these three options towards environmental-related sub-criteria slightly different.

Table 5.4 Summary of key and sub-criteria and weightings

Key criteria	Primary weighting (%)	Sub-criteria	Sub weighting (%)	Scores of options		
				1	2	3
Water supply	20%	Water quantity and security of supply	50%	+4	+3	+2
		Water quality	50%	+3	+4	+3
Risk related issues	15%	Treatment technology	50%	+3	+4	+3
		Reliability, robustness and safety	50%	+3	+4	+3
Operability	10%	Ease of operation	55%	0	–1	0
		System flexibility to upgrade and extend	45%	0	+1	0
Environmental considerations	25%	Volume of waste generated	20%	+2	+1	+1
		Footprint of plant and infrastructure	20%	+1	+1	+2
		Energy use	15%	+1	+1	+2
		Greenhouse gas emission	15%	+2	+1	+1
		Impact on local ecology	20%	+1	+1	+1
		Impact on groundwater	10%	+1	+1	0
Social considerations	15%	Aboriginal, cultural and non-cultural heritage	15%	0	0	0
		Aesthetics	20%	0	+1	0
		Traffic disruption	20%	0	–1	0
		Community/social acceptance	25%	+3	+2	–2
		Community education opportunities	20%	+4	+3	+1
Costs and benefits	15%	Capital cost	50%	–1	–2	0
		Operating cost	50%	–2	–1	0
Total	100%			1.50	1.48	1.22

Additionally, as for primary weighting, since the prime objective of the scheme is to reduce the nutrient loadings on the Hawkesbury-Nepean River system caused by the discharge of treated wastewater, environmental performance has been assigned the highest weight. Other weighting values are based on similar case studies undertaken in the state of Victoria (Coutts, 2006; Muthukaruppan et al., 2011) as well as a survey on the water reuse research priorities in Australia (Dillon, 2000). According to the obtained results, as all three options generated positive scores, they are of great value and can contribute positively to regional sustainability development. The recycled water for a household laundry has been identified as the preferred option that best satisfied the overall criteria. To further validate the results, one-dimensional sensitivity analysis on environmental aspects was performed, given other weights of the remaining criteria were held constant.

As shown in Figure 5.1, the sensitivity analysis indicated that option 1 (recycled water for a household laundry) was an optimum when the weight on environmental considerations was relatively high (greater than 22%) whereas option 2 (recycled water for swimming pools) can be the superior choice if environmental impact was not the major concern. Option 3 (implementation of Level 1 water restriction on the use of recycled water) was the least preferred alternative for any weight combination. Since the environmental concerns on the river system are prime objectives in reality, it is concluded that from a holistic point of view, option 1 is the recommended option. In respect of implementation, more extensive support in terms of organization, decision making and communication as well as external interfaces and consultations are also required.

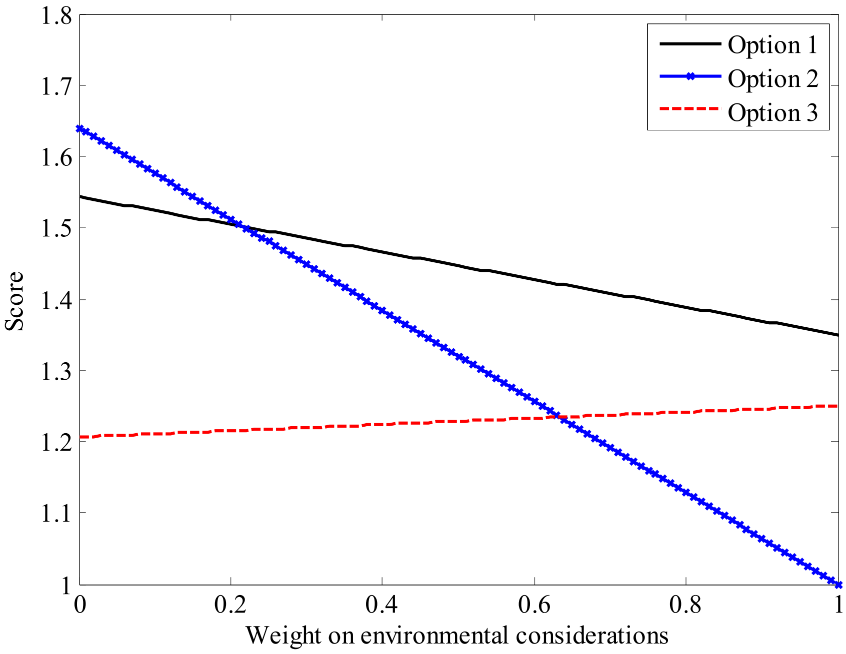

Figure 5.1 A sensitivity analysis of the three management options

5.4.2 Analysis of five management alternatives for recycled water use in household laundry

Having identified the prospects of using recycled water in a household laundry by the MAUT approach for simplified MCA, this study further discussed five different management alternatives on this new end use using quantitative PROMETHEE technique in MCA. Some important strengths and knowledge gaps (e.g., criteria importance and model integration) were recognized, which may need to be addressed by future researchers.

5.4.2.1 Quantification of management alternatives

Table 5.5 Performances of the management options on each criterion

Options Criteria		(1) Baseline	(2) MF+Old[1]	(3) MF+New[2]	(4) MF-GAC+Old[1]	(5) MF-RO+Old[1]
I_{R1} (GL/yr)		2.2	2.9	2.35	2.9	2.9
I_{R2} (5-point)		5	4	3	4	2
I_{R3}	BOD (mg/L)	2	2	2	< 2	< 2
	TSS (mg/L)	4	4	4	2	<0.01
	TN (mg/L)	7.6	7.6	7.6	7.0	5.5
	TP (mg/L)	0.18	0.18	0.18	0.17	0.03
	FC (cfu/100 mL)	<1	<1	<1	<1	<0.1
	Turbidity (NTU)	0.01	0.01	0.01	<0.01	<0.01
	Colour (NTU)	4	4	4	<1	<1
I_{R4} (GL/yr)		0	0.7	0.7	0.7	0.7
I_{R5} (10^3 kWh/d)		7.39[3]	7.39	3.22	7.39	16.1
I_{R6} (10^3 kg CO_2)		7.77	7.77	3.38	7.77	16.9
I_{R7} (5-point)		1	4	2	4	4
I_{R8} (percentage values)		1	0.6	0.5	0.6	0.7
I_{R9} (5-point)		1	3	3	3	2
I_{R10} (5-point)		1	4	5	4	4
I_{R11} (10^3 \$)		0	4,300	34,300	4,300	5,800
I_{R12} (10^3 \$)		198	661	212	661	1,473

[1] Existing washing machines; [2] New washing machines; [3] The energy use for drinking water production is from MF.

Abbreviations: MF= Microfiltration; GAC= Granular activated carbon; RO =Reverse osmosis; BOD= Biochemical oxygen demand; TSS= Total Suspended solids; TN= Total nitrogen; TP= Total phosphorous; FC= Faecal coliforms.

Based on the selected criteria in Table 5.3, five management alternatives regarding the new recycled water end use for a household laundry are considered in RHDA (Table 5.5). The detailed quantification processes on performance scores towards four types of indexes are shown as follows:

(1) Technical performances

According to the Australian average washing machine loading characteristics (loading type, capacity and time), the recycled water demand is estimated to be 36 L/p/d (assuming 2.5 people in each household). Hence, the recycled water supply (I_{R1}) in STP should be increased to 2.9 GL/yr (for existing machines). As the recycled water operability (I_{R2}) is difficult to be quantified in pre-commissioning stages, qualitative scores from 1 to 5 points are employed. Generally, the more complex the treatment system and the washer are, the lower the I_{R1} score will be.

(2) Environmental performances

Considering the recycled water quality (I_{R3}), the recycled water performances by different treatment processes (Table 5.6) are estimated on the basis of former studies (Cooper, 2003; Kim et al., 2009). For simplicity, all newly purchased machines are assumed to be 5-star efficient ones, which can save 28 L/p/d of recycled water, indicating a reduced I_{R1} of 2.35 GL/yr. With regard to energy consumption (I_{R5}), the existing and new washing machines in RHDA consume 5.56×10^3 and 1.39×10^3 Kilowatt-hour per day (kWh/d) respectively, based on an assumption of 0.107 kilowatt-hour per load of a new washer. Since the operation of RO consumes 8.74×10^3 kWh/d of electricity compared with 1.83×10^3 kWh/d of MF, the GHG emissions (I_{R6}) can be estimated by an emission factor of 1.051 kg CO_2-e/kWh. The ecological impact (I_{R7}) can be measured by a qualitative approach and high scores are associated with greater effluent quality and a lesser amount of wastewater discharge.

Table 5.6 Characteristics of recycled water quality produced by different treatment processes

Parameter	NSW guideline	MF	MF-GAC	MF-RO
BOD (mg/L)	5	2	< 2	< 2
SS (mg/L)	8	4	2	<0.01
Total nitrogen, TN (mg/L)	15	7.6	7.0	5.5
Total phosphorus, TP (mg/l)	0.4	0.18	0.17	0.03
Faecal coliforms (cfu/100 mL)	<1	<1	<1	<0.1
Turbidity (NTU)	<2	0.01	<0.01	<0.01
Colour (NTU)	<15	4	<1	<1

Note: Modified from Cooper, (2003); Kim et al., (2009).

(3) Social performances

The community acceptance (I_{R8}) of recycled water for washing machines may be increased from 60% to approximately 70% by adopting more advanced treatment technologies in water quality improvement. However, 44% of residents still showed their unwillingness to use new water efficient washing machines (Dolnicar and Hurlimann, 2010). With respect to the political support (I_{R9}), the charge of recycled water usage is only set at 80% of the drinking water price currently. This proportion will increase up to 85% with the application of RO. Besides, the education opportunities (I_{R10}) are qualitatively measured and the wide use of recycled water as well as new washing machines in households will provide people with more opportunities to understand water and energy conservation.

(4) Economic performances

The capital cost of RO is estimated to be $1,500,000. The total installation fee of recycled water taps for washing machines is $4,300,000 and the total new washing machine purchasing fee is $30,000,000. As for the operational cost (I_{R12}), the annual maintenance fees for existing and new washing machines are assumed to be $20 per washer ($400,000 in RHDA) and 0 respectively and the operational fees of RO and MF are \$0.28/m^3 and \$0.09/m^3 respectively (Côté et al., 2005). The installation and usage fees of GAC are ignored due to relatively low costs.

5.4.2.2 Elicitation of weights

As indicated by Ngo et al. (2009), residents' concerns on using recycled water for a washing machine are mostly on microbial concentrations, hygiene and smell, a fixed group of weights for sub-criteria in I_{R3} has been set as [0.3, 0.2, 0.2, 0.1,…] for FC, turbidity, colour and TSS,…, respectively. The overall water quality performances of the alternatives are then calculated by transformed criteria values from PROMETHEE concave functions multiplied by corresponding weight values.

Figure 5.2 The designed user menu interface using MATLAB

Due to a lack of knowledge about the key criteria, the ROWG has been designed in a MATLAB programme to narrow the field of competing options by evaluating 10,000 combinations of weights via computerized simulation with a given precedence order in the criteria. As can be seen in Figure 5.2, after attaining an importance order of the criteria from the designed menu interface by users, the ROWG is able to assign 10,000 sets of weight values (from largest to smallest) to corresponding criteria values (from most important to least important ones). The full ROWG-PROMETHEE calculation programme is attached in Appendix A.

5.4.2.3 PROMETHEE MCA results

The PROMETHEE MCA calculations were also performed by computerized MATLAB programming technique in this study. Firstly, the key criteria were separated into four groups where different forms of preference functions, and indifference (q) and strict preference (p) thresholds were applied in criteria value transformation. The q and p values for all qualitative criteria were set as 1 and 2.5 respectively, whereas p values for other quantitative criteria were generally similar to the largest distances of a criterion's maximum and minimum values. Reversed preference functions were adopted for criteria such as recycled water demand, energy consumption, GHG emissions and costs, where the smaller utility scores indicate better performances.

Secondly, by incorporating preference function scores with weight values using Equation 5.6, the overall outranking scores of each alternative can be determined. Moreover, as 10,000 sets of weight combinations were employed, the statistical values of optimum options can be finally obtained by finding maximum outranking flows in each weight set. To further illustrate the calculation process, a hypothetical precedence order (criterion I_{R3}> I_{R12}> I_{R6}> I_{R7}> I_{R1}> I_{R4}> I_{R5}> I_{R9}> I_{R8}> I_{R2}> I_{R11}> I_{R10}) was given. The selected order was based on the survey results from Ngo et al. (2009) regarding the residents' most concerned issues.

Figure 5.3 depicts the simulation results, in which the x and y axes represent the weights for the most important and the second most important criterion respectively. Through 10,000 statistics, the option 3 (7414 counts) best satisfies the overall criteria, while the option 4 (2489 counts) and the option 5 (94 counts) can be superior if the weight on recycled water quality is relatively high. However, the options 1 and 2 are the least preferred ones for any weight combination. Consequently, under the current criteria importance order, the least preferred options can be eliminated and the superior options (i.e., MF treated recycled water + new washing machines and MF-GAC treated recycled water + existing washing machines) should be intensively investigated. While this simulation addressed the importance of recycled water quality and operational cost in overall evaluated criteria, people with different knowledge on criteria importance might come up with different management solutions.

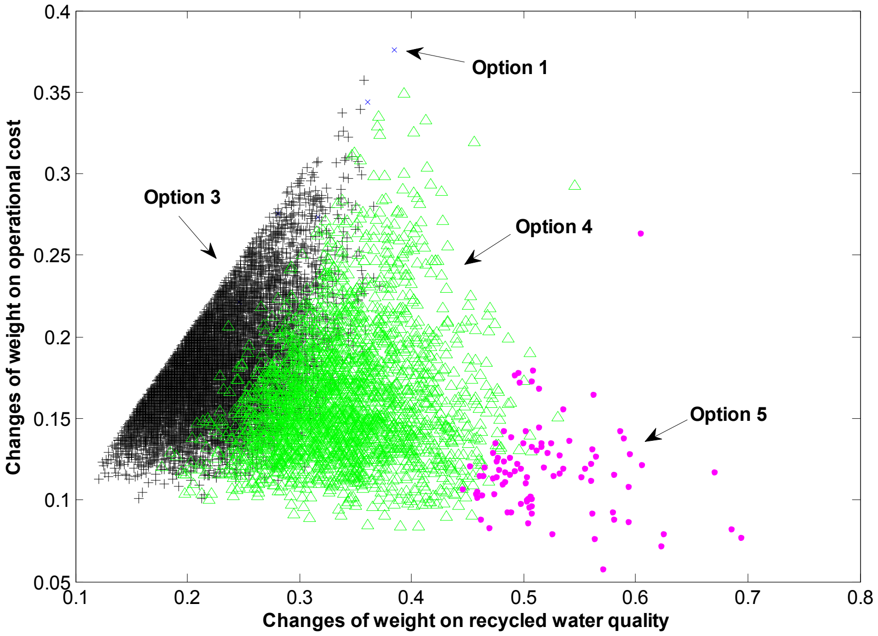

Figure 5.3 Most preferred alternative at all possible weight combinations

5.4.3 Possible improvements on MCA decision making in future research

The following issues might need to be further discussed to modify and improve the MCA analyses in future research:

- Avoid leaving out any key criteria. In the field trials and full application, the proposed criteria in this study should be re-evaluated and the surrounding environmental conditions (e.g., current STP, the nearby catchment and climate conditions) also need to be considered.

