Urban Water Security Research Alliance

Science Forum
and Stakeholder Engagement

Building Linkages, Collaboration and Science Quality

14-15 September 2011
Brisbane, Queensland

Program and Abstracts

Queensland Government

National Research FLAGSHIPS Water for a Healthy Country

CSIRO

Griffith UNIVERSITY

The University Of Queensland AUSTRALIA

UWSRA Science Forum and Stakeholder Engagement – Program and Abstracts ISSN 1839-1230 (Print)
UWSRA Science Forum and Stakeholder Engagement – Program and Abstracts ISSN 1839-1249 (Online)

The Urban Water Security Research Alliance (UWSRA) is a $50 million partnership over five years between the Queensland Government, CSIRO's Water for a Healthy Country Flagship, Griffith University and The University of Queensland. The Alliance has been formed to address South-East Queensland's emerging urban water issues with a focus on water security and recycling. The program will bring new research capacity to South-East Queensland tailored to tackling existing and anticipated future issues to inform the implementation of the Water Strategy.

For more information about the:

UWSRA - visit http://www.urbanwateralliance.org.au/
Queensland Government - visit http://www.qld.gov.au/
Water for a Healthy Country Flagship - visit www.csiro.au/org/HealthyCountry.html
The University of Queensland - visit http://www.uq.edu.au/
Griffith University - visit http://www.griffith.edu.au/

Enquiries should be addressed to:

The Urban Water Security Research Alliance
PO Box 15087
CITY EAST QLD 4002

Ph: 07-3247 3005; Fax: 07- 3405 3556
Email: Sharon.Wakem@qwc.qld.gov.au

Begbie, D.K. and Wakem, S.L. (eds) (2011), *Science Forum and Stakeholder Engagement: Building Linkages, Collaboration and Science Quality*, Urban Water Security Research Alliance, 14-15 September, Brisbane, Queensland.

Copyright

© 2011 CSIRO To the extent permitted by law, all rights are reserved and no part of this publication covered by copyright may be reproduced or copied in any form or by any means except with the written permission of CSIRO.

Disclaimer

The partners in the UWSRA advise that the information contained in this publication comprises general statements based on scientific research and does not warrant or represent the accuracy, currency and completeness of any information or material in this publication. The reader is advised and needs to be aware that such information may be incomplete or unable to be used in any specific situation. No action shall be made in reliance on that information without seeking prior expert professional, scientific and technical advice. To the extent permitted by law, UWSRA (including its Partner's employees and consultants) excludes all liability to any person for any consequences, including but not limited to all losses, damages, costs, expenses and any other compensation, arising directly or indirectly from using this publication (in part or in whole) and any information or material contained in it.

Contents

Welcome from the Chair **5**

The Alliance Research Program and Project Leaders **6**

Alliance Research Projects 2011-2012 - Poster **7**

Program **8**

Papers **13**

Climate Change Impact on Water Availability in the South East Queensland Region 14

A Summary of Social, Technological and Contextual Influences of Residential Water End Use Consumption in South East Queensland 20

Predictors of Household Water Use in South East Queensland 25

Estimating Water Savings from Mandated Rainwater Tanks in South East Queensland 31

Dissecting Rainwater Pump Energy Use in Urban Households 35

A GIS Approach for Identifying the Location and Attributes of Land Parcels for Stormwater Harvesting and Re-Use in Greenfield Urban Development 41

Assessing the Effectiveness of South East Queensland Frequent Flow Management Objectives against a Range of Urbanisation Extents for Tingalpa Creek 47

Evaluation of Reactive Toxicity towards Proteins for Application in Water Quality Assessment 53

Evaluating a Science Communication Workshop as an Educational Tool for the Application of Bioassays 60

Pathogen Decay in a Reservoir Impacted by the January Floods 65

Prevalence of Human Pathogens and Indicators in Stormwater Runoff in Brisbane 68

Potential Health Risks Associated with the Use of Roof-Harvested Rainwater 74

Assessing the Input of Hospitals to the Amount of Pharmaceutical Residues in Municipal Wastewater: A Consumption-Based Approach 80

Investigating the Role of Adsorption and Biodegradation in the Removal of Organic Micropollutants during Biological Activated Carbon Filtration of Treated Wastewater 85

Recent Advances in Field-Based Water Quality Monitoring Techniques 92

Disinfection By-Products in South East Queensland: Assessing Potential Effects of Transforming Disinfectants in the SEQ Water Grid 94

How Does Natural Organic Matter Removal Impact on Disinfection By-Product Formation? 102

Prudent and Efficient: The Use of Extended Cost-Effectiveness Analysis for Total Water Cycle Management Plans 105

Externality Effects and their Monetary Values for Water Servicing Options in South East Queensland 112

Understanding Uncertainty in Multi-Objective Decision Analysis: the Water Planning Context 119

Water-Related Energy and Greenhouse Gas Emissions in Cities - Key Emerging Messages 126

Methane Emissions from Two Reservoirs in a Steep, Sub-Tropical Rainforest Catchment 129

Environmental Risks and Benefits from the Supply of Purified Recycled Water in the Lockyer Valley 136

Assessment of New Chemical Film Monolayers for use as Long-Term Evaporation Reduction Systems 143

Application Systems for Optimal Management of Monolayer Films 150

Effect of a Monolayer on Water Quality at the Air-Water Interface 157

How Long Will a Rainwater Tank Last? Do You Know? 163

Rainwater Tank Adoption in South East Queensland – Factors Influencing Maintenance and Management 170

Electrochemical Treatment of Problematic Reverse Osmosis Membrane Concentrates 175

Poster - Abstracts **183**

Posters - Presented **189**

Unpacking Residential Water End Use Diurnal Patterns to Understand Capital Efficiency Opportunities 190

Development of a Domestic Water End Use Consumption Forecasting Model for South East Queensland 191

Influence of Organic Carbon in Bioassays: Role of Assimilable Organic Carbon 192

Disinfection By-Products (DBPs) in Southeast Queensland: Natural Organic Matter (NOM) Characterisation and Adsorptive DBPs Precursor Removal Strategies 193

Ozonation and Biological Activated Carbon Filtration of Wastewater Treatment Plant Effluents 194

Tracking the Source of Escherichia coli in SEQ Catchments 195

Outcome of the Assessment of Factors Influencing the Decay of Pathogens in South East Queensland Reservoirs 196

Wind and Wave Testing of a New Evaporation Reduction Monolayer System 197

Crop Mapping in the Lockyer Valley using Satellite Imagery 198

Deep Drainage Estimates Under Irrigated Cropping in the Lockyer Valley 199

Salt Loads and Water Storage Changes in the Regolith in the Lockyer Valley 15 Years on - the Ellis Sites Revisited 200

Awards **201**

Delegates **203**

Welcome from the Chair

Water is fundamental to our quality of life, economic growth and the environment. The **Urban Water Security Research Alliance (UWSRA)** is a $50 million partnership over five years between the Queensland Government, CSIRO's Water for a Healthy Country Flagship, Griffith University and The University of Queensland. The Alliance was formed to address Australia's South East Queensland's (SEQ) emerging urban water issues with a focus on water security and recycling. With its booming economy and growing population, SEQ faces increasing pressure on its water resources, compounded by the impact of climate variability and climate change.

The Alliance is the largest regionally focused urban water research program in Australia. The program brings new research capacity to SEQ tailored to tackling existing and anticipated future issues to inform the implementation of the SEQ Water Strategy.

Alliance Research Framework

Since October 2007 when the Alliance was formed, the water situation for SEQ has changed dramatically. The Alliance has responded accordingly to realign its research program, with a greater focus on reducing water grid demand, ensuring the quality of our diverse water sources and planning for efficiency and sustainability.

Research for the Alliance will be delivered under three research themes:

1. Reducing Water Grid Demand
2. Water Source Quality
3. Total Water Cycle Planning and Management to Enhance Sustainability and Efficiency

Research Program Objectives

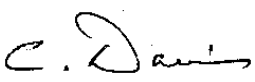
- Undertake research into off-Grid supply sources, water use efficiency and demand management behavioural measures to reduce demand on the SEQ Water Grid by about 35 GL per annum in 2026 and defer new infrastructure by up to five years.
- Undertake research to inform water quality management planning, regulation, guidelines and communication on the nature and level of risk to human health from a wide range of source waters within SEQ.
- Undertake integrated urban water planning and management research to transform to a water smart SEQ region and enhance management efficiencies of the SEQ Water Grid and off-grid supplies.

The Alliance research program currently consists of 18 inter-related projects, each managed by one of the research partners (see page 6 for more details).

The Alliance will seek to align research where appropriate with other water research programs such as those of other local SEQ water agencies, The Australian Water Recycling Centre of Excellence, Water Quality Research Australia Limited (WQRA), the Water Services Association of Australia (WSAA) and Cooperative Research Centres (CRCs). In undertaking projects, consideration will be given to the direction, achievements and gaps in current research and alignment with the focus and goals of the Alliance.

The objectives of the Science Forum are to increase awareness of the science emerging across the Alliance and to increase key stakeholders' understanding of the current research findings of our projects. The intended audience includes key stakeholders, external reference panel experts and scientists undertaking urban water research in SEQ.

It is with great pleasure that I welcome you to the Alliance's 3rd Science Forum being held on Wednesday, 14 and Thursday, 15 September 2011, in Brisbane.



Chris Davis
Chair, Urban Water Security Research Alliance

Research Programs and Projects

PROJECT TITLE	PROJECT DESCRIPTION	PROJECT LEADER	CONTACT INFORMATION
PROGRAM 1 – REDUCING WATER GRID DEMAND			
Stormwater Harvesting and Reuse	This project researches the innovative capture and storage of stormwater for additional water supply in SEQ. It will also research the impact of harvesting stormwater on creek and ecosystem health.	Dr Brian McIntosh CSIRO	07-3123 7766 b.mcintosh@watercentre.org
Decentralised Systems	This project will validate the contribution rainwater tanks can make to water savings targets in SEQ. It will also focus on energy costs associated with tanks and decentralised wastewater treatment and appropriate tank maintenance approaches.	Dr Ashok Sharma CSIRO	03-9252 6151 Ashok.Sharma@csiro.au
Demand Management and Communication Research	This project is researching household water conserving behaviour with demand management interventions.	Dr John Gardner CSIRO	07-3833 5552 John.Gardner@csiro.au
Residential Water End Use Study	This project is quantifying residential water end uses and the impact of urban water demand management strategies in SEQ.	Dr Rodney Stewart GU	07-5552 8778 r.stewart@griffith.edu.au
PROGRAM 2 – WATER SOURCE QUALITY			
Hospital Wastewater	This project will focus on the relative contribution of hospital wastewater to sewage treatment plants.	Dr Kristell Le Corre UQ	07-3346 7207 k.lecorre@awmc.uq.edu.au
Pathogens and Trace Contaminants in Dams	This project will focus on pathogen and trace contaminant attenuation within reservoirs and developing source tracking methods.	Dr Simon Toze CSIRO	07-3214 2698 Simon.Toze@csiro.au
Bioassays and Risk Communication	This project will further develop the scientific, technical and communication basis for implementation of bioanalytical tools in Queensland's water quality monitoring programs.	Dr Beate Escher UQ	07-3274 9180 b.escher@uq.edu.au
Health Risk Assessment of Local Source Waters	This project will research the health risks associated with exposure to rainwater and stormwater.	Dr Simon Toze CSIRO	07-3214 2698 Simon.Toze@csiro.au
Enhanced Treatment	The project will research the effectiveness of biological activated carbon in non MF/RO options to achieve potable water quality.	Dr Julien Reungoat UQ	07-3346 3235 j.reungoat@awmc.uq.edu.au
Assessment of Regulated and Emerging Disinfection By-Products in SEQ Drinking Water	This project will investigate the presence of emerging and regulated disinfection by-products in different SEQ drinking water sources.	Dr Maria José Farré Olalla UQ	07-3346 3233 m.farre@awmc.uq.edu.au
PROGRAM 3 – TOTAL WATER CYCLE PLANNING AND MANAGEMENT TO ENHANCE SUSTAINABILITY AND EFFICIENCY			
Climate and Water	This project will assess and quantify the impact of climate variability and change on water supply over the SEQ region.	Dr Wenju Cai CSIRO	03-9239 4419 Wenju.Cai@csiro.au
PRW in the Lockyer Valley	This project will research the implications of using PRW as an adjunct to groundwater for irrigation in the Lockyer Valley.	Dr Leif Wolf CSIRO	07-3214 2749 Leif.Wolf@csiro.au
Total Water Cycle Planning Framework	The project aims to provide the analytical methods and data required to improve the quantitative assessment and decision-making for the development of total water cycle management plans in SEQ.	Dr Shiroma Maheepala CSIRO	03-9252 6072 Shiroma.Maheepala@csiro.au
Water Quality Monitoring Technology and Information Collection System	This project has focused on developing a proof-of-concept prototype on-line, real-time monitoring system to identify sudden changes in the water matrix at a wastewater treatment plant at Bundamba in SEQ.	Prof Huijun Zhao GU	07-5552 8261 h.zhao@griffith.edu.au
Evaporation Loss	This project is analysing technology to reduce evaporative losses from large SEQ dams and is undertaking detailed field analysis of the effectiveness of monolayers on a large farm dam.	Adj/Prof Stewart Burn CSIRO	03-9252 6032 Stewart.Burn@csiro.au
Institutional Change for Water Management	This project has identified factors and relationships necessary for building effective and efficient long term institutional arrangements for the strategic management of water in SEQ.	Prof Brian Head UQ	07-3346 7450 Brian.Head@uq.edu.au
Water Smart Cities	This project will scope a framework for moving to Water Smart cities.	Dr Sharon Biermann CSIRO	07-3247 3007 sharon.biermann@csiro.au
Human Factors in Urban Water System Safety	This project will scope human elements, such as judgement, decision making, perception, in risk management and quality control systems.	Prof Brian Head UQ	07-3346 7450 Brian.Head@uq.edu.au

Alliance Research Program 2011-2012

Leading water research and knowledge in South East Queensland

Urban Water Security Research Alliance

Queensland Government · CSIRO · Griffith University · The University Of Queensland Australia

1. Climate and Water
Wenju Cai

2. Pathogens and Trace Contaminants in Dams
Simon Toze

3. Evaporation Loss
Stewart Burn

4. Hospital Wastewater
Kristell Le Corre

5. PRW in the Lockyer
Leif Wolf

6. Bioassays and Risk Communication
Beate Escher

7. Demand Management and Communication Research
John Gardner

8. Residential Water End Use Study
Rodney Stewart

9. Assessment of DBP in SEQ Drinking Water
Maria José Farré

10. Enhanced Treatment
Julien Reungoat

11. Water Quality Monitoring Technology and Information Collection System
Huijun Zhao

12. Water Smart Cities
Sharon Biermann

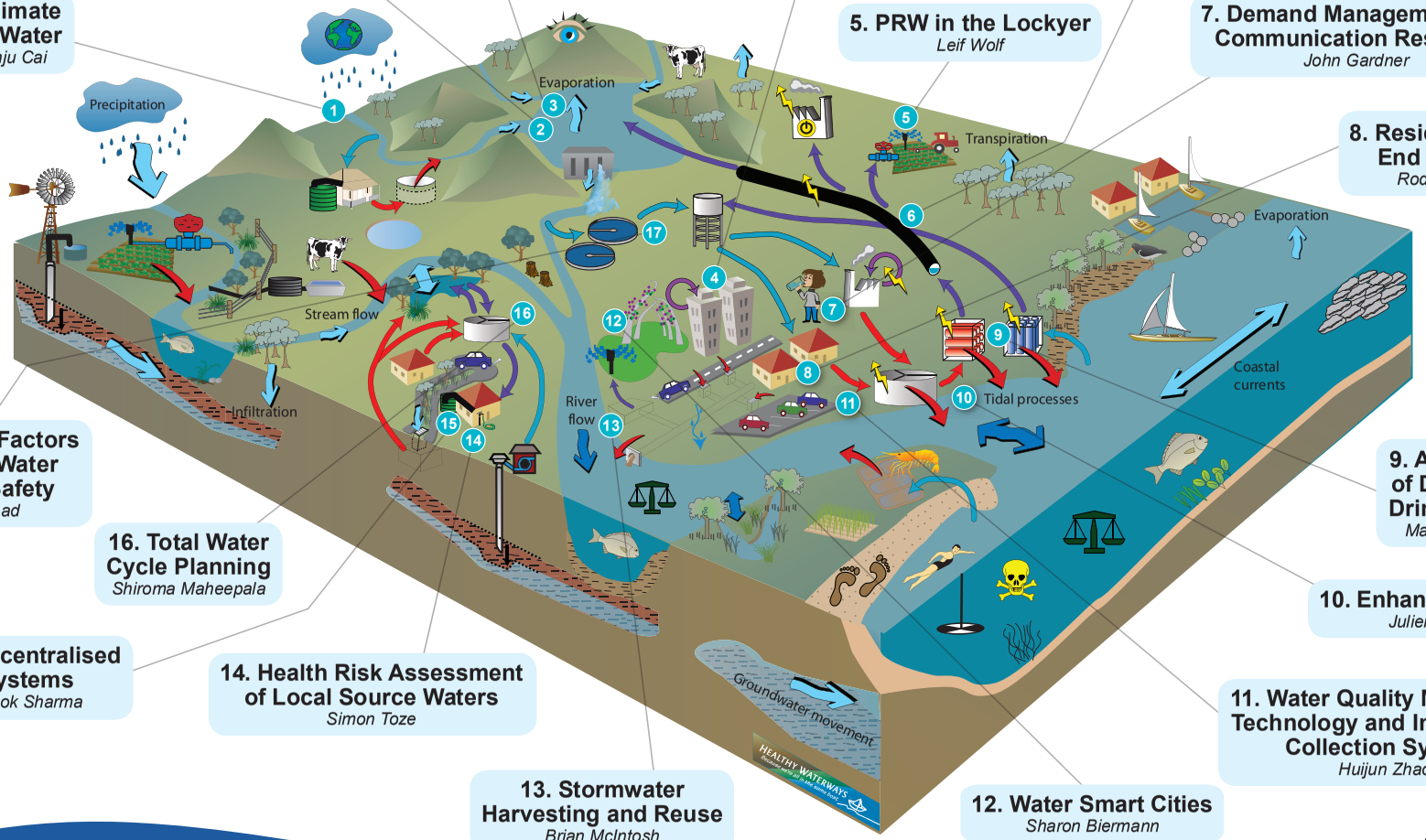
13. Stormwater Harvesting and Reuse
Brian McIntosh

14. Health Risk Assessment of Local Source Waters
Simon Toze

15. Decentralised Systems
Ashok Sharma

16. Total Water Cycle Planning
Shiroma Maheepala

17. Human Factors in Urban Water System Safety
Brian Head



Conceptual model source: Healthy Waterways Partnership

URBAN WATER SECURITY RESEARCH ALLIANCE

Science and Stakeholder Engagement Forum #3

Building Linkages, Collaboration and Science Quality

Wednesday, 14 September 2011, 8:30am – 6:30pm
Thursday, 15 September 2011, 8:00am – 5:00pm

Venue: Burke, Wills and Leichhardt Rooms - Level 2, Mercure Hotel, 85-87 North Quay, Brisbane

Program

DAY 1 – Wednesday, 14 September			
Time	**Session**	**Presenter**	
8:30am	*Registration and Welcome Coffee*		
9:00am	**Welcome and Opening Remarks**	Christine Williams Assistant Director-General Environment and Resource Sciences Division Department of Environment and Resource Management, Qld *Don Begbie, Director, UWSRA (MC)*	
9.05am	**Official Opening**	Hon Stephen Robertson MP Minister for Energy and Water Utilities	
9.30am	**Keynote Speaker**	Paul Doherty Manager Water Security Planning Strategy Planning and Regulation SA Water, Adelaide	
	Reducing Water Grid Demand		
10.00am	Climate Change Impact on Water Availability in the South East Queensland Region		Matthew Gooda
10:30am	*Morning tea*		
11.00am	**SESSION 1 - *(Chair – Vourn Lutton)***		
	A Summary of Social, Technological and Contextual Influences of Residential Water End Use Consumption in South East Queensland		Cara Beal
	Predictors of Household Water Use in South East Queensland		Kelly Fielding
	Estimating Water Savings from Mandated Rainwater Tanks in South East Queensland		Meng Chong
	Panel Discussion / Questions		
1:00pm	*Lunch*		
2.00pm	**SESSION 2 - *(Chair – Ted Gardner)***		
	Dissecting Rainwater Pump Energy Use in Urban Households		Grace Tjandraatmadja
	A GIS Approach for Identifying the Location and Attributes of Land Parcels for Stormwater Harvesting and Re-Use in Greenfield Urban Development		Brian McIntosh
	Assessing the Effectiveness of South East Queensland Frequent Flow Management Objectives against a Range of Urbanisation Extents for Tingalpa Creek		Stephanie Ashbolt
	Panel Discussion / Questions		
3:30pm	*Afternoon tea*		
4:00pm	**SESSION 3 – POSTERS - *(Chair – Rodney Stewart)***		
5:00pm	***Session Drinks and Networking Opportunity***		
6:30pm	**Close – Day 1**		

DAY 2 – Thursday, 15 September				
8:00am	*Registration and Welcome Coffee*			
Time	***Water Quality of Water Sources***	**Presenter**	***Total Water Cycle Planning and Management to Enhance Sustainability and Efficiencies***	**Presenter**
8.30am	**SESSION 4 - *(Chair – Beate Escher)***		**SESSION 5 - *(Chair – Tad Bagdon)***	
	Guest Speaker – Biological tools for assessment of the ecological impact of mixtures of micropollutants - lessons from Europe	Rolf Altenburger	**Guest Speaker -** Total Water Cycle Management Planning for best community outcomes - the South East Water experience	Greg Ryan
	Guest Speaker – *Working Title:* Emerging issues and research priorities for Pharmaceutical and Personal Care Products in the Environment - a perspective from Europe and North America	Alistair Boxall	Prudent and Efficient: The Use of Extended Cost-Effectiveness Analysis for Total Water Cycle Management Plans	Murray Hall
	Evaluation of Reactive Toxicity towards Proteins for Application in Water Quality Assessment	Janet Tang	Externality Effects and their Monetary Values for Water Servicing Options in South East Queensland	Peter Daniels
	Evaluating a Science Communication Workshop as an Educational Tool for the Application of Bioassays	Vicki Ross	Understanding Uncertainty in Multi-Objective Decision Analysis: the Water Planning Context	Magnus Moglia
	Panel Discussion / Questions		*Panel Discussion / Questions*	
10:30am	*Morning tea*			
11.00am	**SESSION 6 - *(Chair – Greg Jackson)***		**SESSION 7 - *(Chair – Wai-Tong Wong)***	
	Pathogen Decay in a Reservoir Impacted by the January floods	Simon Toze	Water-Related Energy and Greenhouse Gas Emissions in Cities - Key Emerging Messages	Steven Kenway
	Characterisation of Human Pathogens and Indicators in Stormwater Runoff in Brisbane	Jatinder Sidhu	Methane Emissions from Two Reservoirs in a Steep, Sub-Tropical Rainforest Catchment	Brad Sherman
	Potential Health Risks Associated with the Use of Roof-Harvested Rainwater	Warish Armed	Environmental Risks and Benefits from the Supply of Purified Recycled Water in the Lockyer Valley	Leif Wolf
	Panel Discussion / Questions		*Panel Discussion / Questions*	
12.30pm	*Lunch*			
1.30pm	**SESSION 8 - *(Chair – Yvan Poussade)***		**SESSION 9 - *(Chair – Mike Taylor)***	
	Assessing the Input of Hospitals to the Amount of Pharmaceutical Residues in Municipal Wastewater: A Consumption-Based Approach	Kristell Le Corre	Assessment of New Chemical Film Monolayers for use as Long-Term Evaporation Reduction Systems	Peter Schouten
	Investigating the Role of Adsorption and Biodegradation in the Removal of Organic Micropollutants during Biological Activated Carbon Filtration of Treated Wastewater	Maxime Rattier	Application Systems for Optimal Management of Monolayer Films	Erik Schmidt
	Recent Advances in Field-Based Water Quality Monitoring Techniques	Huijun Zhao	Effect of a Monolayer on Water Quality at the Air-Water Interface	Pam Pittaway
	Panel Discussion / Questions		*Panel Discussion / Questions*	
3:00pm	*Afternoon tea*			
3.30pm	**SESSION 10 - *(Chair – Paul Burrell)***		**SESSION 11 - *(Chair – Sue Larsen)***	
	Disinfection By-Products in South East Queensland: Assessing Potential Effects of Transforming Disinfectants in the SEQ Water Grid	Maria José Farré	How Long Will a Rainwater Tank Last? Do You Know?	Magnus Moglia
	How Does Natural Organic Matter Removal Impact on Disinfection By-Product Formation?	Niki Knight	Rainwater Tank Adoption in South East Queensland – Factors Influencing Maintenance and Management	Dave Tucker
	Panel Discussion / Questions		*Panel Discussion / Questions*	
4.40pm	**Awards for Best:** Paper, Presentation and Poster - Don Begbie			
5.00pm	**Close**			

OFFICIAL OPENING SPEAKER



Stephen Robertson MP
Minister for Energy and Water Utilities, QLD

Stephen Robertson is Queensland's Minister for Energy and Water Utilities (from 21 February 2011). Stephen is responsible for clean energy, energy efficiency and industry development, electricity, gas, geothermal, nuclear and renewable energy, and bulk water supply, distribution and retail.

Stephen was first elected to the Queensland Parliament in September 1992 as the member for the Brisbane southside electorate of Sunnybank (now Stretton). With the election of the first Beattie Government in 1998, he was appointed Parliamentary Secretary for State Development. Stephen also chaired the Queensland Small Business Council, was a member of the Queensland Innovation Council and the Queensland Food Industry Council. He was first appointed to the Ministry in December 1999 as Minister for Emergency Services. Following the March 2001 election, Stephen was promoted to Minister for Natural Resources and Minister for Mines. Following the 2004 election he became Minister for Natural Resources, Mines and Energy. In July 2005, he was appointed Minister for Health driving significant healthcare reform and the $10 billion Health Action Plan. After the 2009 election, Stephen returned to the portfolio of Natural Resources, Mines and Energy with 'Trade' added to those familiar responsibilities. Since February 2011 he assumed responsibility for Energy and Water Utilities to concentrate on wide reaching reforms in water and energy pricing policies and the government's clean energy agenda.

Stephen was born in Aberdeen Scotland, in 1962. He has a Bachelor of Arts (Hons) from the School of Modern Asian Studies, Griffith University. Prior to his election to parliament, Stephen accrued significant experience in industrial relations, working for a number of industrial organisations, including State Secretary and National President of the United Firefighters Union of Australia. Stephen recently announced that he will be retiring from politics at the next state election due by March 2012.

KEYNOTE SPEAKER



Paul Doherty
Manager Water Security Planning
Strategy Planning and Regulation
SA Water, Adelaide

Paul has spent 35 years in the water industry, the last 15 in water and wastewater planning. He was technical manager in developing the *Water Proofing Adelaide* plan released in 2005, and technical advisor in developing *Water for Good* in 2009, for which his key roles were the urban stormwater harvesting options study for Adelaide and refinements of the water security strategy. His current focus is on long term water security for Adelaide and South Australia's regional areas. Paul has a civil engineering degree and a graduate diploma in project management from the University South Australia, and a masters in infrastructure management from ANU.

GUEST SPEAKERS



Dr Rolf Altenburger

Head, Department of Bioanalytical Ecotoxicology, Helmholtz Centre for Environmental Research – UFZ, Leipzig, Germany.
Visiting Academic, National Research Centre for Environmental Toxicology (ENTOX), University of Queensland, Brisbane, Australia

Since 2005 Rolf is the Head of Department of Bioanalytical Ecotoxicology at Helmholtz Centre for Environmental Research – UFZ, Leipzig, Germany. Rolf also does academic teaching at the Universities of Bremen and Freiberg. Research fellowships to: Visaya State College of Agriculture, Leyte, The Philippines; Roswell Park Cancer Institute, Buffalo, NY, USA; and currently; the University of Queensland, National Research Centre for Environmental Toxicology (ENTOX), Brisbane, Australia. Since 2007: Speaker for HGF programme topic 'Chemicals in the Environment'.

Professional training as a biologist at the Universities of Marburg, Germany; St.Andrews, Scotland; Freiburg, and Bremen, Germany: Diploma 1984, PhD 1990.
Current Research Interests*: m*ode-of-action analysis for toxicants; Mixture toxicity assessment for complex environmental exposure; Bioassays for toxicant identification at multiple contaminated sites.
Professional activities: Member of advisory committees, e.g. EU-Scientific Committee on Ecological and Human Risks (SCHER), DIN/ISO group for Biotest standardisation; Regular reviewer to 15 environmental, toxicological and pesticide journals; Editorial board member to Environmental Pollution.



Dr Alistair Boxall

Reader, Environmental Science
Environment Department, University of York, Heslington, York, YO10 5DD, UK
Visiting Scientist, CSIRO, Adelaide, Australia

Alistair Boxall is a Reader in Environmental Science at the University of York and a Visiting Fellow at the Food and Environment Research Agency. He is based in the EcoChemistry Research Group which is a joint initiative between the University of York and FERA which specialises in the fate, effects and risk assessment of organic chemicals in the environment. With a broad-based training in environmental chemistry and toxicology from Leicester Polytechnic and the University of Sheffield, Alistair had previously worked at the Water Research Centre, Liverpool John Moores University and Cranfield University. He has a particular interest in the fate, behaviour and effects of emerging environmental contaminants (nanoparticles, pharmaceuticals and degradates) and co-ordinates a large number of research projects in this area. He sits on the UK Governments Advisory Committee on Hazardous Substances and is Chair of the SETAC Pharmaceutical Advisory Group. Alistair is currently a Distinguished Visiting Scientist at CSIRO's Adelaide laboratories.



Mr Greg Ryan

Manager, Environment and Technology
South East Water Limited, Melbourne

Greg has worked in the water industry for nearly 20 years, and been with South East Water for just over half of that. He has worked in many areas including stormwater management, treatment of wastewater, supply of fit for purpose water and been involved in establishment of guidelines for drinking and recycled water. He is currently the Manager of Environment and Technology, a strategic group responsible for establishing business direction in relation to Environment and Energy, R&D and knowledge management, he is also working part time for the Water Services Association Australia.

Papers

Climate Change Impact on Water Availability in the South East Queensland Region

Gooda, M.C., Voogt, S. and Vitkovský, J.P.
Department of Environment and Resource Management, Ecosciences Precinct, Dutton Park, Queensland

Summary

The availability of high-resolution dynamically downscaled climate models for South East Queensland (SEQ) has allowed an assessment of the sensitivity to the spatial resolution of climate change projections on water availability. This study uses the Integrated Quality Quantity Model (IQQM) to assess the impact of climate change projections from 11 Global Circulation Models (GCMs) and 12 dynamically downscaled GCMs using the CSIRO Cubic Conformal Atmospheric Model (CCAM). Dynamical downscaling is showing that the temporal pattern of climate change projections may change from the original GCM pattern, in addition to the inherent increase in spatial variability, leading to reduced water availability and increased time between storage filling events. The majority of downscaled GCMs show significantly lower inflow than the original GCM projection, and indicate that climate change assessments for water supply planning purposes in SEQ using GCMs may underestimate the impact on available water resources.

Keywords
Water availability, water security, downscaled climate change projections, CCAM, South East Queensland, IQQM.

Introduction

Recent drought in South East Queensland (SEQ) has prompted further consideration of climate variability and climate change when assessing the security of urban water supplies. This study examines the sensitivity of water availability to the spatial resolution of climate change projections. The Integrated Quality Quantity Model (IQQM) hydrologic model is used to assess the impact of climate change projections from 11 Global Circulation Models (GCMs) and 12 dynamically downscaled GCMs using the CSIRO Cubic Conformal Atmospheric Model (CCAM). This is done by applying climate change projections to long-term historic climate sequences, and analysing the performance of water storages using the IQQM calibrated for the Moreton catchment (Figure 1) by the Department of Environment and Resource Management (DERM).

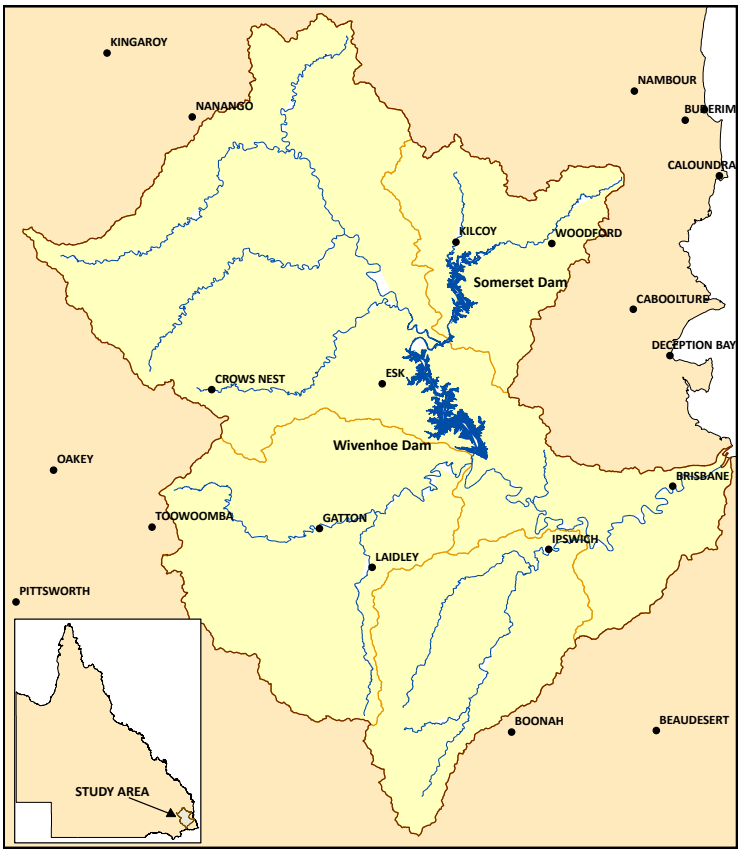


Figure 1. Moreton catchment map.

Climate change impact has been incorporated into the Moreton IQQM by scaling the historic rainfall, evaporation and streamflow inputs, for each climate change projection. The historic rainfall sequences are sourced from the SILO Data Drill dataset (http://www.longpaddock.qld.gov.au/silo/), consisting of daily timeseries at 0.05° x 0.05° grid points across the catchment. This gridded data is used to generate daily average subcatchment rainfall for the ~200 subcatchments used in the model. Historic evaporation data for the model has been extracted from the SILO patched point dataset at two evaporation stations.

Method

Future climate sequences for rainfall and evaporation are produced for each of the GCMs and dynamically downscaled climate models, by scaling the long term historic daily climate sequences (1889 to 2000). The scaling factors are calculated on a monthly basis, for each model, by determining the per degree of global warming response for each climate variable, multiplied by the best estimate of the change in temperature for the year 2050 under emissions scenario A1FI (from Table 4.3 of the Climate Change in Australia Report, 2007). This is consistent with the methodology that has been adopted by DERM for climate change analysis throughout Queensland, and provides a platform for comparing projections from GCMs that have been run under differing emissions scenarios.

Table 1 lists the combinations of models and emissions scenarios analysed. The 12 downscaled GCMs presented are the only models available for this study so far, and the 11 GCMs used have been selected by Queensland Climate Change Centre of Excellence (QCCCE), from the 23 International Panel on Climate Change AR4 GCMs, as the most appropriate for assessing the impact of climate change on rainfall in Queensland. Eight of the models used provide a direct comparison between the original and downscaled GCM climate change projections, with the additional models contributing to the separate GCM and downscaled GCM ensembles.

Table 1. List of GCMs and Downscaled GCMs used in this study

Global Circulation Model (GCM)		Cubic Conformal Atmospheric Model (CCAM) downscaling	
Model	**Emissions Scenario**	**Model**	**Emissions Scenario**
CSIRO Mark 3.0	A2	CSIRO Mark 3.0	A2
CSIRO Mark 3.5	A2	CSIRO Mark 3.5	A2
GFDL CM 2.1	A2	GFDL CM 2.1	A2
MIROC 3.2 (medres)	A2	MIROC 3.2 (medres)	A2
MPI ECHAM 5	A2	MPI ECHAM 5	A2
NCAR CCSM 3.0	A2	NCAR CCSM 3.0	A2
UKMO HADCM3	A2	UKMO HADCM3	A2
UKMO HADGEM1	A2	UKMO HADGEM1	A2
IAP FGOALS 1.0	A1B	INMCM 3.0	A2, A1B
MIROC 3.2 (hires)	A1B	CSIRO Mark 3.0	A1B
MIUB echo	A2	NCAR CCSM 3.0	A1B

Figures 2 and 3 show an example of the spatial variation of climate change effects on rainfall and evaporation from the downscaled CSIRO Mark 3.5 GCM, for the year 2050 under emissions scenario A1FI.

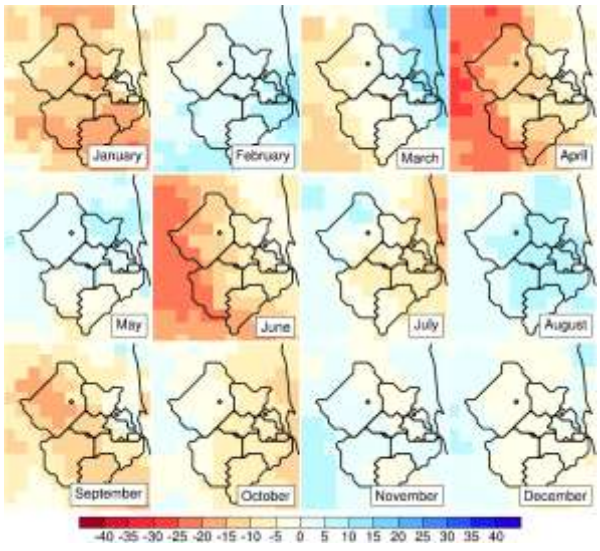


Figure 2. Monthly percentage rainfall change for downscaled CSIRO Mk 3.5 (for year 2050 SRES A1FI) (source Kent, D., CSIRO)

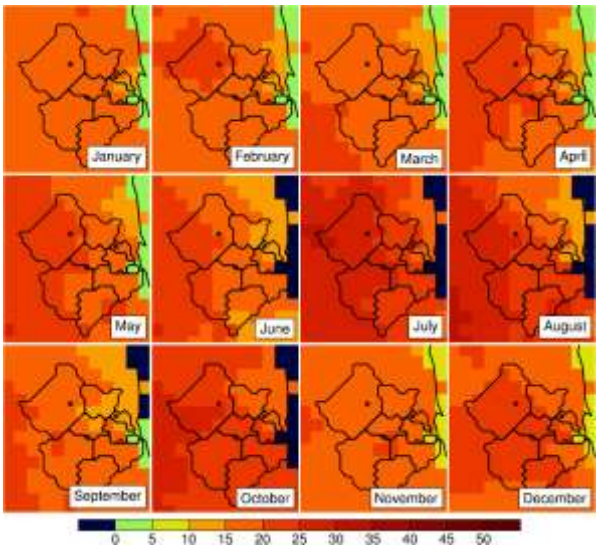


Figure 3. Monthly percentage evaporation change for downscaled CSIRO Mk 3.5 (for year 2050 SRES A1FI) (source Kent, D., CSIRO)

Streamflow inputs are generated by firstly running the climate change adjusted rainfall and evaporation sequences through the previously calibrated Sacramento (Burnash, Ferral and McGuire 1973) rainfall-runoff models. Then the daily flow duration curves from the climate change and historic climate Sacramento sequences are compared for each month, and the IQQM inflow sequence is scaled according to the change in the flow duration curves. This has the benefit of scaling flows of different magnitudes by different amounts, as opposed to applying a constant monthly scaling factor.

Results

Downscaling Effects on Hydrologic Model Inputs

Dynamical downscaling is showing that the temporal pattern of climate change projections may change from the original GCM pattern, in addition to the inherent increase in spatial variability. Figures 4 and 5 show the range of downscaled monthly percentage rainfall change at different locations across the Moreton catchment compared to the original GCM output, for the CSIRO Mark 3.5 and GFDL CM 2.1 GCMs, respectively. The figures clearly show that there is variation in both temporal and spatial rainfall change, which has significant implications to the runoff generation and water availability within the catchment.

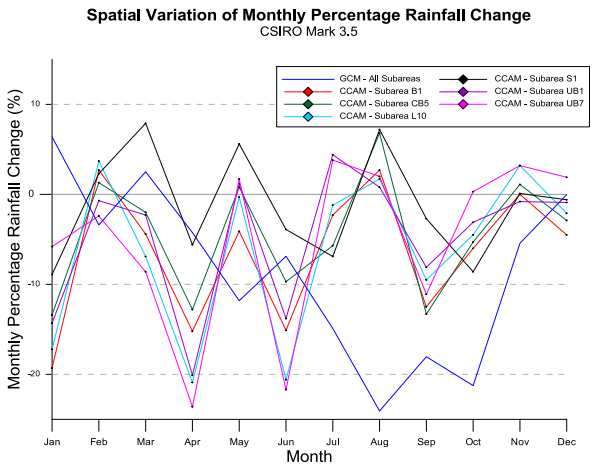


Figure 4. CSIRO Mk 3.5 Percentage Rainfall Change

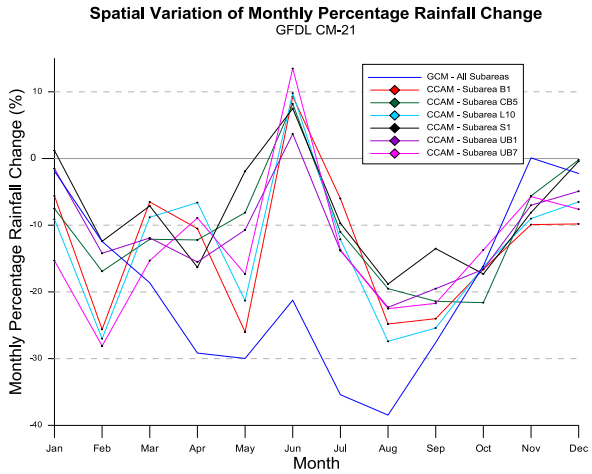


Figure 5. GFDL CM 2.1 Percentage Rainfall Change

Figures 6 and 7 show the range of mean monthly rainfall resulting from the GCM and downscaled simulations respectively for the Upper Brisbane #1 (UB1) subcatchment. It should be noted that the downscaled models have less agreement in rainfall projection than the GCMs in general, but the majority of downscaled models are showing a decrease during the wet season months (Dec-Feb), which are the main contributors to runoff generation.

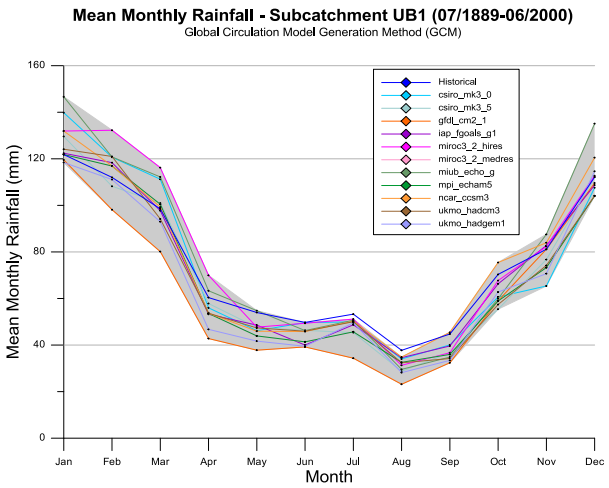


Figure 6. GCM Mean Monthly Rainfall

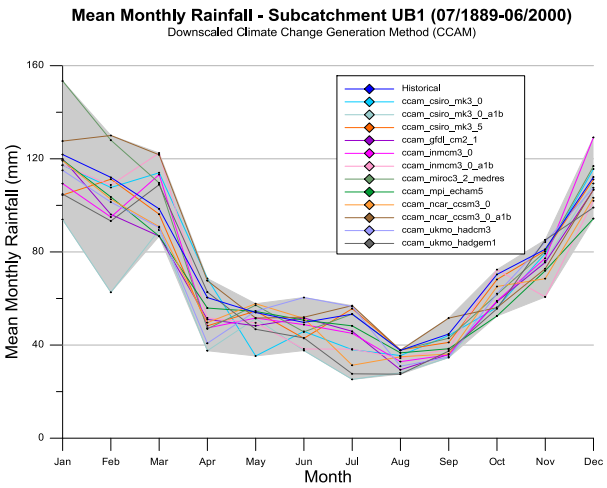


Figure 7. Downscaled GCM Mean Monthly Rainfall

The downscaling effect on evaporation change is more uniform than the rainfall change, and there is good agreement across all models analysed, with regard to the direction and magnitude of that change. The results show generally higher evaporation rates than the original GCM output. Figures 8 and 9 show the effect on mean monthly evaporation rates (as potential evapotranspiration) for both the GCM and downscaled GCM suites respectively.

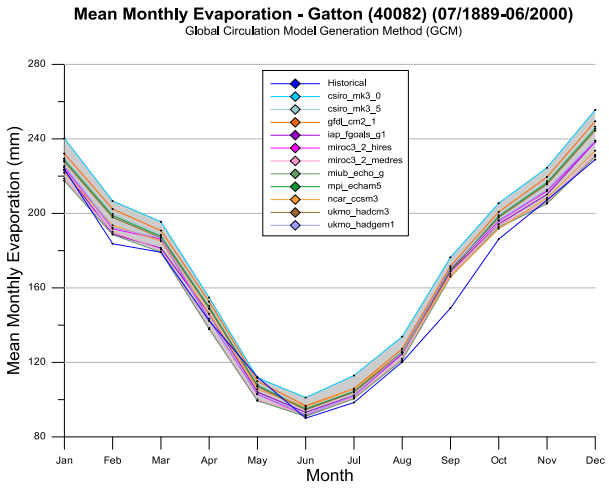


Figure 8. GCM Mean Monthly Evaporation

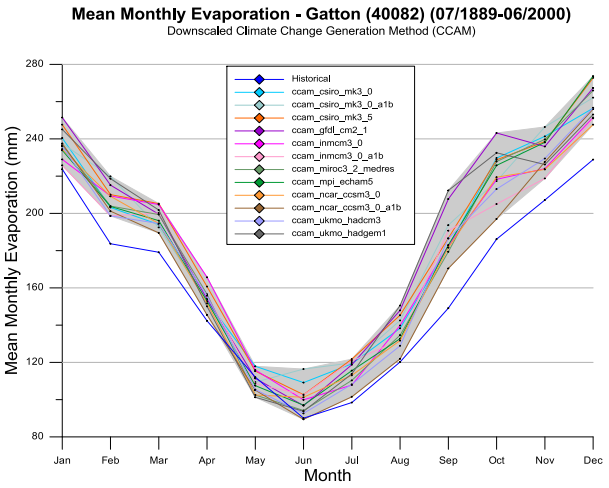


Figure 9. Downscaled GCM Mean Monthly Evaporation

The combined effect of the future climate rainfall and evaporation sequences on subcatchment inflow is shown in Figures 10 and 11. The majority of downscaled GCMs result in significantly lower inflow than the original GCM inflow. This result is expected, considering the effect of downscaling on both rainfall and evaporation sequences.

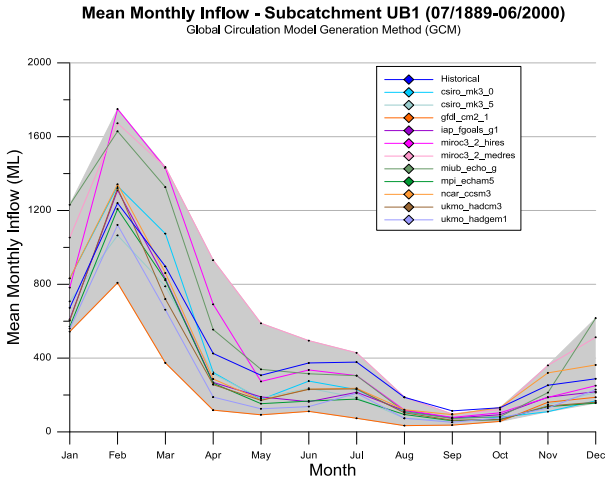


Figure 10. GCM Mean Monthly Inflow

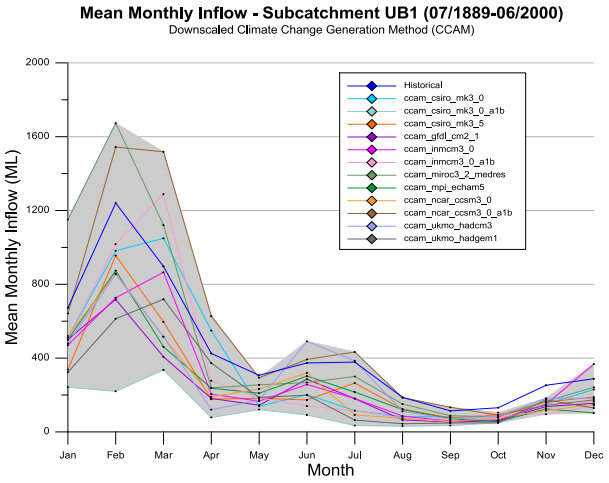


Figure 11. Downscaled GCM Mean Monthly Inflow

Downscaling Effects on Water Availability

For this study, water availability has been assessed using a similar method to the Level of Service (LoS) approach developed for the Regional Water Supply Strategy process in Queensland. The performance of the combined Wivenhoe and Somerset Dams is assessed using the frequency of years where the available storage volume falls below specific trigger levels, expressed as an average recurrence interval. The trigger levels for this analysis have been arbitrarily chosen as 75%, 50% and 25% of the total full supply volume of both storages, and the combined dead storage volume.