- Check for independence of criteria. In some cases, as several sub-indexes are partially correlated, some effects are likely to be double counted indirectly and assigned higher weight than they deserve. For instance, although water quality has been considered in the category of water supply, it is likely to be double counted indirectly in technical criteria. To avoid double counting effects, it is advisable to conduct independence checking in the selection process.

- Measure changes in fundamentals before pricing the recycled water. When the price is set competitively low, greater economic benefits and public acceptability can be attained with the increase of recycled water consumption. However, too low a price would force the authorities to take further actions regarding subsidies and external funding to recover the capital and maintenance costs associated with wastewater treatment works and infrastructure constructions, exacerbating governmental financial burdens.

- The one-dimensional sensitivity analysis might ignore the potential interactions resulted from simultaneous manipulations of multiple weights, which calls for a multi-dimensional sensitivity analysis with high reliability and desirability in comprehensive analyses.

- With respect to the implementation of management solution(s), the comprehensive and holistic MCA needs more consultation, continuous communication among stakeholders, decision makers and community members, public participation and computerized simulation to minimize man-made errors.

5.5 CONCLUSIONS

Recycled water provides a viable opportunity to partially supplement fresh water supplies and substantially alleviate environmental loads. Currently, a large number of recycled water schemes have been successfully conducted in a number of countries and Sydney is one of the leading cities, where considerable effort has been made in applying water reclamation, recycling and reuse. Despite the increasing implementation of recycled water schemes in Sydney for non-potable uses (e.g., agricultural and landscape irrigation, residential, industrial and the environmental uses), the lack of a comprehensive quantitative assessment on new end use development is still a large constraint for long-term sustainability of the recycling project.

Based on the assessment framework and MCA methodology proposed in Chapter 3, this chapter illustrated detailed evaluation processes of the possible recycled water new end uses using the MAUT and ROWG-PROMETHEE algorithms. Based on the case study conducted at RHDA, Sydney, the simplified MCA with MAUT approach indicated that recycled water for a household laundry was the optimum solution which best satisfied the overall evaluation criteria compared with another two management options, namely the recycled water for swimming pools and the implementation of Level 1 water restriction on the use of recycled water.

With the identified strengths of recycled water use in washing machines, five relevant management alternatives: (1) do nothing scenario; (2) MF + existing washing machines; (3) MF + new washing machines; (4) MF-GAC + existing washing machines; and (5) MF-RO + existing washing machines, were investigated specifically. According to the integrated ROWG-PROMETHEE model, the MCA study showed that the MF treated recycled water coupled with new washing machines and the MF-GAC treated recycled water coupled with existing washing machines were preferred options. From the analytical results, good prospects of further expansion, exploitation and development of current and new recycled water end uses can be recognised as an integral part of water planning and management.

With more quantitative assessment data obtained from future onsite investigations, the next step is to perform social surveys, set up experimental equipment and conduct field trials with actual recycled water in washing machines. In this case, more detailed MCA simulations can be performed and the computerized model can be constantly validated and modified to achieve more realistic and reliable outcomes. While the case study focused on one residential recycling scheme in Sydney, the assessment methodology can also help decision makers in making sound judgements on other new end uses in existing or future recycling projects across Australia.

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CHAPTER 6

Analysis of Social Attitude for the New End Uses in Recycled Water Schemes

6.1 INTRODUCTION

Recycled water for a household laundry can be regarded as a promising strategy to alleviate the current demand on scarce water supplies. Public acceptability becomes fairly important to ensure the successful establishment and development of this new end use. To address the issue, this chapter describes the social surveys conducted in three locations of Australia, namely Port Macquarie, Melbourne and Sydney, where respondents were asked 17-18 questions. The statistical analyses provided information regarding the household laundry behaviour, respondents' attitude or knowledge and psychological concerns about recycled water exploitation and consumption. The regression models offered conclusions about which characteristics would be more likely to contribute to the acceptance of recycled water by the society. These findings could drive future research to achieve a better public perception of the new end uses of recycled water.

6.2 RESEARCH BACKGROUND

6.2.1 Current recycled water use situations in Australia

Faced with a fast-growing population and increasing water demands as well as a highly variable climate and serious precipitation imbalances, many locations in Australia have experienced water shortage problems. In the last decades, due to prolonged drought conditions, all Australian capital cities, except Darwin and Hobart, have imposed water restrictions to curtail water use and protect supplies (Ryan et al., 2009; Chen et al., 2012a). Despite the recent flooding rains in eastern Australia, current water consumption practices are widely recognized as unsustainable (Hurlimann and Dolnicar, 2012). These issues have highlighted the importance of exploiting all other possible sources of water before using up limited freshwater supplies. As a consequence, the recycled water that originates from wastewater treatment is increasingly being considered as a realistic option for supplementary water supply. This can help to alleviate the pressure on existing water supplies, protect water bodies from being polluted and on the other hand, provide a more constant volume of water than rainfall-dependent sources (Dolnicar and Schäfer, 2009; Chen et al., 2013c).

There are over 580 different recycled water schemes across Australia at present, which are mostly associated with non-potable uses (e.g., irrigation, industry, residential uses and environmental flows). Particularly, the household use of recycled water has continued to increase, with a 6% growth from 3,106 Megalitres (ML) in 2009-10 to 3,283 ML in 2010-11. However, the amount of treated effluent being reused in Australia (351 Gigalitres (GL) in 2010-11) is still low compared with the total discharged wastewater and there is a considerable inconsistency in water management across local, state and territory governments (ABS, 2012). The NSW Government has set the goal of increasing the water recycling rate from 9.8% in 2009-10 to 14.7% by 2015. However, in Melbourne, Victoria, the recycling rate had already reached 22.8% in 2009-10. Since a number of dual pipe systems have been built in newly developed residential areas, it is anticipated that 40,000 new homes and businesses in Melbourne will use an additional amount of recycled water, which will bring the water recycling to 26.1% by 2015 (Whiteoak et al., 2012). To meet these aggressive recycling targets in the near future, many more recycled water schemes as well as new end uses should be increasingly explored and developed.

As the household water use is the second largest use of water in Australia and approximately 20% of overall Australian household water usage is in the laundry, significant fresh water savings could be achieved if potable-quality water used for clothes washing is replaced with recycled water (Chen et al., 2012a). However, to achieve the goal of saving fresh water and increasing recycled water usage, apart from technical concerns of producing high quality water reliably and economically, research into community attitudes is of great importance. As the Australian community had very little experience of utilising recycled water for uses with relatively high personal contact, some people displayed substantial resistance to several recycling projects. For instance, in Toowoomba, the local community voted against the development of an indirect potable water recycling project in spite of critically low dam levels (Hurlimann and Dolnicar, 2010a). The initial potable water recycling plant in Quaker's Hill, north-west of Sydney, was also put aside owing to public misgiving (McClellan, 1998; Stenekes et al., 2006).

Having recognized the potential great benefits, several social studies and assessments on the use of recycled water have been carried out. Pham et al. (2011) performed research surveys on public attitudes towards recycled water in Sydney and showed that around 60% of the local respondents supported the use of recycled water in a washing machine. Pham et al. (2011) also indicated that the major concerns of the public over this end use are public health, water clarity, cost and machine durability (Chen et al., 2012a). Additionally, Dolnicar et al. (2011) found that the positive environmental attitudes, the positive perceptions of recycled water, the higher influence of other people, more knowledge, experience of water restrictions and watching State TV channels, had increased the stated likelihood of respondents using recycled water (Hurlimann and Dolnicar, 2010b). Nevertheless, what motivates people to participate actively in using recycled water in a laundry is yet to be investigated and understood in detail. Hence, the main purposes of this chapter are to identify further household laundry behaviour, knowledge and attitude about water saving and recycled water use, and the impacts as well as measures that people are concerned about when implementing this new end use. A final regression relationship between these behaviours/concerns and the willingness to accept the use of recycled water in a laundry was proposed. This information, together with other gathered implications, could provide sound suggestions for future research.

6.2.2 Survey locations

The survey in this research was conducted in three places in Australia, namely Port Macquarie, Melbourne and Sydney over a period from early November 2012 to February 2013 by using questionnaires. The need for recycled water in a household laundry and for other uses in these three places stems from the limited water supply and environmental concerns and is encouraged by the city councils and local water suppliers. Figure 6.1 depicts the geographical locations of the study areas.

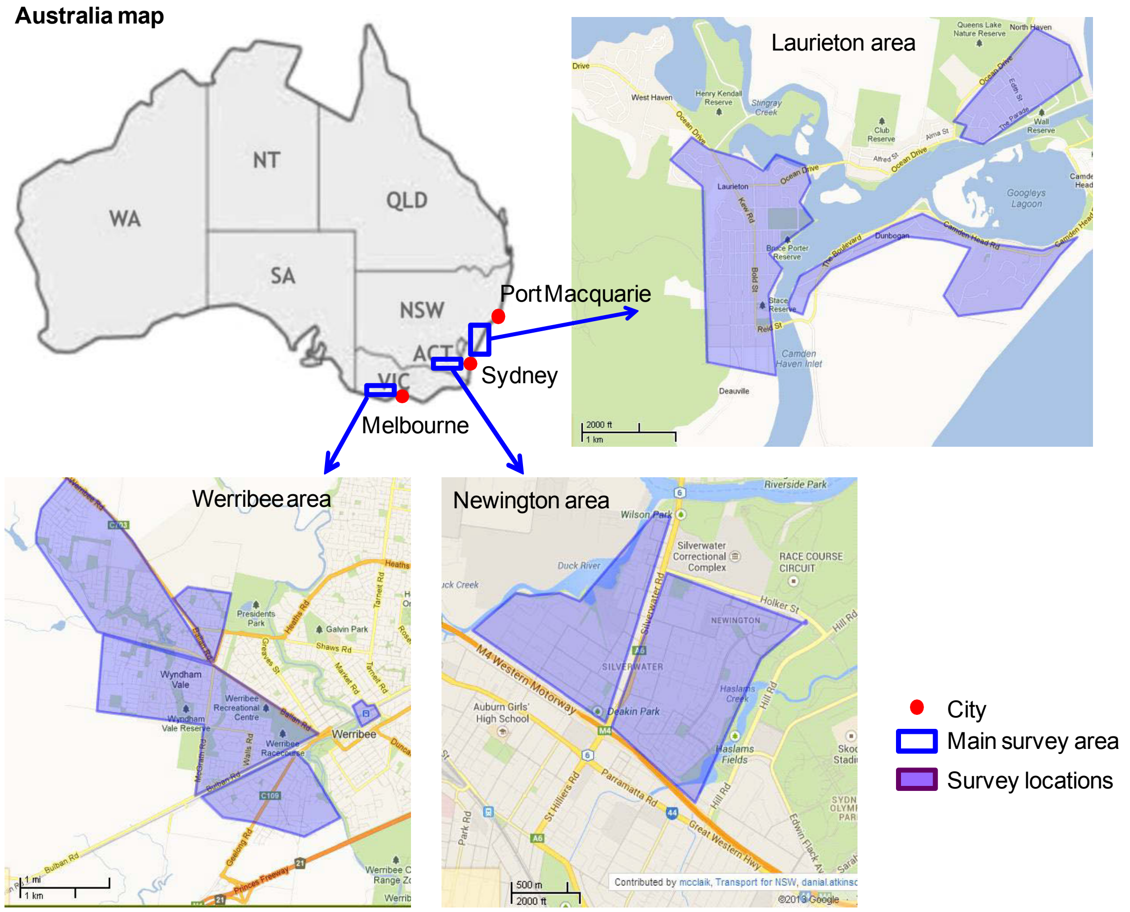

Figure 6.1 Geographical locations of the survey areas in Port Macquarie, Melbourne and Sydney, Australia (Google Map, 2013; The Lost Seed, 2013).

Port Macquarie is a city on the mid north coast of New South Wales (NSW), about 390 km north of Sydney, with a population of 73,000 people. Around 32% of the population was aged 65 years and over in 2011, suggesting a preference of older people for coastal and rural areas after retirement. Overall, 6.9% of this population earned a high income (those earning AU $1,500 per week or more), and 42.7% earned a low income (those earning less than $400 per week), compared with 9.2% and 40.0% respectively for regional NSW. The median individual income was AU $25,000 per year (ABS, 2011b; PMHC, 2013). Currently, only 1.5 Megalitres per day (ML/d) of recycled water from Bonny Hills Sewage Treatment Plant (STP) is used for irrigating Port Macquarie Golf club and horticultural areas. The Port Macquarie-Hastings Council (PMHC) has upgraded the Dunbogan STP to serve 15,000 people in the Camden Haven area since July 2010. The plant employs the Membrane Bioreactor (MBR) treatment to produce an effluent of tertiary treatment quality. The treated effluent is currently discharged to the environment directly rather than reused.

To satisfy any shortfall between water supply and demand in the future, the PMHC is considering the utilization of the Dunbogan STP as a future source for additional recycled water supply, which will then give a combined source capacity of around 2.5 to 3.0 ML/d and up to 5.0 ML/d (PMHC, 2012). Consequently, 181 surveys were performed (151 by interview and 30 by mail) in Port Macquarie, most of which were collected in residential areas near Dunbogan STP (e.g., Laurieton and North Haven) where respondents are likely to be potential customers of recycled water in the near future.

With respect to Melbourne, Victoria, there were an estimated 4.17 million people in 2011 and only 13% of the population was aged 65 years and over. The average annual income in 2008-09 was AU $47,000 with more than 18% of the population earning a high income and 23.1% having a low income (ABS, 2011b). The Western Treatment Plant has been supplying 37 Gigalitres per year (GL/yr) of recycled water for agriculture and landscape irrigation (e.g., golf club, parks, zoos and wetlands) in the Werribee area, a suburb around 32 km southwest of Melbourne's central business district. In the City of Wyndham Vale, Werribee, the dual pipe recycled water systems have already been installed in front/back yards of 2,000 new homes which will be receiving Class A recycled water in December 2013 or early 2014 through the purple pipe. Hence, 152 surveys were conducted (by interview) in the Werribee area, most of which were collected in Wyndham Vale where dual pipe systems have been constructed and recycled water will soon be supplied to new homes (City West Water, 2012).

In terms of Sydney, NSW, the population was 4.63 million in 2011, which was the largest capital city population in Australia. Regarding the age distribution, 31% of people were aged 20- 39 years and less than 18% were aged 65 years and over, indicating a younger age group than that for the rest of NSW. The average annual income in 2010-11 was AU $62,000 (ABS, 2011b). Since 2000, the Water Reclamation and Management Scheme (WRAMS) at the Sydney Olympic Park has been providing continuous recycled water for irrigation, water fountains, domestic and residential uses in Olympic Park and over 2,000 houses in the Newington area. Raw sewage is firstly treated at the water reclamation plant (2.2 ML/d) and then mixed with stormwater to be treated at the water treatment plant (7.5 ML/d), saving more than 800 ML/yr of drinking water (Chapman, 2006).

To examine public attitudes in places where recycled water is being used for multiple household purposes, 151 surveys were conducted (by interview) in Newington area and most of the respondents were current customers of recycled water. The respondents were assured of anonymity. Specifically, to carry out the face-to-face interview, researchers performed door-to-door knocks or stayed at some of the busiest local points (e.g., shopping centres, parks, swimming pools and stations) and randomly selected pedestrians at different hours of the day in the morning and in the afternoon, on a first-to-pass basis (Menegaki et al., 2007).

6.2.3 Questionnaire structure

In Questions 1-5, the participants were asked questions regarding their household laundry behaviour using a numerical rating scale. The questions and response options were: a. Have a washing machine at home: *Yes* (1) *or No* (2); b. Type of washing machine: *Front* (1) *or Top* (2); c. Type of washing detergent: *Powder* (1), *Liquid* (2) or *Mixture of powder and liquid* (3); d. Family size: *1-3 people* (1), *4-6 people* (2), *7-9 people* (3) or *More than 10 people* (4); e. Frequency of doing laundry: *1-2 times/ week* (1), *3-4 times/ week* (2), *5-6 times/ week* (3), or *More than 7 times/ week* (4). In Question 6, the participants' knowledge on recycled water supply was measured by asking "Did you know that your property will be receiving recycled water in the near future". The answer options were: *Yes* (1), *Yes, but was not sure of the date* (2), or *No* (3). Since the residents in the Newington area of Sydney have already received the recycled water supply, this question was removed from the questionnaires conducted in Sydney.

In Questions 7-9, participants were first asked to assess their opinions on "Recycled water is an important alternative to drinking water for non-potable uses" on a four-point scale: *Strongly agree* (1), *Agree* (2), *Disagree* (3), or *Strongly disagree* (4) and then to specify their reasons. Positive respondents were asked to respond to "Why do you agree" by ticking: *Save drinking water* (1), *Provide an alternative to drinking water* (2), *Save money* (3) or *Other* (4) whereas negative respondents were provided with five response options for "Why do you disagree": *Cost too much to produce* (1), *Not clean enough to reuse* (2), *Health reasons* (3), *Desalination plant provides water* (4) or *Other* (5).

In Questions 10-11, the participants were asked to indicate their attitudes to and perceived cost of receiving recycled water at home. The answer options for community attitude to receiving recycled water at the home were: *Very happy* (1), *Quite happy* (2), *Unsure/ don't know* (3), *Not happy* (4) or *Very unhappy* (5). Moreover, the answer options for perceived cost of recycled water compared with drinking water cost were: *Much higher* (1), *Higher* (2), *The same* (3), *Slightly lower* (4), or *Much lower* (5). Next, in Question 12, participants were presented with six options (i.e., flushing toilet, watering garden, washing cars, laundry, filling swimming pool and showering) and were asked "Would you be willing to use recycled water for the following options". These items were answered on a three-point scale: *Yes* (1), *No* (2) or *Unsure* (3).

In Questions 13-14, five possible concerns (i.e., colour, potential damage to clothes, effect on the washing machine, odour and increased cost) and five potential solutions that might add people's confidence (e.g., knowing that recycled water saves drinking water, reading from other customers, recommendations by scientists, having a small-unit for pre-treatment and knowing that recycled water will be supplied soon) were listed respectively. These items were answered on a three-point scale: *Yes* (1), *No* (2) or *Don't know* (3).

In Questions 15-16, participants were asked whether they have received information or updates about the supply of recycled water to their home: *Yes* (1), *No* (2) or *Unsure* (3), and the best method(s) to receive additional information: *Website or email* (1), *Brochures and flyers via mail* (2), *Personal visit by council staff* (3), *Articles or advertisements on newspaper* (4). Finally, they were given the 17th question "Overall, would you support the use of recycled water in a washing machine?" which was assessed on a three-point scale: *Support* (1), *Uncertain* (2), or *Against* (3). Respondents also had the option not to answer a question, which was then marked as *Not Applicable* (0).

Additionally, for residents in the Newington area of Sydney, another two questions have been included regarding the perceived risks and complaints on the quality of current recycled water. The risk perception was answered on a three-point scale: *Yes* (1), *No* (2) or *Don't know* (3). Participants who selected *Yes* (1) were asked to classify the risk into 4 categories: *Danger to children/pets*, *Hygiene*, *Health issues*, and *Cross connection*. Moreover, the specific complaints or concerns were listed for selection: *Colour* (1), *Odour* (2), *Saltiness* (3), *Health issues* (4), *Clearness* (5) or *Cost* (6).

This survey did not address the socio-demographic variables (e.g., gender, age, income and education) as their effects on the public acceptance of recycled water have already been analysed by other papers (Dolničar and Saunders, 2005; Menegaki et al., 2007). Some research showed that people with a higher level of income and education exhibited more willingness to adopt recycled water (Tsagarakis and Georganzís, 2003), while others found that the socio-demographic factors are not good predictors on determining the recycled water acceptance (Marks, 2004; Hurlimann et al., 2008).

Furthermore, to simplify the calculation of the statistical analysis, predictor variables (e.g., behavioural, attitudinal and psychological variables) except the cost variable, were re-coded into a 2-point scale. The positive responses were classified into *Category 1* (0) while the negative and *Unsure/ don't know* responses were included in *Category 2* (1). *Category 2* was regarded as the reference category. The perceived cost proved to play an important role in satisfaction of the recycled water use (Hurlimann et al., 2008). Thus, the cost variable in this study was examined in greater detail and recoded into a 3-point scale: responses 1-2, 3, 4-5 were included in *Category 1* (0), *Category 2* (1) or *Category 3* (2) respectively. In this case, the *Category 3* was considered as the reference category.

Additionally, as the number of *Not Applicable* answers in the survey was small except in Question 13, they were classified into Unsure/ don't know responses. However, the average of *Not Applicable* answers for the psychological variables (i.e., colour, clothes, washing machine and odour) was 7.5% which could not be ignored. In this case, responses with the *Not Applicable* answer(s) in any of these four variables were removed because their correlations with the regression model regressors were insignificant, which led to an exclusion of 10.8% of the respondents.

6.3 GENERAL FEATURES OF LAUNDRY BEHAVIOUR AND WILLINGNESS TO USE RECYCLED WATER

6.3.1 Household laundry behaviours in three survey locations

According to Figure 6.2, the number of respondents using front loading washing machines was lower than using top loaders at all three locations. A higher proportion of households from larger cities (i.e., Sydney and Melbourne) employed the front loaders that are more energy and water efficient compared with households in the small city (i.e., Port Macquarie).

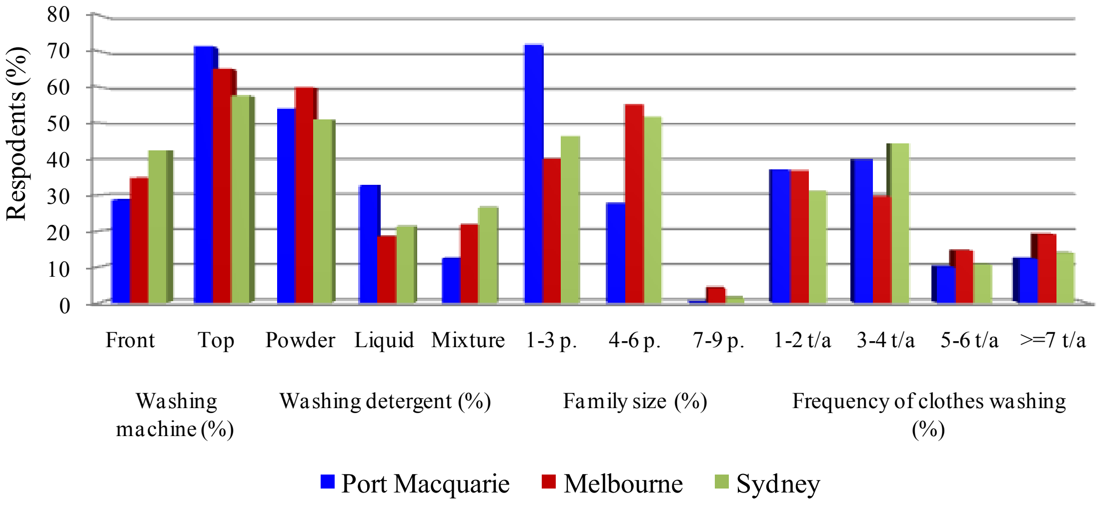

Figure 6.2 Laundry behaviours of respondents in the three survey locations

However, there was a significant difference in survey locations on choosing washing detergent. While powder was still the prime choice, the proportion of respondents using liquid only in Port Macquarie was slightly higher than that of Melbourne and Sydney. They believed that the liquid detergent could be more environmental friendly. Most of the respondents from Port Macquarie were living in a small size family (1-3 people) whereas over 55% of Melbourne respondents and 51% of Sydney residents were living in a medium size family (4-6 people). The majority of the people in the three survey places were likely to use a washing machine 1-4 times/ week. A small proportion of people washed 5 or more loads a week.

6.3.2 Willingness to use recycled water

Figure 6.3 depicts the results on attitudes of participants towards various specific end uses of recycled water in Port Macquarie and Melbourne. It was shown that the percentage of respondents willing to use recycled water decreased gradually from potential low human contact activities (e.g., toilet flushing, garden watering and car washing) to the options involved high personal contact (e.g., clothes washing, swimming and showering). This trend corresponded to previous research findings (Friedler et al., 2006; Hurlimann and Dolnicar, 2010b; Pham et al., 2011).

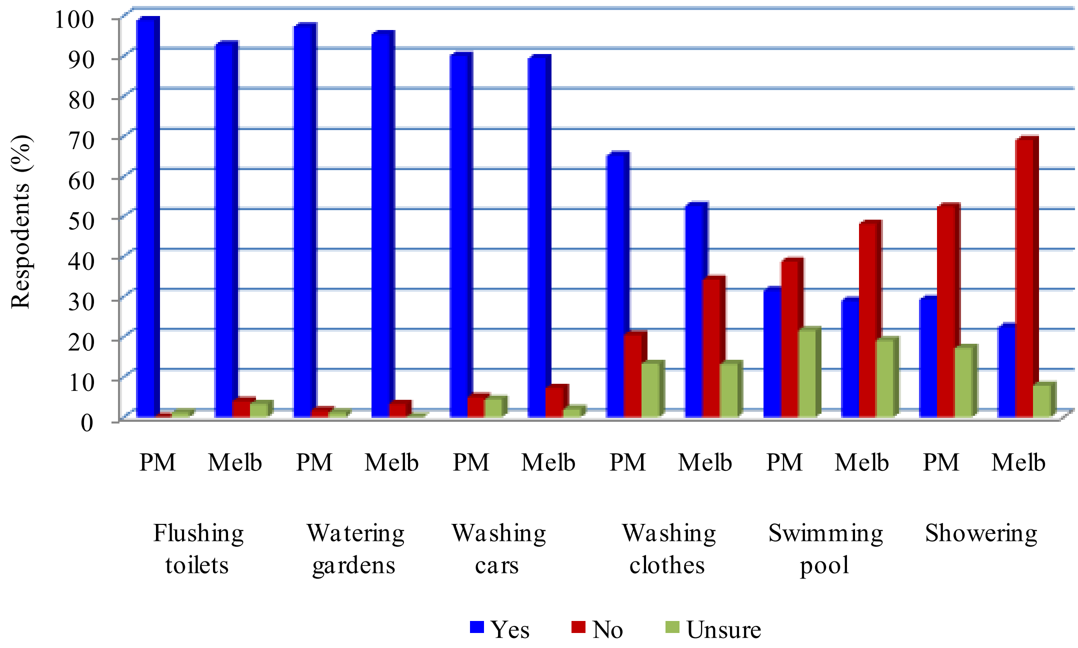

Figure 6.3 Respondents' willingness to use recycled water on various options in Port Macquarie and Melbourne

With respect to the residents in Sydney, most of the respondents have already been using the recycled water for the options including toilet flushing, garden watering and car washing through dual pipe systems installed at each household. They showed high degrees of satisfaction (over 16% of great satisfaction and 44% of satisfaction) towards the current recycled water end uses. However, they expressed less interest in using recycled water for washing clothes, swimming pools and showering compared to those who lived in Port Macquarie and Melbourne.

6.3.3 Preferred ways to receive relevant information

According to Figure 6.4, most of the respondents have not received enough information or updates about the supply of recycled water to the home. "Brochures and flyers via mail" was the most preferred option for people to receive additional information. The second most preferred option for Melbourne and Sydney people was "website or email", compared with "articles or advertisements in newsletters or newspapers" for Port Macquarie participants.

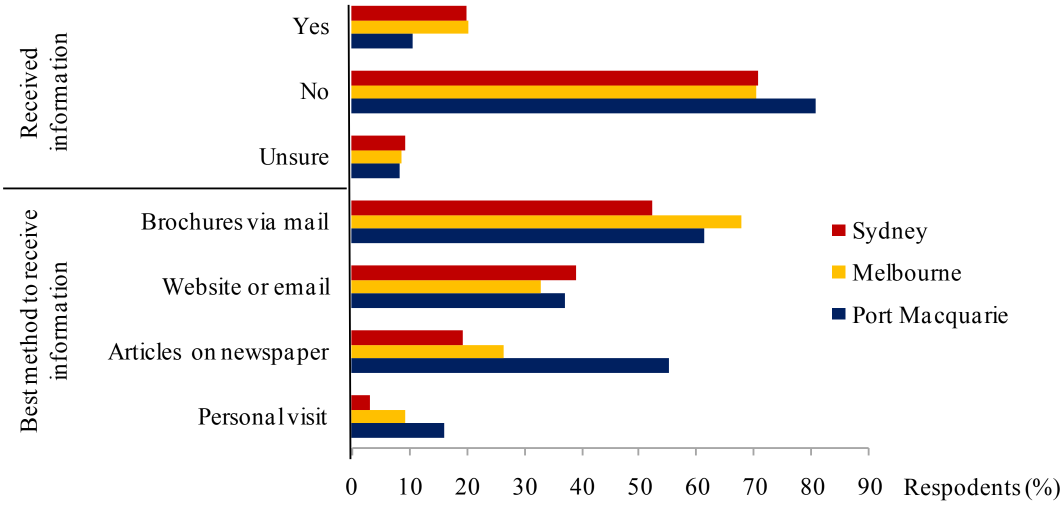

Figure 6.4 Responses to the preferred ways to receive additional information on recycled water

This might be because of the different age composition and family size in the three survey locations. In Port Macquarie, over 60% of respondents were middle-aged and older people (over 40-year-old) and most of them were living in a small size family, they are more likely to use traditional media (e.g., newspaper and TV) to receive information rather than using computers and electronic devices. Comparatively, over

68% of the Melbourne and Sydney respondents were young people (less than 40-year-old). Water-related information can be sent to them efficiently and effectively through website or email.

6.4 INFLUENTIAL FACTORS ON THE PUBLIC ATTITUDES OF FUTURE USERS TOWARD RECYCLED WATER USE IN A LAUNDRY

As the residents in the Newington area of Sydney have already received the recycled water and have been using it for multiple purposes, this section focuses only on opinions from non-users and prospective users in Port Macquarie and Melbourne. Participants from these two survey locations showed their great interest in using recycled water for washing clothes. Consequently, this section discussed their attitudes to or knowledge of recycled water supply in the near future and further analysed the impacts of attitudinal and psychological factors on the final support of recycled water use in a laundry using regression models.

6.4.1 Attitude of future users on receiving recycled water supply

The variations in attitudes and knowledge of respondents from Port Macquarie and Melbourne were analysed by chi-square tests. As can be seen in Table 6.1, there was a significant difference in the two survey locations on knowing the home will be receiving recycled water ($p<0.0000$). Since the dual pipe systems have already been installed at new homes in the survey locations of Melbourne, local residents are likely to have more perceived knowledge and know that the home will be receiving recycled water soon. There was no significant difference in Port Macquarie and Melbourne on other cognitive aspects. The majority of respondents in both survey places recognized the importance of recycled water as an alternative to drinking water and expressed positive attitudes towards receiving recycled water. Of these, more than half (58%) said "agree" rather than "strongly agree", meaning that they were not that confident with their positive attitude. The main reason given for all positive responses was "saving drinking water" (68%). While "save money" was the second priority of Melbourne respondents (17%) for consideration, it was in a lower rank than the option "recycled water is an alternative to drinking water" from Port Macquarie respondents. Additionally, another 6% of respondents in Port Macquarie specified other reasons (e.g., environmental awareness and the experience of water restrictions).