Table 2 shows that, for the majority of models, downscaling is resulting in an increase in the frequency of years where the available storage volume is below the trigger level. Six of the eight directly comparable models show that downscaling results in a decrease in the recurrence interval at the 25%, 50% and 75% trigger levels. The additional downscaled GCMs are also showing significantly decreased recurrence intervals when compared to the additional GCM results.

Table 2. Combined storage performance of Wivenhoe and Somerset Dams using GCM and downscaled GCM projected climate change at various trigger levels.

Average Recurrence Interval of years where storage volume falls below the specified trigger levels								
Model	**GCM**				**Downscaled GCM**			
	Dead Storage Volume	**25%**	**50%**	**75%**	**Dead Storage Volume**	**25%**	**50%**	**75%**
Historical Climate	>111.0	111.0	12.3	2.1	>111.0	111.0	12.3	2.1
CSIRO Mark 3.0 A2	>111.0	13.9	3.2	1.5	37.0	3.8	2.1	1.4
CSIRO Mark 3.5 A2	>111.0	6.9	2.3	1.4	15.9	3.0	2.0	1.4
GFDL-CM21 A2	4.6	2.1	1.6	1.3	6.2	2.5	1.8	1.3
MIROC 3.2 medres A2	>111.0	>111.0	37.0	3.6	>111.0	27.8	6.9	1.7
MPI-ECHAM5 A2	37.0	4.4	2.2	1.4	13.9	3.0	1.9	1.4
NCAR CCSM3 A2	>111.0	27.8	6.9	1.8	55.5	3.8	2.1	1.4
UKMO-HadCM3 A2	55.5	5.8	2.2	1.4	>111.0	12.3	2.8	1.5
UKMO-HadGEM1 A2	22.2	3.6	2.0	1.4	12.3	3.1	1.9	1.3
CSIRO Mark 3.0 A1B					4.1	2.0	1.6	1.2
INM-CM3 A2					>111.0	7.9	2.5	1.4
INM-CM3 A1B					111.0	6.5	2.5	1.4
NCAR CCSM3 A1B					>111.0	111.0	11.1	2.2
IAP FGOALS-G1 A1B	>111.0	13.9	2.8	1.5				
MIROC 3.2 hires A1B	>111.0	55.5	13.9	2.4				
MIUB ECHO-G A2	>111.0	>111.0	18.5	2.8				

The impact of downscaling is also easily visualised by plotting the modelled storage behaviour of the GCM and downscaled GCM projections. Figures 12 and 13 show the range of the combined Wivenhoe and Somerset Dam storage volumes for all GCM and downscaled GCM model results respectively. While the full range of modelled storage behaviour for both methods is similar, the majority of downscaled models result in more frequent and longer lasting periods of reduced available storage, as shown by the median and 80th percentile confidence limits.

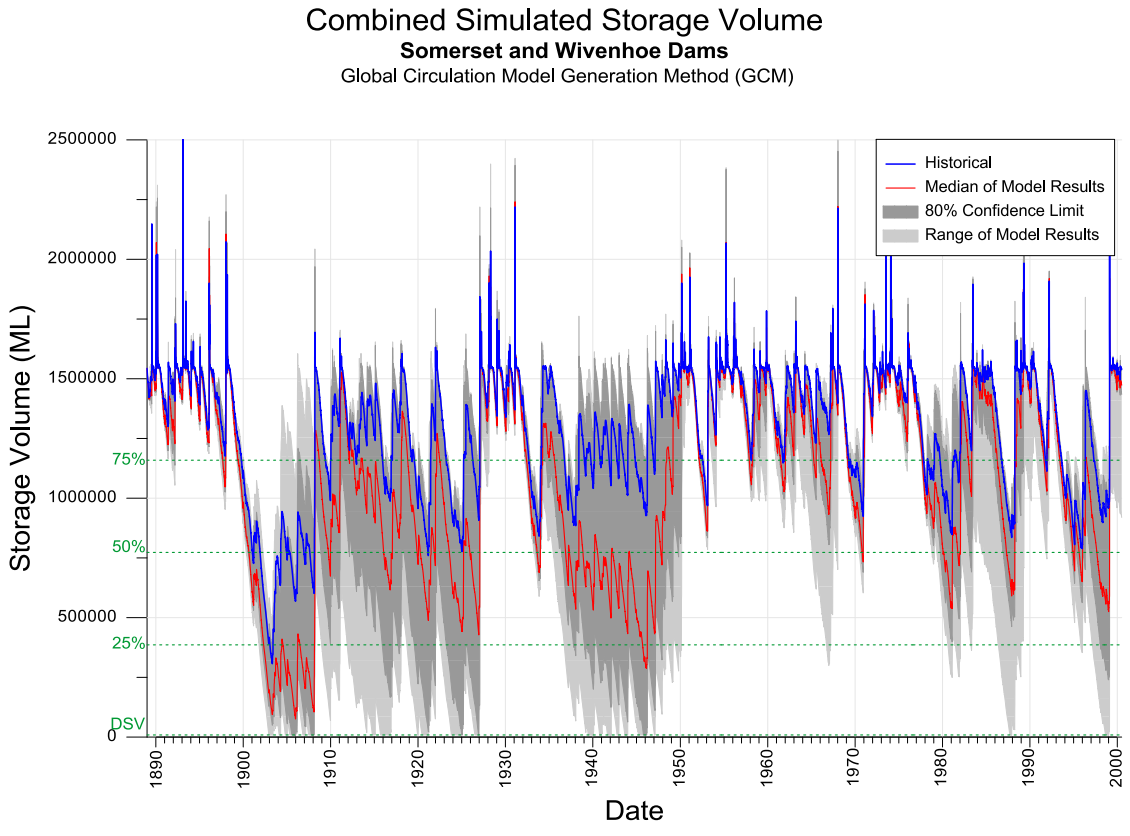


Figure 12. Modelled storage behaviour using GCM projections.

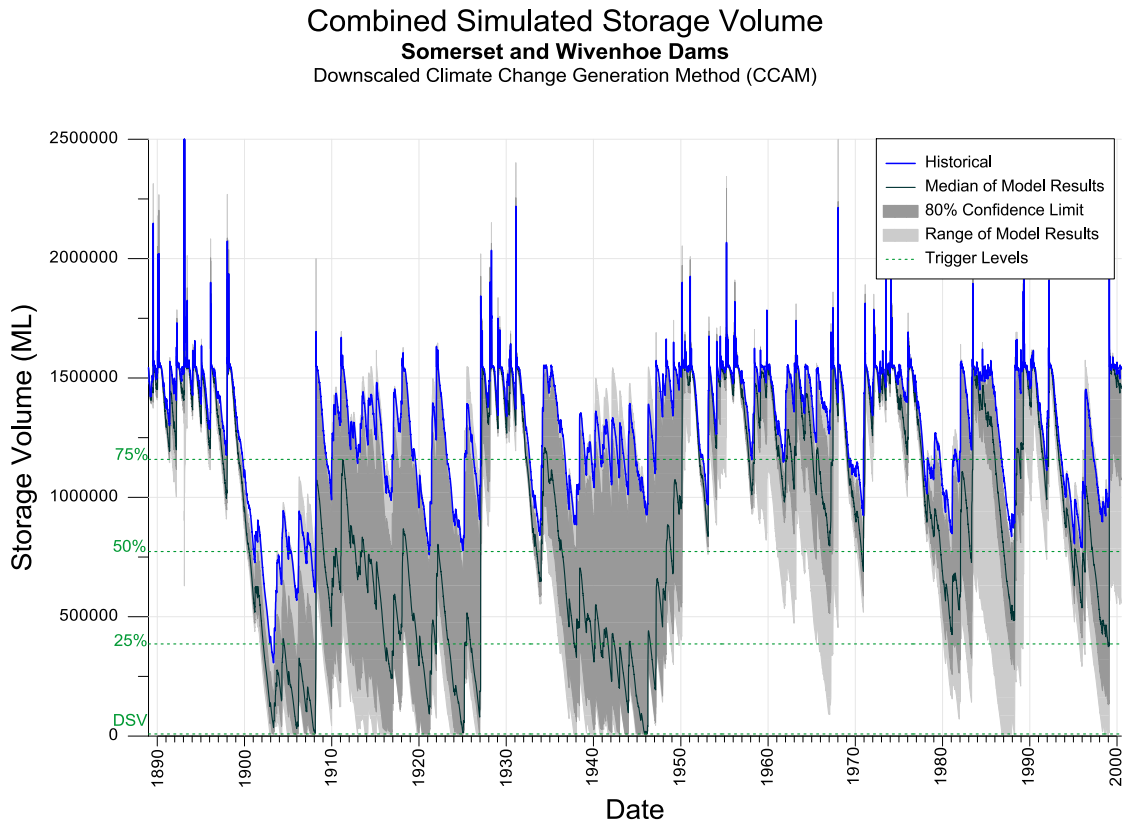


Figure 13. Modelled storage behaviour using downscaled GCM projections.

Conclusions

High-resolution dynamical downscaling climate model output is showing a further reduction in water availability when compared to the performance of the original GCM climate projections. Climate change assessments using Global Climate Models may underestimate the impact on available water resources in SEQ. These reductions are attributed to the spatial and temporal variability of the climate change projections. The likely impact will be reduced inflows to dams and increased time between filling events, and has the potential for significant implications for planning processes and projects that consider climate change impacts on water supply.

References

Burnash, R.J.C., Ferral, R.L. and McGuire, R.A. (1973), A Generalised Streamflow Simulation System: Conceptual Modelling for Digital Computers. Joint Federal-State River Forecast Center, U.S. National Weather Service and California Department of Water Resources, Sacramento, California, U.S.A.

Chiew FHS, Teng J, Kirono D, Frost AJ, Bathols JM, Vaze J, Viney NR, Young WJ, Hennessy KJ and Cai WJ (2008) Climate data for hydrologic scenario modelling across the Murray-Darling Basin. A report to the Australian Government from the CSIRO Murray-Darling Basin Sustainable Yields Project. CSIRO, Australia.

CSIRO and Bureau of Meteorology (2007) Climate Change in Australia. Technical Report, www.climatechangeinaustralia.gov.au

DLWC (1997), Integrated Quantity Quality Model: User Manual, Report No TS96.079. Department of Land and Water Conservation, Australia.

Voogt, S., Giles, S., Harding, P., Vítkovský, J.P. and Loy, A. (2009), Hydrologic Assessment of Regional Water Supply Strategies in Queensland Considering Climate Variability and Climate Change. Department of Environment and Resource Management, Queensland Government, Australia.

A Summary of Social, Technological and Contextual Influences of Residential Water End Use Consumption in South East Queensland

Beal, C.D.[1], and Stewart, R.A.[2]
[1] Smart Water Research Centre, Griffith University, Gold Coast, Queensland
[2] Centre for Engineering and Infrastructure Management, Griffith University, Gold Coast, Queensland

Summary

This paper summarises two years of data from the South East Queensland Residential End Use (SEQREUS) project which measured water consumption from 252 homes in four regions across SEQ (Brisbane, Ipswich, Gold Coast and Sunshine Coast) (Beal *et al.*, 2010). High resolution smart meters have been used in combination with household water appliance stock audits, self-reported water diaries and psycho-social questionnaires to assesses the contextual (eg, seasonal and diurnal), socio-demographic (eg, income, family size) and stock efficiency (eg, star rated clothes washers and low flow shower heads) influences on water consumption.

Keywords
Smart meters, end use, micro-component, demand management, water consumption, forecast modelling.

Introduction

Water end use studies provide the basis for all subsequent demand management work including development of water conservation strategies, demand forecast modelling and infrastructure modelling for new developments. Knowledge of the typical water consumption volumes and the associated end uses (eg, toilet, tap, shower, clothes washer, irrigation, and leak) allows a more proactive approach to demand management and can be used for targeted water conservation strategies and schemes. In Australia, there have been only a few end uses studies using a combination of metering technology, household surveys and end use software (Willis *et al.*, 2011; Athuraliya *et al.*, 2008). End uses of water in residential households include showers, clothes washers, toilets, taps, dishwashers, baths and irrigation (Mayer and DeOreo, 1999). Advances in methods for data capture, transfer, storage and analysis have improved the resolution of water volume data and made transfer and collection of data substantially more time efficient. Factors that influence end use consumption include demographics of the household, water appliances, and residential attitudes and behaviours. The primary aim of the study was to quantify and characterise mains water end uses in a sample of 250+ residential dwellings located within SEQ in different seasonal periods. This paper reports on the baseline (winter) and summer end use analysis conducted in 2010 and 2011.

Methods

A mixed method approach was used, combining high resolution water meters, remote data transfer loggers, household water appliance audits and a self-reported household water use diary (Figure 1).

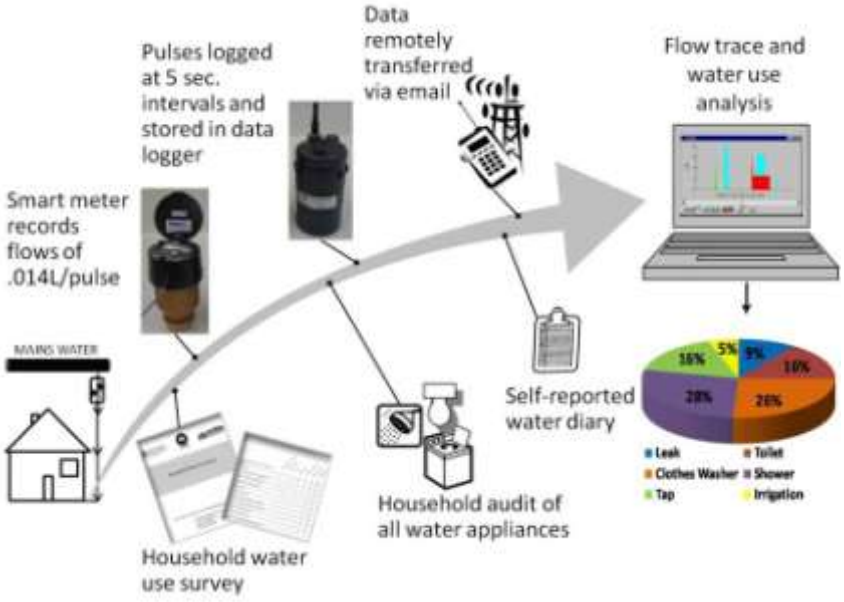


Figure 1. Schematic flow of mixed method approach to acquiring physical and social data.

Existing standard water meters were replaced with high resolution meters that are capable of providing 0.014 L/pulse outputs in 5 second intervals to wireless data loggers. Received data was extracted from the database and disaggregated into all end use events associated with the sampled residential households. A stock appliance survey on the study sample was also conducted to document each household's water fixture/appliance inventory as well as to qualify how householders interact with such stock. Each household was also asked to fill out a water diary, where as many internal and external water use events as possible were recorded over a seven-day period. The logged water flow data was analysed by Trace Wizard® software (Aquacraft, 2010). Winter end use reads were conducted between 14 and 28 June, 2010. Summer end use reads were conducted between 1 December 2010 and 22 February 2011. Historically, a key factor influencing the summer water usage is the Christmas and New Year periods where a proportion of families will either vacate their homes during this period and/or have one or more visitors to stay (Loh and Coghlan, 2003). Additionally, the extended school holiday break and traditional work leave also occurs throughout summer (typically mid December to mid-late January). On the basis of occupancy and water consumption variability, three summer reads were taken to capture a more accurate water usage profile over the summer in SEQ. These occurred between 1 and 14 December, 24 December to 6 January, and 14 to 28 February 2011. Averages of the three reads are reported herein.

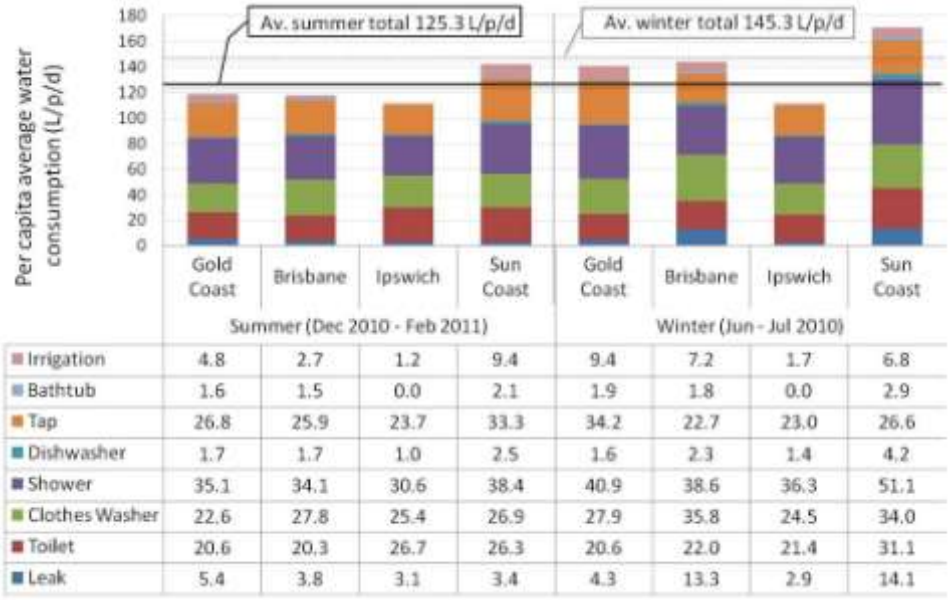
Results

Over the two year data collection period vast amounts of data were collected, analysed and assessed. A selection of some interesting results and trends are summarised below for this paper.

End Use Breakdowns

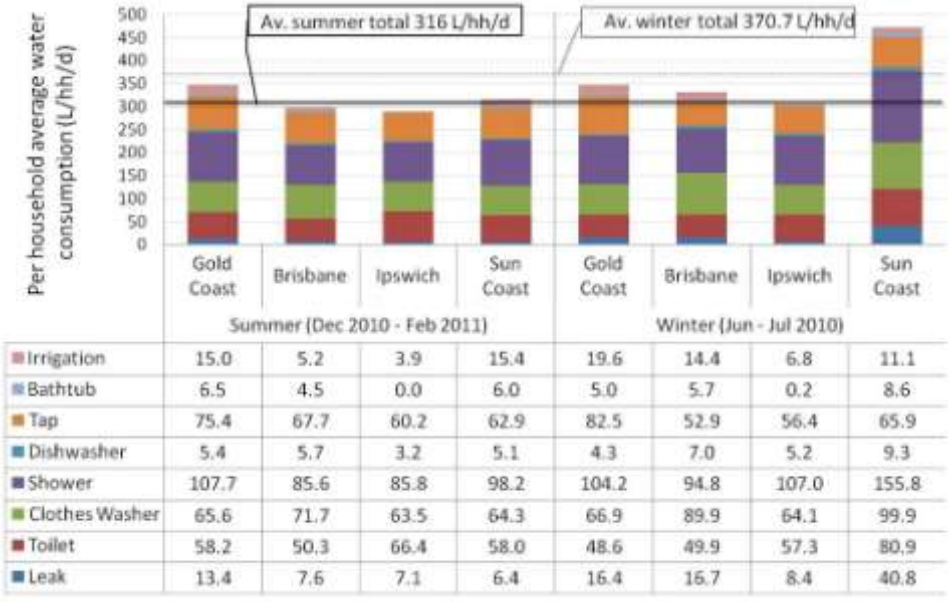
In general, water consumption varied across regions with average total per capita use ranging between 110 litres per person per day (L/p/d) in Ipswich to over 170 L/p/d in the Sunshine Coast (Figure 2). Per household consumption followed the same trends with average daily household use being the least for Ipswich across both seasons at 290 and 305 litres per household per day (L/hh/d) for summer and winter, respectively (Figure 3). Sunshine Coast household usage for winter was the greatest across all four regions (472 L/hh/d), although Gold Coast recorded the highest summer usage at 347 L/hh/d. In general, summer end uses and total consumption were lower than expected for all regions and did not reflect the state government (Queensland Water Commission [QWC]) records which reported average usage for the SEQ region between the periods of recording at 158 L/p/d compared to the SEQREUS figure of 125 L/p/d.

There are several explanations for this reduced summer usage observed in this study. Firstly, the above average rainfall experienced in 2010/11, Queensland's 6th wettest summer on record, clearly resulted in very low outdoor use (irrigation) - the end use which is inherently higher in summer - thus it is the main end use which typically drives the increase in total consumption over summer (Willis *et al.*, 2011; Russell and Fielding, 2010; Roberts, 2005). Secondly, the aligned CSIRO systematic social analysis project conducted an intervention study on a sample which included 80 SEQREUS homes, resulting in significant reductions in water use, and consequently changed consumption behaviour. Thirdly, as part of the intervention process, homes with high leakage rates were alerted to such, with resultant leakage rates reduction following maintenance. Additionally, as for the winter measurements, there will be some bias in the SEQEUS sample due to the smaller size of the sample compared with the QWC database and the possibility of a slight overrepresentation of low water consumers due to their involvement in this study.



	Summer (Dec 2010 - Feb 2011)				Winter (Jun - Jul 2010)			
	Gold Coast	Brisbane	Ipswich	Sun Coast	Gold Coast	Brisbane	Ipswich	Sun Coast
Irrigation	4.8	2.7	1.2	9.4	9.4	7.2	1.7	6.8
Bathtub	1.6	1.5	0.0	2.1	1.9	1.8	0.0	2.9
Tap	26.8	25.9	23.7	33.3	34.2	22.7	23.0	26.6
Dishwasher	1.7	1.7	1.0	2.5	1.6	2.3	1.4	4.2
Shower	35.1	34.1	30.6	38.4	40.9	38.6	36.3	51.1
Clothes Washer	22.6	27.8	25.4	26.9	27.9	35.8	24.5	34.0
Toilet	20.6	20.3	26.7	26.3	20.6	22.0	21.4	31.1
Leak	5.4	3.8	3.1	3.4	4.3	13.3	2.9	14.1

Figure 2. Per capita (L/p/d) break down of end uses for winter and summer readings.



	Summer (Dec 2010 - Feb 2011)				Winter (Jun - Jul 2010)			
	Gold Coast	Brisbane	Ipswich	Sun Coast	Gold Coast	Brisbane	Ipswich	Sun Coast
Irrigation	15.0	5.2	3.9	15.4	19.6	14.4	6.8	11.1
Bathtub	6.5	4.5	0.0	6.0	5.0	5.7	0.2	8.6
Tap	75.4	67.7	60.2	62.9	82.5	52.9	56.4	65.9
Dishwasher	5.4	5.7	3.2	5.1	4.3	7.0	5.2	9.3
Shower	107.7	85.6	85.8	98.2	104.2	94.8	107.0	155.8
Clothes Washer	65.6	71.7	63.5	64.3	66.9	89.9	64.1	99.9
Toilet	58.2	50.3	66.4	58.0	48.6	49.9	57.3	80.9
Leak	13.4	7.6	7.1	6.4	16.4	16.7	8.4	40.8

Figure 3. Per household (L/hh/d) break down of end uses for winter and summer readings.

Nevertheless, indoor end uses were generally consistent across seasons with showers (approx. 40 L/p/d), washing machines (approx. 28 L/p/d) and taps (approx. 27 L/p/d) comprising an average of 29%, 21% and 20% of total household consumption, respectively (Figure 2). These figures are comparable with other Australian domestic end use studies (eg, Willis *et al.*, 2011; Water Corporation, 2011; Athuraliya *et al.*, 2008), further demonstrating the relative homogeneity of indoor end use consumption. External water consumption (eg, irrigation) varied both temporally and spatially across SEQ, with averages considerably lower (2 to 4% of total use) than other reported studies. The above average rainfall experienced in 2010/11 has clearly impacted on irrigation end use, along with a general shift in consumer behaviour and attitude to outdoor watering. Leaks were typically associated with toilet and tap usage and generally comprised about 4 % of total household consumption.

Stock Efficiency and Water Use

A number of observations have been made on the impacts that water use efficient (WUE) technology can have on water consumption and peak flows within a household. Front loading machines used significantly less water (11.3 L/p/d, $p<0.05$) than top loaders and a significantly lower proportion (7%, $p<0.05$) of total household water was required by front loading machines. There was also an observed trend toward lower water consumption on a per capita and per household basis when high efficiency (low flow) shower heads were installed, with results showing that shower end use consumption could be reduced by nearly 20% (Table 1). Willis *et al.* (2010) also reported similar

reductions in consumption as a result of more efficient washing machines and replacement of old, high flow shower heads with more efficient low flow fittings.

Table 1. Showerhead efficiencies cluster comparisons from winter analysis (Beal *et al*., 2011).

Description	Shower Head Efficiency Clusters		
Efficiency Rating (WELS rating)	**Low (A)**	**Medium (AA)**	**High (AAA)**
Daily per capita shower consumption (L/p/d)	49.7	37.7	35.8
Daily household shower consumption (L/hh/d)	139.9	106.7	104.1
Annual per capita shower consumption (kL/p/a)	18.1	14.1	13.1
Annual household shower consumption (kL/hh/a)	51.1	39.0	38.0

An interesting component of this study has been the examination of the impact of WUE technology on average day peak flows. Data in Figure 4 shows how the 50 most WUE homes can have a substantial reduction in flow during the peak daily periods between 7 am to 10 am in the morning and 6 pm to 9 pm in the evening (Carragher *et al.*, 2011, in preparation). These reduced peak demand flow rates suggest substantial implications to water distribution networks. These implications include the potential for capital efficiency opportunities of water supply infrastructure through smaller infrastructure and pipe sizing for green-field developments as well as offset of existing network's inevitable future upgrade costs. Reduced peak demand flow rates also imply that if wide-scale conservation programs were implemented, peaking factors used to scale average day diurnal patterns for network modelling purposes should be extensively reviewed with the possibility of reduction.

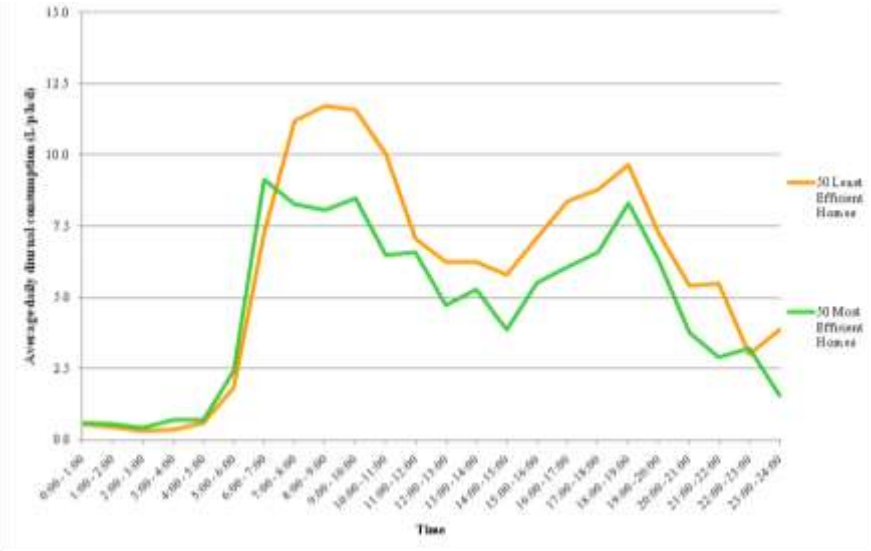


Figure 4. Average day diurnal pattern comparison of 50 least and most efficient homes (Carragher *et al*., 2011).

Socio-Demographical Trends and Perceptions of Water Consumption

Socio-demographic comparisons revealed that larger, younger and high income bracket families tended to have more water efficient stock but use more water than smaller, older and lower income families (Figure 5). This latter demographic also showed a trend for more accurately estimating their actual water use and having greater water conservation intentions (Beal *et al.*, 2011). Analysis of actual water consumption with matched self-reported water use was conducted using multiple univariate tests (ANOVAs) at 99% significance levels. Results showed a trend for self-nominated high water users to consistently use less water than self-nominated medium and low water users (Figure 6). Frequent and high volume water end uses such as clothes washing, showering and tap use were associated with the incorrect estimations of household consumption. Households that overestimate or underestimate their water use also differ in their socio-demographic and psycho-social profile, providing further clues for why the disparity between perceived and actual behaviour arises.

This type of profiling can be used to formulate targeted water policy approaches such as educating citizens on their water use through feedback information and community-based social marketing. Heightened feedback to consumers on their households' water consumption will improve their awareness and understanding of water use and help to encourage future water conservation behaviours. Future research examining a larger sample size, across various seasons, may confirm the conclusion that people who underestimate their water use may be genuinely unaware of their high water use (eg, habits), and/or are less water conservation minded.

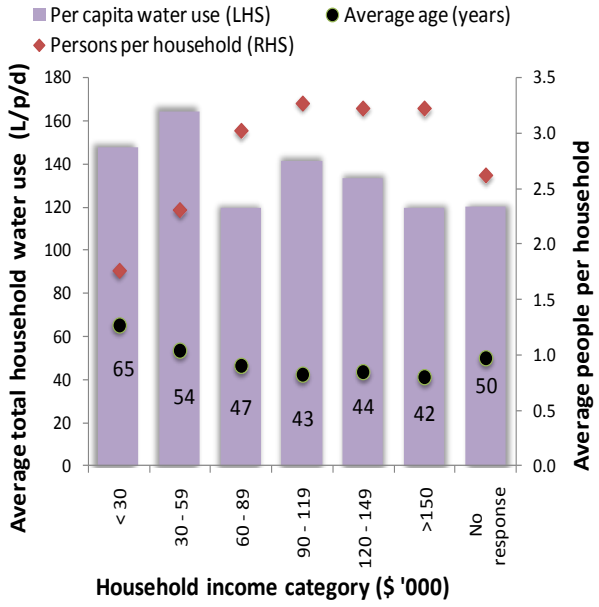
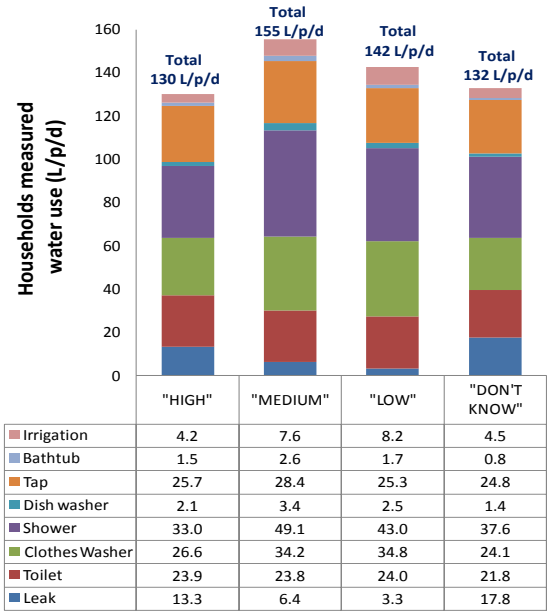


Figure 5. Relationship between income category and average household occupancy.



	"HIGH"	"MEDIUM"	"LOW"	"DON'T KNOW"
Irrigation	4.2	7.6	8.2	4.5
Bathtub	1.5	2.6	1.7	0.8
Tap	25.7	28.4	25.3	24.8
Dish washer	2.1	3.4	2.5	1.4
Shower	33.0	49.1	43.0	37.6
Clothes Washer	26.6	34.2	34.8	24.1
Toilet	23.9	23.8	24.0	21.8
Leak	13.3	6.4	3.3	17.8

Figure 6. Comparisons of actual per capita water use with self-identified users.

Conclusions and Significance

The SEQREUS has generated a substantial data registry of end uses and matching socio-demographical, household stock and psycho-social information. Consequently, this data can provide the fundamental building blocks for water utilities, government agencies and researchers alike in the development of a range of tailored water demand/conservation strategies, and domestic consumption/network distribution forecasting models. As such, there is a strong interest from local water utilities (eg, Unity Water and Queensland Urban Utilities) and state government (eg, QWC) to continue on with the project after the scheduled completion of the Alliance deliverables at the end of June.

References

Athuraliya, A., Gan, K. and Roberts, P. (2008). Yarra Valley Water 2007 appliance stock and usage patterns survey. Yarra Valley Water, Victoria, May 2008.

Beal, C.D., Stewart, R.A., Huang, T.. (2010). South East Queensland Residential End Use Study: Baseline results – Winter 2010. Urban Water Security Research Alliance Technical Report No. 31.

Beal. C.D., Stewart, RA., Fielding, K. (2011) A novel mixed method smart metering approach to reconciling differences between perceived and actual residential end use water consumption. *Journal of Cleaner Production*, (Special Issue, invited paper in review).

Carragher, B.J., Stewart, R.A., Beal, C.D., Determinants of residential peak demand flow rates and contributing end uses: unpacking diurnal patterns to understand capital efficiency opportunities. *Resources, recycling and conservation.* 2011, [under review, July 2011].

Russell, S., Fielding, K., 2010. Water demand management research: A psychological perspective, Water Resources Research, 46, W05302, doi10.1029/2009WR008408.

Water Corporation (2011). Perth residential water use study 2008/2009, Water Forever, Water Corporation, Western Australia.

Willis, R.M., Stewart, R.A., Emmonds, S., (2010) Pimpama-Coomera dual reticulation end use study: pre-commission baseline, context and post-commission end use prediction. Water, Science and Technology: Water Supply, 10(3), 302-314, doi: 10.2166/ws.2010.104.

Willis, R.M., Stewart, R.A., Williams, P.R., Hacker, C.H., Emmonds, S.C., Capati, G., (2011). Residential potable and recycled water end uses in a dual reticulated supply system. *Desalination*, 273(1-3), 201-211. doi: 10.1016/j.desal.2011.01.022.

Predictors of Household Water Use in South East Queensland

Fielding, K.[1], Spinks, A.[2], Russell, S.[3], and Mankad, A.[2]
[1] The University of Queensland, St Lucia, Queensland
[2] CSIRO Ecosystem Sciences, Ecosciences Precinct, Dutton Park, Queensland
[3] Griffith University, Nathan, Queensland

Summary

The current study, framed by an Expanded Theory of Planned Behaviour model, aimed to identify the key psycho-social, demographic and infrastructure predictors of household water use in South East Queensland (SEQ). An online and postal survey was administered to owner-occupiers of free-standing houses in SEQ and permission was sought to access their water billing data. Responses were received from 1,749 householders with 1,055 providing access to their water billing data. Results suggest that a majority of households had installed water efficient appliances. Multiple regression analyses showed that Gold Coast and Sunshine Coast residents and those with more people in the house used more water. Results also suggest that some water efficient appliances are associated with lower water use whereas others are associated with higher water use. Water conservation habits also play a part in reducing water use as does a household culture of water conservation.

Keywords
Residential demand management, household water conservation, Theory of Planned Behaviour, household water culture.

Introduction

As residential water use accounts for the majority of urban water use in South East Queensland (SEQ), achieving the goal of reducing water grid demand hinges on developing a comprehensive understanding of the social and behavioural aspects of household water use. There is a need to understand how people use water and how sustainable levels of water use can be achieved and maintained in the long-term. The Residential Water Demand Management project addresses these issues.

Conceptual Framework

The research draws on the Theory of Planned Behaviour (Ajzen, 1991), a theoretical model that has been used extensively to understand a range of behavioural decision-making contexts, including household water conservation (Clark and Finley, 2007; Kantola *et al.*, 1982; Lam, 1999, 2006). Past research in this area, however, has been marked by methodological and measurement limitations (Russell and Fielding, 2010), the most serious being the reliance on people's self-reports of water conservation intentions or behaviour rather than objective measures of household water use. There has also been a tendency to rely on individual psycho-social data to predict a collective outcome (ie, household water use). The current project is designed to overcome these limitations.

The Theory of Planned Behaviour is a parsimonious model of the informational and motivational influences that combine to predict behaviour. In the Theory of Planned Behaviour, the most immediate predictor of behaviour is an intention (ie, a motivation or plan) to engage in the behaviour. The TPB proposes intention to be determined by the additive effects of attitude, subjective norm and perceived behavioural control. Attitudes refer to the overall positive or negative evaluation of performing the behaviour (eg, water conservation). Subjective norms are based on individuals' perceptions of whether important other people in their life would want them to perform the behaviour, and perceived behavioural control reflects the extent to which individuals consider the behaviour to be within their control. Therefore, according to the Theory of Planned Behaviour, individuals who hold positive attitudes toward water conservation, think that there is support from important others for engaging in water conservation, and who perceive that they can easily engage in this activity, should have strong intentions to conserve water in the household. In addition, to the extent that perceived behavioural control is a proxy for actual control, it may also directly impact on behaviour, such that a greater sense of control over conserving water around the house will be associated with higher levels of actual household water conservation.

An advantage of the Theory of Planned Behaviour is that it allows for the addition of other predictor variables that are relevant to a specific behavioural context, in this case, household water conservation. A review of the literature on household water conservation (Russell and Fielding, 2010) suggests the importance of a number of additional predictors of household water use. As well as subjective norms, which are the standard normative measure included in the Theory of Planned behaviour, we also included measures of descriptive norms and moral norms. Descriptive norms describe typical or normal behaviour – what people *actually* do (Cialdini, Kalgren and Reno, 1990). Cialdini and colleagues (1990) argue that descriptive norms motivate behaviour because they provide information about what is adaptive and effective in a given situation. Essentially: "if everyone's doing it, it must be a good thing". Moral

norms tap into a person's sense of personal obligation to engage in a behaviour. In accordance with past literature (eg, Manstead and Parker, 1995; Nolan, Schultz, Cildini, Goldstein and Griskevicius, 2008), we predict that if people have a sense that others in their community are engaging in water conservation, and that it is a moral duty to conserve water, then they will be more likely to engage in water conservation behaviours.

To assess the collective aspects of water conservation, we incorporated into the Theory of Planned Behaviour measures of identity: *self-identification* (ie, as a water conserver), *household culture* (ie, the extent to which household members think of themselves as a water conserving household and engage in water conservation) and *community identification* (ie, as a member of the SEQ community). There is ample evidence that self-identity, for example as a water conserver, is a powerful motivator of behaviour (Conner and Armitage, 1998). The logic is that once we have incorporated an identity (eg, water conserver) into our self-concept, to not act congruently with that identity creates a state of internal tension (Stryker, 1968; 1980). Therefore, householders who have a strong sense of themselves as water conservers should be more likely to engage in water conservation actions. One of the key ways in which the study tries to overcome the use of individual survey data to predict a collective outcome (ie, household water use) is by including a measure of household water conservation culture. This measure assesses the extent to which the culture (ie, the norms and identity) of the household supports water conservation. Although an individual household member might be committed to water conservation, unless that person's attitudes and actions are shared by the rest of the household, there may be little relationship between individual attitudes and household water use. Finally, we reasoned that people who are more identified with the SEQ community may be more aware of, and concerned about, water security issues, and therefore will engage in greater water conservation.

However, our actions are not always determined by reasoned and rational processes but sometimes they are determined by automatic, habitual processes (eg, Ouellette and Wood, 1998). This is especially likely for actions that are performed on a regular basis (eg, brushing teeth, showering). Habits are usually measured by how often people have engaged in an action in the past. Where past actions are frequent (eg, always turning off tap when brushing teeth) then this is likely a habit and people are more likely to automatically engage in that action in the future. Our Expanded Theory of Planned Behaviour model incorporating descriptive norms, moral norms, community identification, self-identity, household culture and habits (ie, past behaviour) is represented in Figure 1. The variables of this model are all hypothesised to be negative predictors of household water use, that is, more positive water conservation attitudes, norms, etc will be associated with lower levels of household water use.

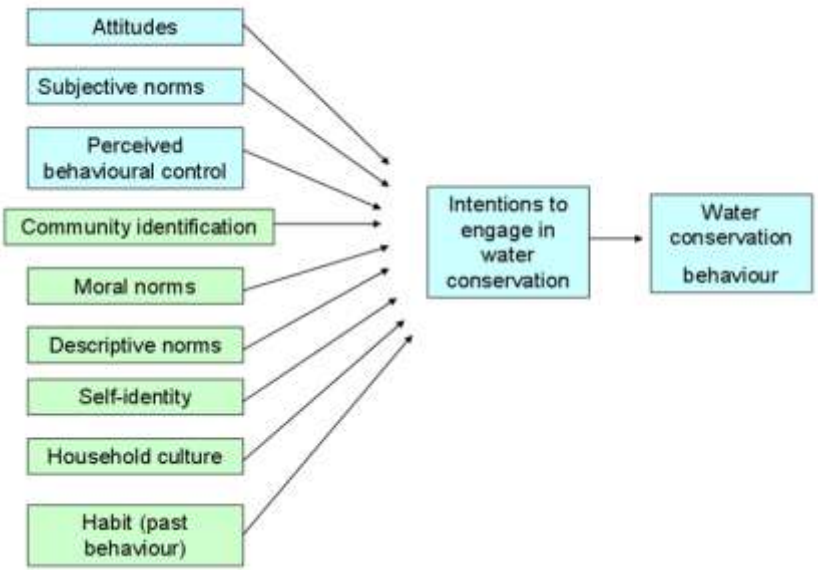


Figure 1. Expanded Theory of Planned Behaviour model.

The research, conducted in late 2009, employed a quantitative survey (delivered in online and postal format) of free standing owner-occupied households from the SEQ community to identify the key predictors of household water use. Respondents were asked to complete the survey and to provide consent to access their historic and future water use data from the appropriate water utility. In addition to the psycho-social variables encompassed in the Expanded Theory of Planned Behaviour, the study also measured demographic variables and household water efficiency infrastructure. The inclusion of this range of variables allowed an assessment of the relative contribution of the psycho-social variables to household water use.

Results

A total of 1,179 households returned a direct mail survey (27% of those invited), while 570 households completed the survey online. Water consumption data were not available for all households completing the survey due to receiving responses from non-targeted households who had shared water meters (ie, multi-dwelling complexes) or were not connected to a central water supply. In addition, not all participants returned a signed consent form to allow for the release of water consumption data by the city councils. In total, water use data was available for 1,055 households.

Household Water Use

Descriptive statistics pertaining to the amount of water used by participating households from October 2009 to March 2010 are shown in Table 1. This table shows both the total household and per person values. It is likely that the extreme minimum and maximum values may be erroneous due to either faulty water meters or substantial leaks (or unoccupied dwellings, for the minimum). Note that exclusion of the outliers in the Multiple Regression Analyses (described below) did not change the results therefore they were included in the analysis.

Table 1. Daily household water use, October 2009 – March 2010.

	Total Daily Household Water Use (Litres)	Per Person Daily Household Water Use (Litres)
Mean	428.48	174.43
Standard Deviation	277.52	124.95
Median	371.95	142.30
Inter-Quartile Range*	245.35 – 540.51	104.46 – 201.11
Minimum	4.75	4.75
Maximum	2,540	1,535

* Inter-quartile range is the 25th-75th percentile

Household Water Use: Temporal Variation

Per person water use increased from late 2008/early 2009 to the billing period of October 2009 to March 2010. When examined across regions, increases in water use per person are most evident in Brisbane and Gold Coast (see Figure 2).

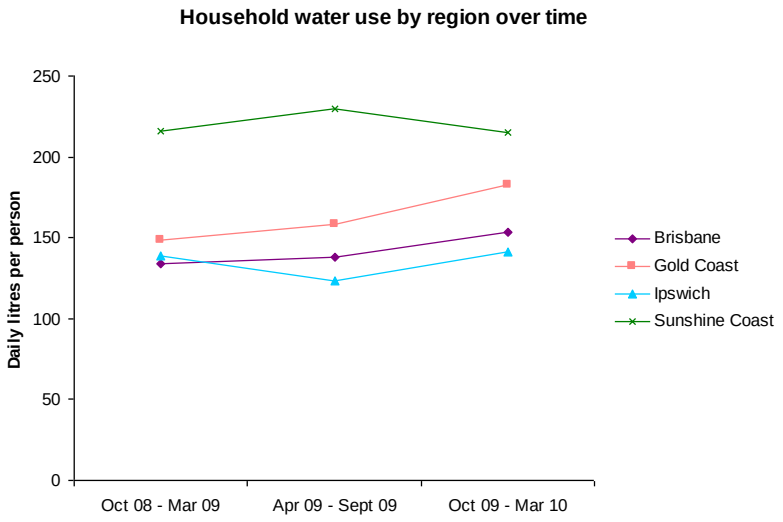


Figure 2. Average per person water use from October 2008 to March 2010 broken down across geographic region.

Household Water Use: Household Type

Daily per person household water use was also influenced by household type ($F = 16.04$, $p < .001$; Table 2). Post-hoc analyses showed that individuals living in single person households used more water per person than residents of all other household types. Families with young children, as well as families with young children and teenagers used less water per person than all other household types.

Table 2. Household water use (per person) by household type, October 2009 – March 2010.

Household Type	N	Mean	S.D.
Single person household	149	234.74	205.88
Two person (adult) household	388	188.45	122.79
Multiple adult household	166	150.41	60.30
Family with young children	185	129.44	59.50
Family with teenagers	66	170.77	105.41
Family with young children and teenagers	47	134.54	89.20

Influence of Installed Infrastructure

Table 3 shows the per person household water usage for households with and without eleven types of water-saving infrastructure devices. It is clear that the majority of respondents had installed low flow taps, trigger hose/timed sprinkler, water-wise plants, dual flush toilets, water-wise dishwasher. T-test comparisons showed significantly less per person water usage in households with low flow taps, pool covers (for those households with swimming pools), shower timers and plumbed rainwater tanks.

Table 3. Household water use (per person) according to installation of water-saving infrastructure, October 2009 – March 2010.

Infrastructure	Installed		Not Installed		t-Value	*p*
	N	Mean (S.D)	N	Mean (S.D)		
Low flow taps	695	167.99 (115.46)	306	189.04 (143.34)	2.46	.014
Pool cover*	93	174.07 (104.59)	235	213.96 (162.93)	2.63	.009
Trigger hose / timed sprinkler	676	175.67 (117.02)	325	171.84 (140.18)	-.45	.650
Water-wise plants	602	169.64 (107.80)	399	181.64 (146.94)	1.40	.162
Dual flush toilet	854	172.72 (125.47)	147	184.33 (121.81)	1.04	.298
Shower timer	444	164.52 (101.92)	557	182.32 (140.22)	2.32	.020
Grey water system	161	169.35 (108.91)	840	175.40 (127.83)	.56	.574
Plumbed rainwater tank	130	137.23 (77.13)	871	179.98 (129.71)	5.30	.000
Non-plumbed rainwater tank	382	165.72 (103.46)	619	179.80 (136.34)	1.85	.065
Water-wise washing machine	452	175.20 (121.94)	549	173.79 (127.48)	-.18	.858
Water-wise dishwasher	632	169.27 (115.17)	369	182.25 (139.84)	1.71	.088

* pool owners only

Predicting Household Water Use

A multiple regression analysis was conducted to assess the significant predictors of household water use in the six months following the survey (October 2009 – March 2010). This analysis allows an assessment of the relative importance of each of the predictors when all others are included in the regression equation (ie, demographic, water efficient appliances, psycho-social determinants). The final model predicted 29% of the variance in total water use ($R = .536$, $F(30,769) = 10.33$, $p < .001$). Demographically, residents in the Gold Coast and Sunshine Coast and households with more adults and children used more water. In terms of water efficient infrastructure, low flow taps, dual flush toilets, and plumbed rainwater tanks were associated with lower water use, while water efficient irrigation systems and water efficient washing machines were associated with higher water use. Behaviourally, households who consistently reported running their washing machine only when full and turning off taps while brushing their teeth, used less water. Finally, household water conservation culture was a significant negative predictor of water use: the more respondents reported that the household, as a whole, was committed to water conservation and engaged in water conservation actions, the less water they used. Surprisingly, the more respondents perceived that others in the community were saving water, the more water they used. This latter finding should be treated with caution as it is contrary to previous findings, although, it may reflect a free-rider effect whereby householders feel that they can do less because others are doing more.

Table 3. Regression coefficients predicting household water use, October 2009 – March 2010.

Variable	Standardised Coefficients (Beta)	*p*
Region: GoldCoast *	**.15**	**.000**
Region: Ipswich *	-.02	.574
Region Sunshine C *	**.19**	**.000**
Age	.06	.140
Household size: adults	.28	**.000**
Household size: children	.29	**.000**
Education category	**-.01**	**.652**
Installed infrastructure: low flow taps	**-.08**	**.024**
Installed infrastructure: Pool cover	.04	.254
Installed infrastructure: trigger hose / timed sprinkler	**.09**	**.012**
Installed infrastructure: water-wise plants	.02	.651
Installed infrastructure: Dual flush toilet	**-.08**	**.027**
Installed infrastructure: Shower timer	-.02	.488
Installed infrastructure: Grey water system	.04	.213
Installed infrastructure: Plumbed rainwater tank	**-.10**	**.001**
Installed infrastructure: Non-plumbed rainwater tank	-.01	.725
Installed infrastructure: Water-wise washing machine	**.09**	**.008**
Installed infrastructure: Water-wise dishwasher	-.02	.613
Scale: Curtailment General Intentions	.10	.052
Scale: Curtailment Attitudes	-.02	.573
Scale: Curtailment Subjective Norms	.02	.574
Scale: Curtailment Moral Norms	-.05	.364
Self Efficacy	.04	.274
Perceived Behavioural Control	.02	.511
Curtailment Descriptive Norm	**.11**	**.001**
Community Identification	.04	.208
Household Culture	**-.12**	**.004**
Self-identity	-.07	.084
Past behaviour: run washing machine if full	**-.09**	**.007**
Past behaviour: turn off taps while brushing teeth	**-.08**	**.019**

* Brisbane is the reference category that other regions are compared against

Conclusions

The current study provides critical insights into the predictors of household water use in SEQ. The incorporation of collective-level variables to assess the contribution of household dynamics to water use and the use of objective measures of household water use means that the research makes a significant theoretical and empirical contribution to the literature. The identification of key predictors of household water use has important applied implications.

It is interesting to note that Gold Coast and Sunshine Coast residents were higher water users than Brisbane and Ipswich residents. One interpretation of this finding is that Gold Coast and Sunshine Coast households did not experience the same drought conditions and associated restrictions and, therefore, did not perceive the imperative to conserve water or develop good water conservation practices. Another interesting finding is that some water efficient appliances were associated with lower water use, whereas others were associated with higher water use. One potential explanation for the counter-intuitive finding that water efficient irrigation and washing machines are associated with higher water use is that people who are more engaged in these actions (eg, gardening, high levels of clothes washing) may be more likely to invest in efficient appliances. It is also possible that efficient appliances give people 'permission' to be less vigilant about their water conservation practices, especially for practices that may be less automatic (Mazar and Zhong, 2010). Finally, the positive influence of a household culture of water conservation highlights the need for water conservation programs to target the household as a whole, to foster good practices and commitment to conserving water as a way of life.

References

Ajzen, I. (1991). The theory of planned behavior, *Org Beh and Hum Dec Proc*, *50*(2), 179-211.

Cialdini, R.B., Reno, R.R., and Kallgren, C.A. (1990). A focus theory of normative conduct: Recycling the concept of norms to reduce littering in public places. *Journal of Personality and Social Psychology, 58(6),* 1015-1026.

Clark, W.A., and J.C. Finley (2007). Determinants of water conservation intention in Blagoevgrad, Bulgaria, *Society and Natural Resources*, *20*, 613-327.

Kantola, S.J., Syme, G.J., and Campbell, N.A. (1982). The role of individual differences and external variables in a test of the sufficiency of Fishbein's model to explain behavioral intentions to conserve water, *Journal of Applied Social Psychology*, *12*(1), 70-83.

Lam, S.P. (1999). Predicting intentions to conserve water from the theory of planned behavior, perceived moral obligation, and perceived water right, *Journal of Applied Social Psychology*, *29*(5), 1058-1071.

Lam, S.P. (2006). Predicting intention to save water: Theory of planned behavior, response efficacy, vulnerability, and perceived efficiency of alternative solutions, *Journal of Applied Social Psychology*, *36*(11), 2803-2824.

Mazar, N., and Zhong, C-B. (2010). Do green products make us better people? Psychological Science, 21(4), 494-498.

Manstead, A.S.R., and Parker, D. (1995). Evaluating and extending the theory of planned behaviour. In W. Stroebe and M. Hewstone (Eds), *European Review of Social Psychology* (Vol. 6, pp. 69-96). Chichester: Wiley.

Nolan, J.M., Schultz, P.W., Cialdini, R.B., Goldstein, N.J., and Griskevicius, V. (2008). Normative Social Influence is Underdetected. *Personality and Social Psychology Bulletin*, 34, 913–923.

Ouellette, J.A., and Wood, W. (1998). Habit and intention in everyday life: The multiple processes by which past behaviour predicts future behaviour. *Psychological Bulletin, 124,* 54-74.

Russell, S.R. and Fielding, K.S. (2010). Water demand management research: A psychological perspective. *Water Research, 46,* W05302, doi:10.1029/2009WR008408.

Stryker, S. (1968). Identity salience and role performance: The importance of symbolic interaction theory for family research. *Journal of Marriage and the Family, 30,* 558-564.

Stryker, S. (1980). *Symbolic interactionism: A social structural version.* Palo Alto CA: Benjamin/Cummings.

Estimating Water Savings from Mandated Rainwater Tanks in South East Queensland

Chong, M.N.[1], Umapathi, S.[1], Mankad, A.[2], Gardner, E.A.[1], Sharma, A.[3] and Biermann, S.[1]
[1] CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland
[2] CSIRO Ecosystem Sciences, Ecosciences Precinct, Dutton Park, Queensland
[3] CSIRO Land and Water, Highett, Victoria

Summary

Under the Queensland Development Code Part MP 4.2 (QDC 4.2), all Class 1 detached dwellings constructed after 1 January 2007 in South East Queensland (SEQ) are required to save 70 kilolitres of mains water per year. This study investigates the potential mains water saving from mandated rainwater tanks installed in Class 1 detached dwellings in SEQ. A benchmark analysis approach was used to estimate the mains water savings from dwellings with mandated rainwater tanks against the published average mains water consumption benchmarks for each local council. The SEQ council areas of interest are Moreton Bay Regional Council (ie, Pine Rivers and Caboolture), Redland City Council and Gold Coast City Council. Individual household occupancy data obtained from a telephone survey was used to match and convert 691 mandated dwellings' mains water consumption rate into per capita water consumption rate (litres/person/day). The benchmark analysis found an average annual mains water savings in the mandated dwellings of 58.8 kilolitres/household/year (kL/hh/yr) for 2009, with a range from 24.5 to 88.5 kL/hh/yr across the SEQ region. When this analysis was repeated for 2010, the average mains water saving was reduced only very slightly to 58.2 kL/hh/yr, with a range from 39.7 to 81.0 kL/hh/yr across the SEQ region. This study clearly demonstrates that dwellings with mandated rainwater tanks can reduce the direct reliance on mains water supplies. However, the full potential of mains water saving has yet to be examined. This is because the benchmark analysis only estimated the relative water savings against the average consumption values, with the average bulk data set for each local council area being comprised of a mix of dwellings, with and without internally plumbed tank (IPT), water efficient appliances and other water saving features.

Keywords:
Water conservation, mandated rainwater tanks, potable water savings, benchmark analysis.

Introduction

The urban population in SEQ is projected to exceed 4.4 million by 2031, approximately 35% higher than the population in 2007 (DLGP, 2010). This poses a challenge to the existing water supply and treatment processes. To help resolve this issue, the SEQ Water Strategy (Queensland Water Commission, 2010) has proposed the collection, treatment and reuse of localised water sources to meet the water standards of different end-use requirements. The QDC 4.2 requires all detached residential dwellings built after 1 January 2007 in SEQ achieve a potable water savings target of 70 kL/hh/yr. One of the simplest ways to achieve this savings target is through the use of a 5 kL rainwater tank that is connected to at least 100 m^2 of roof catchment area and internally plumbed for toilet flushing, a cold water tap for washing machines and at least one external irrigation tap.

Previously, Beal *et al*. (2011) analysed the water savings from dwellings with IPTs using 2008 water billing data. Results from their study estimated average mains water saving of 50 kL/hh/yr across three local government areas (LGAs) of Pine Rivers, Redland and Gold Coast. Beal *et al*. (2011) highlighted that the possible reasons for the difference between the target value of 70 kL per year, and actual mains water savings were: (1) differences in household composition; (2) the absence of tanks in households assumed to have tanks (ie, homes constructed post June 2007), or (3) atypical behaviours which could range from excessive garden watering to very frugal water use.

This present study aims to estimate the potential mains water savings from dwellings with a mandated rainwater tank using a benchmark analysis in conjunction with the published average mains water consumption rates for each local council. The target areas for this analysis are four LGAs within SEQ: Caboolture, Pine Rivers, Redland and the Gold Coast, to replicate the areas targeted in the Beal *et al.* (2011) study. In our analysis, individual household occupancy data will be matched with the water consumption data at each mandated household level, in order to determine the per capita water consumption rate from the first quarter of 2009 to the fourth quarter of 2010. This information is likely to be key in determining whether mains water consumption is being reduced by the implementation of mandated rainwater tanks and, if so, to what extent. It is expected that the successful completion of this study will provide a step-wise improvement in the on-going attempt to accurately estimate and validate the potable water savings target of for post-2007 dwellings with mandated rainwater tanks.

Methods

To further advance the understanding of variations in water savings, this study extends the methodology used by Beal *et al*. (2011) by: (1) ensuring that dwellings assumed to have mandated tanks, do indeed have tanks; and (2) accounting for differences in household occupancy rates. The benchmark analysis of potable mains water savings was carried out through a comparison between mains water usage in IPT dwellings (N = 691), using water billing data for 2009 and 2010, and the corresponding average mains water consumption rates for each LGA provided by the Queensland Water Commission (QWC). The actual household occupancy rates for each IPT dwellings were obtained through a recent telephone survey (Chong *et al*., 2011). This telephone survey was also able to confirm that the majority of dwellings assumed to have mandated tanks (ie, built post-2007) did indeed have tanks. Figure 1 shows the schematic diagram of the benchmark analysis approach to estimate the potential mains water saving from mandated dwellings.

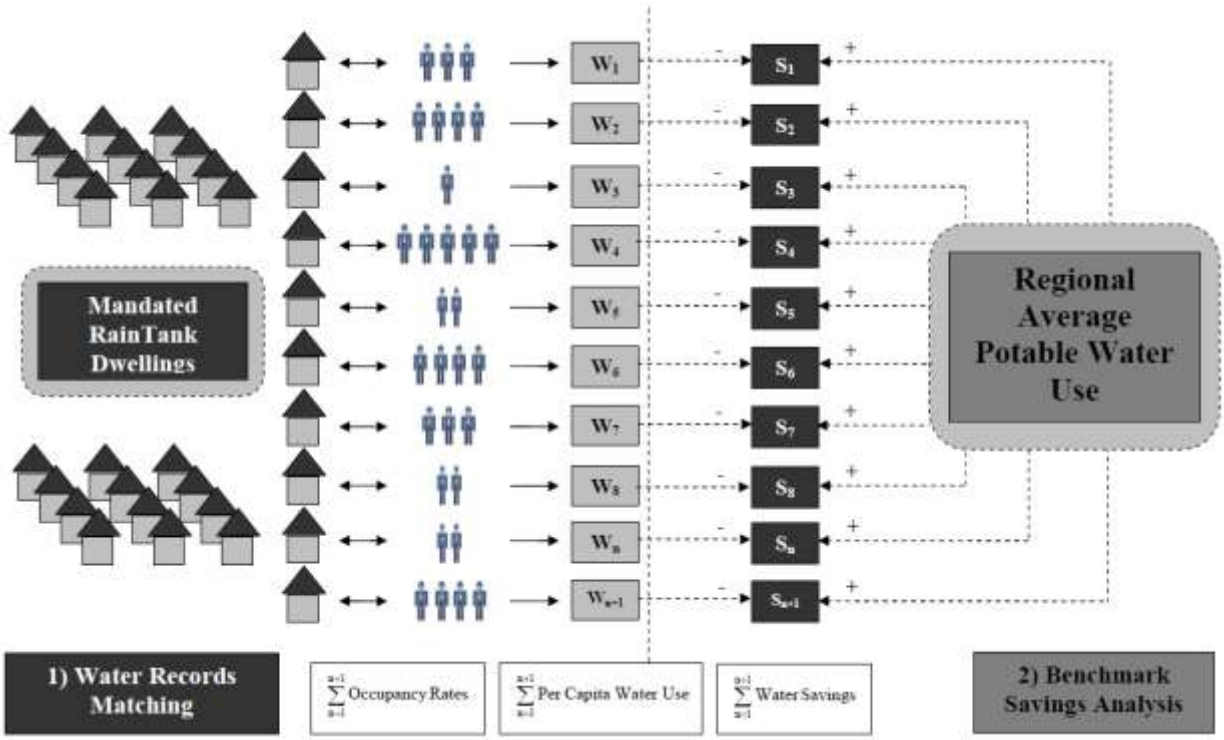


Figure 1. Schematic diagram for the benchmark analysis in estimating the potential mains water saving from dwellings with mandated rainwater tanks.

From Figure 1, the mains water consumption records for each mandated dwelling were matched to their individual household occupancy number obtained from the telephone survey. This was followed by the normalisation of mains water consumption rate to yield the per capita mains water usage (W_X) of each dwelling. The normalised per capita mains water usage data set (W_X) was then individually subtracted from the average mains water use for the respective LGA or region in SEQ to generate the individual mains water saving (S_X). It should be noted that the positive sign notation (+) was used to indicate the resultant mains water savings from mandated dwellings. Hence, any negative values (-) that resulted from the benchmark analysis indicate the mains water consumption at the particular mandated dwellings is actually higher than the regional average mains water consumption value. The annual mains water saving from the mandated dwellings was estimated by summing each of the individual mains water saving (S_X) values. The resultant mains water saving was in litres per person per day (L/p/d). In order to convert this value into kilolitres per household per year (kL/hh/yr), the mean occupancy rate estimated from the mandated dwellings group was used. From the matched water records, it was estimated that the average number of people per household was 3.18 (Redland), 3.20 (Caboolture), 3.21 (Pine Rivers) and 3.34 (Gold Coast).

Results

Figure 2 shows the distribution of normalised per capita mains water usage (L/p/d) for the four LGAs in 2009 and 2010. It is evident that the per capita mains water usage across the targeted LGAs is relatively well-distributed, with the mean usage ranging from 108.3 L/p/d to 138.8 L/p/d. Interestingly, it was observed that the combined mean per capita mains water usage in IPT dwellings for Pine Rivers and Caboolture is approximately 20 L/p/d less than the collective mean of Gold Coast and Redland (Table 1). This might be due to differences in the local rainfall conditions for the studied LGAs or perhaps different behaviours due to the different water restrictions or factors such as water use behaviour in the household.

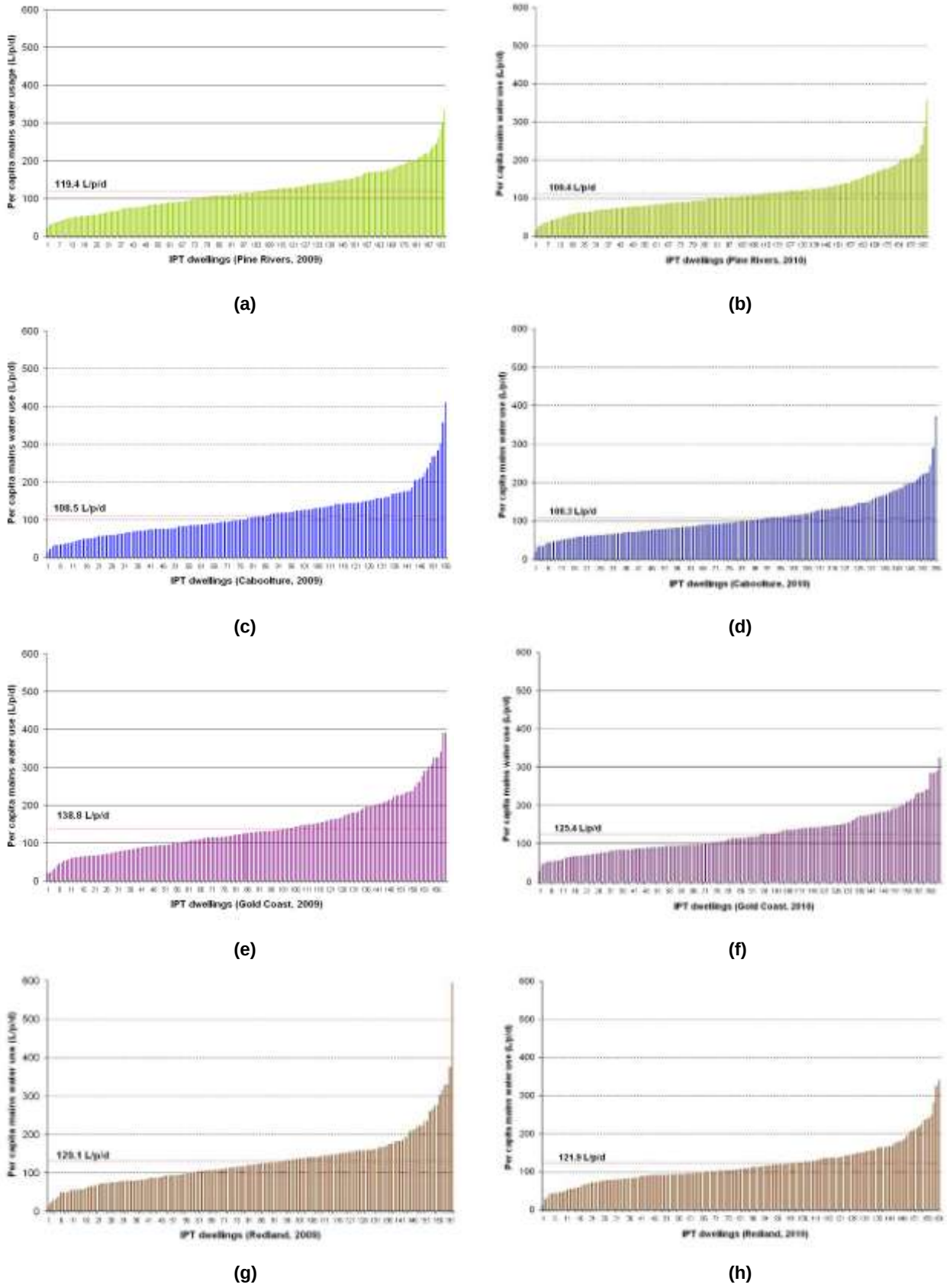


Figure 2. The distribution of normalised per capita mains water usage (L/p/d) in mandated tank dwellings in (a) Pine Rivers, 2009; (b) Pine Rivers, 2010; (c) Caboolture, 2009; (d) Caboolture, 2010; (e) Gold Coast, 2009; (f) Gold Coast, 2010; (g) Redland, 2009; and (h) Redland, 2010.

With these estimated mean per capita mains water usage patterns across the four LGAs, the potential mains water saving from rainwater tanks was analysed relative to the benchmark local average water consumption values reported by QWC. Table 1 shows these data, including estimated mains water savings for IPT dwellings in 2009 and 2010 for the four LGAs of Caboolture, Pine Rivers, Redland and Gold Coast. Results show a per capita reduction in mains water consumption ranging from 15 to 36% for 2009, and from 22 to 33% in 2010. Both the Gold Coast and Redland LGAs showed a larger reduction in mains water consumption (>70 L/p/d), when compared to the published mains water consumption benchmarks in both 2009 and 2010. In contrast, a lower magnitude of mains water savings was found in both Pine Rivers and Caboolture for both 2009 and 2010 with per capita water savings below 41 L/p/d.

Table 1. Estimated mean per capita water consumption (litres per person per day) and average water savings per household (kilolitres per household per year) across four LGAs in SEQ.

Region (Sample Size)	Pine Rivers (197)	Caboolture (158)	Gold Coast (172)	Redland (164)	Across all four LGAs
Benchmark average water consumption in 2009 (L/p/d)	140.4	140.4	211.4	201.5	
Normalised average water consumption in IPT dwellings in 2009 (L/p/d)	119.4	108.5	138.8	129.1	
% savings per person per day in 2009	15%	23%	34%	36%	
Average savings per household per year (kL/hh/yr) in 2009	24.5	37.3	88.5	84.0	58.8
Benchmark average water consumption in 2010 (L/p/d)	143.3	143.3	192.0	183.1	
Average water consumption in IPT dwellings in 2010 (L/p/d)	109.4	108.2	125.7	121.9	
% savings per person per day in 2010	22%	25%	33%	31%	
Average savings per household per year (kL/hh/yr) in 2010	39.7	40.9	81.0	71.0	58.2

Thus for 2009, the average annual mains water savings per household per year ranged from 24.5 kL/hh/yr (lowest, Pine Rivers) to 88.5 kL/hh/yr (highest, Gold Coast), with an average mains water saving of 58.8 kL/household/year across the four LGAs. Similarly, an average mains water saving for 2010 was 58.2 kL/hh/year, with a range from 39.7 kL/hh/yr (lowest, Pine Rivers) to 81.0 kL/hh/yr (Gold Coast) across the same LGAs.

Conclusion

This study has confirmed the effectiveness of mandated rainwater tanks in achieving mains water savings across the four LGAs of Pine Rivers, Caboolture, Gold Coast and Redland in 2009 and 2010. In addition, this study has shown that by accounting for differences in household occupancy rates and validating that IPT dwellings have rainwater tanks installed, a higher mains water savings across LGAs is observed than in the previous study by Beal *et al.* (2011). The remaining variations in mains water savings between LGAs can be explained by other factors such as rainwater tank yield, rainfall, socio-demographic factors (water-wise awareness and household water use behaviour) and the use of water efficient household appliances and fixtures, which could be addressed in future research. The benchmark analysis has provided improved results, mainly due to the normalisation of individual water billing records for the 691 IPT dwellings, confirming that IPT dwellings have tanks installed, as well as the use of updated water billing records for 2009 and 2010. In conclusion, the study has provided the groundwork for further improving the estimation and validation of achieving the potable water savings target of 70 kL/hh/yr for post-2007 IPT dwellings in SEQ.

References

Beal, C., Gardner, T., Sharma, A., Barton, R., 2011. A desktop analysis of potable water reductions from internally plumbed rainwater tanks in South East Queensland, *Urban Water Security Research Alliance Technical Report No. 26 (In publication)*.

Chong, M.,N., Mankad, A., Gardner, T., and Sharma, A., 2011. Baseline demographic analysis of mandated rainwater tank users in South East Queensland. Urban Water Security Research Alliance Technical Report (*In publication*).

Department of Local Government and Planning (DLGP), Queensland Government, March 2010. Regional Planning – South East Queensland. Available from: http://www.dip.qld.gov.au/seq (accessed on 27 April 2011)

Department of Infrastructure and Planning (DIP) 2009. South East Queensland Regional Plan 2009-2031. Department of Infrastructure and Planning, Queensland Government. September 2009.

Sydney Water, 2008. BASIX Monitoring Report: Water Savings for 2007 – 08, Final Report: November 2008, Sydney, New South Wales, Australia.

The State of Queensland (Queensland Water Commission), 2010. South East Queensland Water Strategy. Queensland Water Commission, Brisbane.

Dissecting Rainwater Pump Energy Use in Urban Households

Tjandraatmadja, G.[1], Pollard, C.[1], Sharma, A.[1] and Gardner, T.[2]
[1] CSIRO Land and Water, Highett, Victoria
[2] CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland

Summary

Rainwater tanks are a viable alternative to reduce the demand of mains water and increase the resilience of cities to drought. However, previous studies have found that the energy footprint of rainwater systems can be much higher than the energy footprint for traditional water distribution and is also subject to high variability across various systems in use. This study investigated the operation of pumps used for rainwater supply in a controlled residential environment (a model house) to understand the factors affecting the energy footprint for rainwater supply in urban areas. Pumps commonly installed in urban households operate more efficiently for high flow applications (>15 L/min). However, the majority of household water end uses requires low water flow (<10 L/min), causing the pump to operate in the low energy efficiency range. Energy efficiency varies for each pump. Correct pump sizing and informed selection of system components could improve energy efficiency significantly to <1.5 kWh/kL, lower than or close to the energy footprint of other alternative water sources such as desalination and recycled water.

Keywords
Rainwater tanks, energy, pump system configuration.

Introduction

The widespread uptake of rainwater tanks in urban areas has provided a viable alternative to reduce the demand of mains water and increase the resilience of cities to drought. On the other hand, studies on energy consumption indicate that the energy footprint of rainwater systems, expressed in kilowatt hours per kilolitre (kWh/kL), can be much higher than the energy footprint for traditional water distribution systems (Gardner *et al.*, 2006; Beal *et al.*, 2008; Retamal *et al.*, 2009; SEWL, 2009). Field studies also show that there is significant variability in energy requirements in rainwater supply associated with the various system configurations, technologies, and operation regimens adopted in urban areas, whilst limited information is available on how to optimise their performance (Beal *et al.*, 2008; Retamal *et al.*, 2009; SEWL, 2010; Talebpour *et al.*, 2011). This study investigated the water flow characteristics and the operation of pumps used for rainwater supply in a controlled residential environment (a model house) to understand the factors impacting on the energy footprint for rainwater supply in urban areas.

Methodology

In the model house, rainwater from a tank was supplied by an external pump through ¾ inch polyethylene pipe to the range of common household appliances (Figure 1 and Table 1). Pressure and flow within the water supply system and energy consumed by the pump were monitored in real time at intervals of 0.2s and logged every 1s. To reduce water consumption, water was recycled within the model house.

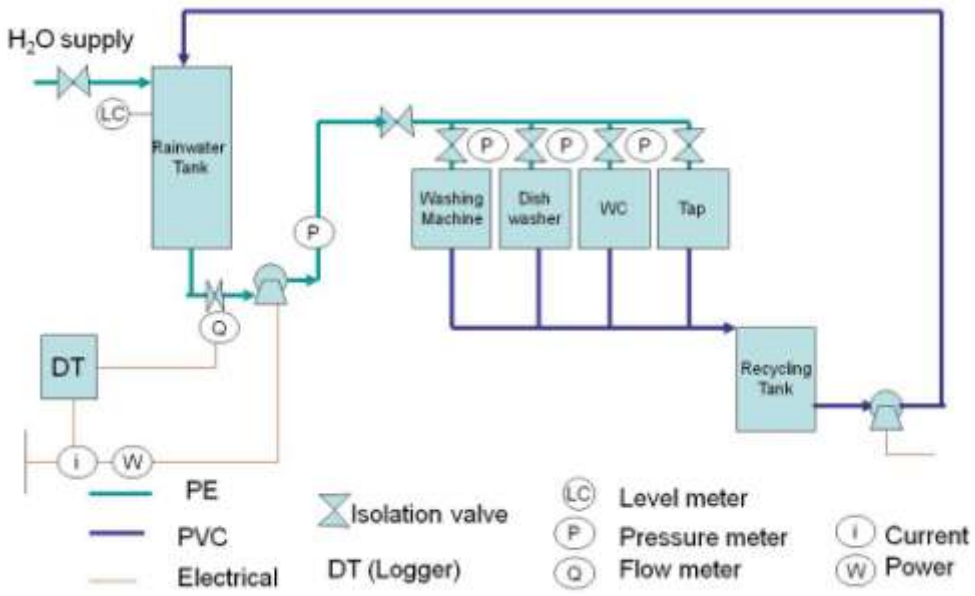


Figure 1. Diagram of rainwater supply system in model house

Table 1. Characteristics of appliances adopted in model house.

Appliance	Average Water Consumption	WELS* Classification
Washing machine (5.5kg capacity)	113 L/wash	4 stars
Dishwasher	15.7 L/wash	2.5 stars
Toilet cistern	Dual flush (6 L/3 L)	n.d.
Tap	Up to 35 L/min	n.d.

Note: *Water Efficiency Labelling Scheme (Australian Government 2011)

Findings

Rainwater End Uses and Water Needs

On rural and peri-urban properties, rainwater is often the sole supply of water adopted for all potable and non-potable end uses. Whilst in urban areas, rainwater is used mainly for non-potable applications, such as outdoor uses, supply of washing machine cold water and filling of toilet cisterns.

Water requirements for household end uses are curtailed within certain limits dictated by design constraints and by the pressure in the water supply system. Water consumption of household appliances and flow rates of fittings, such as taps, showerheads and irrigation devices, are curtailed to increase water efficiency (Australian Government, 2011). In addition, household appliances, such as washing machines and dishwashers are designed to operate within a specific range of flow and/or pressure conditions. Table 2 shows the typical design pressure and flow rates for fittings and some common household appliances sold in Australia. For instance, a washing machine's minimum operating pressure can range from 40 to 100 KPa depending on appliance make and model, and it can sustain pressures up to 1 MPa. Flow from a water-efficient (four to six star Water Efficient Labelling Scheme (WELS)) tap can be restricted from 2 to 7 L/min (Australian Government, 2011).

Table 2. Design parameters for appliance operation

Appliance	Minimum Pressure (kPa)	Maximum Pressure (kPa)	Source
Washing machine	40-100	800-1,000	Products technical specifications
Dishwasher	30-150	800-1,000	Products technical specifications
Toilet cistern	150 *	400*	Standards Australia (1999)
Tap (WELS 4-6 stars)	n.a.	2-7 L/min	Australian Government (2011)

Note: * Standards Australia performance test conditions for water inlet.

In addition, individual end uses are characterised by a typical pattern of water supply. A washing machine and a dishwasher typically require multiple water supply events during a wash cycle, which will require a pump to start multiple times, whilst hand washing and cistern filling tend to be a single continuous event of short duration.

Pump Operation

Earlier studies on tank water supply to dwellings were conducted with a range of pumps with motor power ranging from 0.35 kW to 1.25 kW (Beal *et al.*, 2008; Gardner *et al.*, 2008; Retamal *et al.*, 2009; SEWL, 2009 and 2010). In these studies, the majority of the pumps had motor capacities within the ranges of 0.41 to 0.6 kW (44%) and 0.61 to 0.81 kW (33%).

In this study, three fixed speed external pumps were examined. Pumps A, B and C had capacities of 0.20, 0.55 and 0.75 kW power and could deliver 31, 33 and 45 m maximum head respectively. Fixed speed pumps are the most common type of pump adopted for rainwater systems in urban dwellings (Retamal *et al.*, 2009). Pump selection was conducted to include the two most common pump sizes and a smaller size pump. Pumps A and C were triggered by pressure switches and pump B by a mains water switch. After all pumps were tested, the rig was later modified to incorporate pressure vessels. This paper will incorporate results for pump C and an 18 L pressure vessel as an example.

The energy footprint for rainwater supply for each pump was evaluated for a range of water supply conditions including the operation of each appliance investigated. For any pump, as flow increases the specific energy required for delivery of water decreases, reaching an optimal (minimum) specific energy at higher flow rates (>15 L/min), as shown in the pump curves in Figure 2.

The appliances and end uses tested in this study were toilet cistern filling, hand washing, dishwasher and a washing machine. The operating flow rate requirements during operation were 3.3-4.1 L/min (the dishwasher), 4-6.3 L/min (toilet cistern filling and hand washing) and 9.6-13 L/min (washing machine). Figure 2 shows the range of raintank water flows and the average specific energy measured for water provision by each pump to the washing machine, toilet cistern filling and dishwasher using the three pump sizes in the model house (grey boxes).

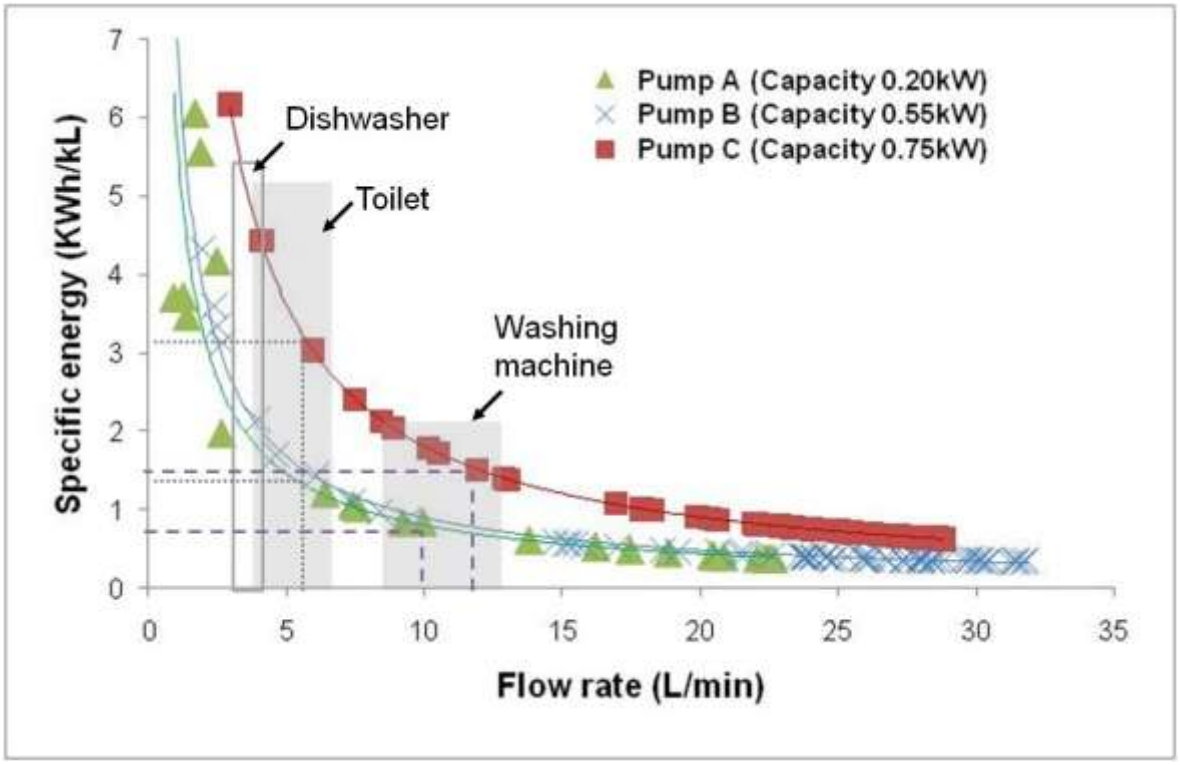


Figure 2. Energy requirements for supply of rainwater to household appliances.

The mean specific energy requirements for each end use ranged from 0.6 to 5.3 kWh/kL for individual end uses and the 3 different size pumps. For instance, pump A supplies rainwater to the washing machine at 10±0.23 L/min and specific energy 0.6±0.05 kWh/kL.

Typical end uses such as toilet, hand washing and dishwashing resulted in high specific energy requirements because of the low flow associated with their operation. The highest flow rate observed was for the washing machine. Thus for the same pump, a rainwater tank connected solely to a toilet cistern would be more energy intensive than a rainwater tank connected solely to a washing machine.

Figure 2 also shows that energy differed across the three pump sizes, with the difference particularly exacerbated in the low flow and high energy use range of the curve (<10 L/min in Figure 2). However, as flow is increased to more than 20 L/min all three pumps displayed optimum performance (optimal energy efficiency). Therefore, reduction of the specific energy for water supply requires either an increase in the water flow requirements during pump operation or adoption of a pump with a low energy versus flow curve.

Energy Distribution

Pressure losses due to friction within the system were minimal (less than 6%) for all three pumps. The majority of the energy was used for tank water transfer. Energy distribution during a pump operation event is comprised of three basic stages: (a) Pump start-up; (b) Flow; and (c) Over run (Retamal *et al*., 2009). Figure 3 illustrates the typical pump operating profile.

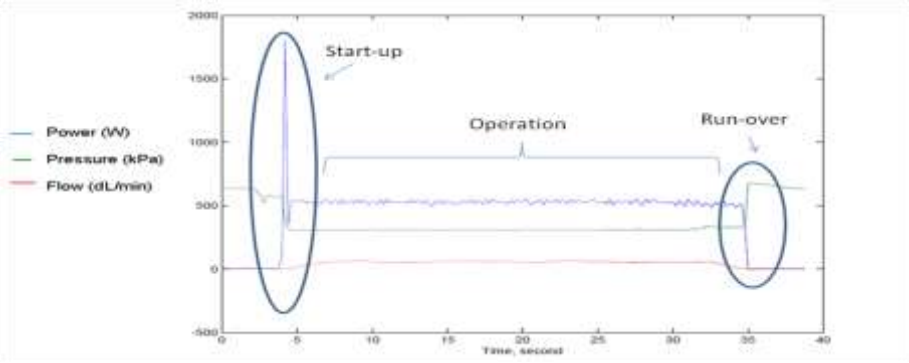


Figure 3. Typical pump operating profile.

As illustrated in Figure 4 for toilet cistern filling, for the larger capacity 0.55 and 0.75 kW pumps B and C, the energy required for pump start-up was minimal (<6% total energy use). However, for the smaller capacity pump A (0.20 kW), a larger proportion of the energy was consumed in the start-up of low flow end uses. For example, the start-up for a full toilet cistern filling consumed 46% of the energy as multiple stop-starts occurred. Energy for the over-run stage also varied across pumps, with negligible energy dissipated in pump A and the most in pump C (25% of total event energy for the half flush).

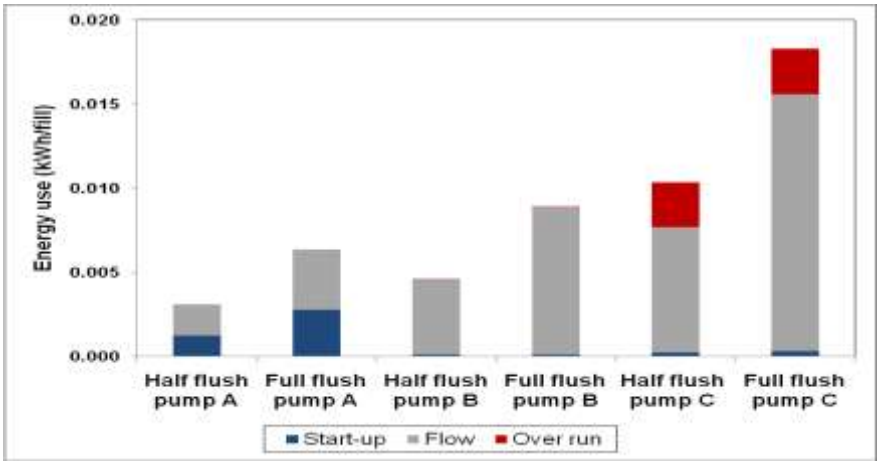


Figure 4. Energy requirements for supply of rainwater to household toilet cistern.

Pump Comparison

Overall, comparison of the specific energy requirements for the three pumps tested showed that the specific energy consumption (Figure 5) and the overall energy consumption for each end use (Figure 6) were lower for the smaller pump, pump A. The variation in specific energy was particularly evident for low flow events such as the dishwasher and toilet cistern filling. However, for end uses with a higher flow of water delivery, such as the washing machine, the differences in specific energy consumption became less marked between pumps, as expected from Figure 2. Hence in this case, choosing the smaller pump A over pump C could reduce the energy for toilet cistern filling by 2/3 (from 3.5kWh/kL to less than 1.5kWh/kL). However, pump design (brand) also needs to be considered in the selection process. Pumps A and B (same manufacturer) produced a very similar specific energy curve despite their power difference (Figure 2). Complementary studies comparing eight different pumps have also shown that the specific energy consumption could vary significantly between pumps of same capacity but of different brands (Hauber-Davidson and Shott 2011, SEWL 2010).

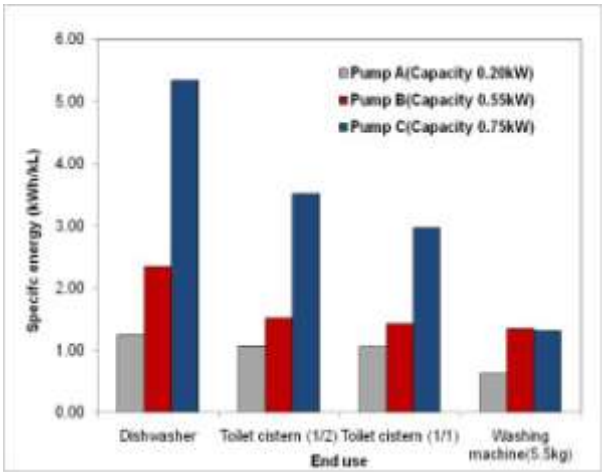


Figure 5. Specific energy requirements for operation of common household appliances.

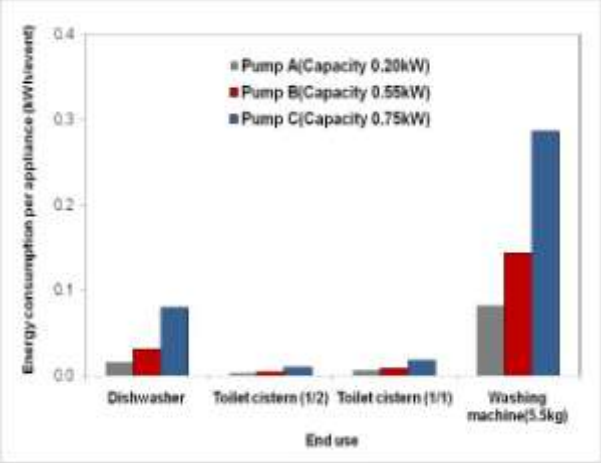


Figure 6. Total energy consumption for pump water supply.

The pressure provided during pumps operation decreased with pump capacity, but all the three pumps tested were able to deliver more than sufficient pressure to operate the appliances tested (Figure 7).

The system configuration adopted (rainwater tank + fixed speed pump + appliance) is the typical set-up adopted for rainwater supply in urban dwellings. These findings confirm some of the initial hypotheses developed through observation of in-situ households by Retamal *et al,* (2009), including that significant energy savings can be achieved by matching end use needs and pump operation. Retamal *et al.* (2009) also concluded that low energy intensity for a rainwater system was not necessarily correlated with lower overall energy consumption in a household. Our outcomes indicate that, for the system adopted, a smaller size pump is able to deliver a satisfactory level of service and use less energy overall for the supply of individual indoor appliances in the same system. The results also show that the energy consumption is not highly influenced by the number of start-ups from a pump, but more often by the flow rate required for water supply.

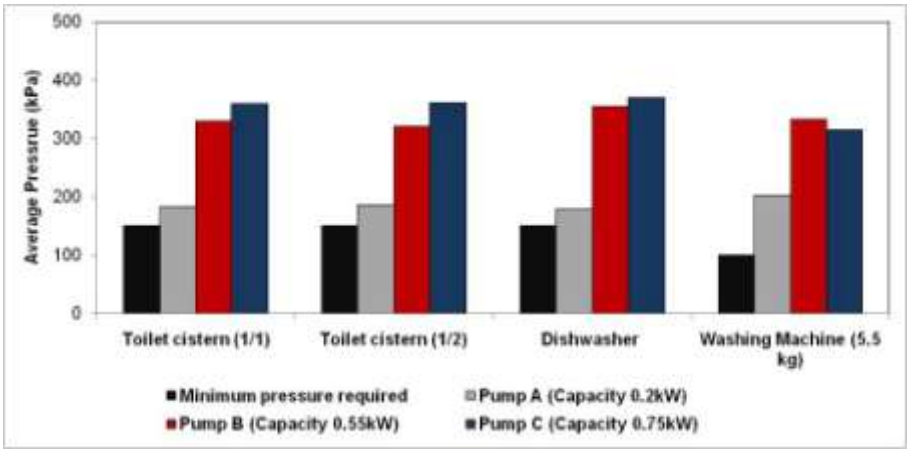


Figure 7. Average pressure provided during pump operation.

The Role of Pressure Vessels

Retamal *et al*. (2009) suspected from the analysis of household energy consumption that pressure vessels could deliver potential energy savings, but were not able to quantify the actual savings. Hauber-Davidson and Shott (2011) verified that a small pressure vessel (5 L capacity) had no effect on pump energy use. Our results indicate that energy savings can be gained when appropriate system configuration, pressure vessel size and end use requirements are considered. The volume of water that a pressure vessel will hold is dictated by the vessel capacity and pump operating pressure. Typically, a pressure vessel holds less water than their nominal volume. Energy savings are generated if the volume of water that the pressure vessel provides is equal to or greater than the total volume of water required for an end use. For instance, this can be illustrated for pump C coupled with an 18 L pressure vessel. An 18 L pressure vessel connected to pump C holds 6.3$\pm$0.09 L, and was able to provide for low volume uses (<6 L), such as hand washing, toilet cistern filling and dishwasher operation as well as intermediate water supply during a

washing machine cycle. As a result, the pump started less often, but more importantly, when the pump filled the pressure vessel it operated at a higher flow rate (average 9.5±0.3 L/min). This led to more efficient pump operation as illustrated in Figure 8, particularly for low volume and low flow end uses.

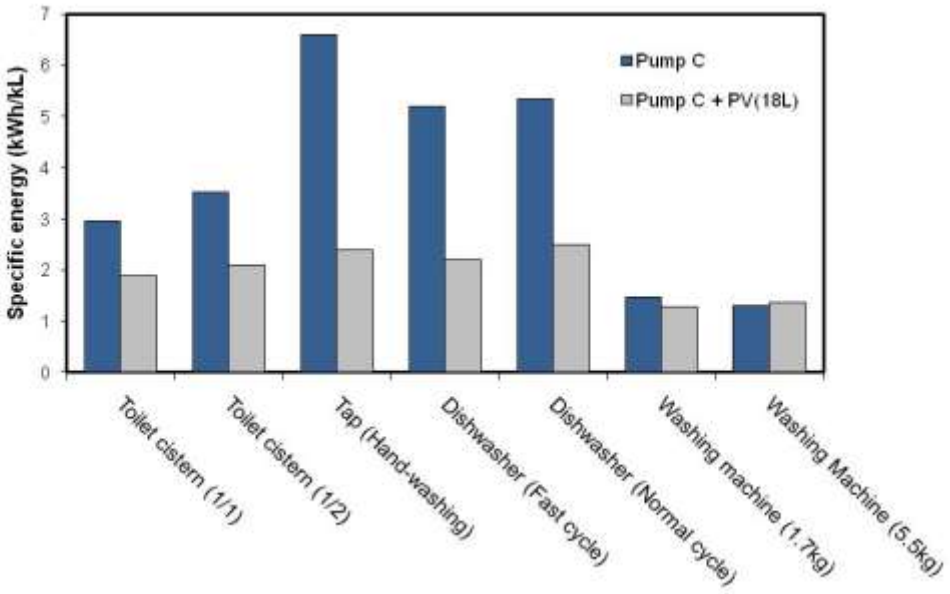


Figure 8. Pump C (0.75 kW) energy consumption with and without an 18 L pressure vessel (PV).

Conclusions

Pumps commonly installed in urban households for rainwater supply operate more efficiently for high flow applications (>15 L/min). However, typical end uses for mandated rainwater supply, such as toilet flushing and washing machine cold water tap, often operate at flows much less than 15 L/min. This in turn causes pumps to operate in their low energy efficiency range resulting in a high specific energy compared with traditional potable water supply, eg, 0.68 kWh/kL for Brisbane (Kenway *et al.*, 2008). Better matching end uses needs and pump size can improve energy efficiency significantly to less than 1.5 kWh/kL, resulting in a lower energy footprint than sea water desalination (3.6 kWh/kL (SEWL, 2009)) or Indirect Potable Reuse (2.8-3.8 kWh/kL (in NSW, 2006)), the common alternatives to supplement dam sourced potable water. Other devices such as pressures vessels can also contribute to reduce energy requirements if adequately sized.

References

Australian Government (2011) WELS Products, http://www.waterrating.gov.au/products/index.html, updated May 2011, accessed May 2011.

Beal, CD, Hood, B, Gardner, T, Lane, J and Christiansen, C (2008). Energy and water metabolism of a sustainable subdivision in South East Queensland: a reality check, *Enviro 2008*, Melbourne.