Table 6.1 Respondents' attitudes to receiving recycled water in two survey locations

Attitudes/Knowledge (%)	Numerical rating scale	Survey location		Chi-square test	
		PM	Melb	χ^2	p-value
Knowing the home will receive recycled water (1= yes; 2= yes, was not sure of the date; 3= no)	1	3.3	17.8	34.99	0.0000
	2	6.1	11.2		
	3	77.9	69.7		
Recycled water is an important alternative to drinking water (1= strongly agree; 2= agree; 3= disagree; 4= strongly disagree)	1	27.6	32.2	6.49	0.1657
	2	58.0	60.5		
	3	10.5	3.9		
	4	3.3	2.0		
Attitude on receiving recycled water (1= very happy; 2= happy; 3= unsure; 4= not happy; 5= very unhappy)	1	22.1	32.2	6.50	0.2609
	2	48.1	47.4		
	3	23.2	16.5		
	4	4.4	2.6		
	5	1.7	0.66		
The cost of recycled water compared to drinking water (1= much higher; 2= higher; 3= the same; 4= slightly lower; 5= much lower)	1	5.0	3.9	3.396	0.6392
	2	21.5	16.4		
	3	18.8	15.8		
	4	27.6	30.9		
	5	18.8	24.3		
Overall, support the use of recycled water in a laundry (1= support; 2= uncertain; 3= against)	1	69.1	67.1	3.222	0.3587
	2	26.0	23.0		
	3	4.4	9.2		

Abbreviations: PM=Port Macquarie, Melb = Melbourne.

On the other hand, negative respondents, who replied "disagree" or "strongly disagree", were also asked the reasons for their response. Health reasons and the sense of the water being not clean were the most commonly mentioned aspects, followed by high cost and the running desalination plants issues. Moreover, approximately 50% of the respondents thought that the price of recycled water would be lower than drinking water. Still, there is a considerable number of missing values (around 10%) from people who had difficulty in deciding on the intention to pay. Overall, over 67% of the participants supported the use of recycled water in a laundry. A high number of "uncertain" responses (24.6%) indicated that respondents were lacking in confidence about this controversial product and wanted to be informed by further analysis.

6.4.2 Regression Model 1 and Model 2

Although respondents from the two places, namely Port Macquarie and Melbourne, demonstrated slightly different household laundry behaviour (e.g., choice of washing detergent) depending on the differences in family size, age composition and living habits, their attitude and knowledge on water related issues exhibited no significant difference.

According to the chi-square tests in Table 6.2, three attitudinal variables and six psychological variables were shown to have a relationship with the final acceptance of recycled water use in a laundry ($p < 0.05$). Therefore, these variables have been selected as predictor variables in Model 1 and 2 (Table 6.3). In Model 1, all predictor variables were regarded as independent to each other so that the interactions between different variables were not taken into account. In contrast, Model 2 added all pairwise interactions between the variables in the model.

Table 6.3 gives the corresponding model coefficient estimates together with the standard errors and odds ratios. The order of estimates is in the sequence each entered the model. When the predictor variable is increased by one unit, the response variable will increase with the estimated unit (Dolnicar et al., 2011). For example, in Model 1, given other predictor variables at reference categories, a one unit increase in attitude variable changes the odds of the acceptance of recycled water use in a laundry multiplicatively by a factor of 4.067. It is worth noting that compared with Model 1, all parameters in Model 2 changed slightly when adding the interaction terms, showing that the combined effect of predictor variables is different from their separate effects. Particularly, the estimated values of coefficients for colour and clothes variables were changed from positive to negative, indicating that the community would be less concerned about these psychological issues when given a series of questions together rather than when facing separate issues.

Besides, it can be observed that in Model 2, when considering the combined effect of two variables on the acceptance level, the interaction effect seemed to offset partially the simple additive effect posed by independent variables. For instance, although the two variables, the positive attitude and the greater confidence by reading from other customers, separately have a positive effect on increasing the community acceptance, their combined effect was reduced if the interaction effect was taken into account (i.e., 2.731 rather than 4.103).

The following variables were significant at the 0.05 level: in Model 1, Recycled water is an alternative to drinking water (RWAlterDW), Attitude, Odour and SmallUnit; in Model 2, RWAlterDW, Attitude, Cost, Odour, Reading, SmallUnit, Reading by SmallUnit, Attitude by Reading, and Colour by Odour. They are the main aspects to change the level of acceptance that respondents would use recycled water in a laundry. With respect to the usefulness of the model, both Model 1 and 2 could predict over 80% of the observations correctly. The higher the overall percentage of correct predictions, the better the model is. Another approach to access the effectiveness of the model is the Negelkerke R^2 test, which gives the proportion of variation in the outcome variable being explained by the model. A small R^2 value means that the model is not a great improvement over the null model with no predictors. As Model 1 and 2 could explain about 41.4 and 45.8% of the variation in data respectively, they were capable of accounting for a substantial amount of the variance.

Moreover, the Hosmer and Lemeshow test was also performed, which is to form groups of cases and construct a "goodness-of-fit" statistic by comparing the observed and predicted number of events in each group (Azen and Walker, 2011). When the p-value in the Hosmer and Lemeshow test is greater than 0.05, there is no significant difference between observed and model-predicted values, implying that the model's estimates fit the data at an acceptable level. Both Model 1 and 2 appeared to fit the data reasonably well. Overall, the model that includes interaction terms (Model 2) is better at predicting the observed data associated with recycled water use in a laundry than the Model with merely independent variables (Model 1).

Table 6.2 Factors found to influence community acceptance of recycled water in a laundry

Variable	Rating	Support (%)	Uncertain (%)	Against (%)	χ^2	p
Attitudinal variables						
Recycled water is an alternative to drinking water	Strongly agree	25.23	3.30	1.20	96.71	0.0000
	Agree	40.84	15.02	3.30		
	Disagree	1.50	5.11	0.60		
	Strongly disagree	0.30	1.20	0.90		
Attitude on receiving recycled water	Very happy	22.52	3.00	1.20	109.3	0.0000
	Happy	36.94	9.01	1.80		
	Unsure	8.41	9.31	2.10		
	Not happy	0.30	2.70	0.30		
	Very unhappy	0.00	0.30	0.90		
The cost of recycled water compared to drinking water	Much higher	1.80	2.10	0.60	33.78	0.0037
	Higher	12.01	5.11	2.10		
	The same	12.31	4.50	0.60		
	Slightly lower	22.52	5.71	0.90		
	Much lower	15.02	4.80	1.50		
Psychological variables						
Be concerned of colour of clothes	Yes	32.43	14.41	3.60	10.85	0.0283
	No	25.23	5.11	1.20		
	Unsure	5.71	3.30	1.20		
Be concerned of damage to clothes	Yes	29.73	12.91	3.90	14.31	0.0064
	No	28.83	6.31	0.90		
	Unsure	4.50	3.00	1.20		
Be concerned of damage to washing machine	Yes	25.83	13.21	3.90	17.89	0.0013
	No	30.63	5.41	1.20		
	Unsure	6.01	3.60	0.90		
Be concerned of odour caused by recycled water	Yes	31.83	17.72	5.11	36.22	0.0000
	No	25.83	2.10	0.00		
	Unsure	5.71	2.40	1.20		
Confidence increased by reading recycled water being used by other people	Yes	48.65	13.21	2.10	47.83	0.0000
	No	9.61	3.90	3.90		
	Unsure	3.30	4.80	0.00		
Confidence increased by having a small unit for pre-treatment	Yes	52.25	13.21	2.40	42.03	0.0000
	No	6.31	3.90	3.00		
	Unsure	3.60	4.80	0.60		

Table 6.3 Logistic regression for recycled water

Predictor variables	Regression Models								
	Model 1			Model 2			Model 3		
	COEF	SE	OR	COEF	SE	OR	COEF	SE	OR
Constant	-1.465*	0.503	0.231	-1.885*	0.745	0.152	-2.274*	0.586	0.103
RWAlterDW (positive)	1.916*	0.465	6.792	2.039*	0.646	7.684	1.901*	0.470	6.692
Attitude (positive)	1.403*	0.312	4.067	2.095*	1.055	8.122	2.216*	0.447	9.171
Cost (higher)	-0.308	0.309	0.735	-0.480	0.328	0.619	-0.261	0.306	0.770
Cost (the same)	-0.607	0.332	0.545	-0.829*	0.354	0.436	-0.414	0.319	0.661
Colour (concerned)	0.484	0.362	1.622	-1.042	0.689	0.353	–	–	–
Clothes (concerned)	0.140	0.380	1.150	-0.095	0.570	0.910	–	–	–
Machine (concerned)	-0.240	0.310	0.787	-0.990	0.663	0.372	–	–	–
Odour (concerned)	-2.077*	0.372	0.125	-2.960*	0.493	0.052	-1.685*	0.285	0.186
Reading (effective)	0.278	0.289	1.320	2.008*	0.655	7.445	1.884*	0.598	6.581
SmallUnit (effective)	0.990*	0.288	2.692	1.942*	0.485	6.971	1.428*	0.435	4.169
Interactions									
Reading by SmallUnit	–	–	–	-1.334*	0.633	0.263	-1.045	0.590	0.352
Attitude by Reading	–	–	–	-1.372*	0.662	0.254	-1.256*	0.591	0.285
Colour by Odour	–	–	–	1.495*	0.721	4.457	–	–	–
Machine by Odour	–	–	–	1.071	0.725	2.918	–	–	–
Attitude by RWAlterDW	–	–	–	-0.009	0.990	0.991	–	–	–
Colour by Clothes	–	–	–	0.591	0.735	1.806	–	–	–
Goodness-of-fit									
Overall percentage correct (%)	80.5			81.1			80.8		
Negelkerke R^2	0.414			0.458			0.422		
Hosmer and Lemeshow	χ^2: 12.827; p-value: 0.076			χ^2: 11.520; p-value: 0.118			χ^2: 8.203; p-value: 0.224		

Abbreviations: COEF= Coefficient; SE= Standard Error; OR= Odds Ratio; RWAlterDW= Recycled water is an alternative to drinking water

*Asterisks indicate significant at the 0.05 level.

6.4.3 Regression Model 3

Nevertheless, as Model 2 involves nine predictor variables along with six different interaction effects between variables, the interpretation of model parameters is somewhat complicated. For ease of understanding, only six variables in Model 2 which were found to have a significant relationship with tendency to use recycled water in a laundry, were included in Model 3. Based on goodness-of-fit tests, Model 3 fits the data well. It can be written as:

$$\begin{aligned}\text{logit}(\pi) &= -2.274 + 1.901 RWAlterDW(0) + 2.216 Attitude(0) - 0.261 Cost(0) \\ &-0.414 Cost(1) - 1.685 Odour(0) + 1.884 Reading(0) + 1.428 SmallUnit(0) \\ &-1.045 Reading.SmallUnit - 1.256 Attitude.Reading\end{aligned} \qquad \text{(Eq. 6.1)}$$

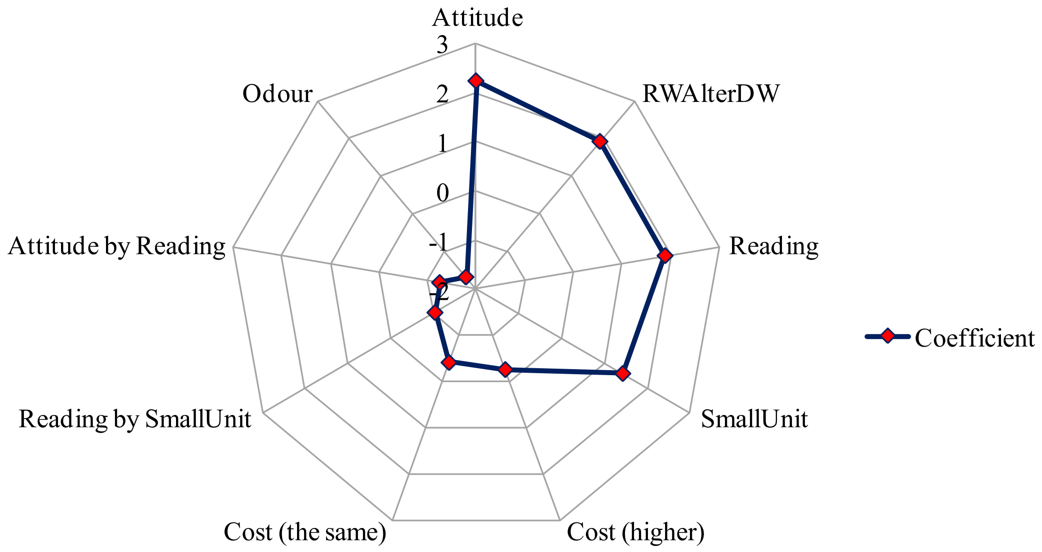

Figure 6.5 Extent of the variable effect in Model 3 on the final acceptance of recycled water use in a laundry

As can be seen from Table 6.3 and Figure 6.5, the effect was strongest for those who exhibited positive attitude to receiving recycled water, followed by the respondents who recognised recycled water as an alternative to drinking water. Each additional score on the Attitude and RWAlterDW scales increases the chance of accepting recycled water use in a laundry by more than 6 times. Besides, enhancing the respondents' confidence by the introduction of successful examples and/or installation of a small unit for pre-treatment is also able to greatly improve the acceptance level of recycled water use in a laundry.

However, the interpretation of the relation between increased confidence by reading from others and the final acceptance also requires the consideration of two-way interactions since the interactions with adding a small unit and presenting positive attitude weakened the direct simple effect posed by a single reading factor. Remarkably, the odour and perceived cost are also key influential factors, and they showed negative impacts on the acceptance. This concern may be closely related to respondents' inherent thoughts on recycled water as the origin of recycled water is odorous wastewater and the cost for wastewater treatment is relatively high owing to the complexity of pollutants.

The findings from regression models have important practical implications as there are very few researches addressing the influences of attitudinal and psychological factors on the community satisfaction of recycled water use in a laundry. Although the major public concerns on recycled water uses can be drawn from previous research surveys, the important drivers that promote the public involvement in new strategies of the scheme are still ambiguous. Hence, this chapter provides guidance to water providers and government policy makers about interventions that are likely to increase the public acceptance of recycled water on new applications with high personal contact.

Specifically, it is advisable to offer more educational campaigns to let the public understand the current water shortage status and the importance/advantage of recycled water as an alternative water resource. People's positive attitude to recycled water can also be established through the relevant information or updates provided by brochures via mail, websites or email and articles in newspapers which are the preferred ways for the community to receive additional information. Besides, apart from a better water quality control program, to reassure the community, particularly regarding the potential odour and perceived high cost issues, it is feasible to encourage personal communications, especially with people who have previous experiences with water reuse. Launching workshops, which allows people to be able to watch treatment processes and experience recycled water, may be another useful strategy. These approaches are likely to be far more effective than blunt public announcements stating that recycled water would be added to the dual pipe systems to households (Dolnicar et al., 2011).

Nevertheless, as the developed regression models were only based on the survey results from two places in Australia, the satisfaction of recycled water use in a laundry seem to be plausible but may not be accurate or true across the whole country. Future research on the refinement of the models include the consideration of additional predictor variables, such as risk perceptions, environmental concerns, information search and TV watching behaviour, and the reduction of regional effects by conducting more case studies. Still, the model does provide insights that the attitudinal and psychological factors have strong relationships with the acceptance level of new end uses of recycled water. This could be beneficial to other recycled water providers, water authorities and policy makers involved in the expansion of recycled water supply.

6.5 FEEDBACK FROM CURRENT USERS ON USING RECYCLED WATER

Although over 66% of respondents in Newington, Sydney, expressed general support of the idea that recycled water would be used for clothes washing purposes, only a few of them displayed strong interest on applying this new end use in their own households. The reason for this is most likely to be the perceived risks involved with close personal contact recycled water end uses. According to the survey results, 44% of interviewees believed that there is some risk for adoption of recycled water in the home. Among them, the hygienic issues are of high priority (39%), followed by health issues, danger to children and/or pets, and the possibility of cross connection. When the participants were asked whether they have any specific complaints about the current recycled water quality, 68% of them indicated no specific complaints.

However, the rest of the participants showed different levels of discontent regarding the colour, salinity and clearness of recycled water. Some of them mentioned the spots that appeared on cars after washing with recycled water while others referred to the suspended soil-like particles seen in the toilet pan after toilet flushing. Consequently, since Newington residents have already established good cognitions on the appearance and cost of recycled water based on current using experiences, they exhibited different areas of concern compared with respondents in the other two survey locations regarding the potential effects that recycled water could have if being used in a laundry. While people in Port Macquarie and Melbourne were primarily concerned with the odour and increased cost, Newington respondents cared more about the colour of clothes and potential damage to washing machines. Nonetheless, similar to other participants, Sydney residents' confidence is likely to be increased by having a small unit for pre-treatment and knowing that recycled water is being used in the laundry by other customers. Thus, corresponding measures, workshops and educational campaigns should also be given to current users in the community. By this means, the dual pipe systems would become the model for all future developments and could possibly be retrofitted to existing building structures.

6.6 CONCLUSIONS

Public acceptability is a prerequisite for society to establish and promote new recycled water end uses. Presently, there has been little achievement of broad public acceptance of and enthusiasm for new end uses of recycled water in Australia. Through extensive social attitude surveys in Port Macquarie, NSW and Melbourne, Victoria, the regression relationships between predicting variables and the public acceptance on recycled water use in a laundry have been established.

The variables significantly contributing to the acceptance of this new end use were: (i) positive attitude on receiving recycled water, (ii) positive opinion on the idea "recycled water is an alternative to drinking water", (iii) increased confidence by reading from other customers or successful examples, and (iv) increased confidence by adding a small unit to improve the water quality. The fear of the potential odour and high cost when using recycled water for a household laundry became the main factors to prevent respondents from being supportive of this new end use. Comparatively, survey results in Sydney, NSW indicated slightly different aspects of concern due to the fact that Sydney residents interviewed have already established good cognitions on the appearance and cost of recycled water from their current using experience.

Overall, the findings from three survey locations in Australia reconfirmed that building the community's knowledge, trust and confidence on water saving and current recycled water status were the critical points in ensuring the smooth expansion of recycled water supply or the introduction of new applications. While the increment of public acceptability is a long-term participatory procedure, some corresponding policy strategies should be primarily taken to guarantee the implementation in a cost-effective way. The methodology and suggestions from this chapter could be applied further in other locations within or outside Australia to obtain holistic community views.

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CHAPTER 7

A New Optional Recycled Water Purification System Prior to Use in the Household Laundry

7.1 INTRODUCTION

With a constantly growing population, water scarcity becomes the limiting factor for further social and economic growth. To achieve a partial reduction in current freshwater demands and lessen the environmental loadings, an increasing trend in the water market tends to adopt recycled water for household laundries as a new recycled water application. The installation of a small pre-treatment unit for water purification can not only improve the recycled water quality further, but also be viable to enhance the public confidence and acceptance level on recycled water consumption. Specifically, this chapter describes column experiments conducted using a 550 mm length bed of zeolite media as a one-dimensional flow reactor. The results show that the zeolite filter system could be a simple, low-cost pre-treatment option which is able to reduce the total hardness level of recycled water significantly via effective ion exchange. Additionally, depending on the quality of recycled water required by end users, a new by-pass controller using a three-level operation switching mechanism is introduced. This approach provides householders sufficient flexibility to respond to different levels of desired recycled water quality and increase the reliability of long-term system operation. These findings could be beneficial to the smooth implementation of new end uses and expansion of the potential recycled water market. The information could also offer sound suggestions for future research on sustainable water management and governance.

7.2 RESEARCH BACKGROUND

Recycled water for household laundry has been increasingly regarded as a prospective new end use since significant fresh water savings could be achieved if potable-quality water used for clothes washing is replaced with recycled water (Chen et al., 2012a). However, despite that many water authorities have encouraged the new applications of recycled water in dual pipe systems and stipulated corresponding policies, guidelines and regulations, the use in practice of recycled water with relatively close human contact is still quite limited. Public concerns on individual health, water clarity, cost and machine durability might be major issues that prevent the society from establishing and promoting the new end use (Pham et al., 2011). Notably, the recent social surveys in three different locations of Australia (i.e., Port Macquarie, Melbourne and Sydney) indicated that the community's confidence on the use of recycled water in a household laundry would be greatly enhanced with the operation of a small unit for pre-treatment of recycled water prior to use in a washing machine (Chen et al., 2013d). Hence, this chapter is to analyse the feasibility and cost-effectiveness of employing a small pre-treatment unit for water purification in the household laundry.

7.2.1 Recycled water quality

The actual recycled water (Class A) samples from the City West Water (CWW) Western Treatment Plant, Melbourne, Australia, have been considered for experimental analyses.

Table 7.1 Performance measure for chemical characteristics

Chemical characteristics	CWW recycled water (mg/L)	ADWG guideline values (mg/L)	
		Health	Aesthetic
Aluminium	0.015	0.1	0.2
Antimony	0.001	0.003	–
Arsenic	0.0015	0.01	–
Barium	0.005	2	–
Beryllium	< 0.001	0.06	–
Boron	0.17	4	–
Cadmium	< 0.0002	0.002	–
Chromium	< 0.001	0.05	–
Copper	0.006	2	–
Iron	0.045	–	0.3
Lead	< 0.001	0.01	
Lithium #	< 0.02	–	–
Manganese	0.024	0.5	0.1
Mercury	< 0.0001	0.001	–
Molybdenum	0.0025	0.05	–
Nickel	0.0075	0.02	–
Selenium	< 0.001	0.01	–
Silver	< 0.001	0.1	–
Zinc	0.025	–	3

Abbreviation: ADWG = Australian Drinking Water Guideline.

The Western Treatment Plant has been supplying 37 Gigalitres per year of recycled water for agriculture and landscape irrigation (e.g., golf club, parks, zoos and wetlands) in the Werribee area, a suburb around 32 km southwest of Melbourne's Central Business District (CBD). The dual pipe recycled water systems have already been installed in front/back yard of 2,000 new homes in Wyndham Vale, Werribee and Class A recycled water will be supplied to local households in December 2013 or early 2014 (City West Water, 2012). Thus, if the pre-treatment unit is demonstrated to be beneficial to ensure the reliability and consistency of recycled water consumption, the system can be widely adopted in real cases. By this means, residents are likely to be more optimistic and confident in establishing and implementing new applications of recycled water. This would further contribute to significant freshwater savings and to achieving water recycling targets toward a sustainable water cycle development.

As can be seen from Table 7.1, according to the Western Treatment Plant recycled water quality monthly reports, all heavy metal concentrations in CWW recycled water are lower than guideline values that are set for Australian drinking water (ADWG, 2011; SRW, 2013). As these chemical parameters are at acceptable levels both from health and aesthetic perspectives, heavy metal removal is not the main target of the pre-treatment unit.

Besides, as shown in Table 7.2, the concentrations of physical, microbial, nutrient and aesthetic indicators in CWW recycled water samples all satisfy the corresponding guideline values (ANZECC-ARMCANZ, 2000; EPA Victoria, 2003; ADWG, 2011). Thus, treatment measures for the further improvement of the physical and microbial performances of recycled water will not be discussed in detail in this chapter.

Table 7.2 Performance measure for water quality indicators

Indicators	CWW recycled water	EPA Victoria guideline for recycled water	ANZECC-ARMCANZ STV Limit[1]	ADWG guideline for drinking water
pH	7-7.5	6-9	–	–
Alkalinity (mg/L as $CaCO_3$)	115-125	–	–	–
BOD/SS (mg/L)	< 2 / 2[2]	< 10 / 5	–	–
Chlorine residual (mg/L)	≥ 1 after 30 min[2]	≥ 1 after 30 min	–	≥0.2 throughout the distribution system
E.coli (per 100 mL)	0[2]	<10	–	–
Total Nitrogen (mg/L)	13-25	–	25-125	–
Total Phosphorous (mg/L)	8.5-10	–	0.8-12	–
Turbidity (NTU)	< 2	< 2	–	–
Colour (PCU)	8-12[2]	–	–	15
Hardness (mg/L as $CaCO_3$)	175-200	–	–	<200

[1] STV is the short-term trigger value of agricultural irrigation water that has minimized risk to the environment up to 20 years.
[2] Note: Adapted from SRW, (2013).

7.2.2 Hardness of recycled water

When it comes to water hardness, the majority of the hardness ions are calcium and magnesium, but small amounts of other ions such as iron and manganese can contribute as well. Hard water minerals in recycled water can react with soap anions which might cause difficulty with soap lathering, decreasing the cleaning efficiency. The insoluble precipitates can also induce scaling problems and serious mechanical failures by forming crusty deposits in household appliances, thereby shortening the life of the washing machines and reducing the machine efficiency (City West Water, 2009; Seo et al., 2010). Other problems not so visible but quite significant are deposits on clothes fabrics, namely soap curd or scum, after they have been washed. Dull whites and colours caused by soap curd might not be easily removed in the rinse cycles. This can also shorten the life of clothes that are washed and worn frequently (Wist et al., 2009).

The Australian Drinking Water Guidelines (ADWG) state that there is no health guideline value for total hardness (referred to as calcium carbonate, $CaCO_3$), but an aesthetic value should not exceed 200 Milligram per litre (mg/L). Comparatively, the

Canadian Guidelines regard the water with total hardness over 200 mg/L as poor quality, and 80-100 mg/L as an acceptable level (ADWG, 2011). The average water hardness level in the drinking water supply from CWW at Werribee area, Melbourne, is approximately 30 mg/L, with maximum levels less than 50 mg/L. As the total hardness of CWW recycled water nearly approaches the ADWG guideline upper limit and is significantly higher than that of drinking water in the local community, the pre-treatment unit is primarily designed to mitigate the current hardness level so as to improve the performance of soaps and laundry detergents, and protect the washing machine from scaling during clothes washing activities.

There are many different methods that have been widely adopted as a means of effective water softening, including chemical precipitation, ion exchange process, membrane techniques (e.g., nanofiltration and reverse osmosis) and electromembrane systems (e.g., electrodialysis, electrodialysis reversal, and electro-deionization reversal). However, in the case of chemical precipitation, the choices of additional chemicals are restricted as some of them might be deleterious to human health. With respect to membrane and electromembrane systems, high power consumption and expenses are required for operation and maintenance of the equipment. Besides, when water hardness level is high, calcium deposits will quickly make the membrane less permeable, causing membrane fouling within a short period of time. The affordability and operability of these advanced techniques are likely to present barriers to their actual application at household levels (Gabrielli et al., 2006; Wist et al., 2009; Seo et al., 2010). Therefore, a cost-effective, simple and low energy consuming approach has been forced on water softening processes.

7.2.3 Potential application of natural zeolites for water softening

As zeolitic minerals have been discovered in many areas of the world, natural zeolites have found a variety of applications in adsorption, catalysis, building industry, agriculture, soil remediation and energy production. Natural zeolites are crystalline microporous minerals with a well-defined open framework structure, consisting of a three-dimensional network of SiO_2 and Al_3O_4 tetrahedra linked together by common oxygen atoms. The mobile non-framework cations are located in cavities and wander inside the hexagonal channel walls within the structure and so natural zeolites possess valuable physicochemical properties, such as high cation exchange and sorption capacities (Sivasankar and Ramachandramoorthy, 2011; Loiola et al., 2012).

Due to their intrinsic properties and significant worldwide occurrence, application of zeolites for water and wastewater treatment has become a promising technique (Wang and Peng, 2010). The effectiveness of applying zeolites on ammonium and heavy metal ion removal from wastewaters has been well documented in the literature (Cooney et al., 1999; Panayotova, 2001; Sarioglu, 2005). However, information regarding the capacity of zeolites for water softening in the process of ion exchange to remove total hardness from highly treated recycled water, which contains much lower concentrations of ammonium and heavy metal ions but relatively high levels of hardness ions, is still very limited (Cinar and Beler-Baykal, 2005). Consequently, considering the practicality, simplicity and economy of the small treatment unit, natural zeolites have been selected as the filtration material in the laboratory scale analyses. To simulate the real operating situations, the results of continuous flow experiments for zeolite column testing will be presented; these results can provide scale-up information to the design of a commercial scale zeolite pre-treatment system for actual application in local households.

7.3 ZEOLITE PRETREATMENT UNIT FOR RECYCLED WATER PURIFICATION

The viability and effectiveness of employing a zeolite column for recycled water purification prior to use in the washing machine have been investigated in detail in the following sections. The actual recycled water from CWW Western Treatment Plant was adopted as the feed solution. The natural zeolite materials were firstly treated by a saturated sodium chloride solution for pre-conditioning purpose and then applied to the column experiment. The specific material preparation and experimental set-up are shown in Chapter 3. After reaching the adsorption equilibrium in the reaction phase, the zeolite column has been regenerated twice to restore the zeolite ion exchange capacity.

7.3.1 Characterization of zeolite samples

The Scanning Electron Microscope (SEM) images demonstrated the changes in surface morphology of the natural zeolites after pre-conditioning and ion exchange reaction.