Gardner, E.A., Millar, G.E., Christiansen, C., Vieritz, A.M. and Chapman, H. (2006). Energy and water use at a WSUD subdivision in Brisbane, Australia, *Australian Journal of Water Resources*, 10(3).

Hauber-Davidson, G. and Shott, J (2011) Energy consumption of domestic rainwater tanks – why supplying rainwater uses more energy than it should, Water,38(3),72-76.

Kenway, S. (2008). Preliminary LCA of the SEQ Water Strategy, Urban Water Security Research Alliance, September 2008.

NSW (2006). A sustainable water supply for Sydney, New South Wales Legislative Council, Sydney.

SEWL (2009). Energy consumption in domestic rainwater harvesting, Prepared by Water Conservation Group for South East Water Limited, South East Water Limited.

SEWL (2010). Energy consumption of domestic rainwater harvesting, Prepared by Water Conservation Group for South East Water Limited, May 2010.

Standards Australia (1999) AS 1172.2 Water closet (WC) pans of 6/3L capacity or proven equivalent- Part 2: cistern, Standards Australia.

Retamal, M, Glassmire, J, Abeysuriya, K, Turner, A, and White, S. (2009). The Water-Energy Nexus: Investigation into the Energy Implications of Household Rainwater Systems, CSIRO-Institute for Sustainable Futures, University of Technology, Sydney.

Talebpour, MR, Stewart, RA, Beal, C, Dowling, B, Sharma, A and Fane, S (2011). Rainwater energy tank end usage and energy demand: a pilot study, Water, March 2011, 97-101.

A GIS Approach for Identifying the Location and Attributes of Land Parcels for Stormwater Harvesting and Re-Use in Greenfield Urban Development

McIntosh, B.S.[1,2,3], Hodgen, M.[1], Laredo, L.[4], Wolf, L.[1], Gardner, T.[6], Chowdhury, R.[7], Maheepala, S.[1]

[1] CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland
[2] Smart Water Research Centre, Griffith University, Gold Coast, Queensland
[3] International WaterCentre Pty, Brisbane, Queensland
[4] CSIRO Ecosystem Sciences, Ecosciences Precinct, Dutton Park, Queensland
[5] CSIRO Land and Water, Highett, Victoria
[6] Central Queensland University, Gladstone, Queensland
[7] University of South Australia, Adelaide, South Australia

Summary

Stormwater harvesting and reuse is a national research priority in Australia. Whilst several studies have indicated its effectiveness in reducing the hydrological impacts of urbanisation on receiving streams, information and methods for the spatial identification of potential suitable sites for stormwater harvesting and reuse are not well developed. This paper reports on progress in the development and application of a method for the identification and characterisation of stormwater harvesting locations in greenfield development sites based on a combination of suitability mapping and location specific supply and cost assessment. Using a combination of hydrological modelling, previous feasibility studies, and expert opinion workshops, a set of rules based on catchment and development characteristics were developed to determine the suitability of parcels of land within urban development sites. The method being developed employs a selection of these rules within GIS to produce a constrained map of potential development locations within greenfield sites. Separately, but also within GIS, the routing of runoff in streams down the catchments within which the development lies, and the location of sub-watershed pour points determined (as potential harvesting locations). The stream network information is then combined with land cover mapping and precipitation data to calculate runoff and harvestable stormwater volumes. The yields and costs of supplying harvested stormwater are then determined for different housing density options and displayed spatially as a set of nested areas with a range of supply reliabilities around each sub-watershed pour point. Progress on the application of the method to the Ripley Valley greenfield development near Ipswich is presented and discussed.

Keywords
Stormwater harvesting, greenfield, urban development, mapping, spatial analysis, suitability.

Introduction

Stormwater harvesting and reuse has recently been identified as a potential alternative for potable water demand reduction (DIP, 2008). Stormwater management philosophy has evolved from the conventional but still important 'flood mitigation' paradigm, to the current 'runoff quality control' approach, and now is progressing towards the 'harvesting and reuse' concept, whilst retaining the previous two targets. Stormwater harvesting has been incorporated into the broad concept of integrated urban water management (IUWM) which emphasises diversification of urban water sources in order to embed resilience into the whole of the urban water (Burn *et al.*, 2011; Maheepala and Blackmore, 2008; Coombes and Kuczera, 2001), whilst providing ecological benefits through mitigating the frequency and magnitude of storm flow events resulting from increased impervious area from urban development (Fletcher *et al.*, 2008). Such hydrological changes are known to constitute one of the primary degrading influences on aquatic ecosystem structure and function (Poff *et al.*, 2009).

Stormwater harvesting and reuse schemes require infrastructure for capture, storage, treatment, maintenance and supply to end users in cost effective ways. Stormwater capture, storage and treatment infrastructure in particular requires space which may preclude uses for other purposes, and may increase the overall costs of any greenfield development, where less space for housing usually equates to lower profit margins for the developer. With 35 identified greenfield development areas in South East Queensland (SEQ) which will add an additional 754,000 new dwellings by 2031 (SEQ Regional Plan, 2009), the key implementation questions are therefore how might we decide where to site stormwater schemes? Can they be implemented anywhere? Which sites are viable in terms of presenting a reliable supply of appropriately treated non-potable water at reasonable cost?

Comparative analysis of stormwater harvesting options across a range of housing densities in SEQ urban developments (Bligh Tanner and Design Flow 2009) has concluded that medium density (40 dwellings per ha) developments of at least 20ha, (and ideally 100ha or more), are preferable from a cost perspective, and that harvesting, storage, treatment and reuse by large, external (off site) non-potable users maximises yields and reduces costs more effectively than reuse for irrigation at allotment or sub-catchment scales. The extent to which stormwater

yields and reliabilities of supply are sensitive to watershed topographic, land cover, precipitation and hydrologic characteristics remains unclear. Current work has employed planned street and development layouts to assess yield, reliability and cost rather than starting from the clean slate of a greenfield development. As a consequence, there are no methods available to show where and how stormwater might be harvested with no knowledge of street, neighbourhood or whole development layout.

To fill this gap, and provide a method for providing an assessment stormwater harvesting potential to feed into the greenfield site structure planning processes, this paper presents a novel GIS and location assessment approach able to identify potential locations for stormwater harvesting in greenfield developments, and to assess their relative suitabilities in terms of yield, reliability and cost. The paper first describes the logic of suitability mapping methods which have been developed and applied elsewhere for other land and water planning and management purposes. Then the derivation of catchment and other rules for identifying stormwater harvesting opportunities will be presented and discussed, and finally, the steps of the method being developed and progress on applying the method to the Ripley Valley greenfield site in SEQ presented.

Suitability Mapping – some Background

The idea of representing an area of land as a set of parcels of regular (eg, a grid) or irregular dimensions, then simulating how the land uses/covers might evolve over time in response to drivers (eg, population growth, economic subsidies, climate change) and constraints (eg, land use zoning, topography, water availability) has been employed since the early 1990s (White and Engelen, 1993) and is now a well-known technique in integrated regional planning (Oxley *et al.*, 2004; Van Delden *et al.*, 2007). In this technique, the idea is that the suitability of a parcel of land for different uses (eg, stormwater harvesting) depends on the characteristics of that land eg, slope, volume of runoff, proximity to natural storage location such as a lake etc. A simple approach is to either say a parcel of land is, or is not, suitable, depending on the characteristics of concern. A more sophisticated approach involves using a scale of 0 (not at all suitable) to 1 (completely suitable) in relation to each characteristic. By combining the suitabilities against each characteristic, a compound map can be generated and used for various purposes. An example to illustrate this idea of overlaying multiple map layers to identify the suitability of parcels of land for particular uses is shown in Figure 1. In this example: (1) shows a set of maps, one for each biophysical characteristic of relevance to the suitability of land for a particular use; (2) shows the form of the relationship between biophysical characteristics and suitability (between 0 and 1) for the land use in question; (3) shows the calculated suitability maps for the land use in relation to each characteristic derived from applying the relationships (equations) of 2 to the maps from 1; and (4) the result of combining (through averaging or more sophisticated means) the set of calculated suitability maps, to provide a compound suitability map for the land use of interest.

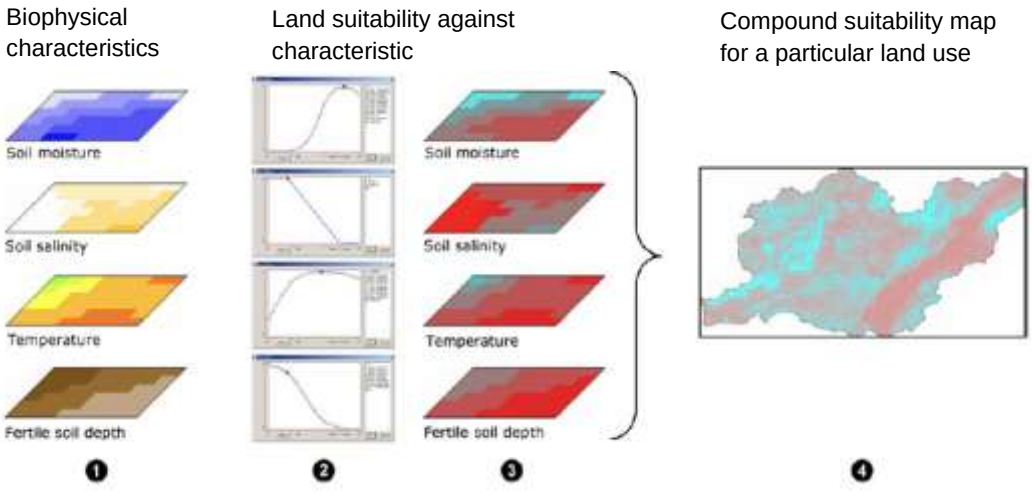


Figure 1. Producing a compound suitability map for a particular land use based on the suitability of that land use in relation to a set of biophysical characteristics (Adapted from van Delden *et al.,* 2007).

A Suitability-Based Mapping Approach for Identifying Stormwater Harvesting Opportunities

Over the last couple of years, several studies in SEQ have attempted to identify different aspects of stormwater harvesting potential. For example, McJannet *et al.* (2008) studied the potential for maximising water storages in SEQ by enhancement of runoff and infiltration. Their study estimated that approximately 15,000 ML/year of water (4% of SEQ water requirements) could be produced by means of runoff enhancement through surface compaction of all

suitable areas within 4 km of Wivenhoe and Somerset storages in SEQ. Surface compaction was recommended as the best option to enhance runoff and the estimated cost was $0.47/kL. Bligh Tanner P/L and Design Flow P/L (2009) identified that a maximum yield of 8 to 10 kL/ha/day is possible from stormwater harvesting in urban areas when there is a large and regular demand for the treated water, or where large storages are available. Their study also revealed that for larger (>20ha, ideally 100ha) developments, stormwater harvesting has both greater supply potential and at lower cost than domestic rainwater harvesting.

However, methods for identifying *where* stormwater harvesting could be employed based on available runoff and stream hydrological modeling, and how those spatially located opportunities compare in yield and cost terms within the scale of new urban developments are not mature. Being able to identify on a map which areas are suitable for stormwater harvesting would constitute an advance in relation to such methods, and should improve the planning of greenfield developments in relation to using alternative water sources.

Based on hydrological modelling of some catchments in SEQ (Chowdhury *et al.*, 2010), previous feasibility studies, and a set of specifically designed expert opinion workshops on stormwater harvesting principles, a set of suitability rules organised into three categories were developed in relation to SEQ. They are presented below, but not all are quantitatively defined at this stage:

(a) Catchment Rules

Catchment rules help identify catchments suitable for stormwater harvesting based on the slope, location, runoff volumes, availability of space for storing water above or in the ground, suitability of the aquifer for storing water underground, and the sensitivity of the surrounding environment. Several catchment rules are listed below:

- Favourable catchment slope (2% ~10%) - too flat or too steep slope incurs additional cost for earthworks in order to establish stormwater collection and distribution conveyance.
- Location of high environmental value (HEV) waterways downstream of greenfield development sites – this links to the national priority to protect stream ecosystem health from urbanisation impacts.
- Available, accessible and cost effective open spaces for storage and treatment.
- Availability of suitable aquifers for storage as an alternative to large tanks.
- Complexity of topography, such as higher stream order or a single exit point. Hydrologically higher stream orders carry more flow than lower stream orders and therefore are more favourable for stormwater harvesting.
- Maximum catchment area per unit of storage volume.
- Land use history to avoid any known detrimental water quality issues such as cattle dips, contaminated industrial land, etc.

(b) Development Rules

Development rules help identify the feasibility of stormwater schemes within a development site:

- Cost effective development size (20 ha ~ 100 ha is favourable) - small development size is not cost effective due to multiple treatment facilities, whilst large development size may not be cost effective for establishing distribution networks(ie dual reticulation).
- Housing density (> 30 dwelling/ha) - ensures a regular and sustained level of internal household demand for stormwater reuse. Ensuring high levels of internal domestic sub-potable demand is an important element, along with siting harvesting points near to large non-domestic sub-potable uses, of ensuring harvested stormwater is actually used, and helps provide independence from external household demand levels (gardening for example) which are known to be climatically dependent and consequently unreliable components of 'base' demand.
- Location within SEQ future greenfield development sites (35 greenfield sites have been indicated in the SEQ Regional Plan 2009).

(c) Facility Rules

Facility rules are existing infrastructure or favourable conditions for implementation of stormwater harvesting schemes within or around the development site.

- Ease of developing diversion structures (barrages or weirs) that require engineering feasibility assessment eg, favourable terrain and soil properties.
- Ease of incorporating bioretention pre-treatment facilities into the scheme layout for preliminary treatment and flow retention.
- Location of a nearby high water user, for example a manufacturing industry or golf course.
- Location of adjacent water impoundments.

Applying a sequence of such rules offers the possibility of being able to locate and characterise stormwater harvesting opportunities in greenfield urban developments. However, such rules can really only act to constrain development space to a smaller sub-space within which stormwater harvesting might potentially work. The application of a suitability-based map overlaying method cannot identify specific locations - points for collection and storage of stormwater – nor can such an approach in itself evaluate the stormwater yields, supply reliabilities or costs of supply at each location, for different densities of housing. Instead one must look to a broader range of tools.

A Combined Suitability Mapping and Location Assessment Approach to Stormwater Harvesting Opportunity Identification

To address these issues a method which combines a set of biophysically based suitability rules to identify, on the basis of constraint, the sub-area of a greenfield development area within which stormwater harvesting could take place. Spatial analysis of harvestable yield and cost across a range of supply reliabilities and housing densities is being developed. Table 1 describes the steps in the method being developed.

Table 1. Steps in a combined suitability mapping and location assessment method for the identification and characterisation of stormwater harvesting opportunities in greenfield urban development sites.

Step	Description	Purpose
1	Create ArcGIS maps and digital elevation model (DEM) of the development area and containing watershed(s) and determine minimum raster size.	To provide the primary spatial data on which to work (including topography), and the size of the basic unit of spatial analysis.
2	Mapping of existing land cover in the development area and containing watershed(s).	To generate a runoff coefficient map for the watershed(s) and to identify pre-existing potential storage locations.
3	Create stream network for watershed(s) using ArcHydro, and identify sub-watershed extraction points.	To determine how runoff flows down the watershed(s) and where stormwater might be harvested (the pour points) and to provide a means of calculating runoff volumes
4	Apply constraining spatial data to the development area with regards to slope (exclude all land with a slope > 15% from urbanisation) and protected natural areas (exclude these from development).	To determine the area in which urbanisation might really take place.
5	Determine daily water volumes at each extraction point over the period of simulation under baseline (existing land cover, without urban development).	To determine the flows of stormwater which should not be harvested ie, the existing creek flows (an estimate for environmental flows).
6	Determine potentially harvestable daily water volumes at each pour point over the simulation period under different spatial patterns and densities of housing development by subtracting the baseline volumes (from step 5) from the higher volumes resulting from urban runoff taking into account the reduction in downstream harvestable volumes.	To determine how much stormwater yield there might potentially be across the development under different harvesting scenarios (lots of smaller harvesting points, or fewer larger downstream points).
7	For each harvesting scenario (multiple smaller harvesting points, or fewer larger downstream points) determine the maximum areas which could be supplied at 70%, 80% and 95% supply reliabilities under the given housing density and any landscape or other irrigation uses.	To determine the location and size of the urban areas which could be supplied using harvested stormwater – the areas will decrease in size as demanded reliability increases shown as different coloured ‘reliability zones’ on the map around each harvesting location.
8	Calculate the storage volumes required to realise the yield required to satisfy the 70%, 80% and 95% supply reliabilities determined under step 7.	Storage equates to approximately 30% of harvesting scheme costs and varies depending on desired reliability of supply – this step provides the basis for calculating this cost and determining in part, scheme feasibility (on the basis of cost or required storage footprint). Storage costs will depend on the sophistication of the option eg, buried tanks (most expensive) through constructed wetlands / ponds to the utilisation of existing landscape features (least expensive).
9	Calculate the treatment and distribution costs for supplying each ‘reliability zones’ taking into account pre-existing storage opportunities.	Treatment and distribution costs are the other major cost elements in scheme construction and will be determined by this step.
10	Associate yield and cost attributes with each ‘reliability zone’ on the development map.	To provide a visual output from the analysis with associated yield and cost attributes.

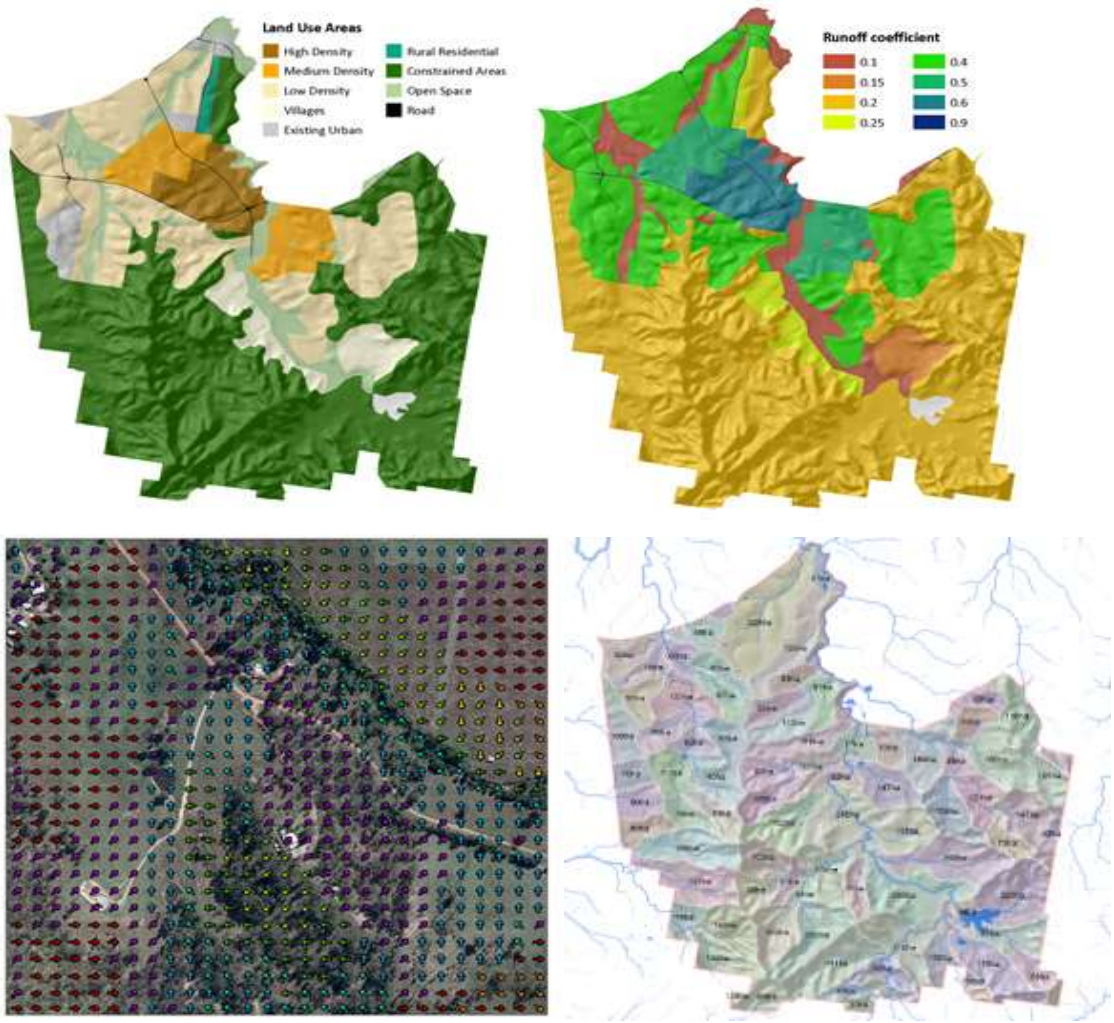


Figure 2. Land use mapping (top left) can be used to generate a runoff co-efficient map (top right), which, with ArcHydro can calculate flow paths and volumes (bottom left) and generate a map of sub-catchments which can then be assessed for their stormwater runoff yield (note: the pattern shown is illustrative only at this time).

Steps 1 through to 4 of the method have been developed and applied at the Ripley development in SEQ, near Ipswich. ArcHydro (Maidment, 2003) has been used to generate runoff maps based on topography from a DEM, land use maps (and therefore runoff co-efficients), and rainfall data. Figure 2 illustrates how land cover is used to map runoff co-efficients, which are then employed by ArcHydro together with a DEM to calculate flow paths then runoff volumes, and subsequently to identify the locations and assess the yield and costs of supply at different reliability levels and housing densities. The resulting set of suitable sub-catchments can then be overlain with further suitability constraints such as a map of protected vegetation areas (which imply no construction), and a map of slope (to exclude those areas with slopes >10%).

Conclusion

Stormwater harvesting offers multiple solutions to urban water systems such as mains water demand reduction, water quality improvement and possibly creek ecosystem health protection (Fletcher *et al.*, 2007, 2008). While several previous studies have identified the benefits of stormwater harvesting for urban streams, methods for practically informing where to employ stormwater as an alternative water source of choice in greenfield sites are not sufficiently well developed. We do not understand how the yields and costs of different harvesting strategies (single centralised collection vs. multiple decentralised collection) vary with the spatial location of harvesting points and uses, nor do we have well developed methods for determining how to better co-locate harvesting and demand during structure planning processes.

An opportunity exists to employ a method incorporating both suitability mapping and location-based assessment of potential harvesting points within such developments as a way of informing development at an early planning stage without the need for detailed hydrological monitoring. Early results from the development and application of this method at the Ripley Valley are promising.

References

Bligh Tanner and Design Flow (2009). *Stormwater infrastructure options to achieve multiple water cycle outcomes*, Report to the Queensland Water Commission, Bligh Tanner Consulting Engineers [executive summary available online http://www.qwc.qld.gov.au/planning/pdf/twc-009-08-31-swh-executive-summary.pdf].

Burn, S., Maheepala, S. and Sharma, A. K. (2011). The role of decentralisation in Integrated Urban water management, Key note speech, *Proceedings of the International Conference on Integrated Water Management*, 2–5 February 2011, Murdoch University, Perth, Australia.

Chowdhury, R., Gardner, T., Gardiner, R. Chong, M. and Tonks, M. (2010). *Catchment hydrology modelling for stormwater harvesting study in SEQ: from instrumentation to simulation*, Science Forum 2010, Urban Water Security Research Alliance, Brisbane, Australia.

DIP (2008). *Water saving targets, Queensland Development Code Part MP 4.2*, Department of Infrastructure and Planning (DIP), Australia.

Fletcher, T., Mitchell, V.G., Deletic, A., Ladson, T.R. and Seven, A. (2007). Is stormwater harvesting beneficial to urban waterway environmental flows?, *Water Science and Technology* 55(4): 265-272.

Fletcher, T., Deletic, A., Mitchell, V.G. and Hatt, B.E. (2008). Reuse of urban runoff in Australia: a review of recent advances and remaining challenges, *Journal of Environmental Quality* 37:116-127.

Maidment, D.R. (2003). *ArcHydro: GIS for Water Resources*, ESRI.

McJannet, D., Cook, F., Hartcher, M. and Burn, S. (2008). Maximising water storage in south-east Queensland reservoirs: evaluating the potential of runoff and infiltration enhancement. CSIRO: Water for a Healthy Country National Research Flagship. Urban Water Security Research Alliance Technical Report No. 5.

Oxley, T., McIntosh, B.S., Winder, N., Mulligan, M. and Engelen, G. (2004). Integrated modelling and decision-support tools: a Mediterranean example. *Environmental Modelling and Software* 19, 999-1010.

Poff, N.L., Richter, B.D., Arthington, A.H., Bunn, S.E., Naiman, R.J., Kendy, E., Acreman, M., Apse, C., Bledsoe, B.P., Freeman, M.C., Henriksen, J., Jacobson, R.B., Kennen, J.G., Merritt, D.M., O'Keeffe, J.H., Olden, J.D., Rogers, K., Tharme, R.E. and Warner, A. (2009). The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards, *Freshwater Biology* 1-24.

SEQ Regional Plan (2009). *South east Queensland regional plan 2009-2031*, The State of Queensland (Department of Infrastructure and Planning), Australia.

Van Delden, H., Luja, P., Engelen, G. (2007). Integration of multi-scale dynamic spatial models of socio-economic and physical processes for river basin management. *Environmental Modelling and Software* 22 (2), 223-238.

White, R., Engelen, G. (1993). Cellular automata and fractal urban form: a cellular modelling approach to the evolution of urban land use patterns. *Environment and Planning A* 25, 1175-1199.

Assessing the Effectiveness of South East Queensland Frequent Flow Management Objectives against a Range of Urbanisation Extents for Tingalpa Creek

Ashbolt, S.[1], McIntosh, B.S.[1,2,3], Gardiner, R.[4], Chowdhury, R.[5], Gardner, T.[6], Aryal, S.[1], Maheepala, S.[1] and Sheldon, F.[7]

[1] CSIRO, Highett, Victoria
[2] International WaterCentre, Brisbane, Queensland
[3] Smart Water Research Centre, Griffith University, Gold Coast, Queensland
[4] Department of Environment and Resource Management, Queensland
[5] University of South Australia, South Australia
[6] Central Queensland University, Queensland
[7] Griffith University, Nathan, Queensland

Summary

'Urban stream syndrome' is a well-recognised set of ecological, river morphology and water quality degradation symptoms caused by changes to stream hydrology resulting from additional impervious area (IA) stormwater runoff. This paper presents an assessment of the impact of reducing stormwater runoff flows from urban IA on catchment hydrology, through the application of the South East Queensland frequent flow management objectives (FFMOs). A calibrated and validated Storm Water Management Model (SWMM) was developed, representing the largely non-urban, naturally vegetated 2785 hectare Tingalpa Creek catchment south east of Brisbane city. The SWMM was used to first simulate the impact of increasing degrees of urbanisation in terms of percentage total IA on a range of creek hydrology components, and to then simulate the application of the FFMOs in terms of reducing or altering the impacts of urbanisation towards pre-development hydrological characteristics. Impacts of urbanisation, and of the application of the FFMOs were characterised by means of flow exceedance curves (showing the percentage of time over a year that each daily flow value is exceeded) and a standard set of, ecologically relevant hydrological parameters.

Keywords
Stormwater harvesting, frequent flow management objectives, Storm Water Management Model, SWMM, Tingalpa.

Introduction

The urban stream syndrome (Walsh *et al.,* 2005) is a well-recognised set of symptoms associated with the biophysical degradation of streams in and around urban areas, primarily as a consequence of changes to flow regime from increased surface runoff from impervious areas. Such increased runoff brings with it a wide range of organic and inorganic pollutants, but importantly changes the hydrology of streams, potentially to a significant extent. Urban impervious area storm runoff can increase the frequency of high flow events; decrease infiltration and base flow; and increase high flow event magnitude, duration and rate of rise. As a consequence of such changes, urban stream syndrome is associated *inter alia* with lower numbers of sensitive invertebrate and fish species and diversities, higher erosion rates and higher nitrogen and phosphorus levels (Walsh *et al.,* 2005). Changes to stream flow regime are held to be a key determinant of aquatic ecosystem structure and function (Poff *et al.,* 2009), and as a consequence, the management and mitigation of hydrologic changes resulting from urban development is a key aspiration for healthy waterways management in the context of urban growth.

A number of technological options are available to developers or local authorities to mitigate hydrological changes, including: water sensitive urban design features such as swales; attenuation or bioretention of collected stormwater flows and release over a longer period of time; and harvesting, treatment and reuse of collected stormwater to satisfy a range of non-potable demands (Fletcher *et al.,* 2008). Stormwater harvesting for the purposes of providing water supply has been shown to be an effective means of mitigating hydrologic changes to urban streams arising from urban impervious areas (Fletcher *et al,.* 2007).

The South East Queensland Regional Plan (2009) establishes a set of objectives for urban developments to maintain streams as close as possible in hydrological terms to pre-urban conditions (also called pre-development or reference conditions). Implementation Guideline Number 7 'Water sensitive urban design: Design objectives for urban stormwater management' (DIP, 2009) provides a suite of design objectives for best practice stormwater management, including a set of frequent flow management objectives (FFMOs). The FFMOs are designed to mitigate the increase in frequency and magnitude of flow associated with increased impervious areas from the urbanisation of catchments. To do so, the Guideline sets the following objectives for developers to meet:

From the proposed development, capture and manage:

- the first 10mm of runoff from impervious surfaces where the total impervious surface is 0 to 40%.
- the first 15mm of runoff from impervious surfaces where the total impervious surface is greater than 40%.

Note: the capacity to capture runoff must be restored within 24 hours of the runoff event.

Compliance with this objective can be demonstrated by providing a total stormwater capture volume, calculated as follows:

Capture volume (m^3) = Impervious area (m^2) x target design runoff capture depth (mm/day) / 1,000.

The spatial distribution of the required capture volume may be adapted to suit individual site conditions, provided that the required volume from all impervious areas is captured before leaving the site. Management of captured stormwater should include one or more of the following:

- Stormwater evaporation;
- Stormwater reuse (including roof rainwater collection and use); and
- Infiltration to native soils or otherwise filtered through an appropriately designed soil and plant stormwater treatment system, such as bioretention.

To determine the effectiveness of the FFMOs this paper presents a simulation modelling analysis of the impact of the FFMOs across a range of urbanisation extents (% total impervious area values) for a currently undeveloped (1% TIA) catchment south east of Brisbane. The model used was calibrated using 18 months of detailed, sub-hourly flow measurements from the creek concerned, Tingalpa.

Methodology

The 2,785 hectare Tingalpa Creek catchment drains to a gauging point located at 25.576 degrees south, 153.182 degrees east near the township of Sheldon. The gauge is located upstream of Tingalpa Reservoir. This catchment is largely undeveloped, with some rural residential properties and roads contributing to a total of approximately 1% impervious area. The impervious area was determined through automated image analysis of geo-referenced colour aerial photographs. Quality continuous water level data has been measured since April 2009 using a pressure transducer at the catchment outlet and combined with rating curves determined from the cross-section profile to obtain streamflow time series. A 0.2mm tipping-bucket raingauge also collects instantaneous rainfall data.

These data were used to simulate the hydrology of the catchment in the Stormwater Management Model (SWMM) (Huber, 2003). SWMM includes capacity to simulate surface runoff and routing using non-linear reservoir routing, losses, and groundwater flows. Model parameters included physically measured parameters (eg, stream cross-section, catchment area, impervious fraction, rainfall intensity, evaporation) and non-measured parameters that were adjusted during a trial and error manual model calibration process. The calibration process was undertaken using measured streamflow and rainfall data for an hourly timestep from 01/04/2009 to 31/03/2010 with a Nash Sutcliffe criterion of efficiency of fit of 0.50 (Nash and Sutcliffe, 1970). Validation was also undertaken as more data became available for the period 01/04/2010 to 06/05/2011, with a Nash Sutcliffe criterion of 0.33. The low validation fit is thought to be due to the short time series of data used, and the occurrence of significant 1 in 100 year flooding event during the period monitored. Action is being taken to recalibrate as more data is gathered.

The catchment is represented in SWMM using a series of subcatchment areas connected to nodes representing junctionsand links representing the stream network. Urbanisation in the model was represented by increasing the impervious area in the catchment with respect to reference or predeveloped conditions (0% imperviousness). The urbanisation was concentrated in subcatchments in the upper portion of the catchment, proceeding downstream with increases in impervious area to represent a range of urbanisation extents. Each subcatchment was urbanised at up to 100% imperviousness (up to a total of 70% of whole catchment), equivalent to produce the total catchment impervious percentage of interest. This urbanisation approach was necessary in the SWMM model so that only impervious runoff would be harvested from the node downstream of urbanised catchments, as impervious and pervious flow could not be separated without affecting groundwater flows. Pervious runoff from an incompletely urbanised catchment was routed to a node adjacent downstream of the proposed harvesting point, so that only impervious flow was harvested. At the harvesting point, a flow diversion representing extraction to an off-stream storage was implemented using a cut-off flow divider set to the required capture volume, calculated as either 10 or 15mm multiplied by the impervious area according to the frequent flow management objectives. Daily required capture volumes were converted from cubic metres to megalitres given the magnitude of flow from this catchment.

To assess the impact of these scenarios of urbanisation and runoff capture across a variety of flow conditions, a longer rainfall record was obtained for the nearly 40 year period 1/11/1971 to 31/03/2011 from Bureau of Meteorology gauge number 040458 at Capalaba Water Treatment Plant, 4.9km from the catchment outlet and 3.4km from the project's rain gauge, which has only 2 years of record. The model was run for each scenario, as outlined in Table 1, on a daily runoff timestep with hourly routing.

Table 1. Scenarios for flow rule testing in SWMM.

Scenario Number	Scenario	Impervious Area (%)	Impervious Area (ha)	Required Capture Volume (ML/day)
1	Pre-development (0% impervious)	0	0	n/a
2	Existing (1% impervious spread throughout catchment)	1	28	n/a
3	1% impervious	1	28	n/a
4	1% impervious with runoff capture	1	28	2.78
5	5% impervious	5	139	n/a
6	5% impervious with runoff capture	5	139	13.90
7	10% impervious	10	278	n/a
8	10% impervious with runoff capture	10	278	27.80
9	20% impervious	20	556	n/a
10	20% impervious with runoff capture	20	556	55.59
11	30% impervious	30	834	n/a
12	30% impervious with runoff capture	30	834	83.39
13	40% impervious	40	1112	n/a
14	40% impervious with runoff capture	40	1112	111.19
15	50% impervious	50	1390	n/a
16	50% impervious with runoff capture	50	1390	208.48
17	70% impervious	70	1946	n/a
18	70% impervious with runoff capture	70	1946	291.87

The impact of each scenario was assessed using the simulation results of flow at the catchment outlet. A water balance test was conducted to ensure that the flow amendments were implemented as planned.

Results

The results are plotted in a flow exceedance curve for selected scenarios, where the plot shows likelihood of a given discharge being equalled or exceeded. Figure 1 demonstrates the impact on runoff of increasing urbanisation in the catchment. Figure 2 demonstrates the impact of flow harvesting on the urbanised catchments' flow regimes. Comparisons can be made of the effect of flow capture in returning to the pre-developed (0% impervious) conditions. Other flow summary statistics are also calculated for comparison of the flow timeseries in Table 2 – a subset of the standard reporting metrics using the River Analysis Package (RAP) (eWater ToolKit - http://www.toolkit.net.au/rap).

Increasing Impervious Area

Mean flows at the catchment outlet increase with catchment impervious area (IA), rising from 44 ML/day (pre-development) to 75 ML/day (70% IA), accompanied by increasing variation in flows from 87.6 ML/day (pre-development) to 139.4 ML/day (70% IA). The total number of days under high flow conditions and the mean value of the high flow spells increases with IA (Table 2). If compared with the pre-development flow threshold, which rises with imperviousness, the frequency of high flows also increases with impervious area. Low flow spells decrease in frequency and mean flow with increasing IA whilst the total number of days under low flow conditions increases. Figure 1 shows a fairly uniform increase in the magnitude of higher flows in correspondence with increasing impervious areas. A similar manner, though smaller magnitude, of increase occurs for lower flows, excepting for 70% imperviousness. In this case, flow is markedly more concentrated in shorter duration 'flashy' peaks, where the high flow spells (Table 2) are significantly increased in magnitude, frequency and duration in comparison to 50% IA, and rate of rise and fall is also greatest.

Stormwater Harvesting

Urbanisation from 40% to 70% imperviousness with runoff capture (ie, harvesting) shows a decrease in magnitude for lower flows with respect to the pre-development (0% impervious) conditions, whereas 5-30% impervious scenarios manage to replicate low and medium flows reasonably well (Figure 2). This is illustrated by the significant reduction in baseflow (Table 2) for these higher imperviousness scenarios with runoff capture.

For medium to high flows (15% exceedance), depicted in Figure 2, runoff capture brings the flows closer to pre-development when compared to scenarios without stormwater harvesting (Figure 1). However, the flows are not as closely matched as for the lower flows. The difference in medium flows is most marked. Indeed, the 15mm runoff harvesting for 50% and 70% imperviousness lowers the medium flows below pre-development levels, but the 10mm rule is not sufficient for 5-40% imperviousness at this point. The rule appears most effective for lower imperviousness (5-20%). For 40% imperviousness, the higher flows are higher than pre-development but the low flows are lower than pre-development with significant effect on low spells (Table 2).

Application of FFMOs appears to successfully reduce increases in high flow frequency and total duration and to mitigate against reductions in low flow frequency and increases in low flow duration caused by increases in %IA.

Conclusions

The simulation scenario results demonstrated that, as would be expected, increasing the impervious area resulted in an increase in the frequency, magnitude and duration of high flows. Low flow spells decreased in frequency but increased in total duration (total number of days under low flow conditions), indicating an increase in flashiness. These results agree in part with those of Fletcher *et al.* (2007), who simulated an increase in high flow frequencies and total duration, but contradict with regards low flows, which they predicted would increase in frequency and stay static with regards to total duration. The FFMOs, implemented through harvesting rules, returned the flow characteristics closer to those at pre-development. However, these rules were not able to closely match the medium to high flows, and the 15mm capture rule (50-70% impervious areas) resulted in a decrease in flows with respect to pre-urban conditions.

Further work will include testing these scenarios with two other undeveloped catchments with calibrated SWMM: Pimpama River at Kingsholme, and Upper Yaun Creek at Coomera, to assess the impact of the flow rules on other gauged catchments. Following this, additional or alternative harvesting measures that may better achieve a return to pre-development conditions will be explored for all three catchments. Additionally, MUSIC software will be assessed as an alternative modelling tool to represent these catchments and stormwater harvesting scenarios, given the limitations in SWMM.

References

DIP (2009), *South East Queensland Regional Plan 2009-2031, Implementation Guideline No. 7, Water sensitive urban design: Design objectives for urban stormwater management*, Queensland Government, Brisbane

Fletcher, T., Mitchell, V.G., Deletic, A., Ladson, T.R. and Seven, A. (2007), Is stormwater harvesting beneficial to urban waterway environmental flows?, *Water Science and Technology* 55(4): 265-272.

Fletcher, T., Deletic, A., Mitchell, V.G. and Hatt, B.E. (2008), Reuse of urban runoff in Australia: a review of recent advances and remaining challenges, *Journal of Environmental Quality* 37:116-127.

Huber, W. C. (2003) Hydrologic Modelling Processes of the EPA Storm Water Management Model (SWMM). World Water Congress 2003, American Society of Civil Engineers (ASCE), 1-10.

Nash, J. E. and Sutcliffe, J.V. (1970) River forecasting using conceptual models, 1. A discussion of principles, *Journal of Hydrology*, 10(1970): 282-290.

Poff, N.L., Richter, B.D., Arthington, A.H., Bunn, S.E., Naiman, R.J., Kendy, E., Acreman, M., Apse, C., Bledsoe, B.P., Freeman, M.C., Henriksen, J., Jacobson, R.B., Kennen, J.G., Merritt, D.M., O'Keeffe, J.H., Olden, J.D., Rogers, K., Tharme, R.E. and Warner, A. (2009), The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards, *Freshwater Biology* 1-24

SEQ Regional Plan (2009). *South east Queensland regional plan 2009-2031*, The State of Queensland (Department of Infrastructure and Planning), Australia

Walsh, C., Roy, A.H., Feminella, J.W., Cottingham, P.D., Groffman, P.M., Morgan II, R.P. (2005), The urban stream syndrome: current knowledge and the search for a cure, *Journal of the North American Benthological Society* 24(3):706-723

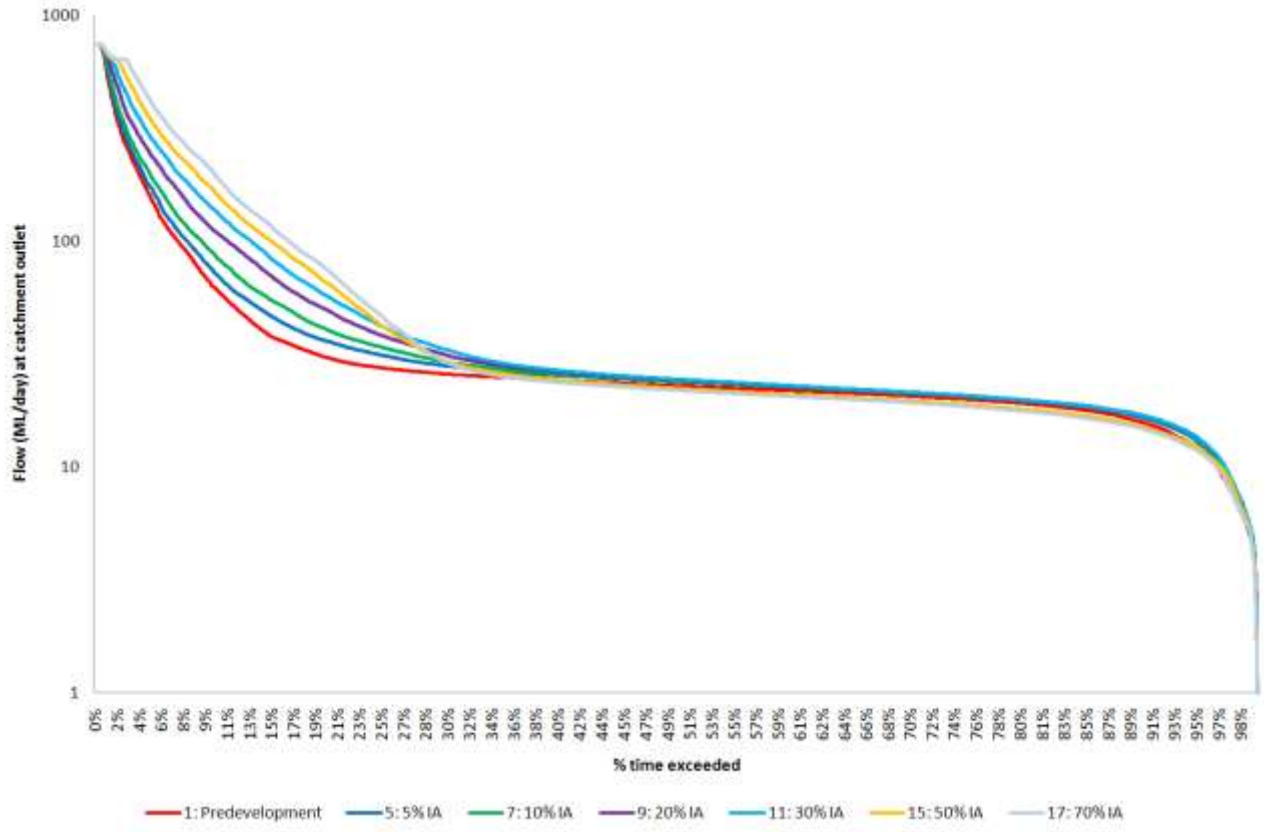


Figure 1. Runoff response at catchment outlet to selected scenarios (Table 1) of increasing catchment percentage impervious area (IA).

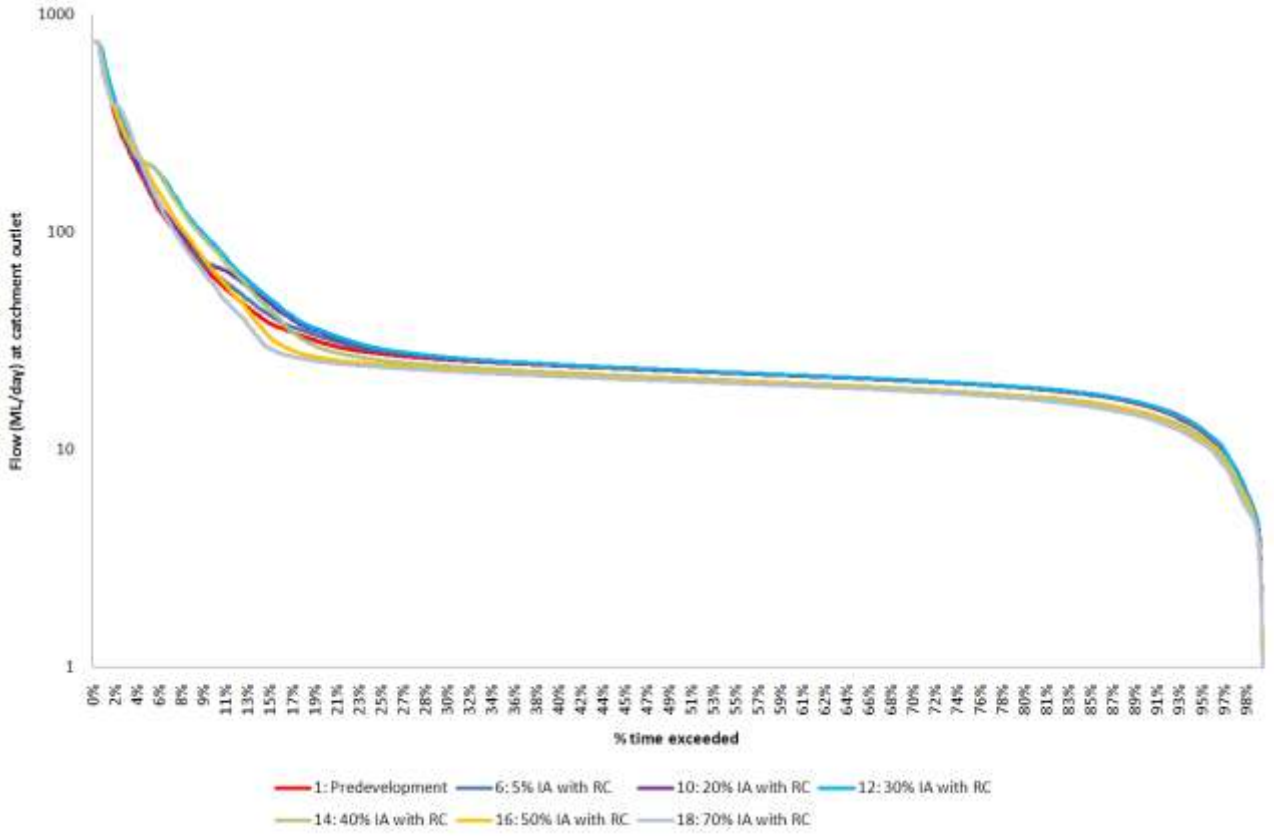


Figure 2. Runoff response at catchment outlet to selected scenarios (Table 1) of increasing catchment percentage impervious area (IA) with runoff capture (RC).

Table 2. Timeseries results for flow at catchment outlet for all scenarios of impervious area (IA) and runoff capture (RC).