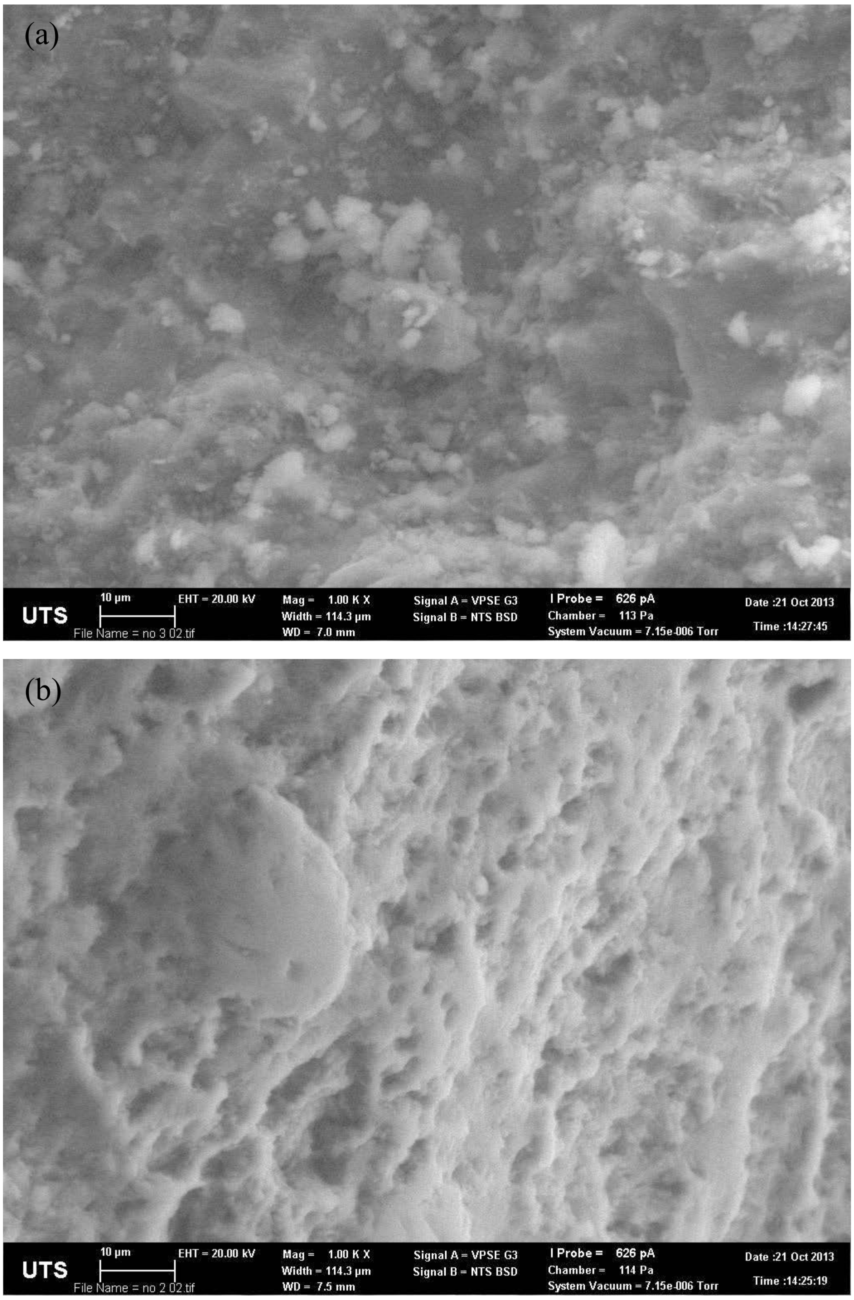

Figure 7.1 SEM images of the raw natural zeolite (a) and after pre-conditioning (b)

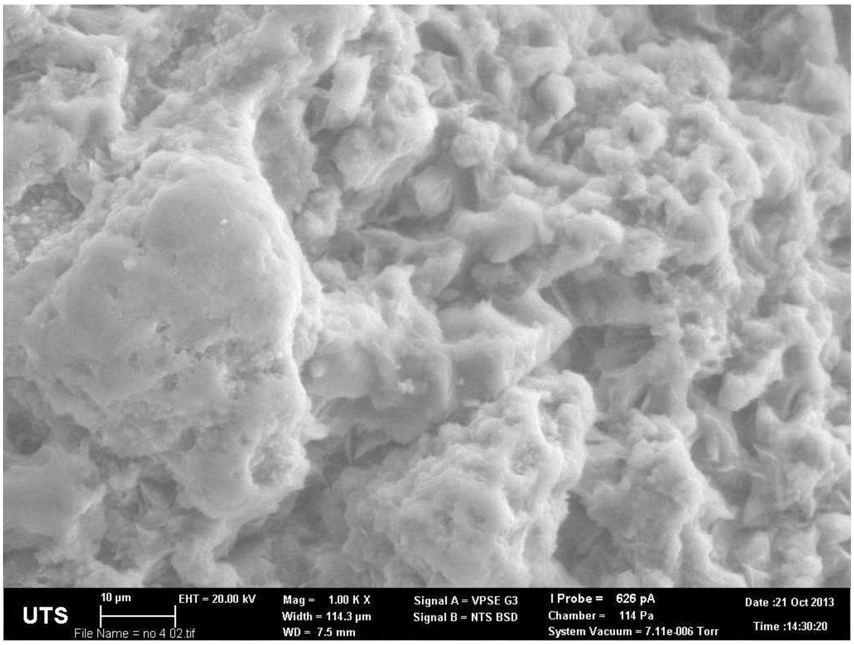

Figure 7.2 SEM image of the zeolite after the column reached saturation

As illustrated in Figure 7.1, with a pre-conditioning stage, the surface of the modified zeolite (Figure 7.1b) became rougher, more irregular and highly porous compared with the surface of the original zeolite (Figure 7.1a). After the ion exchange reaction, as calcium and magnesium ions were preferentially adsorbed, displacing the existing sodium ions, the surface structure of zeolite in the saturated column had further changed and appeared to include larger crystals (Figure 7.2) than the fresh zeolites.

The Energy Disperses X-ray Spectroscopy (EDS) analyses further verified the ion exchange mechanism of hardness ion adsorption by comparing the chemical composition changes of samples in different processes (Table 7.3). After pre-conditioning, sodium content on the zeolite surface increased by 243% whereas calcium and magnesium contents decreased by 78% and 65% respectively. These results indicate that the pre-conditioning using sodium chloride is an effective approach to attain a higher sodium form as well as increase the specific surface and porosity of zeolites, which is beneficial for the subsequent ion exchange reaction (Lin et al., 2013).

Furthermore, when the column reaction reached the adsorption equilibrium, sodium content on the surface of saturated zeolites was detected to be low, while calcium and magnesium contents increased by 565% and 114% respectively compared with that of the pre-conditioned zeolites. With the depletion of exchangeable sodium ions, hardness ions would be unlikely to be attached to the insoluble solids and regeneration of the zeolite resin is required.

Table 7.3 Chemical element compositions of the natural zeolites under raw material, pre-conditioning and adsorption completion conditions by EDS

Chemical elements (weight %)	Natural zeolite	Zeolite after pre-conditioning	Zeolite after adsorption
Oxygen (O)	47.64	47.93	51.09
Silicon (Si)	31.20	33.42	36.99
Aluminium (Al)	8.82	8.84	5.63
Sodium (Na)	1.06	3.64	1.05
Magnesium (Mg)	0.60	0.21	0.45
Calcium (Ca)	1.57	0.34	2.26
Potassium (K)	0.36	0.28	0.31
Carbon (C)	6.85	3.29	2.09
Iron (Fe)	1.77	1.49	–
Others	0.13	0.56	0.13

Note: To minimize the accidental error, each group of zeolite samples from three different conditions were analysed at least three times by EDS.

7.3.2 Optimal contact time and maximum operation capacity

7.3.2.1 Optimal contact time

The operation under the optimum contact time (or service flow rate) will minimize the impact of the film mass transfer resistance and consequently shorten the length of the mass transfer zone (Crittenden et al., 2005). To determine the optimal condition, three breakthrough curves, which represent the evolution of the hardness ion concentrations in the function of adsorption contact time, have been obtained by passing the process stream through a full fresh zeolite column. In these curves, the loading behaviours of total hardness to be adsorbed from solution in the column were expressed in terms of normalized concentrations which is defined as the ratio of effluent ion concentration to its inlet concentration (C/C_0) as a function of time for a given bed height (Sivasankar and Ramachandramoorthy, 2011). During each run, samples of the effluent are collected and analysed until the effluent total hardness concentration reaches 200 mg/L as $CaCO_3$, which is equal to the influent concentration. At this point, no more sodium-form zeolite existed to take up hardness ions so that calcium and magnesium ions made breakthrough into the effluent.

For operational simplicity and ease of observation, the experiments were conducted under scenario 2. The contact time of less than 3 minutes was not evaluated as the high service rate might reduce the mobility of zeolite material inside the column, generating operational problems. As can be seen from Figure 7.3, service time of the zeolite column for a given removal increased as contact time was increased. Breakthrough times ($C/C_0 = 0.1$) were found to be 15, 30 and 390 minutes for the contact time of 3, 4 and 5 minutes with the flow rate of 31.2, 23.5 and 18.6 L/h, respectively.

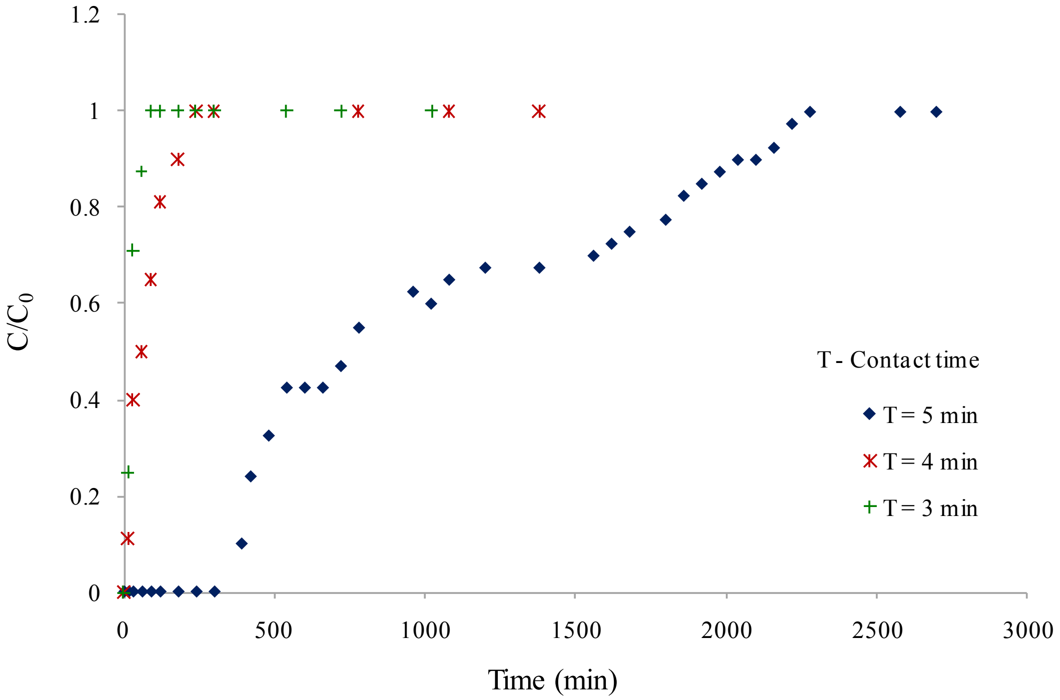

Figure 7.3 Breakthrough curves for recycled water softening process with different contact times

At the contact time of 3 and 4 minutes, the breakthrough was premature and almost immediate so that the column exhibited lower exchange capacity, whereas under a contact time of 5 minutes, the column reached the saturation after 38 hours of service time. This is due to the fact that at a low flow rate of influent, the hardness ions had more time to be in contact with the adsorbent, which resulted in a greater removal of total hardness in the column. Comparatively, at higher flow rates, the adsorption capacity was low as the solute left the column before equilibrium occurred because of the insufficient residence and diffusion time of the solute in the pores of zeolites (Han et al., 2009; Nidheesh et al., 2012).

These results are consistent with another research study conducted by Cinar and Beler-Baykal (2005) which showed that contact times over 5 minutes in the zeolite column could achieve considerable amounts of calcium ion removal. Despite that better performances may be expected for higher contact times and low flow rates in the range of 5-10 Bed Volumes per hour (BV/h), it would be very unlikely for household members to afford a long period time for each laundry activity (Inglezakis, 2005). Considering the time efficiency and practical issues, a contact time of 5 minutes has been regarded as the optimal contact time. Under this contact time, the influent pump was operated at a continuous pumping rate of 18.6 L/h. Hence, the calculated service flow rate was 12.4 BV/h.

7.3.2.2 Maximum operation capacity (MOC)

The MOC of the zeolite column (milligram per gram, mg/g), which is the weight of hardness ions adsorbed per gram of zeolite, can be calculated from Equation 7.1.

$$\mathrm{MOC} = \frac{(\mathrm{V_{eff,T}} - \int_{\mathrm{V_{eff,O}}}^{\mathrm{V_{eff,T}}} X(V_{\mathrm{eff}})dV_{\mathrm{eff}})\mathrm{C_0}}{\rho_z \mathrm{V_c}} \qquad \text{(Eq. 7.1)}$$

where, $\mathrm{V_{eff,\,O}}$ is the effluent volume until the first appearance of the solute in the exit stream, $\mathrm{V_{eff,\,T}}$ is the effluent volume until the exit solute concentration is equal to the inlet feed concentration, $X(\mathrm{V_{eff}})$ is the function of the changing effluent volume versus time, $\mathrm{C_0}$ is the influent concentration, ρ_z is the bulk density of zeolites, and $\mathrm{V_c}$ is the effective volume of the zeolite column (Inglezakis, 2005). At a contact time of 5 minutes, the solute first appeared at the reaction time of 370 minutes and reached the equilibrium at the time of 2280 minutes. Based on other known information, the MOC of the zeolite column under study is calculated to be 35 mg/g (or 1.87 meq/g).

As shown in Table 7.4, the practical MOC of zeolites from this experimental study is lower than the theoretical exchange capacities of the natural zeolites. This is due to the fact that the achieved capacity was influenced by the real operational conditions including the contact time, flow rate and quality, and temperature. Some of the zeolite cations can not be removed from the zeolite structure in specific experimental states because of low mobility and strong bonding forces within the crystal structure of the material (Inglezakis, 2005).

Table 7.4 Ideal exchange capacity of some natural zeolites

Zeolite type	Cation exchange capacity (meq/g[1])
Chabazite	3.84
Clinoptilolite[2]	2.16
Erionite	3.12
Ferrierite	2.33
Heulandite	2.91
Laumontite	4.25
Mordenite	2.29
Phillipsite	3.31
Faujasite	3.39
This study	1.87

Note: Adapted from Inglezakis, (2005).

[1] meq/g is the milliequivalents of negative charges available per gram of zeolite.

[2] Ideal exchange capacity was estimated from the chemical formula of pure species.

Noticeably, the intermittent operation mode of the column, under which the system was implemented continuously for 2 hours followed by a few hours of non-production time, was found to increase the MOC. This is because high speed flows in continuous running mode may suffer from non-ideal states such as channelling, axial dispersion, bed compaction, density currents, leakage, and/or insufficient wetting of the zeolite material (Clifford, 1999). When being operated intermittently, zeolite particles gained opportunities to partially eliminate packing and channelling during the non-production time, thereby facilitating the uniform distribution of the feed solution over the zeolite resin surface for the next running cycle. Nevertheless, since the main compositions of the studied zeolites are clinoptilolite-rich minerals, this practical MOC value was demonstrated to be close to the ideal cation exchange capacity of natural clinoptilolite zeolite.

7.3.3 Zeolite column service life and breakthrough capacity

7.3.3.1 Laundry use frequency and water consumption in a typical household

In the Werribee area of Melbourne, a recent social survey indicated that about 34.8% of local households employed front loading washing machines, compared with 65.2% of top loading machines (Chen et al., 2013d). These two types of washing machines have a number of different characteristics including loading type and capacity, water and energy consumption per wash, brand and model name (ABS, 2011). Front loaders are considered to use less water and energy and require only half the detergent that top-loading machines (1–3 star rating) consume (Gato-Trinidad et al., 2011, Chen et al., 2012a). However, water consumption varies significantly due to large differences of family makeup patterns and wash settings (e.g., size of the load, the number of cycles and clothing texture) in different households. On average, over 37% of households conducted 1-2 loads or less of washing per week, compared with 29% of households with 3-4 loads of washing per week. A small proportion of people washed 5 or more loads a week (Chen et al., 2013d). However, the frequency of use would also change depending on the weather, water restrictions and household income conditions.

In this research, for calculation simplicity, the average volume of water used per wash for a typical washing machine is assumed to be 80 litres (L) and the washing machine use frequency is considered to be 1 time per week. Remarkably, the zeolite ion exchange column can usually produce a hardness-free effluent at the initial treatment stage (e.g., first 360 minutes of reaction time under a contact time of 5 minutes in Figure 7.3) that is much more pure than that required by the guidelines, namely 80-100 mg/L as $CaCO_3$ as the acceptable level. Therefore, to minimize treatment costs, bypass blending can be implemented, which is a common procedure in water treatment applications.