	1: 0% IA	2: Existing	3: 1% IA	4: 1% IA with RC	5: 5% IA	6: 5% IA with RC	7: 10% IA	8: 10% IA with RC	9: 20% IA	10: 20% IA with RC	11: 30% IA	12: 30% IA with RC	13: 40% IA	14: 40% IA with RC	15: 50% IA	16: 50% IA with RC	17: 70% IA	18: 70% IA with RC
General Statistics (ML/day)																		
10th Percentile Flow	15.8	16.2	16.3	16.0	16.6	16.0	16.9	16.0	17.0	16.0	16.9	16.1	15.7	14.7	15.1	14.4	14.8	13.9
90th Percentile Flow	64.7	64.4	66.4	64.4	76.0	67.5	90.1	69.8	114.4	71.0	141.5	92.1	156.3	87.4	172.7	70.3	207.7	60.0
Mean	44.0	43.6	44.7	44.2	47.3	45.0	50.8	45.6	57.3	45.9	63.5	49.5	65.2	46.2	68.4	43.3	75.0	42.4
Median	22.9	23.0	23.2	23.0	23.8	23.1	24.0	23.1	24.4	23.1	24.5	23.2	22.5	21.3	22.3	20.9	21.8	20.5
Standard Deviation	87.6	86.3	88.2	88.0	91.0	88.9	95.6	89.7	105.3	89.5	115.5	95.3	121.1	91.1	127.1	88.3	139.4	89.7
High Flow Spells																		
High Spell Threshold (= 5*mean value)	220	218	223	221	237	225	254	228	287	230	318	248	326	231	342	217	375	212
Number of High Spells	286	287	285	284	281	284	277	285	268	286	271	265	286	253	316	254	367	258
Mean of High Spell Peaks (ML/day)	466	459	470	470	485	474	503	479	534	480	561	508	569	488	579	468	607	486
Total Duration of High Spells (days)	485	484	482	484	489	488	504	492	565	490	617	513	652	507	702	600	774	590
Low Flow Spells																		
Low Spell Threshold (= 0.5*mean value) (ML/day)	22.0	21.8	22.3	22.1	23.7	22.5	25.4	22.8	28.7	23.0	31.8	24.8	32.6	23.1	34.2	21.7	37.5	21.2
Number of Low Spells	2878	2840	2848	2890	2740	2895	2279	2895	1605	2863	1432	2437	1220	2390	1150	2850	1145	2931
Mean of Low Spell troughs (ML/day)	17.1	17.2	17.4	17.3	17.7	17.4	17.5	17.5	16.7	17.5	16.7	17.3	14.4	15.9	14.2	16.2	13.7	16.0
Total Duration of Low Spells (days)	6056	5628	6087	6086	7017	6538	8379	6875	9509	6989	9867	8843	10266	9211	10408	8166	10489	8128
Rise and Fall																		
Mean rate of Rise (ML/day/day)	29.4	25.5	25.2	26.3	23.2	24.1	23.1	23.9	25.3	23.8	30.4	23.1	37.4	21.7	41.4	20.4	52.1	21.9
Mean rate of Fall (ML/day/day)	21.6	21.2	21.9	21.6	22.8	21.8	23.9	22.0	25.7	22.1	27.3	22.0	27.4	19.5	29.2	18.1	34.1	17.6
Base Flow																		
Baseflow Index	0.47	0.48	0.47	0.47	0.46	0.46	0.44	0.46	0.40	0.46	0.37	0.43	0.34	0.42	0.32	0.43	0.28	0.43
Mean Daily Baseflow (ML/day)	20.6	20.7	21.0	20.7	21.7	20.8	22.2	20.9	22.9	21.0	23.4	21.2	21.9	19.3	21.6	18.6	21.3	18.1

Evaluation of Reactive Toxicity towards Proteins for Application in Water Quality Assessment

Tang, J.Y.M., Glenn, E., Thoen, H. and Escher, B.I.
National Research Centre for Environmental Toxicology (Entox), The University of Queensland, Coopers Plains, Queensland

Summary

Reactive organic chemicals comprise a large number of compounds with a variety of reactive moieties. The aim of this project was to develop an *in vitro* test system for the evaluation of hazards and mode of action classification of reactive chemicals for environmental risk assessment. A bioanalytical assay based on genetically modified *Escherichia coli* strains was set up to quantify the specific reactivity toward the important biological nucleophile glutathione (GSH). Seven reference compounds were investigated in the initial phase and environmental extracts from an advanced water treatment plant (AWTP) in South East Queensland (SEQ) were used to evaluate the applicability of the *E. coli* assay for reactive toxicity.

Keywords
Bioanalytical tools, bioassays, reactive toxicity, glutathione.

Introduction

Bioanalytical tools have been widely applied as a routine toxicity testing in compliment to traditional chemical analysis for water quality assessment over the last few years (Muller *et al.*, 2007; Escher *et al.*, 2009; Macova *et al.*, 2010; Reungoat *et al.*, 2010; Chapman, 2011; Escher *et al.*, 2011). Relevant endpoints for dominant chemical groups such as herbicides, insecticides and estrogenic chemicals are well established. However, there remain a number of critical endpoints that are not covered in the current test batteries, and one of these is reactive toxicity. Reactive toxicity results from covalent binding of electrophilic organic pollutants to proteins or endogenous biochemical substrates, and has the potential to initiate severe adverse biological effects (Bohme et al., 2010). Electrophilic compounds can react with nucleophilic groups on proteins and DNA. Various electrophilic xenobiotics react at different nucleophilic sites within the cell and these patterns are governed by the hard/soft acid base principle (Chan *et al.*, 2008). Hard electrophiles have small atomic radii, a high electronegativity and low polarisability, while soft electrophiles are larger and highly polarisable in comparison (Carlson, 1990). Hard electrophiles react preferentially with hard nucleophiles such as amino groups of lysine and DNA; soft electrophiles react preferentially with soft nucleophiles such as the sulfyhydryl group of cysteine or glutathione (Schultz *et al.*, 2006).

While there exist a large number of reactive toxicity assays that target DNA damage, there are no assays that detect reactive protein damage despite it playing an important role in eg, skin sensitisation, nonspecific enzyme inhibition, respiratory allergy, organ toxicity and necrosis (Aptula *et al.*, 2006; Chan *et al.*, 2008). So far, only direct electrophilic reactivity has been use to evaluate the toxic potential of a substance (Aptula *et al.*, 2006). Glutathione (GSH) is one of the most widely used nucleophilic reference molecules in reactivity assays (Schwobel *et al.*, 2011). It is the most prevalent cellular thiol and the most abundant low molecular weight peptide in cells (Aptula *et al.*, 2006). GSH protects cells by detoxifying electrophilic compounds and acts as an antioxidant; it is a soft nucleophile and can be used to sense the reactivity of electrophilic agents toward the thiol group and other electron-rich sites of molecular structures (Bohme *et al.*, 2010). Reactive organic chemicals comprise a large number of compounds with a variety of reactive moieties.

The aim of this project was to adapt an *in vitro* test system that has been developed for the evaluation of hazards and mode of action classification of reactive chemicals for environmental risk assessment to the application with water samples and extracts. Harder *et al.* (2003b) developed a bioanalytical assay based on the growth inhibition of a genetically modified *Escherichia coli*. The parent strain MJF276 (GSH+) is capable of synthesising GSH, while the mutant strain MJF335 (GSH-) lacks both γ-glutamylcysteine synthase and GSH synthase. The comparison of growth inhibition by reactive chemicals in the strain deficient of GSH and its fully functional parent strain indicates the relevance of GSH conjugation as a detoxification step and related toxic effects as well as direct reactivity with cysteine-containing proteins. A quantitative measure of this effect is the ratio of the effect concentrations EC_{50} for growth inhibition of the native strain (GSH+) to the one that lacks GSH (GSH-).

So far, no studies have tested environmental mixtures and water samples using this bioanalytical endpoint, and previous studies (Harder *et al.*, 2003a; Richter and Escher, 2005) were performed in large volume glass tubes instead of a high throughput system. Hence, the aim of this project was to explore this *E. coli* assay as a novel endpoint for identification of reactive chemicals in the environmental matrix. This project was implemented in a stepwise manner: initially by using reference chemicals, then water extracts of reverse osmosis concentrate (ROC) with different electrochemical oxidation from an advanced water treatment plant (AWTP) were investigated to validate its

applicability as a monitoring tool. An analogous strain pair exists also for DNA damage, but as there exists plenty of *in-vitro* bioassays for genotoxicity and the *E. coli* strain pair comparison was less sensitive in a pilot study, we focus in the presented work only on the glutathione *E. coli* strains.

Materials and Methods

Previous studies were conducted on a large volume scale and the first aim of this project was to optimise the bioassay into a 96 well format for future environmental toxicity testing. Different growth conditions including cell density, incubation time and shaking speed were tested on both parent and mutant strains. Experiments were performed at least in duplicate. Growth was monitored by light scattering at 600nm (OD_{600}) and growth related to control was calculated according to:

$$\%\ \text{Inhibition of Growth (Biomass)} = \left(1 - \frac{\Delta OD_{600,sample}}{\Delta OD_{600,control}}\right) = 1 - \frac{OD_{600,t}(sample) - OD_{600,t=0}(sample)}{OD_{600,t}(control) - OD_{600,t=0}(control)}$$

The concentrations resulting in 50% inhibition of growth (EC50) were derived from a logistic fit of the concentration-effect curves using the software Prism (GraphPad Software, San Diego, CA), which computed the best fit for experimental data of all parallels under the prerequisites of fixed minimum at 0% and fixed maximum at 100% growth. Both reference compounds and samples generally followed a sigmoidal log-concentration-effect curve (Figure 1). Sample extracts are normally composed of a mixture of unknown substances at unknown concentrations, so the concentration unit of the sample in the concentration-effect curves is based on the Relative Enrichment Factor (REF) expressed in the units [$L_{water\ sample}$ / $L_{bioassay}$]. The influence of GSH on growth inhibition of the reference compounds was characterised by the toxic ratio (TR_{GSH}) of EC50 values of MJF276 (GSH+) and MJF335 (GSH-):

$$\text{Toxic Ratio} = TR_{GSH} = \frac{EC_{50}\ GSH+}{EC_{50}\ GSH-}$$

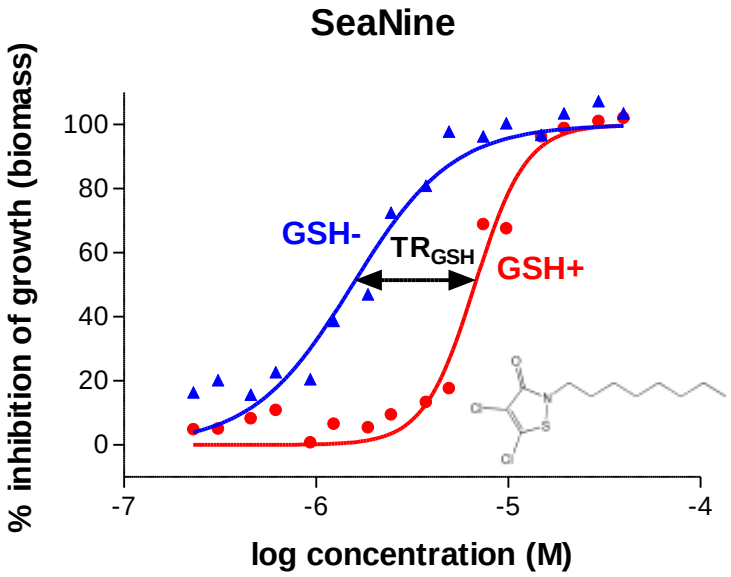


Figure 1. Concentration-effect curve of reference compound Sea-Nine on parent strain MJF276 (GSH+) and mutant strain MJF335 (GSH-). Toxic Ratio (TR_{GSH}) is derived from the ratio of EC50 values between GSH+ and GSH- strains.

Sea-Nine (CAS Registry No. 64359-81-5: 4,5-dichloro-2-n-octyl-4-isothiazolin-3-one) was chosen as a positive reference compound. It is a soft electrophile and reacts with proteins and GSH and thus may also produce oxidative stress. Six additional compounds were selected to evaluate the specificity of the response: 2-aminoanthracene (2AA, CAS Registry No. 613-13-8); 4-nitroquinoline-1-oxide (4NQO, CAS Registry No. 56-57-5); benzo[a]pyrene (BaP, CAS Registry No. 50-32-8); phenol (CAS Registry No. 108-95-2); t-butylhydroquinone (tBHQ, CAS Registry No. 1948-33-0); and methyl methane sulfonate (MMS, CAS Registry No. 66-27-3).

Results and Discussion

Development of the *E. coli* Reactive Toxicity Bioassay

The EC_{50} values for the reference compound Sea-Nine of the parent and mutant strains, GSH+ and GSH-, were evaluated under different growth media to optimise the growth conditions of the cells. Tryptone and yeast were chosen to evaluate the growth condition in addition to minimal medium. Minimal medium consists of KH_2PO_4, K_2HPO_4, $(NH_4)_2SO_4$ and Na_3-Citrate-H_2O. Addition of tryptone and yeast enhanced the growth of cell biomass but decreased the toxic ratio between the parent and mutant strains (Figure 2), while the EC_{50} of the GSH+ strain remained constant. The reason for this is likely direction conjugation of Sea-Nine with proteins in the medium. Hence, only minimal medium will be used for future experiments.

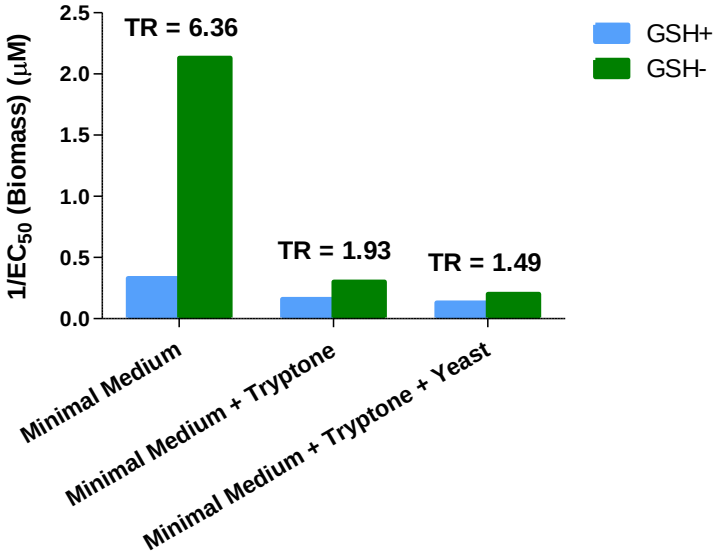


Figure 2. Comparison of the $1/EC_{50}$ values (high value = high toxicity) of the pair GSH+ and GSH- under different growth conditions after 3 hours exposure.

Toxicity Testing of Reference Compounds

If the toxicity of a chemical is related to the reaction with GSH, a significant growth inhibition between GSH producing and not producing *E. coli* strains would be expected (Harder *et al.*, 2003a). To be valid as a selective test, it must also be shown that non-reactive chemicals and those that preferentially attack DNA do not show a difference in toxicity between the two strains. Concentration-effect curves of the investigated compounds are presented in Figures 1 and 3. Benzo[a]pyrene (BaP) did not show any toxicity in both GSH-deficient (GSH-) and GSH-sufficient strain (GSH+), presumably due to its low bioavailability in minimal growth medium (Table 3). Sea-Nine, a known soft electrophile which reacts with proteins and GSH, produced the highest toxic ratio among the test compounds and was therefore chosen as the reference compound for quality control purposes (Figure 1, Table 3). All the other investigated compounds, with an exception of 2-aminoanthracene (2AA), produced similar concentration-effect curves between the GSH+ and GSH- strains and the toxic ratio was around 1 (Table 3), confirming that hard electrophiles and chemicals with other modes of action are not active in this endpoint. It is important for validation that not only the positive controls but also the negative controls act as expected. The toxicities for 2AA were low and could not produce a full dose-response curve but there were no apparent differences between the two strains (Figure 1, Table 3).

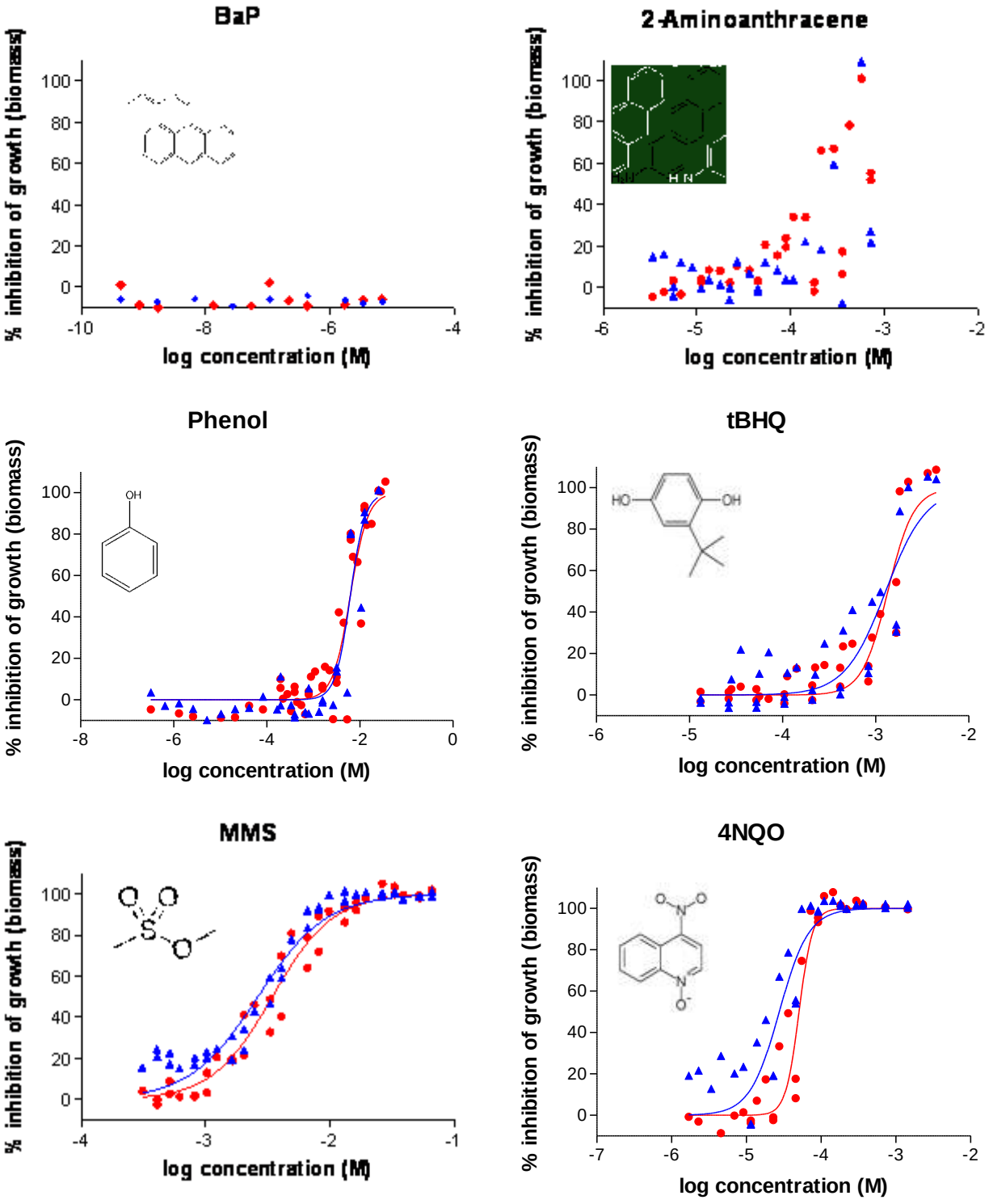


Figure 3. Concentration-response curves of growth inhibition of GSH+ (circles) and GSH- (triangles) of the investigated compounds.

Table 1. EC50 (mM) and Toxicity Ratios (TR) values of the *E. coli* strains GSH+ and GSH- derived from the above concentration-response curves of growth inhibition.

Investigated Compounds	EC50 (mM) GSH+	EC50 (mM) GSH-	TR_{GSH}
BaP	<0.01	<0.01	n.d.
2AA	<1	<1	n.d.
Phenol	6.2060 (5.2410-7.3490)	6.2180 (5.14-7.5230)	1.00
tBHQ	1.3530 (1.1690-1.5660)	1.2810 (1.0090-1.6260)	1.06
MMS	3.4330 (3.1090-3.7920)	2.6290 (2.3310-2.9650)	1.31
4NQO	0.0508 (0.0454-0.0567)	0.0276 (0.0226-0.0338)	1.84
Sea-Nine	0.0042 (0.0037-0.0048)	0.0014 (0.0013-0.0016)	2.93

Reactive Toxicity Testing on Environmental Samples

Reverse osmosis concentrate (ROC) water extracts from an advanced water treatment plant (AWTP) in SEQ were tested on the *E. coli* bioassay to evaluate the applicability of reactive toxicity testing. Electrochemical oxidation was suggested as a means for eliminating pesticides, pharmaceuticals and other organic micropollutants from complex water streams but toxicity experiments using nonspecific endpoints (Microtox assay) have indicated that non-specific toxicity actually increased with time of treatment (Radjenovic *et al.*, 2011). We retested these samples wto validate the bioassay for real (and complex) water matrices. ROC treated with electrochemical oxidation for 0 to 24 hours was extracted with solid phase extraction and subjected to the new bioassay. Initial experiments showed that there was a stimulation of growth by organic matter and other matrix compounds from the water extracts, therefore any data exceeding the growth rate of the control were omitted for concentration-effect analysis (Figure 4).

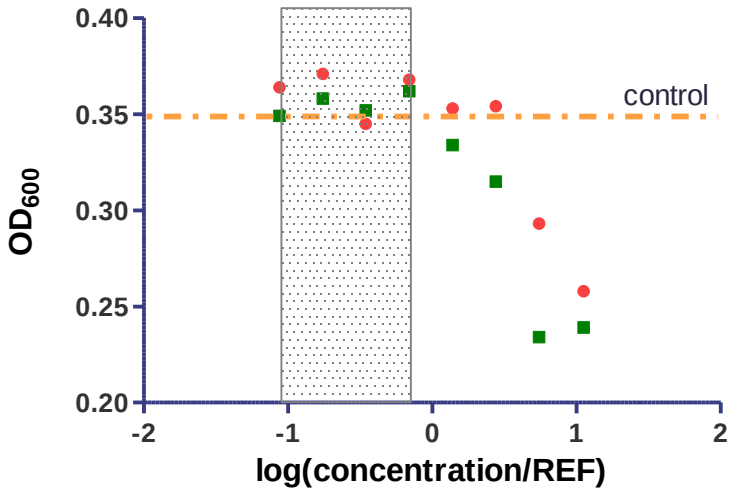


Figure 4. OD_{600} measurements of GSH+ strain of ROC water extract. Grey area denotes data above the control values that had to be omitted for the analysis to avoid any artefacts. ROC water extracts treated for 10 hours (red circles) and 24 hours (green squares).

Extracts of ROC treated for 0 to 8 hours did not show any growth inhibition in the glutathione sufficient strain GSH+ and extracts from ROC treated for 0 to 6 hours have also shown no growth inhibition in the glutathione deficient strain GSH- (Table 4). The extracts of ROC treated for 10 and 24 hours did show appreciable toxicity with EC_{50} values in the range of the unenriched to 10 times enriched sample. The toxic ratios (TR_{GSH}) at 10 hours and 24 hours were similar to the positive reference compound Sea-Nine (Tables 1 and 2). The observed reactive toxicity may be due to the formation of (chlorinated) oxidation byproducts. Radjenvoic *et al.* (2011) found that baseline toxicity increased significantly after 24 hours of treatment, which coincides with our current observation that reactive toxicity increases as treatment times increased.

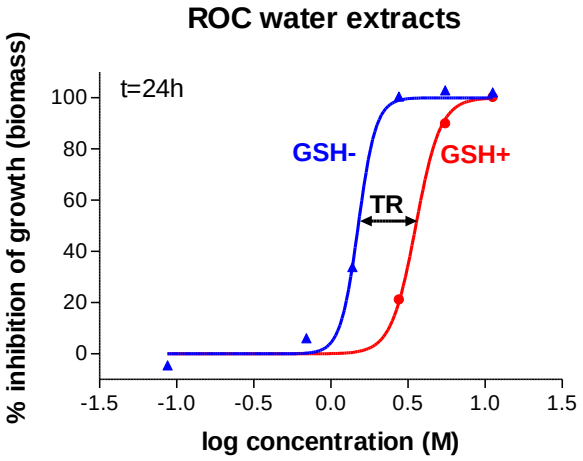


Figure 5. Concentration-response curve of ROC water extract at t=24h.

Table 2. EC50 (REF) and Toxicity Ratios (TR_{GSH}) values of the *E. coli* strains MJF276 and MJF335 tested on ROC water extracts.

Treatment Time	EC_{50} (REF) GSH+	EC_{50} (REF) GSH-	TR_{GSH}
0 h	BDL* (< REF 11.1)	BDL* (< REF 11.1)	BDL* (< REF 11.1)
2 h	BDL* (< REF 11.1)	BDL* (< REF 11.1)	BDL* (< REF 11.1)
4 h	BDL* (< REF 11.1)	BDL* (< REF 11.1)	BDL* (< REF 11.1)
6 h	BDL* (< REF 11.1)	BDL* (< REF 11.1)	BDL* (< REF 11.1)
8 h	BDL* (< REF 11.1)	4.28 (2.32-7.88)	BDL* (< REF 11.1)
10 h	7.29 (2.58-20.6)	2.71 (2.33-3.16)	2.69
24 h	3.56 (2.09-6.05)	1.51 (1.09-2.07)	2.36

* Below detection limit (BDL): highest REF tested was 11.11.

Conclusions

Richter and Escher (2005) evaluated the mixture toxicity of reactive chemicals using the *E. coli* assay and found that the toxicity of mixtures of electrophiles with a single preferred target was consistent with the prediction for concentration addition. So far, there have been no studies using environmental extracts to evaluate the suitability of the *E. coli* assay. In this study, we adapted the protocol to high throughput testing on 96 well plates and to testing of water extracts. Extracts from reverse osmosis concentrate collected from an advanced water treatment plant that was treated by electrochemical oxidation showed reactive toxicity toward glutathione, while the untreated ROC had no effect in this assay. While the EC_{50} values correlated qualitatively with nonspecific toxicity determined with the Microtox assay, the TR_{GSH} clearly indicated that reactive toxicity was responsible for some of the observed effect. While the assay performed well with reference compounds and was highly reproducible, the stimulation of growth by some of the water extracts limits the applicability to those dilutions of samples where no stimulation of growth is observable, thus reducing the window of detection substantially. This might be a drawback for universal application of the assay. Further testing on other environmental samples is required to evaluate the feasibility of the *E. coli assay* for routine water assessment.

References

Aptula, A.O., Patlewicz, G., Roberts, D.W. and Schultz, T.W. (2006). Non-enzymatic glutathione reactivity and in vitro toxicity: A non-animal approach to skin sensitization. *Toxicology in Vitro*, **20**(2): 239-247.

Bohme, A., Thaens, D., Schramm, F., Paschke, A. and Schuurmann, G. (2010). Thiol Reactivity and Its Impact on the Ciliate Toxicity of alpha,beta-Unsaturated Aldehydes, Ketones, and Esters. *Chemical Research in Toxicology*, **23**(12): 1905-1912.

Carlson, R.M. (1990). Assessment of the propensity for covalent binding of electrophiles to biological substrates. *Environ Health Perspect*, **87**.

Chan, K., Poon, R. and O'Brien, P.J. (2008). Application of structure–activity relationships to investigate the molecular mechanisms of hepatocyte toxicity and electrophilic reactivity of α,β-unsaturated aldehydes. *Journal of Applied Toxicology*, **28**(8): 1027-1039.

Chapman, H.F., Leusch, F. D. L., Prochazka, E., Cumming J., Ross, V., Humpage, A., Froscio, S., Laingam, S., Khan, S. J., Trinh, T., McDonald, J.A. (2011). A national approach to health risk assessment, risk communication and management of chemical hazards from recycled water, Waterlines report. Canberra, National water Commission.

Escher, B.I., Bramaz, N. and Ort, C. (2009). JEM Spotlight: monitoring the treatment efficiency of a full scale ozonation on a sewage treatment plant with a mode-of-action based test battery. *Journal of Environmental Monitoring*, **11**(10): 1836-1846.

Escher, B.I., Lawrence, M., Macova, M., Mueller, J.F., Poussade, Y., Robillot, C., Roux, A. and Gernjak, W. (2011). Evaluation of Contaminant Removal of Reverse Osmosis and Advanced Oxidation in Full-Scale Operation by Combining Passive Sampling with Chemical Analysis and Bioanalytical Tools. *Environmental Science and Technology*: null-null.

Harder, A., Escher, B.I., Landini, P., Tobler, N.B. and Schwarzenbach, R.P. (2003a). Evaluation of bioanalytical assays for toxicity assessment and mode of toxic action classification of reactive chemicals. *Environmental Science and Technology*, **37**(21): 4962-4970.

Harder, A., Escher, B.I. and Schwarzenbach, R.P. (2003b). Applicability and Limitation of QSARs for the Toxicity of Electrophilic Chemicals. *Environmental Science and Technology*, **37**(21): 4955-4961.

Macova, M., Escher, B.I., Reungoat, J., Carswell, S., Chue, K.L., Keller, J. and Mueller, J.F. (2010). Monitoring the biological activity of micropollutants during advanced wastewater treatment with ozonation and activated carbon filtration. *Water Research*, **44**(2): 477-492.

Muller, R., Tang, J.Y.M., Thier, R. and Mueller, J.F. (2007). Combining passive sampling and toxicity testing for evaluation of mixtures of polar organic chemicals in sewage treatment plant effluent. *Journal of Environmental Monitoring*, **9**(1): 104-109.

Radjenovic, J., Escher, B.I. and Rabaey, K. (2011). Electrochemical degradation of the beta-blocker metoprolol by Ti/Ru0.7Ir0.3O2 and Ti/SnO2-Sb electrodes. *Water Research*, **45**(10): 3205-3214.

Reungoat, J., Macova, M., Escher, B.I., Carswell, S., Mueller, J.F. and Keller, J. (2010). Removal of micropollutants and reduction of biological activity in a full scale reclamation plant using ozonation and activated carbon filtration. *Water Research*, **44**(2): 625-637.

Richter, M. and Escher, B.I. (2005). Mixture toxicity of reactive chemicals by using two bacterial growth assays as indicators of protein and DNA damage. *Environmental Science and Technology*, **39**(22): 8753-8761.

Schultz, T.W., Carlson, R.E., Cronin, M.T.D., Hermens, J.L.M., Johnson, R., O'Brien, P.J., Roberts, D.W., Siraki, A., Wallace, K.B. and Veith, G.D. (2006). A conceptual framework for predicting the toxicity of reactive chemicals: modeling soft electrophilicity. *Sar and Qsar in Environmental Research*, **17**(4): 413-428.

Schwobel, J.A.H., Koleva, Y.K., Enoch, S.J., Bajot, F., Hewitt, M., Madden, J.C., Roberts, D.W., Schutz, T.W. and Cronin, M.T.D. (2011). Measurement and Estimation of Electrophilic Reactivity for Predictive Toxicology. *Chemical Reviews*, **111**(4): 2562-2596.

Evaluating a Science Communication Workshop as an Educational Tool for the Application of Bioassays

Ross, V. and Chapman, H.
Smart Water Research Centre, Griffith University, Gold Coast, Queensland

Summary

To evaluate the effectiveness of a science communication workshop as an educational tool for the application of bioanalytical tools in water quality monitoring, 11 people representing science, policy, regulation and industry were interviewed both before and after attending the workshop. All interviewees reported that attending the workshop was beneficial in improving their knowledge of bioanalytical tools. Many participants felt more favourably towards the value of bioassays after the workshop and others felt it had helped to improve their understanding and clarify their views about the usefulness of bioassays. A number of constructive recommendations were made, including clarifying some of the more confusing aspects of the presentations, and further developing the workshop to target more specific topics and audiences. An additional aim of the research was to identify any barriers to effective communication within the water sector. Results showed that the main barriers to communication were inconsistent use of recycled water terminology; content-free approaches to policy; staffing changes; and policy, regulation and industry not being "on the same page" when implementing recycled water projects.

Keywords
Risk communication, bioassays, science-policy gap, public confidence, recycled water.

Introduction

One of the major challenges to the implementation of recycled water projects in Australia has been the lack of both public acceptance and consistent policies and regulations for the management of the treated water. As public perceptions of risk have been shown to be influenced by perceptions of the credibility of the responsible authority (Baggett, Jeffrey and Jefferson, 2006; Frewer *et al.*, 1993; Ross, 2009; Tyler and Degoey, 1996), it is vital that policy makers, regulators and water managers provide assurance to the community that they are making decisions informed by the best available science.

Research shows that public confidence can be seriously diminished if authorities are perceived as incompetent or biased (eg, Baggett *et al.*, 2006). It is therefore vital that scientists, policy makers, regulators and water managers are shown to have a good communication process and that policy decisions are made based on reliable science. Given the history of public rejection of indirect potable reuse schemes in Australia, this will be crucial to acceptance of future schemes.

A key project strategy, embedded as Goal 2, the risk communication component of the "Bioanalytical tools and Risk Communication" project is designed to improve communication about the application and interpretation of bioanalytical tools for industry, policy and regulation in Queensland's water quality monitoring programs. The current study aimed to evaluate the effectiveness of a science communication workshop, designed as an educational tool for the application of bioassays in water quality monitoring. The results of this research will provide crucial information to assist in facilitating confidence in the use of bioassays for water professionals and for enhancing the development of future workshops. An additional aim of the research was to identify key issues that need to be addressed to enhance risk communication, with a particular focus on recycled water.

Method

Eleven representatives from science, policy and regulation, and industry were interviewed before and after the bioassays communication workshop using semi-structured, individual, face-to-face interviews. The first set of interviews was conducted prior to the workshop to gain an understanding of participants' current knowledge of, and views about the applicability and limitations of bioassays. The broader aim of the pre-workshop interviews was to identify potential communication barriers between scientists, regulators and policy makers, and industry representatives.

The second set of interviews was conducted with the same participants after the bioassays workshop to provide qualitative feedback and thus assess its effectiveness. Using evaluation criteria, the workshop was assessed based on participants' responses across a number of key points, which included communication of key messages, if and how the workshop had improved knowledge levels about the use of bioassays, whether participants felt better equipped to locate scientific information and experts on bioassays, and whether their views had altered regarding the applicability and limitations of bioassays. The data was analysed using the qualitative method of content analysis. Information

regarding the purpose and scope of the research, including assurance of confidentiality and anonymity, were provided to participants in advance. Ethical clearance was provided by the Griffith University Human Research Ethics Committee.

Results

Pre-Workshop Interviews

Current Level of Knowledge

Participants' self-reported level of knowledge of water quality monitoring and assessment techniques indicated that four of the participants believed they had a high level of knowledge, five people felt their knowledge level was fair, and two felt their knowledge was poor. Similarly, when asked to describe their knowledge of bioassays, the same participants reported their knowledge levels in exactly the same way, that is, four with a high level of knowledge, five with fair knowledge, and two with poor knowledge.

Workshop Expectations

Participants generally expressed a desire to know more about the value of bioassays, how they are used and what they can achieve. A number of interviewees were also keen to discover if bioassays could have coal seam gas water and environmental applications. Several respondents were also interested to know if bioassays could be used to provide an extra level of assurance of safety for the public for the recycled water and thus aid in increasing public confidence in water recycling. Examples of key questions that interviewees hoped to have answered through the workshop were: "*will these tests mean that you can actually assess those standards quicker or faster or easier or cheaper?", "how easy is it to apply to a normal laboratory?", "Do you need experts or …can you have people with laboratory certificates being able to run these assays?", "can a genuine link from bioassays to public health be demonstrated?", "If we were to establish the fact that the processes work… where do we go from here?"*

Perceived Communication Barriers

Recycled water terminology was most commonly cited as a barrier to effective communication. Respondents reported that terminology differed between guidelines, legislation, Acts, and community education documents. In terms of communicating to the public, it was noted that inconsistent use of recycled water terminology results in sending mixed messages, which in turn creates confusion and nervousness. Some participants felt that even within the technical arena there can be substantial differences in the understanding of terminology. As one interviewee suggested *"so I think if you were to get three engineers in the room you could potentially get two very different ideas".*

It was also noted that differences in both terminology and levels of knowledge created communication difficulties between people working on policy and regulations. A trend towards "content free management" where policy writers are not required to have background knowledge on the relevant policy area was identified as a significant barrier to communication between regulators and policy writers. A respondent described a situation where water regulators experienced communication difficulties when working with a group of policy makers and drafters who did not have any technical knowledge on the topic. The lack of technical understanding on the part of the policy team created tension between the regulation and policy groups in a situation where a good working relationship between these groups would have been optimal.

The importance of policy, regulation and industry being "on the same page" in terms of risk communication was also acknowledged. One interviewee explained that a great deal of time and effort goes into writing and requesting information, retesting and sometimes even redesigning recycled water projects. He stressed the importance of a shared understanding during this process, to avoid projects "falling over", resulting in wasted time and investment.

The majority of participants reported that they had few difficulties with locating scientific information and expertise that they required. The importance of professional networks was identified as integral to this process. However, as one interviewee pointed out, staffing changes (which are not always infrequent in government departments) can sever these connections very quickly. A number of people mentioned that they sometimes have difficulties obtaining very specific scientific information. Given that it is a relatively new technology, this was particularly true in the area of purified recycled water (PRW). Several respondents mentioned that they found research conducted by the Urban Water Security Research Alliance to be useful in this area. The issue of commercially confident data was also cited as barrier to communication. Respondents described the frustration of being unable to obtain water quality information. One interviewee stated *"how do you communicate that something is safe if you can't access the data that demonstrates that it is safe?"*

Views on Bioassays Prior to the Workshop

Several participants believed that the link from a bioassay to public health was tenuous, in other words, it would be difficult to infer human health risk, particularly from the cell-based assays. A number of interviewees believed that bioanalytical techniques would be useful as a broad screening tool that was best used to complement, and not replace, traditional methods. It was also suggested that bioassays might have a better application in PRW than in drinking water, and would be useful in providing evidence that recycled water is clean and safe to use.

Concerns were expressed by a number of interviewees about the cost effectiveness and the difficulties of using such a "high tech" method. It was noted that the amount of verification monitoring currently required is very expensive, and there were concerns that introducing bioassays as an additional diagnostic tool could make it prohibitive to get a recycled project "off the ground". In addition, several interviewees cautioned that conducting more and more detailed tests is not necessarily the best way to improve public confidence. Another interviewee also thought that there would be very few laboratories that are actually capable of using bioassays.

Other Issues

Most participants felt strongly about the importance of science communication, particularly in cases where there has been significant investment. As one interviewee stated "*we've invested in this science so the important thing then is to make sure that that investment is used in the maximum way by getting it out there*". Concerns were also expressed that a lot of water related policy and political decisions were being made that were not based on science, and that research results were not having an impact on policy and processes.

It was generally considered by most respondents that effective risk communication about water treatment and safety is a vital element in securing public support. A number of interviewees also considered it important to be able to package up the science in ways that can be used by different audiences.

Post-Workshop Interviews

General Comments

All participants reported that attending the workshop was beneficial to them in terms of improving their knowledge of bioanalytical tools. The workshop presenters were described as having a high level of technical knowledge and experience, and it was noted that they built a good rapport and interaction with the audience. The presentations were described as informative, organised, well-structured, aimed at the appropriate level for the range of people in the audience, as well as providing clear messages and the opportunity for feedback.
Most participants reported that it was useful to obtain a broader knowledge of bioanalytical tools and their application and limitations. Other specific information that respondents reported as being valuable were the range of techniques available, the application of human cells, "myth busting" on the multiplier effect, usefulness with unknown unknowns, and the fact that bioassays could provide information on the mode of toxicity. Several respondents felt that bioanalytical techniques could be useful in promoting public confidence in recycling. It was also noted that resources could be saved by conducting broad screening bioassays, rather than running a battery of independent tests.

A number of interviewees were interested in applying bioassays in water quality assessment of coal seam gas water. Several respondents also noted that PRW was not currently a "hot" issue and that coal seam gas water is a more immediate concern. Most respondents felt that as a result of attending the workshop they were more confident about to whom or where they could go to obtain further information on bioanalytical tools.

Did Views on Bioassays Change after the Workshop?

Five of the participants reported that since attending the workshop their views had altered more favourably towards the value of bioassays. Examples given of the types of information that influenced their views were the depth and variety of tests available, repeatable results, and the development of a toolbox setting and procedures that can be followed by a great number of different people. As one interviewee said "*I was a bit negative beforehand but now I'm sort of coming around a little bit. I can actually see some advantages, particularly the fact that the drinking water didn't light up like a Christmas tree in the bioassays like I thought it would. So I was quietly impressed and surprised"*.

Several interviewees said that they were already supportive of the use of bioassays prior to the workshop. Other respondents felt that though their views had not necessarily altered, the workshop had helped to improve their understanding and clarify their views about the usefulness of bioassays. Another interviewee reported having a much improved understanding about the application of bioassays, but felt there was still a lot of uncertainty in their application.

Risk Perceptions

When asked if their views about the health risks associated with recycled water had altered since attending the workshop, almost all participants stated that their views had not changed because they were already confident with the treatment processes and safety of recycled water. However, one respondent said he found some of the information confusing and that he now perceived the health risks to be higher than he had thought prior to the workshop. Although one interviewee was "*still not 100 per cent convinced on the human applications of bioassays",* a number of respondents felt that the workshop made them more comfortable with how the risks could be assessed. Others felt that the workshop had showed that bioassays could be used to effectively demonstrate and communicate that the health risks in relation to recycled water are low.

Constructive Comments

The most significant feedback from workshop participants was that the component showing comparisons of different water treatment technologies and water quality needed to be explained in context. Several respondents cautioned that some of the graphics on water quality comparisons could be alarming to some people, and that it was important to provide a clear story behind the water quality results from different techniques. One interviewee felt that this was an area of the presentation that was not well packaged and felt there could be a clearer message to give people confidence in water treatment technologies.

A number of participants mentioned that they found the discussion on dose response to be confusing. It was recommended that this component of the presentation could be clarified and that improved graphics could be useful in achieving this. Several people described the workshop handouts as a useful tool to take away, particularly in terms of being able to show the information to other colleagues. However, some were disappointed that some of the detailed information from the power point presentation had been left out and would like to see this included.

Others were keen to know more about how these results could be effectively communicated to the public to provide a more objective picture on the risks. It was noted that more work needs to be done in terms of providing advice for people in the profession on how to communicate risk to the general public.

Most of the participants felt that it would be useful to further develop the workshop for specific audiences to make the information more accessible to a wider variety of people. One recommendation was a series of workshops where the different applications of bioassays could be explored. Other suggestions were additional topic specific type seminars where technical information could be expanded out for the scientific people, and creating two separate workshops for environmental water and drinking water. Several interviewees mentioned that it would have been beneficial to have had more industry representatives in the audience.

Key Messages

During the post workshop interviews, participants were asked if they could describe what they perceived as the key messages that they took home from the workshop. These are summarised below.

- *Bioassays are an important tool to expand our understanding of water quality risks. They can be used to do things that traditional analyses have problems doing, for example, the cocktail effect, unknown unknowns and the mode of toxicity.*
- *Bioassays complement the other methodologies and can help to reinforce the results that are obtained through chemical analysis and biological analysis etc.*
- *Applying bioassays will give us a more comprehensive way of targeting our sampling and a more comprehensive view of what might be there rather than just looking for a single compound.*
- *Bioassays have other applications than just recycled water. They have strengths and weaknesses against traditional techniques.*
- *Bioassays are an underutilised and unexplored resource for determining risk, toxicology or the effects and harm of hazards in the environment.*
- *There are a wide variety of bioanalytical tests available and they cover a range of parameters that can be estimated at different levels. The results are repeatable and reliable.*
- *Bioassays may have a substantial contribution to make, there is a lot of work already happening in the area and still a lot more work to be done.*
- *Bioassays are not the silver bullet that will guarantee public acceptance, but they are powerful tools that could be developed to enhance public confidence.*

As part of this project, the messages taken away by the workshop participants will be compared with key messages still being developed by the research team, and will be used to build a more clear list of communication points for this and future workshops.

Conclusions

Evaluation of the effectiveness of the bioassays workshop will provide vital information for enhancing the development of this and future workshops as science communication tools. The research addresses the project goal of improving communication about the application and interpretation of bioanalytical tools for industry, policy and regulation in Queensland's water quality monitoring programs. The research also identified key issues that need to be addressed to enhance risk communication, which will provide a practical starting point for addressing communication difficulties between scientists, policy makers and regulators and water industry professionals. It is envisioned that this research will ultimately facilitate the implementation of policies and regulations that are based on the best available science, leading to best practice in the industry. An additional set of post-workshop interviews are planned with the same participants in 12 months time to assess the long-term effectiveness of the workshop as an educational tool.

References

Baggett, S., Jeffrey, P. and Jefferson, B. (2006). Risk perception in participatory planning for water reuse. *Desalination*, 187, 149-158.

Frewer, L. J., Howard, C., Hedderley, D. and Shepherd, R. (1995). What determines trust in information about food-related risks? Underlying psychological constructs. *Risk Analysis*, 16 (4), 473-486.

Ross, V., L. (2009). The role of trust in community acceptance of urban water management schemes: A social-psychological model of trust. PhD thesis, University of Queensland.

Tyler, T. R. and Degoey, P. (1996). Trust in organisational authorities. In R. M. Kramer and T. R. Tyler (Eds.), *Trust in Organisations. Frontiers of Theory and Research*. London: Sage.

Pathogen Decay in a Reservoir Impacted by the January Floods

Toze, S., Hodgers, L. and Palmer, A.
CSIRO Water for a Healthy Country, Ecosciences Precinct, Dutton Park, Queensland

Summary

A project has been established to gain a better understanding of the sources and behaviour of microbial pathogens in South East Queensland (SEQ) reservoirs. The results reported here describe the results of a study of the influence of the January 2011 floods on the decay of selected microbial pathogens in Wivenhoe Dam. The decay of several enteric bacteria, adenovirus and *Cryptosporidium* oocysts was determined using diffusion chambers set at the surface and 15 metres depth. The resulting decay rates were compared to decay rates obtained prior to the floods. This comparison showed that there was no noticeable difference for the decay of *E. coli* and *Salmonella* but the rate of *Campylobacter* decay differed at 15 metres depth, adenovirus decayed slower at the surface and the decay of *Cryptosporidium* was slower at both the surface and 15 metres. The results show that the floods influenced the decay of some pathogens, however, the decay rate is still fast enough to demonstrate that factors within Wivenhoe Dam can still be relied upon to actively remove microbial pathogens from the water.

Keywords
Pathogens, decay, reservoirs, flood, water quality.

Introduction

Reservoirs tend to be man-made, large bodies of water used to store water for purposes such as drinking water. Reservoirs often are fed by large catchments which can be sources of a wide variety of pathogens. These pathogens can range from zoonotic pathogens originating from wild and domesticated animals to human-specific pathogens released from townships and septic systems along waterways feeding into the reservoirs.

Traditionally, reservoirs have been used merely as passive storage sources and downstream water treatment plants have been relied upon as the sole source of water treatment. This implies that water treatment plants must therefore be designed to deal with all contaminants under all conditions. Ferguson *et al.* (2003) noted that the production of high quality drinking water should involve the use of high-quality protected catchments. They indicated that using unprotected catchments increased treatment costs and subsequent health risks. Many cities and towns, however, do not have the ability to completely protect their drinking water catchments from sources of microbial pathogens. As the reservoirs are relied upon for passive storage only, this places all reliance on producing high quality water on the treatment plants. The lack of any treatment credits being given to processes within reservoirs can lead to the treatment of water being an over designed process with treatment plants being constructed to deal with contaminants that may have actually already been removed by natural processes within the reservoir. There is a growing acceptance, however, that healthy natural systems such as reservoirs may have the capacity to be used as an active treatment barrier. In addition, a better understanding of the treatment capacity within reservoirs could enable an improved management of potential contaminants, particularly under event-based conditions.

One issue impeding the acceptance of natural treatment systems such as reservoirs as active treatment barriers is that site specificity issues remain a hurdle to be overcome. Not all drinking water catchments can be highly protected, and even catchments and watersheds that have sources that are not impacted by human activity and livestock can be a source of pathogens from local wildlife (Patchanee *et al.*, 2010; Till *et al.*, 2008). This means that pathogens remain a persistent risk for all reservoirs and water treatment systems. At present, all sites would need to be extensively tested before even being considered to have a potential role as a treatment barrier. There is little information in the literature that can be considered able to be transferred between sites, even sites within similar geographical locations. The need for the validation of each individual site poses challenges as there are few documented tests of treatment capacity of reservoirs, let alone attempts to validate the treatment capacity of these systems. In addition, the time the water is present in the reservoir, the residence time, is a critical parameter influencing the extent of pathogen removal. Understanding the movement and residence time of water in a reservoir is a complex hydrodynamic issue with site specific issues relating to reservoir shape and topography details, water usage from the reservoir, along with influences from local climatic and seasonal conditions. This information then needs to be combined with an understanding of the behaviour and fate of different pathogens to obtain an estimate of removal efficiencies of each pathogen tested.

This project was established to gain a better understanding of the sources and behaviour of microbial pathogens in SEQ. The research has focused on Wivenhoe Dam as it is the major water source providing Brisbane with drinking water. The research has involved a series of experiments studying the factors that could have a major influence on the fate of pathogens in Wivenhoe Dam. The January 2011 floods provided an opportunity to determine if such a major event, which has significantly changed conditions in Wivenhoe Dam, can have a corresponding influence on

pathogen survival in the reservoir. This paper provides details on the outcomes of the first experiment undertaken post-flood and compares these results with decay rates determined in earlier pre-flood experiments.

Methods

Decay experiments were undertaken *in-situ* in Wivenhoe Dam from April to June 2011 using decay chambers as described in Toze *et al*. (2010). The experiment was undertaken in Wivenhoe Dam at the research location at the mouth of Logans Inlet, which has been used for all of the pathogen decay experiments undertaken as part of the Urban Water Security Research Alliance.

The post-flood experiment tested the survival of the bacteria *E. coli, Salmonella enterica* and *Campylobacter jejuni*; the enteric virus adenovirus; and *Cryptosporidium* oocysts. The bacteria were suspended in reservoir water to approximately 10^6 cfu mL^{-1}. Adenovirus and *Cryptosporidium* oocysts were suspended in a second volume of reservoir water to approximately 10^4 oocysts or virus PCR detectable units (pdu) per mL. The bacterial and virus/*Cryptosporidium* oocysts suspensions were then used to fill a series of diffusion chambers. The filled chambers were then placed in the reservoir at the research site at the mouth of Logans Inlet.

Chambers were set at one metre below the surface and at 15 metres depth. Three replicate chambers were collected at Time 0 and on days 4, 7, 14, 21, 35, 49 and 63. The collected chambers were immediately placed in individual ziplock plastic bags and stored on ice bricks in a cold box for transport to the laboratory. Once received in the laboratory, the water within each of the chambers was retrieved as described in Toze *at al.* (2010) and placed into sterile 15 mL tubes.

The numbers of detectable target microorganisms were determine using the methods outlined in Toze *at al.* (2010). In brief, the bacteria were isolated on selective media, adenovirus detected using quantitative PCR, and viable *Cryptosporidium* oocysts identified using vital stains and fluorescent microscopy. The bacteria were reported as colony forming units (cfu) mL^{-1}, the adenovirus as PCR detectable units (pdu) mL^{-1}, and *Cryptosporidium* as viable oocysts mL^{-1}. The resulting numbers of each studied microorganism were collated, log transformed, and plotted over time. A linear regression line was then fitted to each resulting plot and the slope of the regression line used as the decay rate (kd) (log day^{-1}). The inverse of each decay rate was then calculated to provide the time for a 1 log reduction (T90) (days). The results from this post-flood experiment were then compared to the outcomes from earlier experiments which were undertaken prior to the January 2011 floods.

Results and Discussion

The results of the experiments undertaken in pre- and post-flood conditions are presented in Table 1.

Table 1. T90 decay times (days) for enteric bacteria in Wivenhoe Dam under pre- and post-flood conditions.

Tested Microbes	T90 Times (days)			
	Pre-Flood		Post-Flood	
	Surface	Depth	Surface	Depth
E. coli	3	3	3	3
Salmonella	5	5	3	7
Campylobacter	4	14	3	3
Adenovirus	11	ND	27	12
Cryptosporidium	13	ND	46	46

The results obtained show that the decay of *E. coli* and *Salmonella* in Wivenhoe Dam was the same at both the surface and at 15m depth under both pre- and post-flood conditions. This suggests that the changed conditions in Wivenhoe Dam due to the floods either had no impact on their decay, or that any impacts were so minor that they did not have any significant influence on an already fast decay rate. The 1 log decay times for these bacteria in all the experiments undertaken to date have been 7 days or less (data not shown). In contrast, the decay of *Campylobacter* in the pre-flood experiment had been observed to be significantly slower at 15m depth than at the surface (14 vs 4 days). After the extreme weather and resulting floods, the decay of *Campylobacter* at the 15m depth was found to be the same as at the reservoir surface, both times being almost the same as determined for *Campylobacter* at the surface in the pre-flood experiment. This suggests that conditions or individual factors at the 15 m depth in the reservoir have changed since the January floods to have a similar influence to the conditions at the surface. These changes in conditions have not changed any factors that have a major influence on the decay of *E. coli* and *Salmonella* but can influence the survival of *Campylobacter*. *Campylobacter* is known to be sensitive to oxygen concentrations with too much oxygen inhibiting growth, however, there may be other factors that have also changed

with a corresponding impact on *Campylobacter* survival. Analysis remains to be undertaken on the changes in a range of measured water quality characteristics from pre- to post-flood conditions to determine which of these factors could be influencing this change in decay rate.

The results obtained in the post-flood experiment also suggested that there was some potential increase in cell numbers of *E. coli* and *Salmonella* between Time 0 and day 4 at 15 m below the surface (Figure 1). While the increase observed only occurred within the first 7 days and was less than a 1 log increase, such an increase had not been previously observed in any of the experiments for either of these bacteria. No increases in *Campylobacter* cell numbers were observed (data not shown). More analysis is required, however, to determine the reason for this brief increase in cell numbers. Regardless of the reasons for this increase, it was only a minor change in cell numbers and a very transient event and thus, there would be no discernable changes in the health risk from either *E. coli* or *Salmonella*.

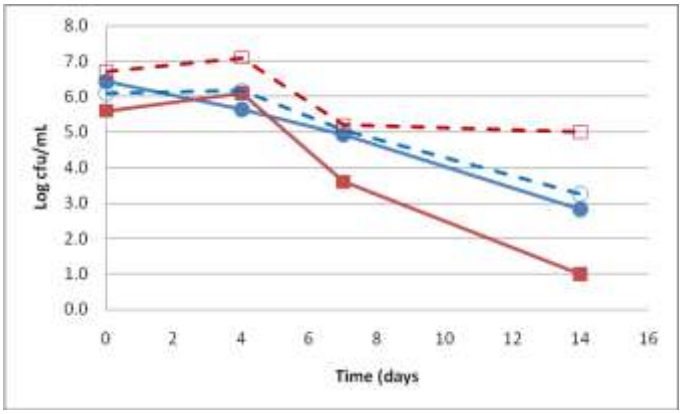


Figure 1. Decay of *E. coli* and *Salmonella* in Wivenhoe Dam at 15 m depth after the January floods where ● is *E. coli* pre-flood; ■ is *E. coli* post-flood; ○ is *Salmonella* pre-flood; and □ is *Salmonella* post-flood.

In contrast to the decay of the bacteria, both adenovirus and *Cryptosporidium* oocysts were observed to persist longer in the reservoir after the January floods than prior to the floods. Adenovirus had a decay time at 15m depth post-flood that was virtually the same as at the surface pre-flood, but the decay at the surface, post-flood was more than double the time in the pre-flood conditions (27 vs 11 days). The decay of the *Cryptosporidium* oocysts was also much slower at the surface post-flood then in the pre-flood conditions. Unlike adenovirus, however, *Cryptosporidium* had a time for a 1 log loss that was the same at the 15m depth as at the surface (45 days). The decay times for adenovirus and *Cryptosporidium* at the 15m depth are not available in the pre-flood conditions so it is unclear whether the changed conditions in the reservoir have influenced the decay of these pathogens at 15m compared to the surface. The reasons for these variations in decay times for adenovirus and *Cryptosporidium* are not clear and will require a detailed analysis of changes in water chemistry and conditions to determine what factors may be able to be linked to these different decay times.

Conclusions

The results obtained for this research have shown that the changes in the conditions in the reservoir had no observable impact on the decay rates of *E. coli* or *Salmonella* and actually increased the decay rate of *Campylobacter* at depth. In contrast, the decay of adenovirus and *Cryptosporidium* oocysts were slower at the reservoir surface post-flood than in the pre-flood conditions. Despite this, adenovirus and Cryptosporidium still had 1 log removal times in the post-flood reservoir that can be considered reasonably short in relation to the residence times of the water in Wivenhoe Dam. Thus, even with the impact from the January 2011 floods, Wivenhoe Dam has the capacity to act as an active treatment barrier for the removal of any pathogens that may enter the dam from upstream contamination sources. It is intended to repeat this experiment, particularly for *Campylobacter*, adenovirus and *Cryptosporidium* to confirm the results and to further determine the factors influencing the differences between the measured pre- and post-flood decay rates.

References

Ferguson, C., Husman, A.M.D., Altavilla, N., Deere, D. and Ashbolt, N. (2003) Fate and transport of surface pathogens in watersheds. Crit. Rev. Environ. Sci. Tech. 33, 299-361.

Patchanee, P., Molla, B., White, N., Line. D.E. and Gebreyes. W.A. (2010) Tracking Salmonella Contamination in Various Watersheds and Phenotypic and Genotypic Diversity. Foodborne Path. Dis. 7, 1113-1120.

Till, D., McBride, G., Ball, A., Taylor, K. and Pyle, E. (2008) Large-scale freshwater microbiological study: rationale, results and risks. J. Wat. Health, 6, 443-460.

Toze S, Hodgers L, Sidhu J, Morrow D and Udy J (2009) Determining the persistence of enteric pathogens in SE QLD reservoirs. In Proc. of OzWater'10, Brisbane, September 2010.

Prevalence of Human Pathogens and Indicators in Stormwater Runoff in Brisbane

Sidhu, J.P.S.[1], Hodgers, L.[1], Ahmed, W.[1] and Toze, S.[1,2]
[1] CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland
[2] School of Population Health, The University of Queensland, Herston, Queensland

Summary

Elevated enteric pathogen numbers in receiving waters following a storm event can be considered to be a serious public health concern as there is evidence of link between higher levels of pathogens and disease outbreaks. Twenty four surface water samples were collected from six sites across Brisbane and analysed for the presence of a selected group of indicators and pathogens and faecal indicator bacteria (FIB) *E. coli* and *Enterococcus* spp. The levels of FIB in the stormwater samples were found to have exceeded recreational water quality guidelines, frequently by several orders of magnitude. *E. coli* numbers were at least ten-fold lower after the storm event compared to *Enterococcus* spp. numbers, which ranged from $7x10^3$ to $10x10^3$ $100mL^{-1}$. Human adenovirus and polyomavirus was detected in the collected water samples after the storm event. However, both viruses were also found in the samples collected during the dry period. Presence of human-specific HF183 *Bacteroides* marker found during both the wet and dry period suggests the ubiquitous nature of sewage contamination at all sites.

Keywords
Stormwater, pathogen detection, adenovirus, virus concentration.

Introduction

Urban stormwater is considered to be a major potential source for alternate water that can be used within cities as a substitution for potable water sources. Captured stormwater has the potential to be used for both non-potable and portable purposes, although the current major interests are for non-potable uses. The potential presence of faecal contamination in stormwater runoff (Parker *et al.*, 2009), however, can negatively impact public health. Concerns over this issue are heightened by the reported links between the occurrence of storm events, microbial inputs from nonpoint sources and increase in the incidence of waterborne disease (Curriero *et al.*, 2001).

Presence of chemical and biological pollutants in the stormwater might pose a health risk for humans. In particular, faecal pollution presents a significant hazard since it might contain enteric pathogens that infect humans. Human enteric pathogens can find their way into aquatic environments through leaking sewer systems, sewer pumping station overflows, septic systems, agricultural runoff and discharge of treated wastewater into aquatic environments (Gaffield *et al.*, 2003; Noble *et al.*, 2006; Rajal *et al.*, 2007). High numbers of enteric virus, bacteria and protozoa have been reported in storm water runoff indicating the presence of sewage pollution (Noble *et al.*, 2006; Rajal *et al.*, 2007; Sercu *et al.*, 2009; Cizek *et al.*, 2008). Hence, monitoring for faecal pollution in stormwater is required to make an assessment of the extent of treatment required prior to its use as recycled water. Although detection of faecal pollution indicators *E. coli* and enterococci in water is relatively straightforward, presence of indicator bacteria does not necessarily correlate with the presence of viral and protozoan pathogens (Selvakumar and Borst 2006). The adequacy of monitoring for faecal pollution indicator bacteria has also been questioned in recent years as viral and protozoan pathogens are known to be more persistent in the environment than bacteria (Tree *et al.*, 2003; Robertson and Nicholson 2005).

Recent advances in molecular techniques have made it possible to detect enteric pathogens in surface water (Rajal *et al.*, 2007; Muscillo *et al.*, 2008, Dyke *et al.*, 2009; Mull and Hill 2009), hence, direct surveillance for pathogens such as enteric viruses may be a better approach for protecting public health. Several studies have suggested that adenovirus and polyomavirus are present in higher numbers in wastewater and are found to be prevalent in rivers and coastal waters, and thus, may be suitable indicators for viral pollution in stormwater (Muscillo *et al.*, 2008; Bofil-Mas *et al.*, 2006; Jiang, 2006).

Since viral and protozoan pathogens are usually present in low numbers in stormwater, an efficient concentration method coupled with sensitive detection is required for successful detection and quantification of pathogens. However, due to the high suspended solids content and complex physical and chemical properties of surface waters, pathogen concentration techniques may yield variable results (Loge *et al.*, 2002). In addition, existing methodologies for extraction and purification of nucleic acid from these sample types are limited to very small starting volumes (Fuhrman *et al.*, 2005) which may also lead to false negative results due to PCR inhibition. The hollow fibre ultrafiltration system (HFUFS) combined with PCR has been previously used for the detection of enteric virus in drinking, estuarine and stormwater with some success (Hill *et al.*, 2005; Hernandez-Morga *et al.*, 2009; Rajal *et al.*, 2007; Mull and Hill 2009). In this study, we have used this approach for the detection and quantification of pathogens of concern in surface water and stormwater.

The aim of this study was: (i) to develop an understanding of how pathogen and indicator numbers change between the dry and wet periods; and (ii) to assess the utility of HFUFS for the concentration of higher volume samples that would permit real-time PCR for the detection of enteric pathogens in stormwater samples.

Results and Discussion

This study was designed to compare changes in the FIB and selected enteric pathogens in urban creek water during dry and wet weather conditions, in order to understand how rainfall runoff affects water quality. Approximately 20 L water samples were collected from six sites around Brisbane (Brisbane River near Auchenflower drain, Oxley Creek, Cabbage Tree Creek, Fitzgibbon stormwater harvesting site, Enoggera Creek and Samford) to determine microbial contaminant numbers in both base-flow and stormwater runoff (Figure1 and Table 1). Four grab samples were collected from each site; two samples in the dry period (no significant rain in 72 hours prior to sampling) and two samples after a significant rainy period.

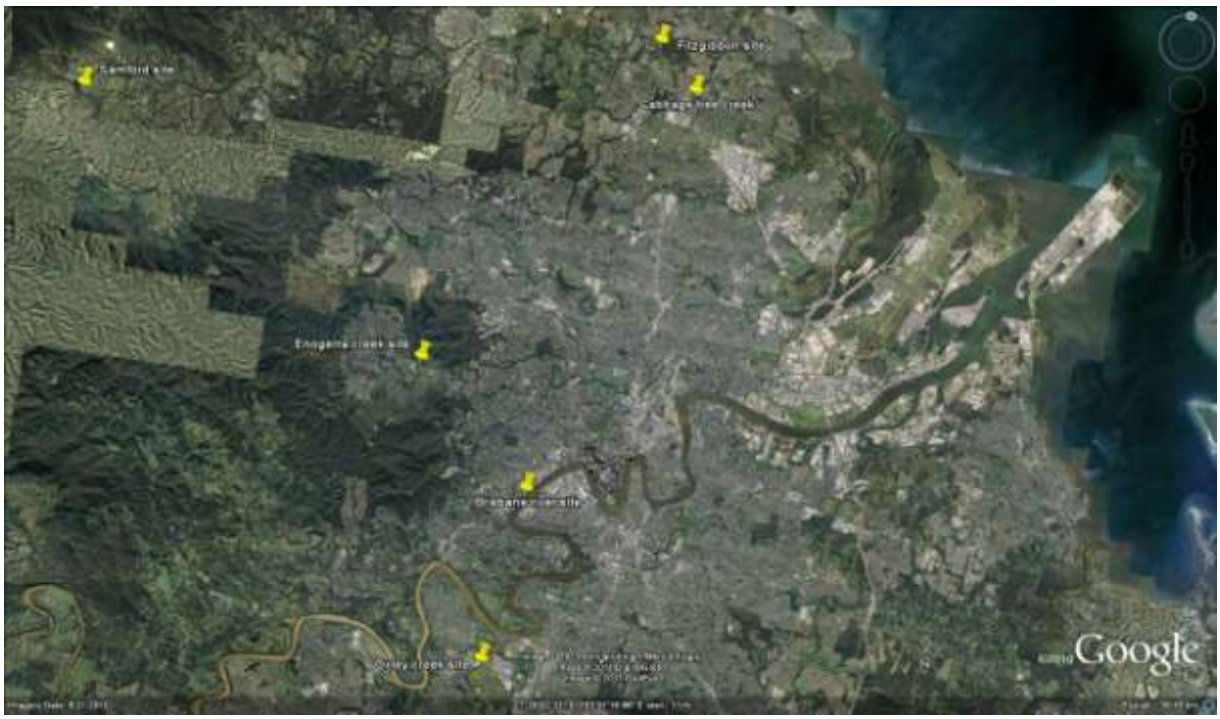


Figure 1. Location of sampling sites across north Brisbane.

Table 1. Sampling sites description and GPS coordinates.

Site Name	Description	GPS Coordinates
Cabbage Tree Creek	Flows thorough populated suburbs in north Brisbane	27º20´59.7"S ; 153º02´06.6"E
Fitzgibbon Drain	Low density population, large blocks, some animal input	27º20´08.7"S ; 153º.01´14.5"E
Oxley Creek	Industrial area	27º32´07.8"S ; 152º59´31.4"E
Enoggera Creek	Moderately populated area	27º26´41.96"S S; 152º57´16.90"E
Samford Site	Rural area with large block size, potential animal input	27º21´31"S ; 152º52´51"E
Brisbane River	Stormwater drain outlet form city area	27º28´49.1"S S; 152º.59´54.1"E

Changes in FIB Numbers during the Wet and Dry Periods

Collected samples were analysed for FIB numbers and results were plotted after $\log_{10}$ transformation of raw data (Figure 2). Mean of $\log_{10}$-transformed numbers from dry and wet weather from all sites were compared. Both *E. coli* and *Enterococcus* spp. numbers were significantly higher during the wet period than the dry period. The mean *E. coli* numbers during the wet period (3.54 colony forming units (cfu) $100mL^{-1}$) was higher than the dry period (2.32 cfu $100mL^{-1}$). The corresponding means for *Enterococcus* spp. were 3.58 cfu $100mL^{-1}$ and 2.43 cfu $100mL^{-1}$ respectively.

No significant difference ($p<0.05$) was observed between *E. coli* and *Enterococcus* spp. numbers either during the dry period or the wet period. The *E. coli* numbers varied between $8x10^1$ to $1x10^3$ 100 mL^{-1} between the dry and wet periods whereas, the corresponding numbers for *Enterococcus* spp. varied between $4x10^1$ to $3x10^4$ $100mL^{-1}$. *E. coli* and *Enterococcus* spp. numbers from all six sites have a better correlation ($R^2 = 0.61$) during the dry period compared to the wet period ($R^2 = 0.42$).

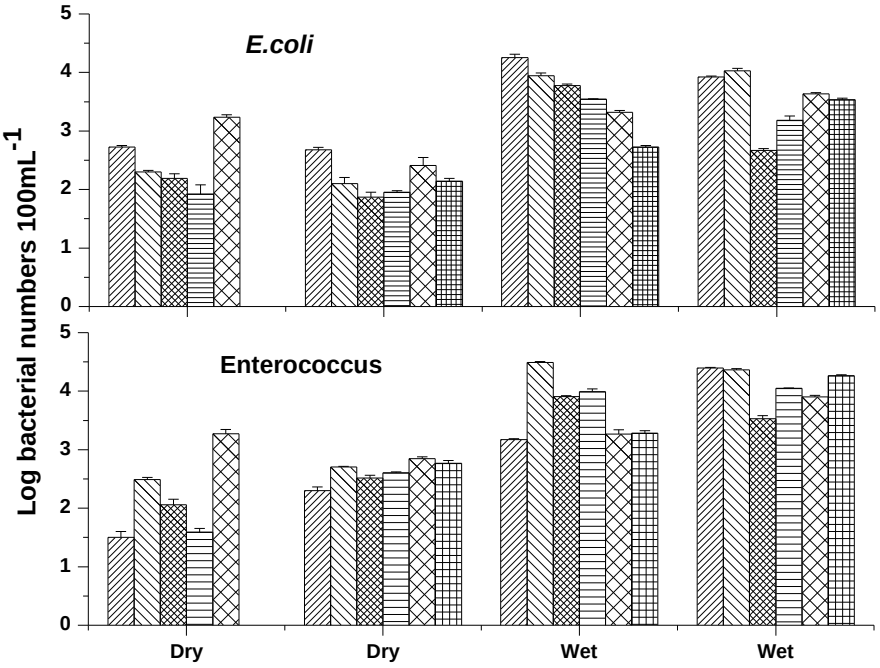


Figure 2. Changes in the *E. coli* and *Enterococcus* from dry to wet period at six sites across Brisbane.
In the figure Cabbage Tree Creek, Fitzgibbon Site, Brisbane River, Oxley Creek, Enoggera Creek, Samford Site.

Pathogen Detection in Collected Water Samples

PCR-based methods were used for the detection of *Campylobacter* spp., *C. jejuni*, *C. coli*, and adenovirus with published primer sets (Lund *et al.*, 2004; Heim *et al.*, 2003). The presence of sewage contamination at the sampling site was investigated by using PCR-based assays for the detection of *esp* gene of *Enterococcus faecium* and human-specific HF183 *Bacteroides* marker (Scott *et al.*, 2005; Seurinck *et al.*, 2005). Nucleic acid was extracted from the samples using a Power soli DNA isolation kit (MoBio) as per manufacturer instructions and stored at -80°C prior to analysis.

Stormwater samples can have a high amount of particulate material and other compounds that can interfere with DNA amplification by PCR. In this study, a variable degree of PCR inhibition was observed in nearly all samples, with samples collected from the Brisbane River showing the least inhibition and samples collected from the Enoggera Creek and Fitzgibbon site showing highest inhibition, with no amplification of seeded adenovirus at 100-fold dilution of extracted nucleic acid. Similar findings have also been reported previously by Rajal *et al.*, (2007) with stormwater samples collected in California. To overcome random PCR inhibitor effects, duplex PCR was carried out for the detection of enteric pathogens. The first amplification was carried out with 5 µL of DNA extracted from the sample and then 1 µL of amplified product from the first run was used for the detection of target genes. This approach gave a presence/absence test for the selected group of pathogens. The PCR results from the processed samples are presented in Table 2.

All samples tested positive for the presence of *Campylobacter* spp., the primer set used in this study primarily detects three species (*C. jejuni*, *C. coli* and *C. lari*) which are known to cause campylobacteriosis in humans. Further investigation of the samples with *C. jejuni* (17 samples) and *C. coli* (12 samples) specific primers revealed frequent presence of these pathogens in the stormwater samples. Human specific adenovirus and polyomavirus samples were detected in 15 and 8 samples respectively. Polyomavirus was primarily detected after the storm event; however, samples collected during the dry period were also positive for both polyomavirus and adenovirus. Human-specific HF183 *Bacteroides* marker was widely found in the water samples tested, suggesting human sewage input during the wet and dry period at the six sites. Whereas, *E. faecium esp* gene was mostly found after the storm events.

Table 2. Summary of results.

Site	Event	*E. coli*	*Enterococcus* .spp.	*Campylobacter* spp.	*C. jejuni*	*C. coli*	Adenovirus	Polyomavirus	esp	HF183	Rainfall (mm)
Enoggera Creek	dry	1.73E+03	1.93E+03	ND	ND	ND	ND	ND	-	ND	0
	dry	2.83E+02	7.10E+02	+	-	-	+	ND	-	+	0
	wet	2.10E+03	1.90E+03	+	-	-	+	ND	+	+	21.4
	wet	4.33E+03	8.00E+03	+	+	-	+	+	+	+	18.4
Cabbage Tree Creek	dry	4.83E+02	2.03E+02	+	+	+	ND	ND	-	+	0
	dry	5.33E+02	4.57E+02	+	+	+	+	+	-	+	2.6
	wet	8.37E+03	2.50E+04	+	+	+	+	+	-	+	19.4
	wet	1.82E+04	1.48E+03	+	+	-	ND	ND	+	ND	134.6
Fitzgibbon Site	dry	3.60E+02	5.10E+02	+	+	-	ND	ND	-	+	0.4
	dry	1.33E+02	1.27E+02	+	+	-	ND	ND	-	-	2.6
	wet	8.93E+03	2.23E+04	+	+	+	+	ND	-		19.4
	wet	1.07E+04	3.11E+04	+	+	+	+	+	+	+	134.6
Brisbane River	dry	1.60E+02	1.20E+02	+	-	-	+	+	-	-	0
	dry	7.67E+01	3.30E+02	+	+	-	+	+	-	-	0
	wet	6.00E+03	8.10E+03	+	+	-	+	+	+	+	4.6
	wet	4.67E+02	3.43E+03	+	-	+	+	-	+	+	10.2
Oxley Creek	dry	9.33E+01	4.00E+01	+	-	+	-	-	-	-	0
	dry	9.00E+01	4.03E+02	+	+	-	-	-	+	+	0
	wet	3.50E+03	9.93E+03	+	+	+	+	+	-	+	7
	wet	1.57E+03	1.12E+04	+	+	+	+	ND	-	+	15
Samford Site	dry	1.40E+02	5.93E+02	+	+	+	-	-	-	-	0
	wet	5.33E+02	1.93E+03	+	+	+	+	-	+	+	40
	wet	3.43E+03	1.84E+04	+	+	+	+	-	-	+	33.2

ND= not done,

Conclusions

The primary objective of this study was to conduct a preliminary evaluation of the presence of pathogens and indicator microorganism in surface water samples collected in Brisbane during dry and wet periods. Presence of faecal pollution in the stormwater runoff presents a significant hazard since it might contain enteric pathogens that infect humans.

All water samples collected during the dry and wet periods had high numbers of faecal indicator bacteria (FIB) enterococci and *E. coli*. *Enterococcus* spp. and *E. coli* numbers in the stormwater samples were found to have exceeded recreational water quality guidelines, frequently by several orders of magnitude regardless of the amount of rainfall occurring prior to sampling. The Australian Guidelines for Recreational Water Quality recommended limit for *Enterococcus* spp. for secondary contact activities, such as boating and fishing, is 230 cfu $100mL^{-1}$ (ANZECC and ARMCANZ, 2000). Significantly high numbers of *Enterococcus* spp. ($3x10^4$ $100mL^{-1}$) and *E. coli* ($1x10^4$ $100mL^{-1}$) were detected after a storm event at the Fitzgibbon site (Table 2). The increased number of FIB found after the storm event is potentially due to the stormwater runoff, as high rainfall has previously been linked to increased number of FIB in receiving water (Brownell *et al.*, 2007; Parker *et al.*, 2010).

PCR-based methods were carried out to give a presence/absence test for the selected group of enteric pathogens. *Campylobacter* spp. was detected from all water samples collected, showing their extensive prevalence in Brisbane surface waters. There is documented evidence of extensive existence of *Campylobacter* spp. in surface waters (Saville *et al.*, 2001; Dorner *et al.*, 2007; Dyke *et al.*, 2009). This is expected as, apart from humans, there are other sources of *Campylobacter* such as domestic animals and birds (Dyke *et al.*, 2010). We also found presence of *C. jejuni* and *C. coli* after subsequent PCR assays. Presence of *Campylobacter* in the surface water needs further investigation to assess the viability status PCR positive detects.

Adenovirus and polyomavirus was also detected in the surface water samples from all sites. Adenovirus was found to be more prevalent than polyomavirus which was mostly detected in water samples collected after the storm event. Human adenovirus and polyomavirus have been reported previously in surface water and stormwater (Hamza *et al.*, 2009; Muscillo *et al.*, 2007; Rajal *et al.*, 2007). Efforts are currently made to limit or remove the effect of PCR inhibitors which enables quantification of virus numbers. In the published literature, adenovirus and polyomavirus numbers in surface water have been reported in the range of $1x10^2$ to 10^5 L^{-1} (Hamza *et al.*, 2009; Muscillo *et al.*, 2007).

The presence of human-specific HF183 *Bacteroides* marker in the water samples collected during the dry and wet period suggest the ubiquitous nature of sewage contamination. This corroborates with the previous study where HF183 *Bacteroides* were found to be widely present in the stormwater (Sauer *et al.*, 2011). *Enterococcus faecium esp* gene was mostly found after the storm events which suggests low presence of *esp* gene in the environment.

This study shows that stormwater routinely contains high numbers of FIB and other enteric pathogens and hence some degree of treatment of captured stormwater would be required to use it for non-potable purpose.

References

ANZECC and ARMCANZ (2000). Australian and New Zealand Guidelines for Fresh and Marine Water Quality.

Bofill-Mas, S., Albinana-Gimenez, N., Clemente-Casares, P., Hundesa, A., Rodiguez-Manzano, J., Allard, A., Calvo, M., Girones, R. (2006). Quantification and stability of human adenoviruses and polyomavirus JCPyV in wastewater matrices. Appl. Environ. Microbiol. 72 (12), 7894–7896.

Brownell, M.J., Harwooda, V.J, Kurzb, R.C., McQuaiga, S.M., Lukasikc, J. Scott, T.M. (2007). Confirmation of putative stormwater impact on water quality at a Florida beach by microbial source tracking methods and structure of indicator organism populations. Water Res. 41; 3747 – 3757.

Cizek, A. R., Characklis, G W., Krometisa, L., Hayes, J. A., Simmons, O. D., Di Lonardo, S., Alderisio, K. A., and Sobsey, M. D. (2008). Comparing the partitioning behavior of *Giardia* and *Cryptosporidium* with that of indicator organisms in stormwater runoff. Water Res. 42; 4421 – 4438.

Curriero, F.C., Patz, J.A., Rose, J.B., Lele, S. (2001). The association between extreme precipitation and waterborne disease outbreaks in the United States, 1948–1994. Am. J. Publ. Health 91 (8), 1194–1199.

Dong, Y., Kim, J. Lewis, G. D. (2009). Evaluation of methodology for detection of human adenoviruses in wastewater, drinking water, stream water and recreational waters. J Appl Microbiol 108, 800–809.

Dorner, S.M., Anderson, W.B., Gaulin, T., Candon, H.L., Slawson, R.M., Payment, P. and Huck, P.M. (2007). Pathogen and indicator variability in a heavily impacted watershed. J Water Health 5, 241–257.

Dyke, M. I., V., Morton, V. K., McLeelan, N L., and Huck, P. M. (2009). The occurrence of Campylobacter in river water and waterfowl within a watershed in southern Ontario, Canada. J. Appl. Micro. 109, 1053-1066.

Fuhrman, J.A., Liang, X.L., Noble, R.T., (2005). Rapid detection of enteroviruses in small volumes of natural waters by real-time quantitative reverse transcriptase PCR. Appl. Environ. Microbiol. 71 (8), 4523–4530.

Gaffield, S.J., Goo, R.L., Richards, L.A. and Jackson, R. J. (2003). Public health effects of inadequately managed stormwater runoff. Am. J. Public Health 93 (9), 1527-1533.

Hamza, I. A., Jurzik, L., Stang, A., Sure, K., Uberla, K. and Wilhelm, M. (2009). Detection of human viruses in rivers of a densly-populated area in Germany using a virus adsorption elution method optimised for PCR analyses. Water Res. 43, 2657-2668.

Heim, A., Ebnet, C., Harste, G. and Pring-Akerblom, P. (2003). Rapid and quantitative detection of human adenovirus DNA by real-time PCR. J. Med. Virol. 70, 228-239.

Hill, V. R., Polaczyk, A. L., Hahn, D., Narayanan, J., Cromeans, T. L., Roberts, J. M. and Amburgey, J. E. (2005). Development of a rapid method for simultaneous recovery of diverse microbes in drinking water by ultrafiltration with sodium polyphosphate and surfactants. Appl. Environ. Microbiol. 71, 6878–6884.

Jiang, S. C. (2006). Human adenoviruses in water: Occurrence and health implications: A critical review. Environ. Sci. Technol. 40, 7132-7140.

Loge, F.N., Thompson, D.E., Call, D.R., 2002. PCR detection of specific pathogens in water: a risk-based analysis. Environ. Sci. Technol. 36 (12), 2754–2759.

Lund, M., Nordentoft, S., Pederson, K. And Madsen, M. (2004). Detection of *Campylobacter* spp. in chicken faecal samples by real-time PCR. Appl. Environ. Microbiol. 42(11), 5125–5132.

Mull, B. and Hill, V. R. (2009). Recovery and detection of *Escherichia coli* O157:H7 in surface water,using ultrafiltration and Real-Time PCR Appl. Environ. Microbiol. 75(11), 3593–3597.

Muscillo, M., Pourshaban, M., Iaconelli, M., Fontana, S., Di Grazia, A., Manzara, S., Fadda. G., Santangelo, R. and La Rosa, G. (2008). Detection and Quantification of Human Adenoviruses in Surface Waters by Nested PCR, TaqMan Real-Time PCR and Cell Culture Assays. Water Air Soil Pollut 191, 83–93.

Noble, R.T., Griffith, J.F., Blackwood, D. A., Fuhrman, Gregory, J. B., Hernandez, X., Liang, X., Angie A. Bera, A. A. and Schiff, K. (2006). Multitiered approach using quantitative PCR to track sources of faecal pollution affecting Santa Monica Bay, California. Appl. Environ. Microbiol. 72 (2), 1604-1612.

Parker, J.K., McIntyre, D., and Noble, R.T. (2010). Characterising faecal contamination in stormwater runoff in coastal North Carolina, USA. Water Res. 44 (14), 4186-4194.

Rajal, V.B., McSwain, B.S., Thompson, D. E., Leutenegger, C.M. and Wuertz, S. (2007). Molecular quantitative analysis of human viruses in California stormwater. Water Res. 41 (19), 4287-4298.

Robertson, B.H. and Nicholson, J.K.A. (2005). New microbiology tools for public health and their implications. *Annu Rev Pub Heal.* 26, 281–302. Sauer, E. P., VandeWalle, J. L., Bootsma, M. J., McLellan, S. L. (2011) Detection of the human specific *Bacteroides* genetic marker provides evidence of widespread sewage contamination of stormwater in the urban environment. Water Res. 45(14); 4081-4091.

Scott, T.M., Jenkins, T.M., et al. (2005). Potential use of a host associated molecular marker in Enterococcus faecium as an index of human faecal pollution. Environ. Sci. Technol. 39 (1), 283–287.

Savill, M.G., Hudson, J.A., Ball, A., Klena, J.D., Scholes, P., Whyte, R.J., McCormick, R.E. and Jankovic, D. (2001). Enumeration of Campylobacter in New Zealand recreational and drinking waters. J Appl Microbiol 91, 38–46.

Selvakumar, A. and Borst, M. (2006). Variation of microorganism concentrations in urban stormwater runoff with land use and seasons. J Wat Heal 4(1), 109-124.

Sercu, B., Van De Werfhorst, L.M., Murray, J. And Holden, P. A. (2009). Storm drains are sources of human faecal pollution during dry weather in three urban southern California watersheds. Environ. Sci. Technol.43 (2), 293-298.

Seurinck, S., Defoirdt, T.,Willy Verstraete, W. and Siciliano, S. D. (2005). Detection and quantification of the human-specific HF183 *Bacteroides* 16S rRNA genetic marker with real-time PCR for assessment of human faecal pollution in freshwater. Environ. Microbiol. **7**(2), 249–259.

Tree, J.A., Adams, M.R., Lees, D.N. (2003). Chlorination of indicator bacteria and viruses in primary sewage effluent. Appl. Environ. Microbiol. 69, 2038–2043.

Potential Health Risks Associated with the Use of Roof-Harvested Rainwater

Ahmed, W., Hodgers, L., Sidhu, J.P.S. and Toze, S.
CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland

Summary

Roof-harvested rainwater (RHRW) is one of the major potential alternative water sources that can be used in South East Queensland (SEQ). The most significant issue in relation to RHRW reuse is the potential public health risks associated with pathogenic microorganisms. This research project reports on the preliminary research outcomes toward gaining detailed information on the type, prevalence and source of pathogens that can be present in RHRW tanks within SEQ. Among the 80 RHRW samples tested, 72% and 92% were positive for *E. coli* and enterococci, respectively. Among the 80 samples, 10 (8%), 1 (1.3%), and 5 (6.3%) were positive for *Campylobacter* spp., *Salmonella* spp. and *G. lamblia*, respectively. To identify the sources of these pathogens, possum (*n*=39) and bird (*n*=21) faecal samples were screened for the above pathogens. A number of possum and bird faecal samples were positive for bacterial and protozoa pathogens suggesting that these animal species are being the dominant sources of faecal contamination in RHRW tanks. To date, several disease outbreaks and clinical cases associated with rainwater consumption have been reported. The results of the current study suggest that it would be prudent to disinfect RHRW, such the installation of a UV treatment unit, boiling, or other forms of effective treatment prior drinking. Maintenance of good roof and gutter hygiene and elimination of overhanging tree branches and other structures where possible to prevent the congregation of possums and birds are highly recommended for the improved quality of water.

Keywords
Roof-harvested rainwater, faecal contamination, zoonotic pathogens, PCR, Quantitative Microbial Risk Assessment.

Introduction

Roof-harvested rainwater (RHRW) has been used as potable and non-potable water sources in many countries (Despins *et al.*, 2009; Evans *et al.*, 2006; Uba and Aghogho, 2000). For example, 10% of Australian people currently use RHRW as a major source of their drinking water, and an approximate additional 5% use RHRW as potable replacement for showering, toilet flushing, and clothes laundering (ABS, 2007). The most significant issue in relation to RHRW for potable use is the potential public health risks associated with pathogenic microorganisms (Simmons *et al.*, 2001; Ahmed *et al.*, 2008). Wild animals such as birds, mammals (ie, bats, possums) and reptiles are the most likely sources of faecal contamination in RHRW as these animals can get access to the roof surface. Consequently, faecal matter from wild animals and other organic debris originating from overhanging trees on the roof-surface could be transported to the tank via roof runoff following rain events.

The microbiological quality of RHRW is generally assessed by monitoring faecal indicator bacteria such as faecal coliforms, *Escherichia coli*, and enterococci which are commonly found in the gut of warm-blooded animals including humans (Ahmed *et al.*, 2008; Pinfold *et al.*, 1983; Sazakil *et al.*, 2007; Spinks *et al.*, 2007). In addition, a number of studies on the microbial quality of RHRW detected specific zoonotic bacterial and protozoa pathogens in individual or communal RHRW systems (Ahmed *et al.*, 2008; Birks *et al.*, 2004; Crabtree *et al.*, 1996; Lye, 2002; Simmons *et al.*, 2001). However, it has to be noted that the majority of the studies reported in the literature assessed the quality of the RHRW on the basis of the presence or absence of the specific pathogens, with little information available regarding the actual numbers of these pathogens in RHRW.

Little is also known regarding the prevalence of pathogenic microorganism in wild animals such as birds and possums which are most likely the dominant sources of faecal contamination in RHRW tanks. Possums can get access to the roof via over hanging trees, electricity cable or climbing on to the roof via walls or other structure attached to the house. Birds can get access to the roof via overhanging trees or mounted structure on the roof such as TV aerial. It is documented that faecal contamination of RHRW appears to be limited to systems that are not well maintained and do not undergo any disinfection procedures.

The aims of this study were to: (i) assess the microbiological quality of RHRW tanks in SEQ, Australia; (ii) investigate the prevalence and numbers of faecal indicators (*E. coli* and enterococci) zoonotic bacterial (*Campylobacter* spp., and *Salmonella* spp.) and protozoa (*Cryptosporidium parvum* and *Giardia lamblia*) pathogens in RHRW tanks; and (iii) investigate the prevalence of above mentioned pathogens in faecal samples from possums and birds. Conventional culture based methods were used to enumerate *E. coli* and enterococci and quantitative PCR (qPCR) were used to obtain numbers of pathogens. These were done to obtain a better understanding regarding the magnitude of health risk associated with the drinking of RHRW and to obtain insights on how to manage such risks.

This research project reports on the on the preliminary research outcomes on the type, prevalence and source of pathogens that can be present in RHRW tanks from a range of different locations within SEQ.

Results and Discussion

RHRW Tanks Survey Results

In all, 80 RHRW tanks were selected for this study representing Currumbin Ecovillage, Bridgeman Downs, East Mt. Gravatt, Ashgrove, Eaton Hill, The Gap, Holland Park, Beaudesert, Versdale, Gleneagle, St. Lucia, Yeronga, Coochiemudlo Island, Yeerongpilly, Brookefield, Marburg, West Mount Cotton, Eastern Heights, Tingalpa, Algester, West End, Kenmore, McDowell, Annerley, Mitchelton, Coorparoo, New Market, Bellbird Park, Sherwood, Indooroopilly, Auchenflower, Graceville, Jamboree Heights and Albany Creek areas/suburbs in Brisbane and Gold Coast Region in SEQ, Australia. A sanitary survey was performed to identify various factors that may contribute to the faecal contamination of the selected tanks. The size of the selected tanks ranged from 5,000 to 30,000 L and aged between one to 20 years. Among the 80 tanks surveyed, 28 (35%) had overhanging trees, 43 (54%) tanks had visible sign of faecal droppings on the roof and 58 (73%) RHRW tanks were never desludged. 42 (53%) tanks had a first flush diverter installed. Among the 80 tanks, 44 (55%) were used for both potable and non-potable uses and the remaining 36 (45%) were used for only potable purposes. Among the households tested, 24 (30%) treated the water before consumption.

PCR Inhibitors

An experiment was conducted to determine the potential presence of PCR inhibitory substances in DNA extracted from RHRW samples (*n*=3), possums (*n*=3) and birds (*n*=3) faecal samples. For the spiked distilled water, the mean C_T value for the human-specific adenovirus DNA was 25.6 ± 0.4. For RHRW samples (*n*=3), possum (*n*=3) and bird (*n*=3) faecal samples, the mean C_T value was 28 ± 0.3, 27 ± 1.1, and 26 ± 0.4, respectively when undiluted DNA was spiked (Table 1). Values obtained for 10-fold and 100-fold dilutions are shown in Table 3. One-way analysis of variance (ANOVA) was performed to determine the differences between the C_T values obtained for distilled water and those obtained for RHRW samples, possum and bird faecal samples. No significant differences were observed between the C_T values for spiked distilled water, undiluted DNA, and serially diluted DNA, thus indicating that the tested samples were free of PCR inhibitors.

Table 1. Evaluation of PCR inhibition on the PCR detection of spiked sewage-associated adenoviruses in RHRW and animal faecal samples as opposed to spiked distilled water samples.

Samples	Threshold cycle (C_T) value (mean ± SD) for PCR		
	Undiluted DNA	10-fold dilution	100-fold dilution
RHRW tanks			
RHRW1	27.1 ± 0.3	26.4 ± 0.3	26.9 ± 0.3
RHRW10	27.2 ± 0.6	26.3 ± 0.4	26.6 ± 0.5
RHRW15	26.7 ± 0.2	26.5 ± 0.5	26.7 ± 0.4
Possum faeces			
P3	26.0 ± 0.3	25.8 ± 0.6	25.8 ± 0.7
P14	26.0 ± 0.6	25.4 ± 0.8	25.9 ± 0.3
P26	28.0 ± 0.1	25.9 ± 0.5	25.9 ± 0.6
Bird faeces			
B1	26.0 ± 0.4	26.0 ± 0.4	25.9 ± 0.9
B5	26.4 ± 0.5	25.6 ± 0.7	25.4 ± 0.4
B11	25.6 ± 0.2	25.2 ± 0.8	25.2 ± 0.6

Number of Faecal Indicators in RHRW Tanks

The membrane filtration method was used to process the RHRW tank samples for bacterial enumeration. Sample serial dilutions were made and filtered through 0.45 µm pore sized (47 mm diameter) nitrocellulose membranes (Advantec, Tokyo, Japan) and placed on modified mTEC agar (Difco, Detroit, MI), membrane-Enterococcus indoxyl-β-D-glucoside (mEI) agar (Difco) for the isolation of *E. coli* and enterococci, respectively.

Of the 80 samples tested, 27.5% had no measurable *E. coli* and the remaining 72.5% had > 1 *E. coli* per 100 mL. Similarly, 7.5% had no measurable enterococci and the remaining 92.5% had > 1 enterococci per 100 mL (Table 2). The number of *E. coli* in water samples (*n*=80) from RHRW ranged from 0 CFU per 100 mL to 4800 ± 1410 CFU per 100 mL of water. For enterococci, these figures were 0 CFU per 100 mL to 3233 ± 450 CFU per 100 mL.

Table 2. Percentage of RHRW samples positive for faecal indicators.

CFU/100 mL	*E. coli*	Enterococci
	RHRW tanks	RHRW tanks
< 1	28	8
1-10	22	20
11-100	26	42
101-1,000	21	26
>1,000	3	4



Figure 1. Dominant sources of faecal contamination in RHRW tanks.

Number of Pathogens in RHRW Tanks and Animal Faecal Samples

qPCR based methods were used to detect bacterial and protozoa zoonotic pathogens in RHRW samples, possum and bird faecal samples. Among the 80 households, ten (8%), one (1.3%), and five (6.3%) were positive for *Campylobacter* spp. 16S rRNA, *Salmonella invA*, and *G. lamblia* β-giardin genes, respectively (Table 3). The numbers of *Campylobacter* spp. 16S rRNA genes ranged from 1.5×10^1 to 3.0×10^2 genomic copies per L of water.

The numbers of *Salmonella invA* gene was 7.3×10^3 genomic copies per L of water. The numbers of *G. lamblia* β giardin genes ranged from 1.9×10^3 to 9.2×10^3 genomic copies per L of water. For the estimation of pathogen numbers, the number of the genomic copies (determined by qPCR) of each pathogen was converted to bacterial cells or protozoan cysts. The *G. lamblia* β-giardin gene is expressed as a single-copy gene within the nucleus of each trophozoite (Holberton and Marshall, 1995). Cysts of *Giardia* contain two trophozoites that have undergone multiple steps of nuclear division, resulting in 16 copies of total genetic information within each cyst (Bernader *et al.*, 2001). Therefore, there are 16 copies of the β-giardin gene per *Giardia* cyst (Guy *et al.*, 2003). After conversion of the number of genomic copies to number of cells, the number of *Campylobacter* spp. in RHRW samples ranged from 5.0×10^0 to 1.0×10^2 cells per L of water. Similarly the estimated number of *Salmonella* spp. was 7.3×10^3 cells per L of water. The numbers of *G. lamblia* cysts ranged from 1.2×10^2 to 5.8×10^2 per L of water. None of the samples were positive for *C. jejuni* and *C. parvum*.

Although certain householders are potentially exposed to pathogenic bacteria and protozoa, no increase has been reported in cases of salmonellosis, campylobacteriosis and giardiasis (giardiasis is not notifiable) in SEQ. This could be due to the fact that, there is a naturally high incidence of gastroenteritis in the community which may mask the actual disease (Hellard *et al.*, 2001). Before the disease can be reported in the Notifiable Diseases Surveillance System Database, it must first be identified, and not every individual will seek medical attention if the illness is mild and lasts only for a few days. Another factor is the possibility of individuals acquiring immunity to certain pathogens due to frequent exposure. It is acknowledged that the qPCR methods used in the study do not provide information on what fraction of PCR detected cells or cysts being viable and infective. This is one of the major limitations of PCR based methods. A number of possum (n=5) and bird (n=3) faecal samples were tested for the presence of *Campylobacter* spp. using both conventional and PCR based methods. Four possums and all three bird faecal samples were positive for *Campylobacter* spp. with both methods (data not shown).

Among the 39 possum faecal samples tested, the *Campylobacter* spp. 16S rRNA, *C. parvum* COWP, and *G. lamblia* β-giardin genes were detected in 24 (62%), three (8%), and nine (23%) samples, respectively. The numbers of *Campylobacter* spp. 16S rRNA gene ranged from 6.0×10^5 to 6.0×10^7 genomic copies per gm of faeces. *G. lamblia* β-giardin gene was detected in nine samples, however, seven were quantifiable. The numbers of *G. lamblia* β-giardin gene ranged from 3.3×10^2 to 2.5×10^3 genomic copies per gm of faeces. *C. parvum* COWP gene was not quantifiable and *Salmonella invA* gene could not be detected in DNA from possum faecal samples. After conversion of number of genomic copies to number of cells, the number of *Campylobacter* spp. and *G. lamblia* organisms in possum faecal samples ranged from 2.0×10^5 to 2.0×10^7 (for *Campylobacter* spp.) and 2.1×10^1 to 16×10^2 cysts (for *G. lamblia*).