7.3.3.2 Mass balance of the pre-treatment system

As shown in Figure 7.4, to produce a final effluent with total hardness level approaching the guideline value for satisfactory water quality (i.e., 100 mg/L as $CaCO_3$), part of the feed water can be bypassed around the process (through valve C) and blended with the treatment effluent under scenario 3.

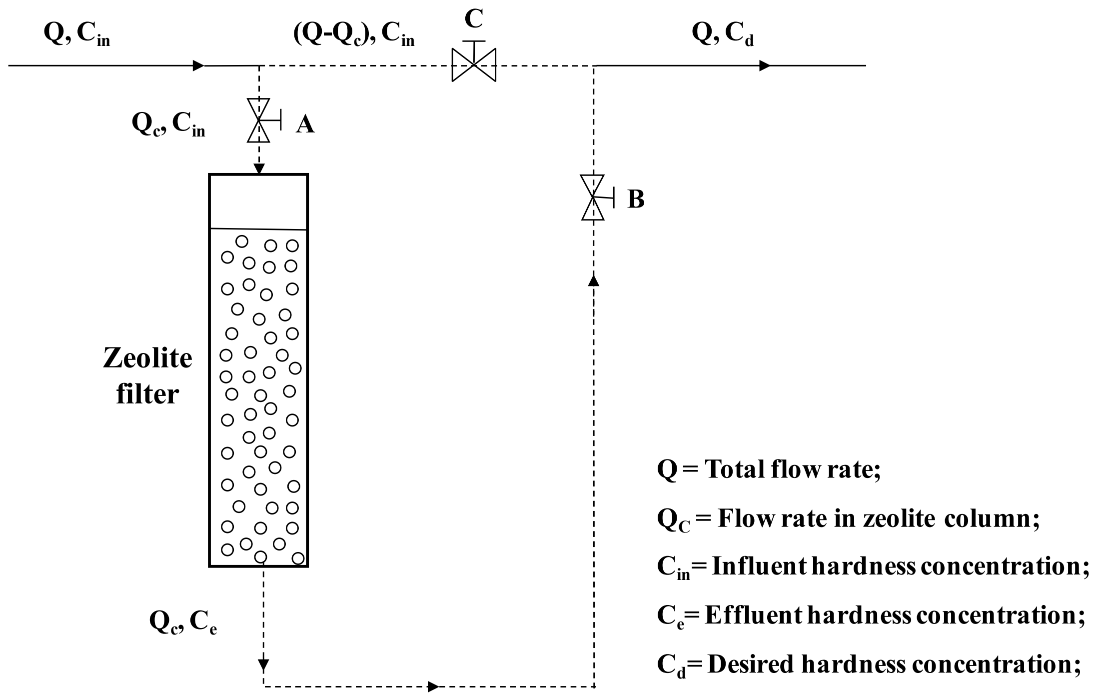

Figure 7.4 Schematic diagram of water flows in the zeolite pre-treatment system

To calculate the percentage of bypass volume, the mass balance equation (Equation 7.2) can be employed (Clifford, 1999).

$$Q_c \cdot C_e + (Q - Q_c) \cdot C_{in} = Q \cdot C_d \qquad \text{(Eq. 7.2)}$$

where some assumptions were made; there is no water accumulation and no reactions at the pipe joint, the influent hardness concentration (C_{in}) is 200 mg/L as $CaCO_3$, and the desired hardness concentration in blended product water (C_d) is 100 mg/L as $CaCO_3$.

During the first 360 minutes of the reaction time, the effluent hardness concentration from column (C_e) was zero. The bypass fraction can be easily obtained from Equation 7.3.

$$f_B = \frac{Q - Q_c}{Q} = \frac{C_d}{C_{in}} \qquad \text{(Eq. 7.3)}$$

The fraction that must be treated by the zeolite column is:

$$f_F = 1 - f_B = 0.5 \qquad \text{(Eq. 7.4)}$$

In this situation, the designed volume of recycled water per load of washing that should be passed through zeolite column is calculated to be 40 L. To attain an overall hardness level of less than 100 mg/L as $CaCO_3$ prior to use in the washing machine, it would take approximately 2 hours in households to purify the recycled water in the zeolite column. The column can maintain a high service capacity up to 3 loads of washing for a typical washing machine.

7.3.3.3 Calculation of zeolite column service life

However, after the breakthrough point which is defined as the critical time when the exit hardness concentration equals approximately 10% of the inlet concentration (i.e., the reaction time of 390 minutes), the effluent concentration increased very rapidly with the increment of time (Inglezakis, 2005). This indicated that to maintain the same quality of blended product water, the bypass fraction should be reduced and an additional amount of feed water was required to be delivered through the zeolite column. When C_e reached 90 mg/L as $CaCO_3$, around 90% of raw recycled water (72 L) should be treated through the column for one laundry activity. As depicted in Figure 7.3, by the reaction time of 660 minutes, the blended product water can still meet the desired hardness level. Nevertheless, for the 4th load of washing, due to a surge in water purification demand, the laundry activity can last for 4 hours. Although the column could continue to absorb the hardness ions in the feed water, the treated effluent could then no longer satisfy the guideline value. Consequently, given that the overall use frequency of the washing machine is once a week, the zeolite column service life without regeneration was shown to be one month.

7.3.3.4 Breakthrough capacity (BC)

The BC of the zeolite column (mg/g), which is the loading of the zeolite column when the hardness concentration in the effluent reaches the breakthrough point, can be evaluated using Equation 7.5.

$$\mathrm{BC} = \frac{(\mathrm{V}_{\mathrm{eff,B}} - \int_{\mathrm{V}_{\mathrm{eff,O}}}^{\mathrm{V}_{\mathrm{eff,B}}} X(V_{\mathrm{eff}})dV_{\mathrm{eff}})\mathrm{C}_0}{\rho_{\mathrm{z}}\mathrm{V}_{\mathrm{c}}} \qquad \text{(Eq. 7.5)}$$

where, $\mathrm{V}_{\mathrm{eff,\,O}}$ is the effluent volume until the first appearance of the solute in the exit stream, $\mathrm{V}_{\mathrm{eff,\,B}}$ is the effluent volume until the breakthrough point (Inglezakis, 2005). Accordingly, the BC is estimated as 14 mg/g. Compared with MOC, the BC only considers the reaction time from the beginning of the experiment to the breakthrough point. This might be of greater importance for the design of column operations and determination of selectivity series, as after the breakthrough point, the treatment unit often becomes far less effective and time-consuming and therefore makes insignificant contributions in practice.

7.3.4 Zeolite column regeneration

The regeneration process can restore the capacity of the zeolite column and can make it ready for reuse in the next running period. The driving force for this reaction is the large excess of sodium ions in the regenerant solution so as to flush the retained calcium and magnesium ions to the waste stream. Thus, the waste brine after exiting contained a mixture of calcium chloride, magnesium chloride along with excess sodium chloride. When regeneration proceeded toward completion, the calcium and magnesium ion concentrations became lower (Wist et al., 2009). Figure 7.5 gives three breakthrough curves corresponding to the column performances after zero, first and second regeneration cycle. As can be seen, after each run, the breakthrough curve possessed similar shape and features as previous ones and no large difference was observed. Only a slight deterioration of hardness removal was noted during the second regeneration cycle. This may be because some ionic sites associated with slow diffusion within the zeolite structure were saturated with hardness ions in previous service cycles. The hardness ions might have difficult access to exchange with sodium ions, especially when regeneration occurred over a short time frame (Sivasankar and Ramachandramoorthy, 2011). Notwithstanding, the zeolite material can be reused several times before it must be replaced due to irreversible fouling.

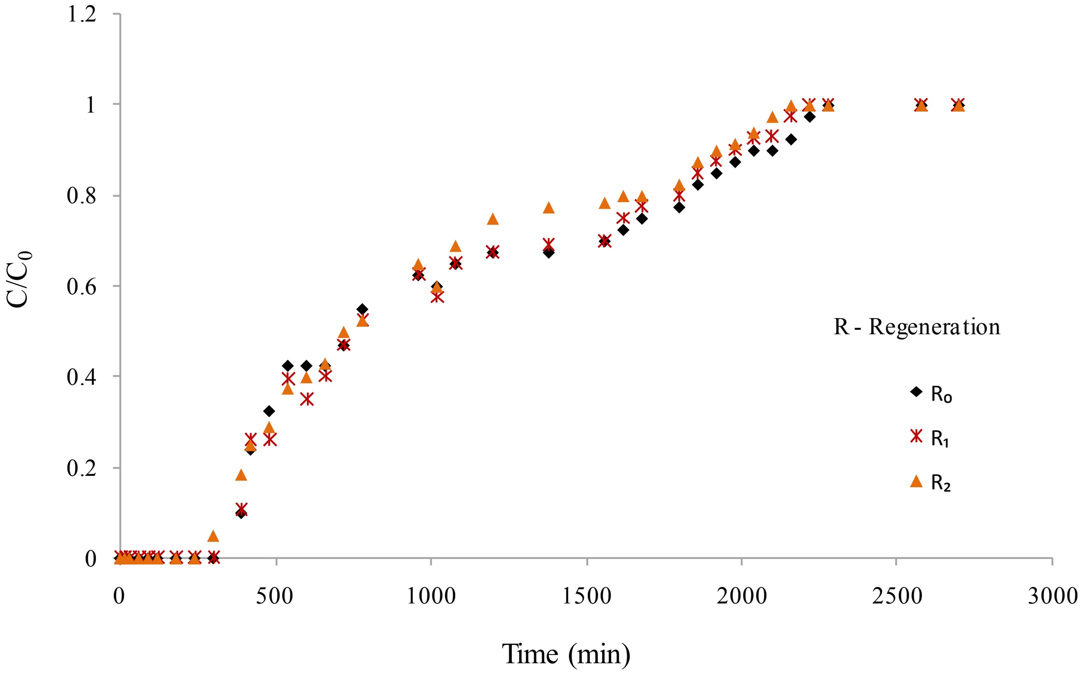

Figure 7.5 Breakthrough curves for recycled water softening process under the contact time of 5 minutes with successive regenerations

Although the regeneration process fully demonstrated the cost effectiveness and durability of the zeolite material, there are also some challenges regarding the storage and treatment of spent regenerant. While reuse of spent regenerant is a good means of reducing costs and minimizing waste disposal, the concentration of trace contaminants is likely to be built up as the number of regenerant reuse cycles increases, triggering difficulties in environmental remediation (Wist et al., 2009).

7.3.5 Pilot-scale column design and considerations

The experimental data derived from this small column study form the basis for the field trial operations and onsite scale-up pilot studies. To extend the current service life of 1 month to a longer period, the column height and/or diameter can be scaled up directly. Based on the existing column breakthrough data, when the pilot study is conducted under the same operating flow rate (i.e., 18.6 L/h), the increase of column volume would not change the shape of the breakthrough curve nor enhance the breakthrough capacity (Crittenden et al., 2005). Alternatively, the adsorption columns can be connected in series to improve the service life and regenerant usage. It is worth noting that the MOC and BC obtained in pilot studies might be slightly higher than the values from experimental analyses owing to the intermittent operation mode coupled with prolonged non-production time in real-life situations of clothes washing. Overall, the optimal design option and specific parameters as well as the potential practical problems encountered in the pilot-scale studies could lead to precautionary procedures in the further development of full-scale designs and following application, maintenance and replacement in local households.

7.4 CONCLUSIONS

This chapter demonstrates the feasibility, operability and cost-effectiveness of applying the natural zeolite material as an effective ion-exchange resin for recycled water softening prior to use in washing machines. The column experiment results revealed that the pre-conditioning of natural zeolites contributed to improved surface morphology and increased sodium contents on particles, resulting in higher ion exchange capacity. Under the contact time of 5 minutes, the process was shown to be a promising water purification pre-treatment unit with a considerable hardness ion removal. Moreover, both maximum operational capacity and breakthrough capacity data were generated to present a quantitative assessment of the proposal, which were found to be 35 and 14 mg hardness ions /g of zeolites respectively. The zeolite column service life for a typical washing machine was shown to be one month without material regeneration. The regeneration process further guaranteed the simplicity of maintenance and durability of operation for actual application at household levels. After accomplishing a series of scale-up pilot studies, the pre-treatment unit can be put into practice, which is likely to play a significant and positive role in changing the public perception on the safe use of recycled water.

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CHAPTER 8

Conclusions and Recommendations

8.1 CONCLUSIONS

The growing trend in water supply and management field in the world is to consider the water reuse practice as an essential component of integrated water resource management. The exploitation and development of recycled water schemes in many countries are closely related to water scarcity, effluent discharge and freshwater contamination control, energy consumption reduction and obtaining alternative water resources. Consequently, it is of great importance to ensure the long-term sustainable application of recycled water in practice, including the expansion of existing schemes and the establishment of new end uses. This can be achieved by adopting advanced treatment technologies, implementing adaptive risk control and management methods, and conducting reliable economic appraisals. Additionally, improving environmental quality and obtaining wide community acceptance may also contribute largely to recycled water sustainability. The thesis addresses these multiple issues by building assessment models and conducting statistical and experimental analyses under a systematic decision-making framework.

The purpose of this chapter is to summarise the main findings regarding the optimal recycled water management strategies and the effective ways to improve the viability, reliability and acceptance of recycled water use in households. Moreover, the future research directions, improvements and potential opportunities for practical implementation of research findings are presented as well.

8.1.1 Special findings

The major findings drawn from this thesis are outlined in the following sections. The first section refers to recycled water end uses and assessment model situations. The findings from sections 2 and 3 are related to systematic decision-making system for sustainable development and multi-factorial evaluations. The brief analytical results of social surveys and experimental designs are shown in sections 4 and 5, respectively.

1. Critical review studies on end uses and assessment models of recycled water schemes

- Although agricultural irrigation and industrial application represent the dominant uses of recycled water presently on a global scale, considerable progress has been made on other end uses such as urban and residential applications and groundwater recharge, especially in developed countries;

- High value end uses with potential close human contact such as recycled water for household laundries, swimming pools and indirect and direct potable uses are promising but still somewhat ambiguous due to strong public misgivings;

- The recycled water market in many developing countries moves towards a higher level of wastewater treatment and more stringent recycled water quality guidelines and regulations;

- Integrated assessment models for recycled water schemes outweigh individual evaluation models due to their comprehensiveness, accuracy and reliability. However, integrated tools might suffer from complexity and partial conflicts in model integration, making the understanding and efficient decision-making difficult;

- Risk control and minimization strategies can be effective in guaranteeing the safety and reliability of recycled water use on public health and the environment in recycled water schemes. The corresponding solutions may include the source separation, wastewater treatment and quality improvement, Hazard Analysis and Critical Control Points (HACCP) control, and exposure control.

2. Conceptual principle and assessment for analysis of new end uses in recycled water schemes

- Recycled water can contribute to considerable freshwater savings, reduced effluent discharge, dual pipe system expansion, and minimized health risks and system failures through water quality improvement. The qualitative feasibility analysis indicated that these strengths and opportunities of recycled water could benefit the development of three potential new end uses in practice, namely household laundries, livestock feeding and servicing and swimming pools;

- There were also some constraints limiting the establishment and smooth implementation of these three new recycled water end uses. The weaknesses and threats might include additional costs for dual pipe system connection, the lack of comprehensive quantitative assessment, and public concerns on colour, odour, disease transmission and increased cost;

- The quantitative assessment concept allowed the multiplicity of perspectives, technologies and management practices in sustainable recycled water management to be explored, which could facilitate the decision making with a clear, sound and reliable strategy further.