Among the 21 bird faecal samples tested, the *Campylobacter* spp. 16S rRNA, *Salmonella invA*, *C. parvum* COWP, and *G. lamblia* β-giardin genes were detected in three (14%), two (10%), one (5%), and two (10%) samples, respectively. The numbers of *Campylobacter* spp. 16S rRNA, *Salmonella invA*, and *G. lamblia* β-giardin genes ranged from 1.2×10^4 to 2.5×10^7, 6.3×10^2 to 1.8×10^3, and 1.3×10^4 to 3.0×10^4 genomic copies per gm of faeces. *C. parvum* COWP gene was not quantifiable. After conversion of number of genomic copies to number of cells, the number of *Campylobacter* spp. *Salmonella* spp. and *G. lamblia* organisms in bird faecal samples ranged from 6.6×10^4 to 6.6×10^6 (for *Campylobacter* spp.), 6.3×10^2 to 1.8×10^3 (for *Salmonella* spp.) and 1.3×10^0 to 1.0×10^2 cysts (for *G. lamblia*).

Previous research studies also reported the presence of *G. lamblia* in possum and bird faeces in the North Island, New Zealand (Marino *et al.*, 1992; Chilvers *et al.*, 1998). In this study, three possum and one bird faecal samples were also positive for *C. parvum*. However, the prevalence of *C. parvum* was lower than *G. lamblia* in possum and bird faecal samples. Chilvers *et al.* (1998) reported similar findings and suggested that this could be due to the fact that the duration of *Cryptosporidium* infection much shorter than *Giardia* infection.

Table 3. Numbers of bacterial and protozoa pathogens in RHRW tanks and animal faecal samples.

Samples	No. of samples tested	No. of PCR positive samples (%)				
		Campylobacter spp. 16S rRNA	*C. jejuni mapA* gene	*Salmonella invA* gene	*C. parvum* COWP	*G. lamblia* β-giardin
RHRW	80	10 (8)	ND	1 (1.3)	ND	5 (6.3)
Possums	39	24 (62)	ND	ND	3 (8)	9 (23)
Birds	21	3 (14)	3	2 (10)	1 (5)	2 (10)

ND: Not detected.

Conclusions

In this study, RHRW samples were tested for the number of traditional faecal indicators, using conventional culture methods, as well as the presence of potential pathogenic microorganisms, using qPCR detection. One-off water samples were collected within one to seven days after rainfall when faecal and other organic matter deposited on the roof enters the tank via roof-run off. This was done to obtain information regarding the magnitude of faecal pollution in the worst-case scenario. Among the 80 RHRW sampled, 72% of the RHRW tanks exceeded Australian Drinking Water Guidelines of zero CFU *E. coli* per 100 mL water, suggesting tank water may not be suitable for drinking. For the detection and quantification of pathogens, qPCR based methods were used due to the fact that these methods are rapid and can detect microorganisms that are difficult to grow using conventional culture techniques. In addition, PCR has the potential to improve detection limits since a larger volume of water can be screened for the presence of pathogens (Ahmed *et al.*, 2008).

Among the pathogens tested, *Campylobacter* spp. *Salmonella* spp. and *G. lamblia* were detected in RHRW tanks. For *Campylobacter* spp., 80% human infections are caused by *C. jejuni* and 10% infections are caused by *C. coli.* Because of this, all *Campylobacter* spp. PCR positive samples were further tested for the presence of *C. jejuni*. None of the RHRW and household tap water samples was positive for *C. jejuni*.

The presence of *G. lamblia* in five RHRW tanks is alarming because of the low infectious dose of *Giardia* to cause illness. The number of *G. lamblia* appears to be one order of magnitude higher in RHRW samples in this study compared to our previous study (Ahmed *et al.*, 2010). It has to be noted that in this current study, 20 L of water samples were tested whereas in our previous study, a small volume (ie, 2-2.5 L) of water samples tested. Concentration of large volume of water samples may have increased the detection sensitivity.

Cryptosporidium parvum could not be detected in any of the RHRW samples tested, however, the presence of *Cryptosporidium* spp. in RHRW tanks has been reported in US Virgin Islands and Denmark (Albrechtsen, 2002; Crabtree *et al.*, 1995).

To identify the dominant sources of bacterial and protozoa pathogens in the RHRW tanks and household taps, a number of possum and bird faecal samples were tested for the presence of *Campylobacter* spp. *Salmonella* spp, *G. lamblia* and *C. parvum*. A large number of possum faecal samples were positive for *Campylobacter* spp. However, none of the samples were positive for *C. jejuni*. In contrast, three bird faecal samples were positive for *Campylobacter* spp. Further PCR analysis identified these three samples as *C. jejuni* (data not shown). A number of possum and bird faecal samples were also positive for *G. lamblia*. Three possum and one bird faecal samples were also positive for *C. parvum*. The results suggest that possums and birds carry both *G. lamblia* and *C. parvum* and have the potential to contaminate RHRW tanks. It has to be noted that *Giardia* cysts were also detected in faecal samples from cats, rats and mice, and therefore, these animals may also contribute *Giardia* to RHRW tanks. Other animals such as lizards, frogs and flying foxes that have access to the roof cannot be ruled out as possible sources of bacterial and protozoa pathogens in RHRW tanks.

In conclusion, RHRW samples tested in the study appear to be highly variable and of poor microbiological quality. The presence of one or more faecal indicators along with the potential bacterial and protozoa pathogens suggest RHRW may not be suitable for drinking without proper treatment methods implemented. The high prevalence of bacterial and protozoa pathogens in possum and bird faecal samples indicates these animal species as being the dominant sources of faecal contamination in RHRW tanks. Around 10% of the population in Australia currently use RHRW as a major source of their drinking water and use is increasing with the shortage of potable water (ABS, 2007). To date, several disease outbreaks and clinical cases associated with rainwater consumption have been reported (Ashbolt and Kirk, 2006; Brodribb *et al.*, 1985; Merritt *et al.*, 1999). The results of this study suggest that it would be prudent to disinfect RHRW prior to drinking, such as with the installation of a UV treatment unit, boiling, or other forms of effective treatment. Maintenance of good roof and gutter hygiene and elimination of overhanging tree branches and other structures where possible to prevent the congregation of possums and birds are highly recommended for the improved quality of water.

Acknowledgements

This research was undertaken and funded as part of the Queensland Urban Water Security Research Alliance, a scientific collaboration between the Queensland government, CSIRO, The University of Queensland and Griffith University. We thank residents of SEQ who provided access to their houses for collecting samples. We also thank "Peter the Possum Man" for providing possum faecal samples.

References

ABS 2007. Environmental issues: people's views and practices (no. 4602.0). Australian Bureau of Statistics, Canberra, Australia.

Ahmed, W., F. Huygens, A. Goonetilleke, and T. Gardner. (2008) Real-time PCR detection of pathogenic microorganisms in roof-harvested rainwater in Southeast Queensland, Australia. Appl. Environ. Microbiol. 74, 5490-5496.

Ahmed, W., A. Vieritz, A. Goonetilleke, and T. Gardner. **(**2010) Health risk from the use of roof-harvested rainwater in Southeast Queensland, Australia, as potable or nonpotable water, determined using quantitative microbial risk assessment. Appl. Environ. Microbiol. 76, 7382-7391.

Albrechtsen, H.J. (2002). Microbiological investigations of rainwater and gray-water collected for toilet flushing. Water Sci. Technol. 46, 311-316.

Ashbolt, R., and M.D. Kirk. **(**2006) *Salmonella* Mississippi infections in Tasmania: the role of native Australian animals and untreated drinking water. Epidemiol. Infect. 134, 1257–1265.

Bernander, R., J.E.D. Palm, and S.G. Svärd. **(**2001) Genome ploidy in different stages of the *Giardia lamblia* life cycle. Cell. Microbiol. 3, 55–62.

Birks, R., J. Colbourne, S. Hills, and R. Hobson. 2004. Microbiological water quality in a large in-building water recycling facility. Water Sci. Technol. 50, 165-172.

Brodribb, R., P. Webster, and D. Farrell. (1995). Recurrent *Campylobacter fetus* subspecies fetus bacteraemia in a febrile neutropaenic patient linked to tank water. Commun. Dis. Intell. 19, 312–313.

Chilvers, B.L., P.E. Cowan, D.C. Waddington, P.J. Kelley, and T.J. Brown. (1998). The prevalence of infection of *Giardia* spp. and *Cryptosporidium* spp. in wild animals on farmland, southeastern North Island, New Zealand. Int. J. Health Res. 8, 59-64.

Crabtree, K.D., R.H. Ruskin, S.B. Shaw, and J.B. Rose. (1995). The detection of *Cryptosporidium* oocysts and *Giardia* cysts in cistern water in the U.S. Virgin Islands. Water Res. 30, 208–216.

Despins, C., K. Farahbakhsh, and C. Leidl. (2009) Assessment of rainwater quality from rainwater harvesting systems in Ontario, Canada. J. Water Supply Res. Technol. AQUA. 58, 117-134.

Evans, C.A., P.J. Coombes, and R.H. Dunstan. (2006) Wind, rain and bacteria: The effect of weather or the microbial composition of roof-harvested rainwater. Water Res. 40, 37-44.

Guy, R.A., P. Payment, U.J. Krull, and P.A. Horgen. (2003) Real-time PCR for quantification of *Giardia* and *Cryptosporidium* in environmental water samples and sewage. Appl. Environ. Microbiol. 69, 5178-5185.

Hellard, M.E., M.I. Sinclair, A.B. Forbes, and C.K. Fairley. **(**2001) A randomised blinded, controlled trial investigating the gastrointestinal health effects of drinking water quality. Environ. Health Pers. 109, 773–777.

Holberton, D.V. and J. Marshall. (1995) Analysis of consensus sequence patterns in *Giardia* cytoskeleton gene promoters. Nucleic Acids Res. 23, 2945–2953.

Lye, D.J. (2002) Health risks associated with consumption of untreated water from household roof catchment system. J. Am. Water Res. Assoc. 38, 1301–1306.

Merritt, A., R. Miles, and J. Bates. (1999) An outbreak of *Campylobacter* enteritis on an island resort, north Queensland. Commun. Dis. Intell. 23, 215–219.

Marino, M.R., T.J. Brown, D.C. Waddington, R.E. Brockie, and P.J. Kelly. (1992) *Giardia intestinalis* in North Island possums, house mice and ship rats. N. Z. Vet. J. 40, 24–27.

Pinfold, J.V., N.J. Horan, W. Wiroganagud, and D. Mara. (1993) The bacteriological quality of rainjar water in rural northeast Thailand. Water Res. 41, 2039-2047.

Sazakil, E., A. Alexopolous, and M. Leotsinidis. (2007) Rainwater harvesting quality assessment and utilisation in Kelafonia Island, Greece. Water Res. 41, 2039-2047.

Simmons, G., V. Hope, G. Lewis, J. Whitmore, and G. Wanzhen. (2001) Contamination of potable roof-collected rainwater in Auckland, New Zealand. Water Res. 35, 1518–1524.

Spinks, J., S. Phillips, P. Robinson, and P. Van Buynder. (2006) Bushfires and tank rainwater quality: A cause for concern? J. Water Health. 4, 21–28.

Uba, B.N., and O. Aghogho. (2000) Rainwater quality from different roof-catchments in the Port Harcourt District, Rivers State, Nigeria. J. Water. Supply Res. Technol. AQUA 49, 281-288.

Assessing the Input of Hospitals to the Amount of Pharmaceutical Residues in Municipal Wastewater: A Consumption-Based Approach

Le Corre, K.[1], Kateley, D.[2], Allen, B.[2], Ort, C.[1,3], Keller, J.[1]
[1] Advanced Water Management Centre (AWMC), The University of Queensland, St Lucia, Queensland
[2] Medication Services Queensland, Queensland Health, Herston, Queensland
[3] Swiss Federal Institute of Aquatic Science and Technology (Eawag), CH-8600 Dübendorf, Switzerland

Summary

Hospitals are often pointed at as a major source of pharmaceutical residues in influents of sewage treatment plants and as such should undergo separate treatment prior to discharge. Recent case studies however concluded that the impact of a hospital on the loads of active pharmaceutical ingredients was limited. The question is whether experimentally measurable pharmaceuticals are the most relevant ones to focus our attention on. The current study investigates a novel consumption-based approach to predict the contribution of hospitals to the loads of 589 pharmaceuticals in municipal wastewater with the aim of identifying compounds of potential concerns. In addition, the possibility of hospital-specific substances being present in concentrations that may present a hazard for human health is evaluated.

Keywords
Audit data, hospital wastewater, pharmaceuticals, source control.

Introduction

The detection of an ever increasing number of pharmaceutical contaminants throughout the water cycle has raised scientific and public concerns regarding their potential impact on the environment and human health. Among the sources responsible for the presence of pharmaceuticals in the environment including household wastewater, agriculture (manure, fruit plantations) and pharmaceutical industries (Kümmerer, 2004), hospitals are often seen as major contributors to pharmaceutical loads in influents of sewage treatment plants (STP) (Hawkshead, 2008). As a result, the collection of hospital wastewater combined with municipal wastewater has been criticised and a dedicated (pre-) treatment of hospital wastewater has been recommended (Verlicchi *et al.*, 2010, Gupta *et al.*, 2009). However, recent studies concluded that the contribution of an individual hospital to the total load of a selection of pharmaceuticals in the influent of the corresponding STP was unlikely to exceed 15% (Thomas *et al.*, 2007, Ort *et al.*, 2010). But are experimentally measurable pharmaceuticals the most relevant ones to focus our attention on?

Indeed, due to various issues associated with the lack of readily available analytical methods, limited toxicity information and sampling restrictions (Murray *et al.*, 2010, Ort *et al.*, 2010), the analysis of a comprehensive list of compounds is hardly feasible (if not impossible). Therefore, the question is a matter of prioritisation and, ultimately, answering it requires finding methodologies to identify and quantify pharmaceuticals of concerns. To this end, the current study has investigated a novel consumption-based approach to predict the contribution of a selection of public hospitals located in the catchment area of advanced water treatment plants (AWTP) in South East Queensland (SEQ) to the loads of 589 pharmaceuticals in municipal wastewater. In addition, the possibility for hospital-specific substances of being present at levels that may present a hazard for human health is evaluated.

Methodology

Hospitals Investigated

The current study focuses on six hospitals in the catchment area of AWTP in SEQ. The hospitals selected are public hospitals located in three distinct catchments (Table 1). They vary in size and diversity of health services provided. QEII hospital is the smallest of the six with a total number of 132 beds. It provides a wide range of services including general medicine, orthopedics, urology, gynecology and general surgery. RBWH is the largest hospital with 882 beds. RBWH is a general and teaching hospital. It offers a comprehensive set of medical services such as medicine, surgery, orthopedics, psychiatry, oncology, trauma and comprehensive women's and newborn services. The volumes of water consumed at these 2 hospitals in 2008 were $95m^3$ day^{-1} and $627m^3$ day^{-1} respectively. QEII discharges its effluent to the Oxley Creek STP, which in 2008 treated on average 55,336 m^3 day^{-1}. RBWH hospital discharges its effluent to the Luggage Point STP, with an average of 148,622 m^3 of wastewater treated per day in 2008. All STPs, except Caboolture STP, are part of the Western Corridor Recycled Water scheme and their effluents undergo advanced water treatment including microfiltration, reverse osmosis and H_2O_2/UV advanced oxidation processes to produce purified recycled water (PRW).

Table 1. Characteristics of the investigated hospitals and catchments.

Hospital Name	Queen Elizabeth II Jubilee	Caboolture	Ipswich	The Prince Charles	Princess Alexandra	The Royal Brisbane & Women's Hospital
Acronym	QEII	CAB	IPS	PC	PA	RBWH
Number of beds*	132	190	296	533	754	882
Population in catchment area	280,000	45,000	75,000	572,000	572,000	572,000
Number of beds per 1,000 inhabitants	0.5	4.2	3.9	0.9	1.3	1.5
Hospital water consumption* (m^3 day^{-1})	95	126**	176***	541***	773***	627
STP	Oxley Creek	Caboolture	Bundamba	Luggage Point	Luggage Point	Luggage Point
Average raw influent flow rate to STP* (m^3 day^{-1})	55,300	16,100**	15,000	148,600	148,600	148,600
Ratio of hospital wastewater volume to influent wastewater volume at STP	0.002	0.008	0.012	0.004	0.005	0.004

* In 2008; ** Ort et al. (2010); ***Water consumption data only available for the year 2009

Audit Data Evaluation

Pharmaceutical Consumption Databases

Queensland hospitals audit data
Annual pharmaceutical consumption audit data at the six hospitals investigated were collected by Medication Services Queensland (Queensland Health, Clinical and Statewide Services Division, Queensland Government) for the year 2008. The database lists medicines dispensed annually to in-patients in public hospitals by generic name, strength, brand, form (ie, tablets, injections etc.) and quantity. In 2008, the database comprised 70,319 entries. As the audit data is a list of all items used by each hospital, it includes prescription drugs but also non-prescription drugs and other general items such as dispensers and needles.

National Consumption data
Consumptions of pharmaceuticals by the general public in Australia were compiled by the Drug Utilisation Sub-Committee (DUSC, Department of Health and Ageing, Australian Government) for the year 2008. The information provided in the database includes:

- The number of prescriptions submitted for payment of a subsidy under the Pharmaceutical Benefits and Repatriation Pharmaceutical Benefits Schemes (PBS/RPBS) supplied by Medicare Australia.
- Estimates of non-subsidised medicines calculated from continuous data on all prescriptions dispensed from a validated sample of community based pharmacies.

Data on prescription medicines dispensed to in-patients in public hospitals, non-prescription medicines (ie, available over the counter) and compounds sold in supermarkets are not taken into account. In 2008, the database comprised 928 entries. These correspond to single and combined pharmaceutical compounds and their respective mass consumption per year.

Prediction of the Contribution of a Hospital to the Load of Pharmaceuticals in Municipal Wastewater

Based on the information provided in the hospital audit data base, a list of 589 pharmaceutically active ingredients (API) to evaluate was generated. This list excludes naturally occurring substances such as hormones, sugars and enzymes as well as gaseous substances. Compounds available over the counter in Australia (TGA, 2011) are also excluded from the list since information on consumption for these substances is not available. The contribution of the six hospitals to the load of each individual API in municipal wastewater was predicted by comparing annual API consumptions in this hospital with its consumption by the general public.

Potential Impact of Hospital-Specific Substances on Human Health

In order to assess if hospital-specific compounds (ie, 100% contribution) may present a hazard for human health, estimated concentrations in both hospital wastewater and influent of the corresponding STP were compared to effect thresholds (ET). These ET (µg L^{-1}) were calculated based on the method used in the Australian guidelines for water recycling: managing health and environmental risks, augmentation of drinking water supply (NRMMC, 2008).

ET values for the entire list of compounds to evaluate were determined based on acceptable daily intakes (ADI, - µg kg_{BW}^{-1} d^{-1}). When not available, a substitute ADI (S-ADI - µg kg_{BW}^{-1} d^{-1}), was determined using the Lowest Daily Therapeutic Doses (LDTD- µg d^{-1}) available in the Australian Medical Information Management System (MIMS Australia, 2011).

A margin of exposure (MOE) was then determined to compare concentrations expected in hospital wastewaters and in influents of STPs with these ET values. A MOE > 100 means that the pharmaceutical concentration predicted in either wastewater type (hospital wastewater or STP influent) is more than 100-fold below the calculated exposure threshold values. In that case, the pharmaceutical is unlikely to present a risk of reaching drinking water sources at relevant concentrations and hence to affect human health.

Results

Contribution of the Six Hospitals

For this study, the contributions of six hospitals to the load of 589 pharmaceutically active ingredients in the influent of the STP to which they discharge their effluents were evaluated. Figure 1 shows the distribution of the contributions determined for the year 2008. The distribution patterns obtained are similar at each of the six hospitals. For 63 to 84% of the compounds, the contributions of an individual hospital are likely to be less than 15%. This suggests that for a large amount of the compounds investigated, hospitals would not be a major point source of pharmaceutical residues in municipal wastewater. This means that, for these compounds, at least 85% of the pharmaceutical load would still reach the corresponding STP even if hospital effluents were treated separately. For 16 to 37% of the evaluated compounds, the predicted contributions of hospitals could vary between 15 and 100%. The percentage of compounds used solely at a hospital, (ie, returning a 100% contribution) reaches 10% at the smallest hospital (ie, QEII hospital - 132 beds) and 20% at the largest (ie, RBWH hospital – 882 beds).

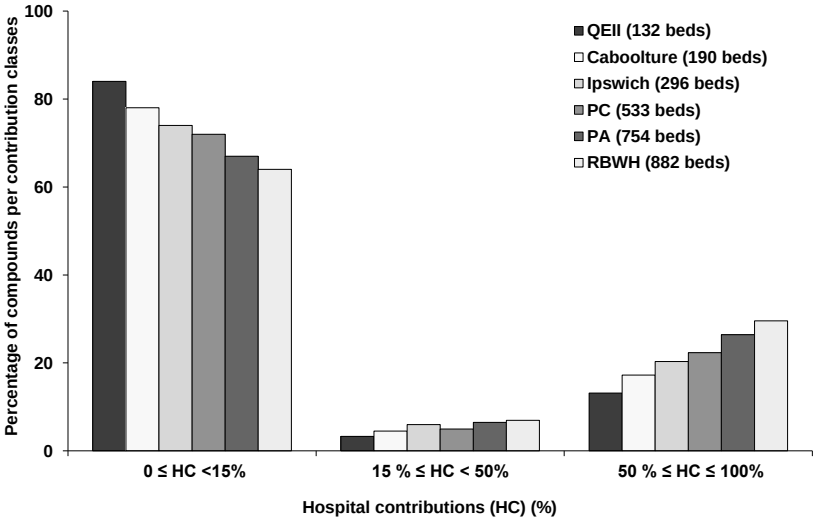


Figure 1. Percentage of compounds per hospital resulting in hospital contributions values in the ranges 0-15%, 15-50% and 50-100%.

When comparing predicted contributions obtained for a set of compounds in the contribution category 0≤HC<15% with experimental data available in the literature, the predicted contributions seems to correlate well with measured contributions. Indeed, all predicted values fall in the contribution ranges obtained experimentally by Langford and Thomas (2009) and Ort *et al.*, (2010) for metoprolol, atenolol, tamoxifen, paroxetine, sertraline, simvastatin and atorvastatin. For instance, predicted contributions for the β-blocker metoprolol ranged from 0.4% at QEII hospital to 2.5% at CAB hospital, while contributions measured at 2 hospitals in Norway by Langford and Thomas (2009) were 0.7% and 0.8%. Similarly, the 2.5% contribution predicted here at CAB hospital for metoprolol is also in good agreement with the results obtained experimentally by Ort *et al.* (2010) for that same hospital which varied from 2-7%.

Therapeutic Classes of Pharmaceuticals per Contribution Classes

Overall, there are 109 distinct therapeutic classes of drug figuring in the list of compounds to evaluate. The substances for which the hospitals would be a minor point source (ie, contribution <15%) are very diverse, with for example QEII and RBWH covering respectively 97 distinct classes in this contribution category. These classes range from common antibiotics (eg, cyprofloxacin), antidepressants (eg, paroxetine) and β-blockers (eg, atenolol) to more specific classes such as antineoplastics (eg, fluorouracil, tamoxifen) and antivirals (eg, lamiduvine). In the contribution category 15≤HC<100%, the number of compounds was typically lower (ie, 26 to 78 substances), so is the number of drug classes. This contribution category covers 11 and 31 distinct drug classes at QEII and RBWH hospitals respectively. At both hospitals, the most frequent therapeutic class in that contribution range are antibiotics. Finally, for compounds used solely in hospitals (100% contribution), the therapeutic classes include more specific substances. These include substances used in surgery such as anaesthetic agents and muscle relaxants (rocuronium) but also antivirals such as abacavir and ritonavir.

Hospital-Specific Compounds

As seen previously, the results of the audit data evaluation suggest that for a majority of the pharmaceutically active substances investigated, the hospital would not be a major contributor to the load of pharmaceutical residues in municipal wastewater. In contrast, the results show that 138 distinct pharmaceuticals resulted in 100% contributions for at least one of the six hospitals investigated. However, high hospital contributions may not be necessarily associated with high consumption values, and consequently high concentrations in municipal wastewater. For example the antiviral abacavir, which is used at four of the six hospitals, is not prescribed outside hospitals. For this compound, consumptions vary from 0.06g y^{-1} bed^{-1} at IPS hospital to 0.18g y^{-1} bed^{-1} at RBWH hospital. Based on water consumption data and assuming no metabolism, concentrations expected in these hospitals effluents would be 0.3 µg L^{-1} and 0.7 µg L^{-1}. This would lead to levels of 0.003 to 0.004 µg L^{-1} in the influent of the corresponding STP given flow rates data for the period investigated.

The comparison of concentrations predicted in hospital effluents with the calculated effect thresholds for compounds originating exclusively from hospitals show that - depending on the hospital investigated - between 54 and 75% of these compounds are expected in concentrations more than 100-fold lower than the calculated ET. In STP influents, the percentages of compounds for which concentrations are more than 100-fold lower than the ET increases to values in the range 90 to 100%. This indicates that only a small percentage of compounds originating from hospitals may require specific attention.

For instance, at the smallest of the hospital, QEII, 15 of the hospital-specific compounds concentrations were found to be less than 100-fold lower than the calculated ET values. Indeed, MOEs for these compounds vary from 1 for the local anaesthetic agents ropivacaine and oxybuprocaine to 70 for the antibiotic meropenem and the anaesthetic agent ketamine. However, when determining MOE values in the influent of the corresponding STP, none of these 15 compounds return a MOE below 100. The expected concentrations in STP influent (based on flow rates data and assuming no metabolism) would all be more than 500 times lower than the ET values, making these compounds unlikely to increase significantly risks of human exposure to any of these hospital specific substances.

At the largest hospital, RBWH, MOEs in hospital effluent are below 100 for 42 out of the 123 hospital-specific compounds. At this hospital, the cytotoxic drug vincristine sulphate, the beta-lactam inhibitor tazobactam, and the anaesthetic agent propofol are among the compounds with the lowest MOE (close or equal to 0) while the analgesic alfentanil, the antineoplastic agent anagrelide and the muscle relaxant atracurium have MOE just below 100 - 85, 92 and 98 respectively. In the influent of the STP to which RBWH hospital discharges its effluent, 10 out of 126 hospitals-specific compounds with MOE below 100 remain. These still include vincristine sulphate (MOE =0.5), tazobactam (MOE=3) and propofol (MOE=68).

Overall, the methodology used here for hospital specific substances show that 58 distinct substances across the six hospitals investigated would results in MOE values below 100 when calculated for hospital effluent. When calculated for influent of the STP to which each hospital discharge its effluent, the number of hospital-specific substances with an MOE value below 100 would decrease to 13. Anesthetic agents (local or general) are among the compounds most frequently listed and for which MOE in hospital effluent would be the lowest. For example, where used, concentrations predicted in hospital effluent for the general anesthetic propofol are very close to the calculated ET values (ie, 1≤ MOE ≤3). Though, when reported in STP influents to which the hospitals investigated discharge, concentrations for this compound is already 69-fold lower than the calculated ET values.

The second most represented therapeutic class is antibiotics. Here, nine distinct hospital-specific antibiotics (ampicillin, aztreonam, cefazolin, ceftazidime, ertapenem, meropenem, piperacillin, tazobactam and pentamidine) were used at one hospital at least and found to have an MOE below 100 based on concentrations expected in hospital effluents. However, the MOE calculations done here in hospital effluents are based on predicted concentrations assuming that no metabolism occurs. Therefore, MOEs in effluent of CAB and IPS hospitals are likely to increase

above 100. The list of distinct hospital-specific antibiotics is further reduced when MOE are estimated in STP influents. The compounds remaining are cefazolin, piperacillin and tazobactam.

Finally, seven antineoplastic agents presented MOE values below 100 in the hospital effluents at four of the six hospitals investigated (anagrelide, capecitabine, procarbazine, carmustine, vincristine, busulfan and mitomycin). For these substances, consumption values lead to concentrations varying from a minimum of 1.10^{-5} µg L^{-1} predicted for mitomycin in the effluent of PC hospital to a maximum of 14 µg L^{-1} for capecitabine at IPS hospital. However, concentrations in the corresponding STPs drop significantly making vincristine the only cytotoxic compound remaining with a MOE below 100 at PA and PC hospitals with concentrations below 0.012 µg L^{-1}. Although this study shows that cytostatic agents are typically hospital-specific compounds, the impact of hospitals discharge on the load of these compounds in municipal wastewater is difficult to assess. Indeed the administration of some of these compounds to out-patients as well as the slow excretion of some of these substances mean that significant fractions of cytotoxic drugs may be excreted at home (Weissbrodt *et al*., 2009).

Conclusion

In conclusion, it is important to note that raw wastewater concentrations are compared here with effect thresholds derived for augmentation of drinking water supply. Concentrations of pharmaceuticals in raw wastewater (from hospital or domestic sources) are likely to be significantly reduced after conventional wastewater treatment and advanced water treatment. The MOE determined here are then conservative values. This suggests that hospital-specific compounds are unlikely to be present in STP effluent at levels representing a risk to humans.

Nevertheless, the 13 compounds identified as having an MOE<100 in the influent of the STP warrant more detailed investigation including environmental and human toxicity, biodegradation and treatment/control options. Overall, the current approach seems to offer a unique opportunity of screening pharmaceuticals used in hospitals and identify potential compounds of concern that may require monitoring and/or specific treatment or disposal. Being based on consumption and loads discharged by hospitals in municipal wastewater, this approach is an additional step towards prioritisation of pharmaceuticals originating from hospital wastewater.

Acknowledgement

The authors acknowledge the following institutions and individuals for their contribution to this study: the Urban Water Security Research Alliance for their financial support; the Hospital Wastewater project team for the discussions and comments on this study; Queensland Health and the Medication Services Queensland (MSQ) for providing hospital audit data and support regarding the understanding and processing of the audit data; Vanna Mabbott and Chris Raymond from the Drug Utilisation Sub-Committee (DUSC) (Department of Health and Ageing, Australian Government) for providing statistics on Australian pharmaceutical consumptions and their support during the data processing; Queensland Urban Utilities for providing information on water consumptions of hospitals and flow data of the STPs investigated.

References

Gupta, P., Mathur, N. Bhatnagar, P., Nagar, P. and Srivastava, S. (2009). Genotoxicity evaluation of hospital wastewater. *Ecotox Environ Safe,* **72**(7), 1925-1932.

Hawkshead, J. (2008). Hospital wastewater containing pharmaceutically active compounds and drug resistant organisms: a source of environmental toxicity and increased antibiotic resistance. J Residuals Sci Tech, **5**(2), 51-60.

Kümmerer, K. (2004) Pharmaceuticals in the environment: Sources, fates, effects and risks. Second Edition, Springer, Berlin, Germany. 527p.

Langford, K.H. and Thomas, K.V. (2009). Determination of pharmaceutical compounds in hospital effluents and their contribution to wastewater treatment works. *Environ Int*, **35**(5), 766-770.

MIMS Australia (2011). MIMS Online, available at: https://www.mimsonline.com.au. Accessed in February 2011.

Murray, K.E, Thomas, M.S. and Bodour, A.A. (2010). Prioritizing research for trace pollutants and emerging contaminants in the freshwater environment. *Environ Pollut*, **158**(12), 3462-3471.

NRMMC (2008). Australian guidelines for water recycling: managing health and environmental risks (phase 2) – Augmentation of drinking water supplies, published by the Natural Resource Management Ministerial Council (NRMMC), the Environment and Heritage Council (EPHC) and the National Health and Medical Research Council (NHMRC), Canberra, Australia, 159p.

Ort, C., Lawrence, M.G., Reungoat, J., Eaglesham, G., Carter, S. and Keller, J. (2010). Determining the fraction of pharmaceutical residues in wastewater originating from a hospital. *Wat Res*, **44**(2), 605-615.

TGA (2011). Therapeutic Good Administration, Australia. Data provided in 21st March 2011, http://www.tga.gov.au/

Thomas, K.V., Dye, C., Schlabach, M. and Langford, K.H. (2007). Source to sink tracking of selected human pharmaceuticals from two Oslo city hospitals and a wastewater treatment works. *J Environ Monitor*, **9**(12), 1410-1418.

Verlicchi, P., Galleti, A., Petrovic, M. and Barceló, D. (2010). Hospital effluents as a source of emerging pollutants: an overview of micropollutants and sustainable treatment options. *J Hydrol*, **389**(3-4), 416-428.

Weissbrodt, D., Kovalova, L., Ort, C., Pazhepurackel, V., Moser, R., Hollender, J., Siegrist, H. and Mcardell, C.S. (2009). Mass flows of x-ray contrast media and cytostatics in hospital wastewater. *Environ Sci Technol*, **43**(13), 4810-4817.

Investigating the Role of Adsorption and Biodegradation in the Removal of Organic Micropollutants during Biological Activated Carbon Filtration of Treated Wastewater

Rattier, M.[1], Reungoat, J.[1], Gernjak, W.[1], Joss, A.[2], and Keller, J.[1]
[1] Advanced Water Management Centre (AWMC), The University of Queensland, St Lucia, Queensland
[2] Swiss Federal Institute of Aquatic Science and Technology (Eawag), CH-8600 Dübendorf, Switzerland

Summary

Municipal water recycling processes are potential human and environmental exposure routes for low concentrations of trace organic contaminants. This study assesses biological activated carbon (BAC) filtration for the removal of pharmaceuticals, pesticides and other organic micropollutants (MPs). The results show that BAC has a good potential for the removal of dissolved organic carbon (DOC) (40%) and MPs (60-95%). The primary objective of this research on BAC was to investigate in detail the removal mechanisms of representative MPs at environmentally relevant concentrations. Respective contribution of biodegradation and adsorption as removal mechanisms for 28 compounds with a wide range of physicochemical properties has been evaluated. Overall, average removal of compounds by adsorption on GAC achieved 85±10% removal with no influence of azide treatment. Average BAC removal reached 72±15% and diminished to 60±24% after inhibition of the microbial activity: a clear confirmation, that biological activity is important for BAC performance. The decrease in the MPs removal after inhibition of BAC biomass is discussed, providing further information about the biodegradability of targeted organic compounds. The long prior-operation time (>4 years and 35,000 bed volumes) of the tested BAC media suggests that adsorption equilibrium was achieved in the tested BAC grains, allowing the study of the role of biodegradation in the maintenance of the performance. The decrease in removal of some compounds by BAC showed that biomass present on BAC is potentially capable of removing a number of MPs. Sustained DOC removal by BAC after treatment with azide showed that BAC maintains sorption capacity, ie, by biological regeneration, also potentially responsible for the sustained removal of most recalcitrant compounds. These results highlight the advantage of combination of adsorption and biodegradation as compared to other biofiltration techniques for the long term attenuation of MPs.

Keywords
Adsorption, biodegradation, organic micropollutants, microbial activity, wastewater reuse.

Introduction

Over the last decade, organic contaminants such as endocrine disrupting compounds (EDCs) and pharmaceuticals active compounds (PhACs) have been detected in drinking water at very low levels, ng L^{-1} to µg L^{-1} concentrations (Kümmerer, 2009), suggesting that these compounds resisted removal through wastewater treatment processes. The removal of organic compounds during conventional activated sludge treatment and membrane bioreactors has been shown to be highly variable (Petrovic *et al.*, 2009) and many of these compounds reach the aquatic environment. Under recharge conditions, polar pharmaceuticals such as clofibric acid, carbamazepine, primidone, diclofenac or iodinated contrasts agents, can leach through the subsoil and have also been measured in groundwater samples (Heberer, 2002). Whether the long-term exposure to trace concentrations is of any public health significance is currently unknown. Nevertheless, the simple fact that they have been observed and reported is a cause of scientific and public concern. The presence of antibiotics in wastewater and their persistence through wastewater treatment processes may contribute to the prevalence of resistance to antibiotics in bacterial species (Baquero *et al.*, 2008; Richardson, 2009). Reusing treated effluents for non-potable or potable purposes increases the range of human and environmental exposure scenarios to bacteria potentially harbouring antibiotics resistance. The question of mixture toxicity has also recently gained more and more interest and additive effects are to be expected from mixtures of EDCs (Kümmerer, 2009).

Depending mainly on their sorption behaviour and biodegradability, chemicals bind to matter, can be transformed or mineralised. The most desirable removal mechanism is biodegradation for its low financial and environmental cost and acceptability as it can result in non-toxic transformation products or even inorganic end products. Limitations to biodegradation by wastewater treatment plants (WWTPs) have paved the way for the development of advanced treatment processes. Currently available technologies for the efficient removal of MPs from wastewater are based on membrane filtration, activated carbon, oxidation, managed aquifer recharge, and slow sand filtration. Activated carbon adsorption is very effective for MP removal but is limited by its adsorption capacity and has to be renewed regularly or dosed continuously. Nonetheless, when filtering secondary effluents containing microbes and effluent organic matter (EfOM), the media invariably become colonised by microorganisms. Indeed, the adsorption capacity of granular activated carbon (GAC) filters is generally deemed to be exhausted after approximately six months or 5,000 bed volumes (Simpson, 2008; Yapsakli and Çeçen, 2010). A type of surface biofilm appears and adds an additional mass transfer resistance layer thus reducing film diffusion. As a consequence for micropollutants, both

GAC adsorption capacity and rates are substantially reduced due to increasing preloading from organic matter, leading to a much shortened bed service life (Knappe *et al.*, 1999; Yu *et al.*, 2009). The preloading effects can be primarily attributed to competition for adsorption sites and pore blockage or constriction. Electrostatic interactions between biofilm and the investigated compounds are also proposed to contribute to the reduction of film diffusion.

The advantage of biofilms growing in biological activated carbon (BAC) filters is the presence of a diverse microbial community (Zhang *et al.*, 2011) that can participate in the treatment by removing biodegradable dissolved organic carbon (BDOC), and may degrade MPs present in secondary treated wastewater. A similar effect has been observed for cyanotoxins in drinking water treated with BAC (Wang *et al.*, 2007). In fact, the capability of diverse biological systems to degrade certain MPs has been well described in WWTPs biological processes (Onesios *et al.*, 2009; Tadkaew *et al.*, 2011) and the most common biofiltration technologies (Petrovic *et al.*, 2009; Rauch-Williams *et al.*, 2010; Baumgarten *et al.*, 2011), but only few studies have been published specifically on the MP removal capability of engineered BAC filters (Kim *et al.*, 1997; Gerrity *et al.*, 2011; Reungoat *et al.*, 2011). However, pilot scale BAC filters showed high performance in term of DOC and MP removal (28-68% and >90% respectively) (Reungoat *et al.*, 2011). A previous study on these filters showed that DOC removal had reached a steady state two years prior to the study (Reungoat *et al.*, 2011), indicating that adsorption process is under equilibrium conditions and suggesting that biodegradation is responsible for the removal observed. Clear evidence on the mechanism by which biological activity enhances MP removal by GAC is currently lacking. A synergistic effect between adsorption and biological activity on BAC filters is postulated to extend GAC's lifetime. In this paper, the removal of a broad range of MPs by BAC media is reported. In addition to biodegradation, the contribution of adsorption to MP removal was evaluated. For proper understanding of the removal of PPCPs during BAC treatment, the contribution of adsorption, biodegradation and the role of physico-chemical properties of the compounds with respect to their behaviour during BAC filtration are discussed.

Experimental Method

Pilot Scale BAC Filters

A sample of BAC media was collected from the top 10 cm of a pilot scale BAC filter set up in December 2006 at the South Caboolture Water Reclamation Plant (van Leeuwen *et al.*, 2003; Reungoat *et al.*, 2010). The reclamation plant receives water from a 40,000 equivalent people biological nutrient removal WWTP using a sequencing batch reactor process which achieves partial nutrient removal. The column is 3 m high and 22.5 cm internal diameter; it consist of 80±1 cm filtering bed (Acticarb BAC GA1000N, supplied by Activated Carbon Technologies Pty Ltd, Australia), supported by a 20 cm layer of gravel at the bottom operated with an empty bed contact time (EBCT) of an hour and a flow rate of 30 L hr^{-1}. The top of the columns are filled with water and compressed air is injected above the filtering bed to ensure a high level of dissolved oxygen to support biological activity. The BAC grains sampled have been filtering treated effluent for more than 4 years for a total of approximately 35,000 bed volumes. As the GAC media particles slowly become exhausted, it was observed that the rough porous surface of fixed bed GAC granular media was amenable to microbial colonisation (Figure1).

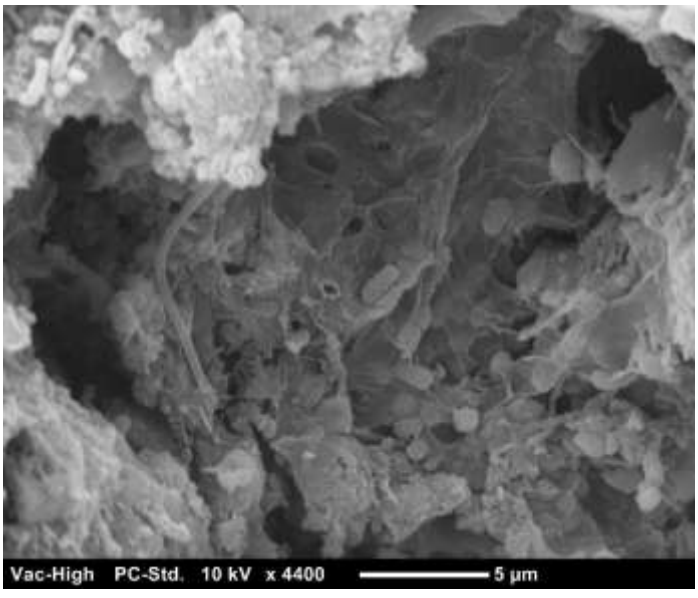


Figure 1. Scanning electron microscopy of the pore entry of a BAC particle exposed to 35,000 bed volumes of tertiary treated wastewater.

Adsorption and Biodegradation Batch Experiments

Batch experiments were conducted in two parallel amber glass reactors by adding 11.5 g of BAC and 2 L of feed water sampled at the top of the pilot scale filter. Reactors were capped and kept for 5 days at room temperature (23±1°C) and hand shaken twice a day. To discriminate between biodegradation and adsorption, the inhibition of the biomass activity (blockage of the ATP synthesis) was achieved by adding 300 mg L^{-1} sodium azide in the second reactor (BAC+azide). The azide anion blocks electron transfer and thereby inhibits energy production in the cells, which will cease energy dependant intracellular activities. It has been successfully used in several studies at concentrations ranging between 0.04 and 400 mg L^{-1} (Nishijima and Speitel, 2004; Uhl *et al.*, 2006; Rossner *et al.*, 2009; Yamamoto *et al.*, 2009; Lin *et al.*, 2010), seemingly without cell lysis as a complete recovery of azide suppressed bacteria was measured four hours after inhibitor was removed from a batch test with BAC in preliminary experiments. Two other parallel reactors were setup in the exact same conditions with the exception of media used; GAC instead of BAC was added to the reactors in order to verify that sodium azide did not influence adsorption. In order to ensure that conditions are kept aerobic during the batch experiment, the dissolved oxygen (DO) concentration was measured in the biologically active batch (BAC reactor) and aeration of every batch was completed when DO concentration reached 4mg L^{-1} in the BAC reactor.

Analytical Methods

Samples from the reactors after five days of contact time were compared to the feed for performance evaluation. Method for the assessment of MP concentration consisted of solid phase extraction (SPE), elution, concentration, and analysis by liquid chromatography coupled with tandem mass spectrometry (LC/MS–MS) (Reungoat *et al.*, 2011). 200 mL samples were extracted on a Visiprep manifold system (SigmaAldrich, U.S.A.) using Oasis HLB cartridges (60 mg, 3 mL) from Waters Corporation (U.S.A.), previously conditioned with 9 mL of methanol and 9 mL of deionised water (HPLC grade). Analyses of non-purgeable organic carbon (NPOC) were completed. The samples were filtered prior to all measurements using 0.45 mm filters, thus the determined NPOC is equivalent to DOC.

Results and Discussion

Dissolved Organic Carbon (DOC) Removal in GAC and BAC

As expected, the performance in terms of DOC removal achieved by adsorption or/and biodegradation by BAC reactors (30-40% DOC removal) was lower than by fresh GAC (75% DOC removal) as shown on Figure 2. This is due to the decreased adsorption capacity with increasing carbon load known to reduce high energy adsorption sites and blocking pores (Knappe *et al.*, 1999). Removal of the organic compounds in the BAC reactor took place accompanied by a significant reduction of oxygen concentration. This strongly suggests that part of the DOC removal in the BAC system is due to biodegradation.

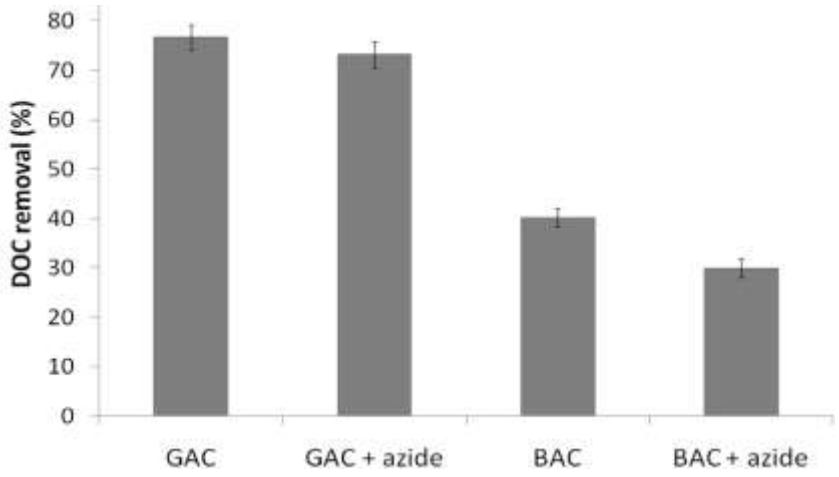


Figure 2. Percentage dissolved organic carbon (DOC) removal. Bars are the value of a single measurement; error bars represent the error on the removal value using the variance obtained from the measurements of 5 replicate samples.

To discriminate the adsorption and biodegradation removal mechanisms in BAC, sodium azide was added to inhibit microbial activity. The removal of DOC in the presence of the inhibitor was due to adsorption capacity remaining on BAC. Bioregeneration is a mechanism known to renew activated carbon media by microbial activities (Aktas and Cecen, 2007). After sampling, the microbial activity continued to reduce the DOC load on the BAC surface by bioregeneration, renewing a limited but significant adsorption capacity before the inhibition experiment. The observed advantage in term of DOC removal for the non-inhibited BAC compared to inhibited BAC at the end of the batch test is probably due to: (1) continued bioregeneration of adsorption sites during the five days of batch experiments by biodegradation of adsorbed organic substances; (2) direct uptake of BDOC from the feed by the biomass; and (3) replacement due to adsorption competition by non-BDOC of adsorbed BDOC subsequently desorbed and biodegraded. However, the third mechanism is less likely to be significant as it was shown previously that BDOC was degraded before adsorption (Nishijima and Speitel, 2004).

Micropollutants (MPs) Removal in GAC and BAC - Influence of Inhibition

In the same test, analysed MPs were present in the feed at concentrations below 300 ng L^{-1} with the exception of atenolol, carbamazepine, sulfamethoxazole, tramadol and venlafaxine (411, 660, 558, 1256, 1100 ng L^{-1} respectively). As expected during the first months of service of a GAC filter, adsorption by fresh GAC proved to have a great capacity to bind compounds with a broad range of physico-chemical properties by adsorption as greater than 80% of the tested compounds were removed. Practically, the efficiency of activated carbon mainly relies on its accessible surface area where physical adsorption takes place, on the heteroatom content and the properties of the adsorbates. Adsorption from solution is essentially an exchange process, and hence molecules adsorb not only because they are attracted by solids but also because of their low affinity with the solvent (Moreno-Castilla, 2004). Thus, the characteristics of the solution chemistry are also parameters influencing the adsorption process (Schreiber *et al.*, 2005). Interestingly, the presence of sodium azide in the water did not influence the removal of organic MPs by GAC adsorption. Important physico-chemical characteristics of the compounds, such as their octanol-water distribution coefficient, log D, can be used for predicting the tendency of a chemical to partition between water and solids and/or solid organic matter. A general sense is that compounds with a higher log D value should have higher sorption affinity on activated carbon. However, the results obtained in the present investigation at ng L-1 concentrations confirmed that this may not be the case in very dilute solutions, as it was previously observed by Yu *et al.* (2009). The performance and removal mechanisms of activated carbon in natural water are also significantly impacted by EfOM present in the water.