3. Multi-criteria analysis towards the new end use of recycled water schemes for a household laundry: a case study in Sydney

- Based on the case study conducted at Rouse Hill development area, Sydney, the simplified Multi-criteria Analysis (MCA) with the Multi-attribute Utility Theory (MAUT) technique showed that recycled water for a household laundry was the optimum solution. This new end use option best satisfied the overall evaluation criteria compared to another two management options, namely the recycled water for swimming pools and the implementation of Level 1 water restriction on the use of recycled water;

- The complex MCA study was achieved by the integrated Rank Order Weight Generation plus Preference Ranking Organization Method for Enrichment Evaluation (ROWG-PROMETHEE) model. It was demonstrated that the MF treated recycled water coupled with new washing machines and the MF-Granular Activated Carbon (GAC) treated recycled water coupled with existing washing machines were preferred options.

4. Analysis of social attitude for the new end uses in recycled water schemes

- Through extensive social attitude surveys in Port Macquarie, New South Wales (NSW) and Melbourne, Victoria, the regression relationships between predicting variables and the public acceptance on recycled water use in a household laundry were established. The variables significantly contributing to the acceptance of this new end use include: i) positive attitude on receiving recycled water, ii) positive opinion on the idea "recycled water is an alternative to drinking water", iii) increased confidence by reading from other customers or successful examples, and iv) increased confidence by adding a small unit to improve the water quality. The fear of the potential odour and high cost when using recycled water for a household laundry became the main factors preventing respondents from being supportive of this new end use;

- Sydney residents have already established good cognitions on the appearance and cost of recycled water from their current using experience. They were more concerned about the colour of clothes and potential damage to washing machines. Notably, similar to other participants, Sydney respondents' confidence is likely to be increased by having a small unit for pre-treatment and knowing that recycled water is being used in the laundry by other customers;

- The community's knowledge, trust and confidence on water saving and current recycled water status can be built through several effective ways such as corresponding measures, workshops and educational campaigns. Strong public supports are beneficial to the smooth expansion of recycled water supply or the introduction of new applications.

5. A new optional recycled water purification system prior to use in the household laundry

- The recycled water column experiment results revealed that the pre-conditioning of natural zeolites contributed to improved surface morphology and increased sodium contents on particles, resulting in higher ion exchange capacity;

- Under the contact time of 5 minutes with a flow rate of 18.6 L/h, the zeolite column was shown to be a promising water purification unit prior to use in a washing machine with a considerable hardness ion removal. With the optimal contact time condition of 5 minutes, the maximum operational capacity and breakthrough capacity data were found to be 35 and 14 mg hardness ions /g of zeolites respectively;

- The zeolite column service life for a typical washing machine was shown to be one month without material regeneration. The regeneration process further guaranteed the simplicity of maintenance and durability of operation for actual application at household levels.

8.1.2 Final Conclusions

The thesis highlights that in both planning and assessment of recycled water schemes, adaptive management strategies for sustainable development need to be considered. Consequently, the thesis calls for a systematic framework and comprehensive MCA models in detailed characterization of options provision and decision making. On the other hand, the thesis addresses public attitudes of recycled water application in households through extensive social surveys. Accordingly, viable approaches for enhancing the public confidence have been established, including the adoption of zeolite column for additional recycled water quality improvement. The whole research could provide powerful guidance and advice for sustainable water reuse governance and management in the long term.

Notably, the exploration of practical tools for quick, simple and effective decision making in sustainable water systems remains an urgent goal. There is a pressing need to continue the research work on the refinement of models regarding the recycled water sustainability assessment and social attitude analysis. Such an effort will enable the models to work with other technologies in water resources market more smoothly and/or to be incorporated in integrated water resources planning and management.

8.2 RECOMMENDATIONS

Despite increasing implementation of recycled water schemes over the world, the insufficient and/or improper project planning and management still exist, leaving much room for further development. Based on the present study, the following recommendations for the future work are proposed:

a) As the benefits and high potential of applying recycled water for a household laundry have been demonstrated, the actual implementation of recycled water for clothes washing should be done over the country of Australia. By the means of site investigation, data collection and performance analysis, the MCA assessment models could be validated, modified and refined continuously so as to achieve more realistic and reliable outcomes for future recycled water decision making;

b) The developed regression models for social attitude analysis in two survey locations only take into account of several key predictor variables. To enhance the reliability, accuracy and effectiveness of the models further, other variables such as public risk perceptions, environmental concerns, information search and watching behaviour can be considered in future research. Furthermore, the regional effects on model outcomes can be reduced to some extent by implementing extensive case studies in other locations within or outside Australia;

c) The zeolite columns for recycled water purification prior to use in a washing machine were only investigated in the laboratory. The future field trial operations and on-site studies can be conducted in residential areas where recycled water has been supplied for indoor uses. The columns with different design volumes and flow rates can be connected with the recycled water pipe and the washing machine, and then tested further so as to facilitate the operation as well as the extension of service life in actual households.

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Appendix

Appendix A The full ROWG-PROMETHEE calculation programme

The commercial software version: MATLAB R2012b.

Description: The designed programme has been used for the full ROWG-PROMETHEE calculation in Chapter 5.

```
clear;
clc;
%% Water quality
load WaterQualitySubData.mat
Scenarios=textdata(:,1);

P=[1, 2, 1.5, 0.1, 0.9, 0.01, 2];
Sub_weight=[0.066 0.1 0.067 0.067 0.3 0.2 0.2];   % Sub-weighting of water quality
% m is the number of option, j is the number of criteria, i is the number of option
% 3.Concave shape
  %if d<=0 Pd=0;
  %elseif d<=p  Pd=-sqrt(1-(1-d./p).^(2));
  %else Pd=1;
for m=1:5;
  for j=1:7; % Here the preference function for price is V-shape.
 for i=1:5;
     d=-(data(m,j)-data(i,j));
    if d<=0 PV(i,j,m)=0;
   elseif d<=P(j) PV(i,j,m)=sqrt(1-(1-d./P(j)).^(2));
   else PV(i,j,m)=1;
    end
 end
  end
end
%
for m=1:5;
for j=1:7;
 for i=1:5;
     d=-(data(i,j)-data(m,j));
    if d<=0 PM(i,j,m)=0;
   elseif d<=P(j) PM(i,j,m)=sqrt(1-(1-d./P(j)).^(2));
   else PM(i,j,m)=1;
    end
 end
end
end
for i=1:5
    WaterQuality_positive(i)=sum(PV(:,:,i)*Sub_weight')/4;
    WaterQuality_negative(i)=sum(PM(:,:,i)*Sub_weight')/4;
end
```

```
WaterQualityScore=WaterQuality_positive-WaterQuality_negative;

load TotalScoreData.mat
TotalScoreData(:,3)=WaterQualityScore';
TotalScoreData=TotalScoreData';
Criteria2=Criteria';

%% Ask decision maker to choose the importance order of the criteria

CI_1 = menu('Choose the 1st important criterion',Criteria2);
reranked_TSD=TotalScoreData(CI_1,:);

Criteria2{CI_1}=' ';
CI_2 = menu('Choose the 2nd most important criterion',Criteria2);
reranked_TSD=[reranked_TSD;TotalScoreData(CI_2,:)];

Criteria2{CI_2}=' ';
CI_3 = menu('Choose the 3rd most important criterion',Criteria2);
reranked_TSD=[reranked_TSD;TotalScoreData(CI_3,:)];

Criteria2{CI_3}=' ';
CI_4 = menu('Choose the 4th most important criterion',Criteria2);
reranked_TSD=[reranked_TSD;TotalScoreData(CI_4,:)];

Criteria2{CI_4}=' ';
CI_5 = menu('Choose the 5th most important criterion',Criteria2);
reranked_TSD=[reranked_TSD;TotalScoreData(CI_5,:)];

Criteria2{CI_5}=' ';
CI_6 = menu('Choose the 6th most important criterion',Criteria2);
reranked_TSD=[reranked_TSD;TotalScoreData(CI_6,:)];

Criteria2{CI_6}=' ';
CI_7 = menu('Choose the 7th most important criterion',Criteria2);
reranked_TSD=[reranked_TSD;TotalScoreData(CI_7,:)];

Criteria2{CI_7}=' ';
CI_8 = menu('Choose the 8th most important criterion',Criteria2);
reranked_TSD=[reranked_TSD;TotalScoreData(CI_8,:)];

Criteria2{CI_8}=' ';
CI_9 = menu('Choose the 9th most important criterion',Criteria2);
reranked_TSD=[reranked_TSD;TotalScoreData(CI_9,:)];

Criteria2{CI_9}=' ';
CI_10 = menu('Choose the 10th most important criterion',Criteria2);
reranked_TSD=[reranked_TSD;TotalScoreData(CI_10,:)];

Criteria2{CI_10}=' ';
CI_11 = menu('Choose the 11th most important criterion',Criteria2);
```

```
reranked_TSD=[reranked_TSD;TotalScoreData(CI_11,:)];

Criteria2{CI_11}=' ';
CI_12 = menu('Choose the 12th important criterion',Criteria2);
reranked_TSD=[reranked_TSD;TotalScoreData(CI_12,:)];
reranked_TSD=reranked_TSD';
TSD=TotalScoreData';
%%
CI=[CI_1,CI_2,CI_3,CI_4,CI_5,CI_6,CI_7,CI_8,CI_9,CI_10,CI_11,CI_12];
reranked_Criteria=Criteria(CI);
PP=[0.6,2.5,0.5,0.5,4000,4500,2.5,0.15,2.5,2.5,1000000,20000];
Q=[NaN, 1, NaN, NaN,NaN,NaN,1,NaN,1,1,10000,1000];
PPranked=PP(CI);
Qranked=Q(CI);

%% Generate radom weight for each parameter
nsamples=10000;
ww=[];
for i=1:12;
    w{i}=rand(1,nsamples);
    ww=[ww;w{1,i}];
end

    ww=sort(ww,'descend');

    i=2;
     for j=1:nsamples;
         k(i-1,j)=1-ww(i-1,j);
         k(i,j)=ww(i-1,j)-ww(i,j);
     end
     for i=3:11;
         for j=1:nsamples;
         k(i,j)=ww(i-1,j)-ww(i,j);
    end
     end
     i=12;
     for j=1:nsamples;
         k(i,j)=ww(i-1,j)-0;
     end
     kk=sort(k,'descend'); % Rank weights of each parameter
% m is the number of option, j is the number of criteria, i is the number of option


%% Positive
for m=1:5;
    for j=1:12;
  if CI(j)==1||CI(j)==5||CI(j)==6 % Here the preference function for price is V-shape.

      for i=1:5; % V-shape-Min
     d=-(reranked_TSD(m,j)-reranked_TSD(i,j));
```

```
    if d<=0 PVV(i,j,m)=0;
   elseif d<=PPranked(j) PVV(i,j,m)=d./PPranked(j);
   else PVV(i,j,m)=1;
    end
    end
  elseif CI(j)==3||CI(j)==4||CI(j)==8

for i=1:5; % V-shape-Max
     d=reranked_TSD(m,j)-reranked_TSD(i,j);
    if d<=0 PVV(i,j,m)=0;
   elseif d<=PPranked(j) PVV(i,j,m)=d./PPranked(j);
   else PVV(i,j,m)=1;
    end
end
  elseif CI(j)==2||CI(j)==7||CI(j)==9||CI(j)==10

      for i=1:5;%Level
     d=reranked_TSD(m,j)-reranked_TSD(i,j);
    if d<=Qranked(j) PVV(i,j,m)=0;
   elseif d<=PPranked(j) PVV(i,j,m)=1/2;
   else PVV(i,j,m)=1;
    end
 end
  elseif CI(j)==11||CI(j)==12

for i=1:5;%Linear
     d=-(reranked_TSD(m,j)-reranked_TSD(i,j));
    if d<=Qranked(j) PVV(i,j,m)=0;
   elseif d<=PPranked(j) PVV(i,j,m)=(d-Qranked(j))./(PPranked(j)-Qranked(j));
   else PVV(i,j,m)=1;
    end
 end
  end
    end
end

P1_P=sum(PVV(:,:,1)*kk)/4;
P2_P=sum(PVV(:,:,2)*kk)/4;
P3_P=sum(PVV(:,:,3)*kk)/4;
P4_P=sum(PVV(:,:,4)*kk)/4;
P5_P=sum(PVV(:,:,5)*kk)/4;

PT_positive=[P1_P;P2_P;P3_P;P4_P;P5_P];


 %%  Negative
for m=1:5;
    for j=1:12;
  if CI(j)==1||CI(j)==5||CI(j)==6 % Here the preference function for price is V-shape.
```

```
     for i=1:5; % V-shape-Min
    d=-(reranked_TSD(i,j)-reranked_TSD(m,j));
   if d<=0 PMM(i,j,m)=0;
  elseif d<=PPranked(j) PMM(i,j,m)=d./PPranked(j);
  else PMM(i,j,m)=1;
   end
   end
 elseif CI(j)==3||CI(j)==4||CI(j)==8

for i=1:5; % V-shape-Max
    d=reranked_TSD(i,j)-reranked_TSD(m,j);
   if d<=0 PMM(i,j,m)=0;
  elseif d<=PPranked(j) PMM(i,j,m)=d./PPranked(j);
  else PMM(i,j,m)=1;
   end
end
 elseif CI(j)==2||CI(j)==7||CI(j)==9||CI(j)==10

     for i=1:5;%Level
    d=reranked_TSD(i,j)-reranked_TSD(m,j);
   if d<=Qranked(j) PMM(i,j,m)=0;
  elseif d<=PPranked(j) PMM(i,j,m)=1/2;
  else PMM(i,j,m)=1;
   end
 end
 elseif CI(j)==11||CI(j)==12

for i=1:5;%Linear
    d=-(reranked_TSD(i,j)-reranked_TSD(m,j));
   if d<=Qranked(j) PMM(i,j,m)=0;
  elseif d<=PPranked(j) PMM(i,j,m)=(d-Qranked(j))/(PPranked(j)-Qranked(j));
  else PMM(i,j,m)=1;
   end
 end
 end
   end
end

P1_N=sum(PMM(:,:,1)*kk)/4;
P2_N=sum(PMM(:,:,2)*kk)/4;
P3_N=sum(PMM(:,:,3)*kk)/4;
P4_N=sum(PMM(:,:,4)*kk)/4;
P5_N=sum(PMM(:,:,5)*kk)/4;

PT_negative=[P1_N;P2_N;P3_N;P4_N;P5_N];

Pi=PT_positive-PT_negative;

%%
 [~, ind]=max(Pi);
```

```
option1=0;
option2=0;
option3=0;
option4=0;
option5=0;

k1=kk(1,:); % Highest weight is given to the most important parameter
k2=kk(2,:); % Second Highest weight is given to the second most important parameter
%%
figure
 for i=1:length(ind);

   if ind(i)==1
              plot(k1(i), k2(i), 'bx');  hold on; option1=option1+1;
   else
   end
 end
 for i=1:length(ind);
      if ind(i)==2
              plot(k1(i), k2(i), 'r.');  hold on; option2=option2+1;
      else
      end
 end
 for i=1:length(ind);
      if ind(i)==3
             plot(k1(i), k2(i), 'k+');  hold on; option3=option3+1;
      else
      end
 end
 for i=1:length(ind);
      if ind(i)==4
              plot(k1(i), k2(i), 'g^');  hold on; option4=option4+1;
      else
      end
 end
 for i=1:length(ind);
      if ind(i)==5
             plot(k1(i), k2(i), 'm.');  hold on; option5=option5+1;
      else
      end
 end

 option=[option1 option2 option3 option4 option5]
 option_percent=option./nsamples.*100
```