After several months of filtrations, the initial performance of the GAC is expected to decrease due to accumulation of EfOM, and formation of biofilm on the media now termed BAC. Indeed, limitation of diffusion by the biofilm, competition for adsorption sites with the EfOM and pore blockage by the EfOM and biomass are expected to decrease MP removal with an increasing number of bed volumes filtered (Yu *et al.*, 2009; Corwin and Summers, 2011; de Ridder *et al.*, 2011). Initial breakthrough of trace organic compounds in the filtrate of loaded GAC was previously shown to occur after filtering 3,000 to 80,000 bed volumes depending on the compounds and the type of GAC (de Ridder *et al.*, 2011). As mentioned before, the action of inhibiting the biomass (BAC + azide reactor) decreased the DOC removal by only 10% compared to the BAC reactor, so the DOC removal by adsorption was still high (60%). As a consequence, a substantial removal of MPs by BAC and inhibited BAC is to be expected. Removal of target organic compounds by conventional BAC (BAC reactor) and inhibited BAC (BAC + azide reactor) are presented in Figure 3. As expected, loading of the media with organic matter and MPs not only reduced the removal of DOC but also the removal of MPs. Nevertheless, a significant reduction of the MP concentrations was achieved by the BAC reactor with most of the quantified compounds removed from the liquid phase by more than 60%. Interestingly, compounds such as carbamazepine, citalopram, diclofenac, sulfamethoxazole and trimethoprim, manifesting resistance to biological transformation previously (Suarez *et al.*, 2010), and persistence through subsoil filtration (Heberer, 2002) were removed to a high extent by the BAC. The removal obtained by the inhibited BAC was attributed to sorption only on the BAC grains. These results confirm the high potential of BAC filtration for the long-term removal of MPs during advanced treatment of WWTP effluents.

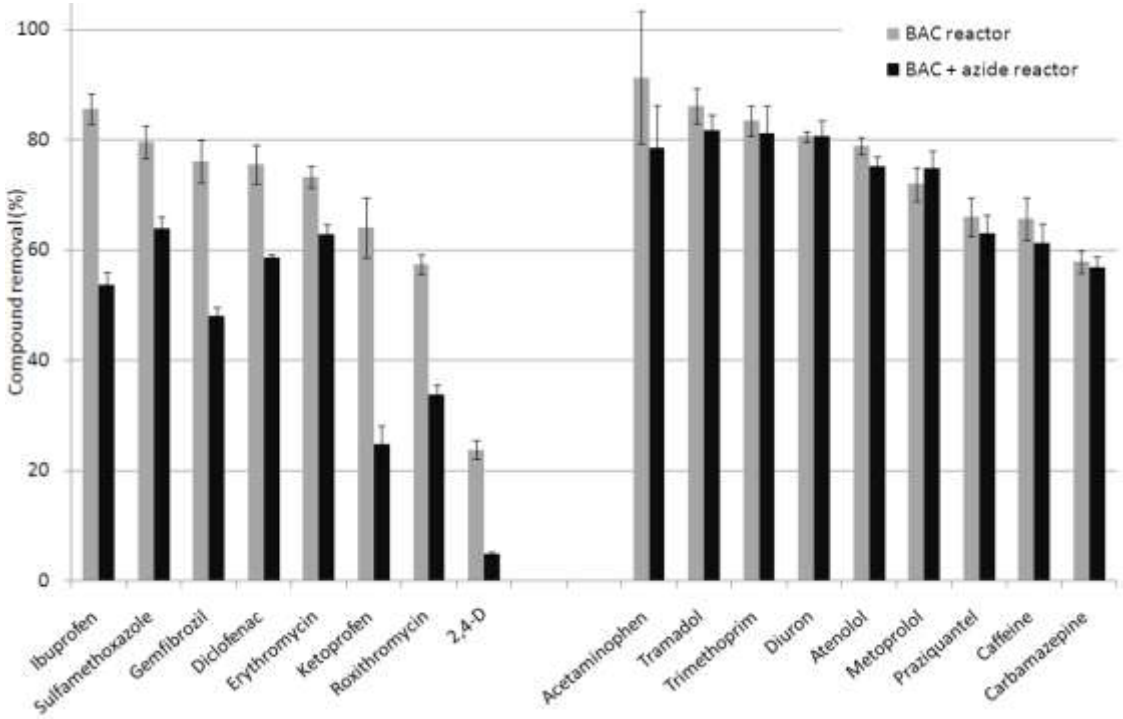


Figure 3. Removal of MPs by BAC and BAC + 300 mg L^{-1} sodium azide (MPs arranged in 2 groups depending on the influence of inhibition on their removal. MPs classified within groups from left to right in decreasing removal order). Bars represent the mean of duplicates. Error bars represent standard deviation observed for duplicate measurements of the same sample.

A decrease in the removal of several compounds due to inhibition was attributed to the absence of biodegradation and bioregeneration within the BAC grains. Further interpretation of results can be done by separating compounds in two different parts; on one side, the compounds showing a decrease in removal after inhibition and on the other the compounds showing no decrease in removal after inhibition.

Most of the compounds showing a decrease in removal due to the addition of sodium azide were known to be biodegradable, and as such, biodegradation of compounds by the biomass present within the BAC is a plausible explanation. The porous structure of activated carbon may facilitate growth and maintenance of special degrading microbes which may contribute to enhanced removal of varied compounds. On the other hand, sulfamethoxazole, diclofenac and hydrochlorothiazide shown previously to be relatively persistent through biological systems (Kimura *et al.*, 2005; Castiglioni *et al.*, 2006; Maeng *et al.*, 2011), were also significantly less removed when the biomass was inhibited. Surpisingly, these results suggested biodegradation as a mechanism potentially aiding the removal of these recalcitrant compounds. A longer biomass retention time has already been suggested to allow for the enrichment of slower growing bacterial species, and therefore to provide greater diversity of enzymes, some of which are capable of breaking down pharmaceutical compounds (Jones *et al.*, 2005; 2007). The characteristics of the DOC also need to be considered in the case of biodegradation of relatively recalcitrant compounds. Indeed, Lim *et al.* (2008) observed that contaminants such as sulfamethoxazole were less difficult to remove in BDOC derived from aquatic plants, compared to BDOC derived from wastewater effluent dominated surface water. The variation in biotransformation rates of test compounds in biofilms receiving different sources of BDOC was attributed to the differences in microbial diversity and functionality among the microcosms. This implies that compounds being termed recalcitrant in secondary treatment could be amenable to degradation by the microorganisms growing on BAC.

Another explanation for the retention of recalcitrant compounds is the removal by adsorption on the BAC surface regenerated by the biomass (Herzberg *et al.*, 2003). The principal theory regarding bioregeneration is the hypothesis of desorption by exoenzymatic reaction (Nath and Bhakhar, 2010). Indeed, adsorbed molecules are included in the biofilm and probably more exposed to enzymes produced, maybe being responsible for elimination of compounds by co-metabolism. Being much smaller than a bacterial cell, these exoenzymes can diffuse into the mesopores and micropores of the activated carbon. Exoenzymes are believed to act on the adsorbed compound inducing its hydrolytic transformation. The new products generated by this process may have a lower affinity to the activated carbon and therefore desorb renewing adsorption capacity. The absence of bioregeneration is a plausible explanation for the decrease in removal of recalcitrant compounds analysed in this study.

Compounds for which the removal did not vary with and without inhibition suggest that biological transformation was slow or nonexistent toward these compounds. Surprisingly, well biodegradable compounds such as caffeine and acetaminophen (Heberer, 2002; Reungoat *et al.*, 2011) are found in this category. No adaptation of the BAC microbial community to the compounds degradation can be shown in this case. Similar observation can be made for acetaminophen which was easily degraded and removed by the WWTPs (Heberer, 2002; Reungoat *et al.*, 2011). On the other hand, relatively recalcitrant MPs were expected to not show any variation in removal after biomass inhibition. Compounds such as trimethoprim and carbamazepine, shown previously to be persistent in the aquatic environment (Drewes *et al.*, 2002; Maeng *et al.*, 2011), were still well removed in the presence of sodium azide (>60%).

Conclusions

In a reactor containing BAC, the removal of DOC and eight MPs were reduced by 10% and between 8 and 39% respectively after the addition of azide, indicating a substantial influence of biodegradation. Results of inhibition also showed that adsorption was still partly responsible for every MPs removal in BAC even after four years filtering of treated secondary effluent (35,000 bed volumes).

The removal of DOC and some organic MPs are significantly enhanced by the action of microbial biomass in BACs by different potential mechanisms. Comparison to previous studies on biological removal of the same organic compounds showed that biodegradation of the recalcitrant compounds by the biomass present on BAC is a plausible explanation. The removal of compounds known to be recalcitrant was attributable to either acclimatisation or bioregeneration of adsorption.

Adsorption capacity recovered by bioregeneration can be responsible for the extension of MP removal from the liquid phase by BAC. Removal of compounds, including highly stable compounds such as bezafibrate, carbamazepine, and trimethoprim, were not significantly impacted by the inhibition, but a substantial adsorption capacity remained and enabled removals greater than 60%. The important advantage of BAC filtration over other biological filtration for MP removal lies in the sustainable high adsorption capacity even after several years of filtration.

References

Aktas, O. and F. Cecen (2007). "Bioregeneration of activated carbon: A review." *International Biodeterioration and Biodegradation* **59**(4): 257-272.

Baquero, F., J.-L. Martínez and R. Cantón (2008). "Antibiotics and antibiotic resistance in water environments." *Current Opinion in Biotechnology* **19**(3): 260-265.

Baumgarten, B., J. Jährig, T. Reemtsma and M. Jekel (2011). "Long term laboratory column experiments to simulate bank filtration: Factors controlling removal of sulfamethoxazole." *Water Research* **45**(1): 211-220.

Castiglioni, S., R. Bagnati, R. Fanelli, F. Pomati, D. Calamari and E. Zuccato (2006). "Removal of Pharmaceuticals in Sewage Treatment Plants in Italy." *Environmental Science and Technology* **40**(1): 357-363.

Corwin, C. J. and R. S. Summers (2011). "Adsorption and desorption of trace organic contaminants from granular activated carbon adsorbers after intermittent loading and throughout backwash cycles." *Water Research* **45**(2): 417-426.

de Ridder, D. J., A. R. D. Verliefde, S. G. J. Heijman, J. Q. J. C. Verberk, L. C. Rietveld, A. L. T. J. van der, G. L. Amy and J. C. van Dijk (2011). "Influence of natural organic matter on equilibrium adsorption of neutral and charged pharmaceuticals onto activated carbon." *Water Science and Technology* **63**(3): 416-423.

Drewes, J. E., T. Heberer and K. Reddersen (2002). "Fate of pharmaceuticals during indirect potable reuse." *Water Science and Technology* **46**(3): 73-80.

Gerrity, D., S. Gamage, J. C. Holady, D. B. Mawhinney, O. QuiÃ±ones, R. A. Trenholm and S. A. Snyder (2011). "Pilot-scale evaluation of ozone and biological activated carbon for trace organic contaminant mitigation and disinfection." *Water Research* **45**(5): 2155-2165.

Heberer, T. (2002). "Occurrence, fate, and removal of pharmaceutical residues in the aquatic environment: a review of recent research data." *Toxicology Letters* **131**(1-2): 5-17.

Herzberg, M., C. G. Dosoretz, S. Tarre and M. Green (2003). "Patchy Biofilm Coverage Can Explain the Potential Advantage of BGAC Reactors." *Environmental Science and Technology* **37**(18): 4274-4280.

Jones, O. A. H., N. Voulvoulis and J. N. Lester (2005). "Human pharmaceuticals in wastewater treatment processes." *Critical Reviews in Environmental Science and Technology* **35**(4): 401-427.

Jones, O. A. H., N. Voulvoulis and J. N. Lester (2007). "The occurrence and removal of selected pharmaceutical compounds in a sewage treatment works utilising activated sludge treatment." *Environmental Pollution* **145**(3): 738-744.

Kim, W. H., W. Nishijima, A. U. Baes and M. Okada (1997). "Micropollutant removal with saturated biological activated carbon (BAC) in ozonation-BAC process." *Water Science and Technology* **36**(12): 283-298.

Kimura, K., H. Hara and Y. Watanabe (2005). "Removal of pharmaceutical compounds by submerged membrane bioreactors (MBRs)." *Desalination* **178**(1-3): 135-140.

Knappe, D. R. U., V. L. Snoeyink, P. RÃ¶che, M. J. Prados and M. M. Bourbigot (1999). "Atrazine removal by preloaded GAG." *Journal / American Water Works Association* **91**(10): 97-109.

Kümmerer, K. (2009). "The presence of pharmaceuticals in the environment due to human use - present knowledge and future challenges." *Journal of Environmental Management* **90**(8): 2354-2366.

Lim, M.-H., S. A. Snyder and D. L. Sedlak (2008). "Use of biodegradable dissolved organic carbon (BDOC) to assess the potential for transformation of wastewater-derived contaminants in surface waters." *Water Research* **42**(12): 2943-2952.

Lin, A. Y.-C., C.-A. Lin, H.-H. Tung and N. S. Chary (2010). "Potential for biodegradation and sorption of acetaminophen, caffeine, propranolol and acebutolol in lab-scale aqueous environments." *Journal of Hazardous Materials* **183**(1-3): 242-250.

Maeng, S. K., S. K. Sharma, K. Lekkerkerker-Teunissen and G. L. Amy (2011). "Occurrence and fate of bulk organic matter and pharmaceutically active compounds in managed aquifer recharge: A review." *Water Research* **45**(10): 3015-3033.

Moreno-Castilla, C. (2004). "Adsorption of organic molecules from aqueous solutions on carbon materials." *Carbon* **42**(1): 83-94.

Nath, K. and M. Bhakhar (2010). "Microbial regeneration of spent activated carbon dispersed with organic contaminants: mechanism, efficiency, and kinetic models." *Environmental Science and Pollution Research*: 1-13.

Nishijima, W. and G. E. Speitel (2004). "Fate of biodegradable dissolved organic carbon produced by ozonation on biological activated carbon." *Chemosphere* **56**(2): 113-119.

Onesios, K. M., J. T. Yu and E. J. Bouwer (2009). "Biodegradation and removal of pharmaceuticals and personal care products in treatment systems: a review." *Biodegradation* **20**(4): 441-466.

Petrovic, M., M. J. L. De Alda, S. Diaz-Cruz, C. Postigo, J. Radjenovic, M. Gros and D. Barcelo (2009). "Fate and removal of pharmaceuticals and illicit drugs in conventional and membrane bioreactor wastewater treatment plants and by riverbank filtration." *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences* **367**(1904): 3979-4003.

Rauch-Williams, T., C. Hoppe-Jones and J. E. Drewes (2010). "The role of organic matter in the removal of emerging trace organic chemicals during managed aquifer recharge." *Water Research* **44**(2): 449-460.

Reungoat, J., B. I. Escher, M. Macova and J. Keller (2011). "Biofiltration of wastewater treatment plant effluent: Effective removal of pharmaceuticals and personal care products and reduction of toxicity." *Water Research* **45**(9): 2751-2762.

Reungoat, J., M. Macova, B. I. Escher, S. Carswell, J. F. Mueller and J. Keller (2010). "Removal of micropollutants and reduction of biological activity in a full scale reclamation plant using ozonation and activated carbon filtration." *Water Research* **44**(2): 625-637.

Richardson, S. D. (2009). "Water Analysis: Emerging Contaminants and Current Issues." *Analytical Chemistry* **81**(12): 4645-4677.

Rossner, A., S. A. Snyder and D. R. U. Knappe (2009). "Removal of emerging contaminants of concern by alternative adsorbents." *Water Research* **43**(15): 3787-3796.

Schreiber, B., T. Brinkmann, V. Schmalz and E. Worch (2005). "Adsorption of dissolved organic matter onto activated carbon--the influence of temperature, absorption wavelength, and molecular size." *Water Research* **39**(15): 3449-3456.

Simpson, D. R. (2008). "Biofilm processes in biologically active carbon water purification." *Water Research* **42**(12): 2839-2848.

Suarez, S., J. M. Lema and F. Omil (2010). "Removal of Pharmaceutical and Personal Care Products (PPCPs) under nitrifying and denitrifying conditions." *Water Research* **44**(10): 3214-3224.

Tadkaew, N., F. I. Hai, J. A. McDonald, S. J. Khan and L. D. Nghiem (2011). "Removal of trace organics by MBR treatment: The role of molecular properties." *Water Research* **45**(8): 2439-2451.

Uhl, W., F. Persson, G. Heinicke, M. Hermansson and T. Hedberg (2006). Removal of geosmin and MIB in biofilters - on the role of biodegradation and adsorption. Recent Progress in Slow Sand and Alternative Biofiltration Processes. L. IWA Publisshing, UK.**:** 360-368.

van Leeuwen, J., C. Pipe-Martin and R. M. Lehmann (2003). "Water Reclamation At South Caboolture, Queensland, Australia." *Ozone: Science and Engineering* **25**(2): 107 - 120.

Wang, H., L. Ho, D. M. Lewis, J. D. Brookes and G. Newcombe (2007). "Discriminating and assessing adsorption and biodegradation removal mechanisms during granular activated carbon filtration of microcystin toxins." *Water Research* **41**(18): 4262-4270.

Yamamoto, H., Y. Nakamura, S. Moriguchi, Y. Nakamura, Y. Honda, I. Tamura, Y. Hirata, A. Hayashi and J. Sekizawa (2009). "Persistence and partitioning of eight selected pharmaceuticals in the aquatic environment: Laboratory photolysis, biodegradation, and sorption experiments." *Water Research* **43**(2): 351-362.

Yapsakli, K. and F. Çeçen (2010). "Effect of type of granular activated carbon on DOC biodegradation in biological activated carbon filters." *Process Biochemistry* **45**(3): 355-362.

Yu, Z., S. Peldszus and P. M. Huck (2009). "Adsorption of selected pharmaceuticals and an endocrine disrupting compound by granular activated carbon. 1. Adsorption capacity and kinetics." *Environmental Science and Technology* **43**(5): 1467-1473.

Zhang, D., W. Li, S. Zhang, M. Liu, X. Zhao and X. Zhang (2011). "Bacterial community and function of biological activated carbon filter in drinking water treatment." *Biomedical and Environmental Sciences* **24**(2): 122-131.

Recent Advances in Field-Based Water Quality Monitoring Techniques

Zhao, H.[1], O'Halloran, R.[2], Zhang S.[1] and Winnel, M.[2]
[1] Centre for Clean Environment and Energy, Griffith University, Gold Coast, Queensland
[2] CSIRO Land and Water, Clayton, Victoria

Introduction

Generally, the majority of existing online sensors/sensing systems perform well in a laboratory environment for pre-treated samples. Nevertheless, only a few have demonstrated satisfactory performance in field environments, especially for applications involving a difficult analytical environment or sample matrix. Despite enormous effort, limited improvement on the applicability of online instruments for field-based analyses has been achieved. This could be attributed to the fact that most existing online instruments were developed by direct adoption of classic laboratory-based analytical methods that were not originally developed for field-based applications. These methods exhibit an intrinsic flaw of requiring strictly-defined measurement conditions. As a result, they may work well in a lab environment, where sample manipulation is allowed, but become problematic for field-based online applications where the circumstances do not allow the sample to be artificially manipulated before assay. In this regard, two general strategies can be used to develop analytical methods suitable for field-based applications: to innovatively use analytical signals of existing sensors; and to purposely develop new field-based analytical principles. This presentation will review recent advances in field-based analytical methods developed in the Centre for Clean Environment and Energy, Griffith University and CSIRO Land and Water laboratories.

Keywords
Photocatalysis-based COD analysis, artificial electron acceptor-based BOD analysis, diffusive gradients in thin films technique, real-time event detection.

Results

The first part of the presentation will introduce a suite of purposely developed field-based analytical methods including: photocatalysis-based COD analysis (PeCOD^{TM}); artificial electron mediator-based BOD analysis (F-BOD); and a new diffusive gradients in thin films technique (DGT).

The standard COD method involves a time-consuming (2-4 hour) reflux process, requires the used of expensive (Ag_2SO_4), highly corrosive (concentrated H_2SO_4), and extremely toxic (Hg(II) and Cr(VI)) reagents, and is only suitable for samples with a concentration grater than 10 ppm. PeCOD^{TM} technology uses the strong oxidation power of illuminated nanostructured TiO_2 photocatalysts to achieve rapid and complete mineralisation of organic pollutants, while the accompanied photocatalysis processes are used to quantify the COD value. The results to date have demonstrated that PeCOD^{TM} is an absolute method that requires no calibration. It requires only 0.5–5 min to complete an assay and only consumes $NaNO_3$[1-3].

The standard five-day biochemical oxygen demand (BOD_5) method has been the most widely used method to determine biodegradable organic pollutants but the method has drawbacks. It requires five days to complete an assay, involves tedious procedures, and needs highly skilful personnel to obtain reproducible results. The five-days assay time is mainly due to the insufficiency of the electron acceptor resulting from the poor water solubility of O_2. F-BOD method uses a highly water soluble artificial electron acceptor (eg, Ferricyanide, 10,000 time more soluble than O_2) to replace the natural electron acceptor (O_2) to achieve rapid biodegradation. The results confirmed that the F-BOD method is capable of reducing the assay time from five days to 30 min[4,5].

DGT is a powerful environmental monitoring tool. However, the traditional DGT employs problematic hydrogel-based binding phase and diffusive layer, greatly limited its applicability to determine wide spectrum of analytes. In this work, we present a revolutionised new DGT based on a liquid binding phase and solid membrane diffusive layer[6,7]. The results demonstrated that the new DGT overcomes many problems of the traditional hydrogel-based DGT and has been widely adopted for environmental monitoring applications.

The second part of the presentation will introduce a real-time sewage quality information collection system (SQICS) with the capability of near real-time detecting anomalous sewage quality changes (an event), developed through the UWSRA. By innovatively utilising commercially available sensors combined with the new mathematic event detection model enables the system to reliably function in the sewage environment over three months without the need for calibration and maintenance. The developed SQICS system has shown a great potential as an effective tool for wastewater source control and management.

Conclusions

A suite of field –based analytical methods have been presented. The results demonstrated that the innovative use of analytical signals of existing sensors and development of new field-based analytical principles are effective strategies to develop suitable analytical methods for field-based applications.

References

1. Zhao H., Jiang D., Zhang S. and John R., *Analytical Chemistry*, 76 (2004) 155.
2. Zhang S., Jiang D., Zhao H., *Environmental Science and Technology*, 40 (2006) 2363.
3. S. Zhang, L. Li and H. Zhao, *Environmental Science and Technology*, 43 (2009) 7810.
4. Catterall K., Zhao H., Pasco N., John R., *Analytical Chemistry*, 75 (2003) 2584.
5. Morris K., Zhao, H.; John, R., *Australian Journal of Chemistry*, *Invited review*, 58 (2005) 237.
6. Li W., Zhao H., Teasdale P.T. and John R., *Analytical Chemistry*, 75 (2003) 2578.
7. J. Panther, P. Teasdale, W. Bennett and H. Zhao, *Environmental Science and Technology*, 44 (2010) 9419.

Disinfection By-Products in South East Queensland: Assessing Potential Effects of Transforming Disinfectants in the SEQ Water Grid

Farré, M.J.[1], King, H.[1], Keller, J.[1], Gernjak, W.[1], Knight, N.[2], Watson, K.[2], Shaw, G.[2], Leusch, F.D.L.[2], Sadler, R.[2], Birt, J.[3], Bartkow, M.[4] and Burrell, P.[5]

[1] Advanced Water Management Centre (AWMC), The University of Queensland, St Lucia, Queensland
[2] Smart Water Research Centre (SWRC), Griffith University, Gold Coast, Queensland
[3] Bond University, Gold Coast, Queensland
[4] Seqwater, Brisbane, Queensland
[5] SEQ Water Grid Manager, Brisbane, Queensland

Summary

The formation potential of disinfection by-products (DBPs) from chlorination and monochloramination of source water for South East Queensland (SEQ) drinking water was determined. Moreover, the effect on DBP formation of transforming disinfectants in the SEQ Water Grid was simulated at the bench scale. Samples were collected after coagulation and before disinfection at Molendinar, Capalaba and Mt Crosby water treatment plants (WTP). Samples were filtered at the laboratory to simulate sand filtration. Reverse osmosis permeate before disinfection was taken at Tugun desalination plant. DBPs measured were trihalomethanes (THMs), halonitromethanes (HNMs), haloacetonitriles (HANs), haloaldehydes (HAs), haloketones (HKs) and iodo-THMs (I-THMs).

In general, the highest concentration of DBPs formed was observed when disinfecting source water from Capalaba WTP with chlorine. Molendinar and Mt Crosby WTPs generated lower and similar concentration of DBPs, even though Mt Crosby showed a higher concentration of bromine containing DBPs. Desalinated water from Tugun generated extremely low concentration of DBPs as a result of the low total organic carbon (TOC) present in the water. Compared to chlorine, monochloramination generally resulted in lower concentrations of DBPs with the exception of 1,1-dichloropropanone (1,1DCP) and I-THMs. Chloramines used after chlorine do not generate additional DBPs to the ones previously generated during the chlorination, except for the 1,1DCP. However, chloramines used before chlorine alter natural organic matter (NOM) and affect the final DBPs formed compared to the application of chlorine alone. Some DBPs such as THMs are reduced when employing this strategy, particularly with Capalaba and Molendinar source water, while the final concentration of HAN, 1,1,1-trichloropropanone (1,1,1-TCP), chloral hydrate (CH) and trichloronitromethane (TCNM) increased in comparison to the use of chlorine alone.

A disinfection strategy needs to be discussed in order to account for unexpected DBPs being generated if disinfectants are alternated in the SEQ Water Grid.

Keywords
Chlorine, chloramines, DBP, distribution, residual disinfection, transformation.

Introduction

The Queensland Government's South East Queensland (SEQ) Water Grid is a network of treatment facilities and pipes that move multiple sources of water across the region for drinking water purposes. Water is transported in pipes that can be operated two-way to enable alleviation of localised water needs in a flexible way, drawing upon the regions supplies. It provides SEQ with numerous water sources, both climate dependant (dams) and rainfall and climate resilient (desalination and purified recycled water). The SEQ Water Grid transports water over long distances via the water distribution system in the presence of disinfectants. Disinfectants are added to ensure that the water is safe to drink from a microbiological perspective, preventing acute health implications. Disinfectants can, however, generate potentially hazardous disinfection by-products (DBPs) when mixed with organic matter present in drinking water sources. The presence of DBPs in drinking water is of concern as some DBPs have been linked to potential health effects after long exposure periods.

The SEQ Water Grid covers a number of regions which use different disinfection regimes - chlorination and chloramination. It has been shown that alternative disinfectants such as chloramines or ozone can generate different DBPs compared to those generated by chlorine (Richardson *et al.*, 2007). However, regulated DBPs have historically been selected as indicators of DBPs formed when disinfecting water with chlorine, although these capture only a fraction of all by-products of chlorination and often are not suitable indicators for the generation of DBPs associated with other disinfectants, which are often referred to as "emerging DBPs".

Chlorine reacts with naturally occurring organic matter and/or anthropogenic compounds to produce a mixture of DBPs such as trihalomethanes (THMs) and haloacetic acids (HAA). Regulated THMs include chloroform, bromodichloromethane (BDCM), dibromochloromethane (DBCM) and bromoform. Total THMs (TTHMs) refers to the sum of these four compounds. The USEPA regulates TTHMs at 80 µg/L (EPA, 1998), while the value for the Australian Drinking Water Guidelines is 250 µg/L (ADWG, 2004 and 2010). HAAs include nine compounds including iodine containing HAAs, the most common being dichloroacetic acid (DCAA) and trichloroacetic acid (TCAA). Other species, found generally at lower levels, are bromochloroacetic acid (BCAA), dibromoacetic acid (DBAA), monochloroacetic acid (MCAA) and monobromoacetic acid (MBAA). In the US, the sum of DCAA, TCAA, DBAA, MCAA and MBAA is commonly denoted as HAA5 and it is currently regulated at 60 µg/L as HAA5 (EPA, 1998). In Australia, only MCAA, DCAA and TCAA are regulated at 150, 100 and 100 µg/L (ADWG, 2004). Other DBPs that may be generated at lower concentrations are halogenated acetonitriles, chlorinated phenols, chlorinated furanones, halonitromethanes, cyanogen halides, haloketones and haloaldehydes. However, the sum of the halogenated DBPs measured as total organic halogens (TOX) identified so far when disinfecting water with chlorine accounts for only 50% of the total formed (Reckhow and Singer, 1984; Weinberg, 1999).

Chloramine DBPs include cyanogen halides formed at much higher concentration than with chlorine, halonitromethanes, N-nitrosamines such as N-nitrosodimethylamine (NDMA), dihaloacetonitriles, dihalogenated acetic acids, dihalogenated acetaldehydes and halogenated furanones among others (Krasner *et al.*, 2006; Yang *et al.*, 2007). Hua and Reckhow (2007) found that chloramines produced a higher percentage of unknown TOX than free chlorine (ie, 80% in comparison to the 50% for chlorination). Therefore, while the formation of DBPs by free chlorine is relatively well understood, much less is known about the formation of DBPs by means of chloramination and even less is known about the transformation of those DBPs when alternating disinfectants. As a result of the way the SEQ Water Grid system works, water of different qualities employing different disinfectants may be mixed, requiring the conversion of disinfectant from one form into another at conversion stations along the grid. The potential effects of this mixing process on the formation of DBPs is presently not well understood.

The aim of this work is to evaluate the potential for DBP formation of four different source waters in SEQ when being disinfected with chlorine or monochloramine (MCA). Moreover, the effect of converting from chlorine to chloramines and vice versa will be assessed.

Methodology

Source water from Molendinar water treatment plant (WTP), Capalaba WTP and Mt Crosby WTP was taken after coagulation and before disinfection in acid washed amber glass bottles and was kept cool until shipped to the laboratory. Samples were taken on the 31st of January 2011 (sample 1), 1st of March 2011 (sample 2) with the exception of Capalaba, taken on the 2nd March 2011 (sample 2), and 7th of April (sample 3). The water samples were filtered (Wathman 1, pore diameter 11 µm) to simulate sand filtration at the WTP. Combined reverse osmosis permeate was taken before final disinfection at Tugun desalination plant. Table 1 shows the average water characteristics for the waters after filtration in the lab including their chlorine and MCA demand.

Table 1. Average properties of the source waters selected for the study. Intervals correspond to the standard deviation between the values obtained from three independent sampling events. na=not analysed.

Source Water	TOC (mg/L)	NO_3^- (mg/L)	DON (mg/L)	Conductivity (µS.cm)	pH	SUVA (L/mg.m)	Chlorine Demand (mg/L)	MCA Demand (mg/L)
A) Molendinar (n=3)	3.1 ± 1.8	0.03 ± 0.02	0.25 ± 0.04	122 ± 46	6.7 ± 0.3	2.2± 1.2	3 ± 1	3 ± 0
B) Capalaba (n=3)	12.9 ± 4.8	0.45 ± 0.47	0.43 ± 0.27	167 ± 63	6.3 ± 0.2	1.2 ± 0.9	10 ± 1	6 ± 2
C) Mt Crosby (n=3)	2.64 ± 0.90	0.34 ± 0.06	0.28 ± 0.02	315 ± 59	6.5 ± 0.1	2.5 ± 0.6	5 ± 3	4 ± 2
D) Tugun desal (n=1)	<1.00	0.02	na	69	9.6	na	0	1

To evaluate the formation potential of DBPs, first the specific chlorine demand was determined for each of the four waters following Standard Methods 5710 (Clesceri *et al.*, 1998). To this aim, each buffered water was spiked with 100 mg/L of HOCl and was left in contact for four hours. After this time, free and total chlorine were measured and the chlorine demand was calculated by subtracting the residual chlorine measured from the 100 mg/L spiked concentration. From these data, the dosing for the formation potential tests were calculated to obtain a residual chlorine concentration similar to what would be found in a real application of drinking water disinfection. We targeted the residual chlorine at 2 mg/L and MCA at 3.5 mg/L to simulate SEQ Water Grid conditions. Two aliquots of 240 mL for each sample were buffered at pH 7 using 5 mL phosphate buffer (6.81 g potassium dihydrogen phosphate (anhydrous) and 1.17 g sodium hydroxide into 100 mL). Free chlorine (as sodium hypochlorite, NaOCl) was added to one aliquot and MCA was added to the other. Samples were kept headspace free in contact with the disinfectants for three and six days in the dark and at constant temperature (23±2^{o}C). After this time, the samples were quenched using ascorbic acid and stored headspace free in the fridge until extraction. Another batch of samples was used for the disinfectant conversion experiments. To this aim, chlorine was added to the samples containing MCA via breakpoint chlorination to obtain the desired residual dose. Ammonia was added to the chlorine samples topping up with MCA if necessary to obtain the required residual dose. After three days of contact time, the samples were quenched using ascorbic acid and stored headspace free in the fridge until extraction. The following DBPs were measured: trichloromethane (TCM), bromodichloromethane (BDCM), dibromochloromethane (DBCM), tribromomethane (TBM), dichloroiodomethane (DCIM), bromochloroiodomethane (BCIM), dibromoiodomethane (DBIM), chlorodiiodomethane (CDIM), bromodiiodomethane (BDIM), trichloroacetonitrile (TCAN), dichloroacetonitrile (DCAN), bromochloroacetonitrile (BCAN), dibromoacetonitrile (DBAN), chloral hydrate (CH), trichloronitromethane (TCNM), 1,1-dichloropropanone (1,1-DCP) and 1,1,1-trichloropropanone (1,1,1-TCP).

DBPs were liquid-liquid microextracted with methyl tert-butyl ether (MtBE) and analysed with an Agilent 7890A Series gas chromatograph (GC) with electron capture detector (ECD) using a DB-1 column of 30 m length x 0.25 mm inner diameter, 1.0 μm film thickness for quantification and a DB-5 column of 30 m length x 0.25 mm inner diameter for confirmation. The method followed the procedure described by Krasner *et al.* (2006). In this method, 30 mL of sample is acidified to pH 3.5 with 0.2 N sulphuric acid. 3 mL of MtBE with internal standard (1,2-dibromopropane 99% from Sigma-Aldrich) is added to the sample. Finally, 10 g of pre-baked sodium sulphate is added to each 30 ml sample and vortexed for one minute. After the sample has settled for 5 minutes, the organic phase is transferred to a vial for injection. A calibration curve was also included in each extraction batch to correct for recovery losses during the extraction. 2 μL samples were splitless injected into the GC at 200^{o}C. The initial temperature of the oven was 40^{o}C for 25 min. A ramp was then programmed at 5^{o}C/min to 145^{o}C. Temperature was finally increased progressively to 260^{o}C. The final run time was 64 min. Helium gas at 1 mL/min was used as carrier gas and nitrogen at 29 ml/min was used as make up gas. The lowest level for quantification was 0.10 μg/L for all the DBPs measured.

Results

DBP formation was measured after three and six days of contact time between source water and disinfectant to estimate the effect of this parameter (data not shown). From this data, it was seen that most of the DBPs were already formed after three days. Therefore, we compared herein the DBPs generated after four specific scenarios: (i) three days contact time with chlorine; (ii) three days contact time with MCA; (iii) three days contact time with chlorine followed by three days contact time with MCA; and finally (iv) three days contact time with MCA followed by three days contact time with chlorine.

THMs in conjunction with haloacetic acids (HAAs) are the most prevalent DBPs in drinking water, formed as a result of the reaction between chlorine and natural organic matter. Figure 1 shows the average formation of TCM, BDCM, DBCM and TBM following disinfection strategies i-iv for the four source waters selected for this study. Error bars represent the inter-day variability of the three sampling events. Individual data on the DBP formation for each sampling event (s1, s2 and s3) is plotted in the inset graph. Tugun desalinated water was only sampled once and therefore no inset graph or error bars are plotted. As mentioned earlier, the ADWG regulation for TTHMs in drinking water is 250 μg/L (ADWG, 2004).

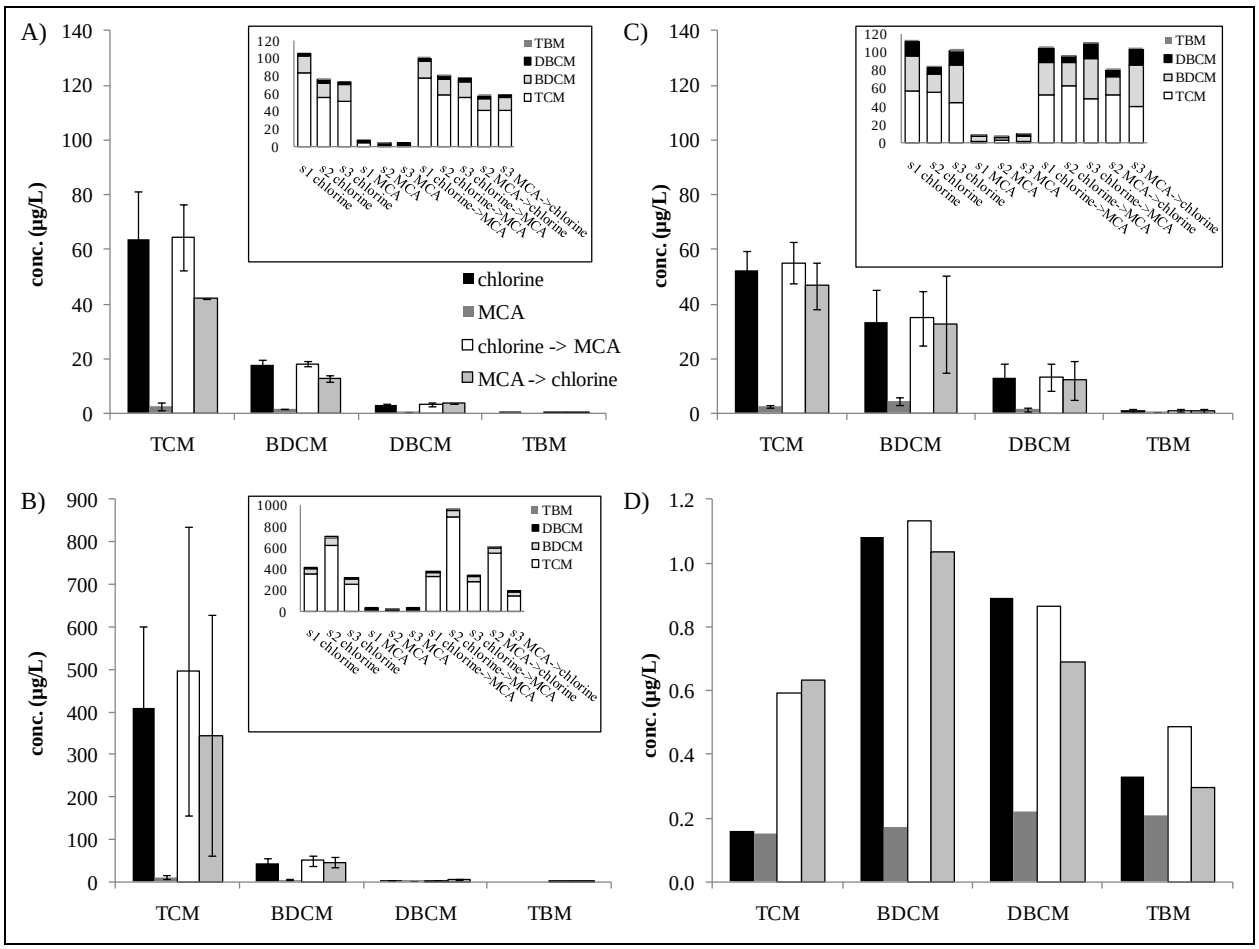


Figure 1. Trihalomethane formation potential of four different source waters for SEQ drinking water. pH=7, T=23±2°C. A=Molendinar, B=Capalaba, C=Mt Crosby, D=Tugun desalinated water. Disinfectant dose=disinfectant demand + 2 mg/L (chlorine) and/or 3.5 mg/L (monochloramine). n=3. Bars correspond to the standard deviation of three sampling events (individual results plotted in the inset graph). Samples were taken on the 31st of January 2011 (sample 1), 1st of March 2011 (sample 2) except for Capalaba, where the second sample was taken on the 2nd of March 2011, and 7th of April (sample 3).

In general, the highest concentration of THM formation was observed when disinfecting source water from Capalaba WTP (B) with chlorine. The high standard deviation is caused by the high formation of DBPs from water from the second sampling campaign. As an example, the concentration of TCM generated was 360, 630 and 250 µg/L when disinfecting Capalaba water with chlorine from sampling event 1, 2 and 3, respectively. This high concentration of THMs (in particular TCM) can most likely be attributed to the high total organic carbon (TOC) measured on the second sampling campaign (TOC=16.2 mg/L, SUVA=0.95 L/mg•m) since this is directly proportional to THMs formed during disinfection (Krasner *et al.*, 1994). More research is needed to understand the TOC variations at this specific WTP to be able to correlate this parameter with a high risk of THM formation. Molendinar (A) and Mt Crosby source waters (C) generated lower and similar concentration of DBPs, though Mt Crosby showed an increased concentration of bromine containing DBPs, which are in many cases more toxic than their chlorinated analogues. Mt Crosby WTP also showed a lower variability between sampling days. The Tugun desalinated water (D) generated extremely low concentrations of DBPs as a result of the low TOC present in the water, and therefore we did not extend the study on DBP formation using this specific source water. From Figure 1, we can also see that MCA generates significantly less THMs than chlorine, as expected (Krasner, 2009). Hua and Reckhow (2007) found that chloramines produced a higher percentage of unknown total organic halogens (TOX) than free chlorine. This means that because chloramines are much weaker oxidants than free chlorine, organic matter may be halogenated but organic molecules may not be broken down to the extent to generate THMs or other small DBPs. No differences were observed between the DBPs generated after disinfecting the waters with chlorine (i) and with chlorine followed by MCA (iii), except for the desalination plant. However, at this plant TCM was found at a concentration lower than 1 µg/L and this result could be due to an analytical artifact as analysing TCM at such low concentration is extremely difficult because of its common presence in solvents and the laboratory environment. On the other hand, a common trend for THMs reduction could be seen when using disinfection strategy (iv) (MCA followed by chlorine) to disinfect Molendinar and Capalaba source waters.

Figure 2 shows the average formation of haloacetonitriles (HAN) following disinfection strategies i-iv for the four source waters selected for this study. Bars represent the inter-day standard deviation of the three sampling campaigns. Individual data on the DBP formation for each sampling event (s1, s2 and s3) is plotted in the inset graph. Tugun desalinated water was only sampled once and therefore neither inset graph nor standard deviation are plotted. There is no guideline value in the ADWG for any of the HAN included in the study, however, the World Health Organisation (WHO) recommended values for DCAN and DBAN are 20 and 70 µg/L, respectively (WHO, 2011).

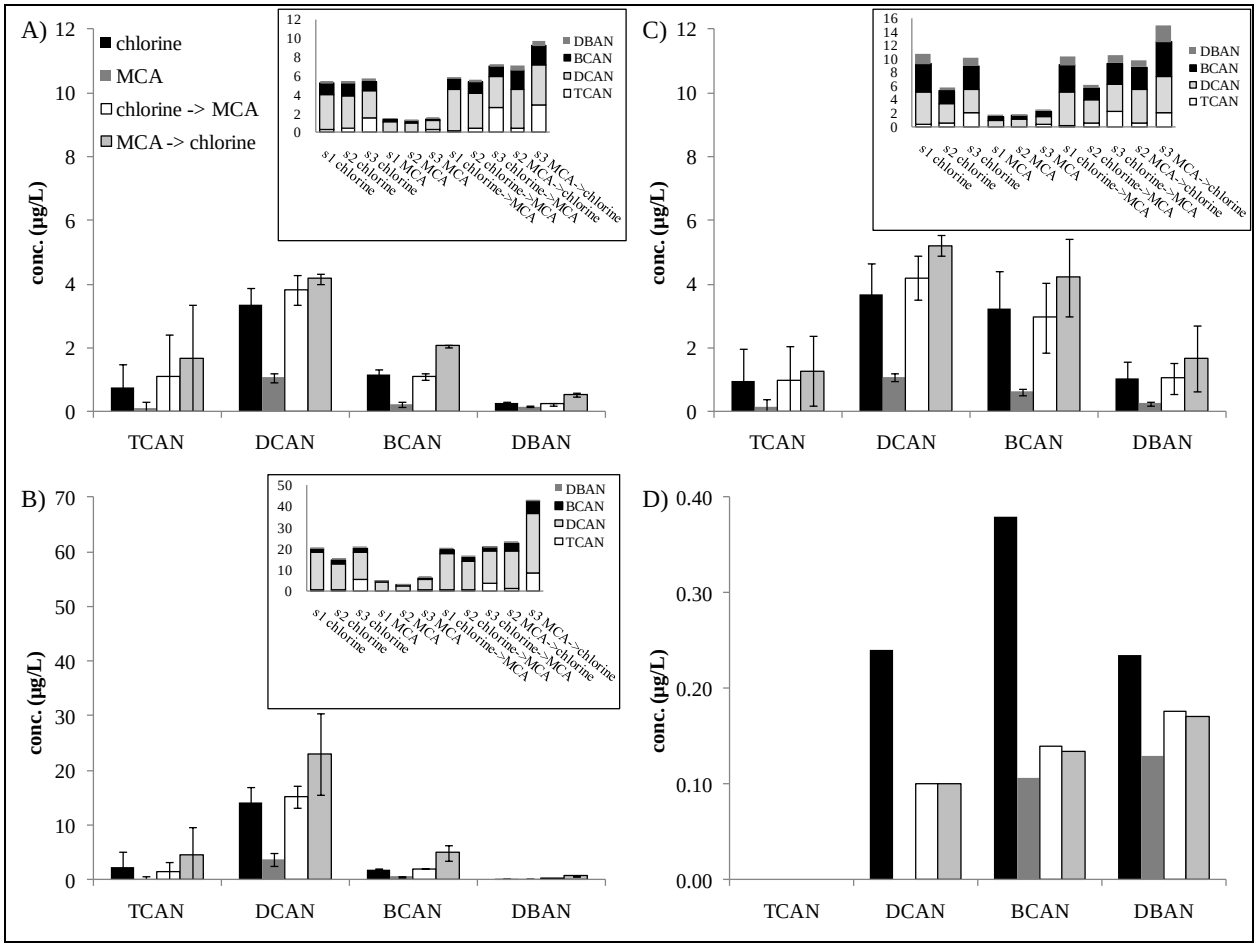


Figure 2. Haloacetonitrile formation potential of four different source waters for SEQ drinking water. pH=7, T=23±2°C. A=Molendinar, B=Capalaba, C=Mt Crosby, D=Tugun desalinated water. Disinfectant dose=disinfectant demand + 2 mg/L (chlorine) and/or 3.5 mg/L (monochloramine). n=3. Bars correspond to the standard deviation of three sampling events (individual results plotted in the inner graphs). Samples were taken on the 31st of January 2011 (sample 1), 1st of March 2011 (sample 2) with the exception of Capalaba, taken on the 2nd of March 2011, and 7th of April (sample 3).

Total HAN concentrations in Molendinar, Mt Crosby and the desalination plant were in line with the findings of Bougeard *et al.* (2010), who reported levels of HANs between approximately 0.023 and 5.5 µg/L when using formation potential tests. However, concentrations measured from Capalaba WTP source water were extremely high, in particular results obtained for DCAN, though this specific DBP has previously been the major HAN observed (Bougeard *et al.*, 2010). Dihalogenated HANs are reported to be more stable than the trihalogenated HANs (Peters *et al.*, 1990). In addition, TCAN can undergo base-catalysed hydrolysis at pH higher than 5.5 (here, the pH was 7.0) which is the likely explanation as to why it was rarely detected in this study (Croue and Reckhow, 1989).

Disinfecting with MCA rather than chlorine (ii), decreased the concentration of HANs by at least a factor of three. Hua and Reckhow (2007) also found that concentrations of HAN were reduced when using MCA and low concentrations of dihalogenated HANs (<1 µg/L) were formed. No significant differences were observed when comparing the DBPs generated after disinfecting the waters with chlorine and with chlorine followed by MCA (strategies i and iii in Figure 2), except for the desalinated water. However, most of the DBPs generated from desalinated water were measured close to the limit of detection LOD (0.1 µg/L). A common trend of HAN increase could be seen for the disinfection strategy iv (MCA followed by chlorine) in comparison to disinfection strategy i (chlorine) and iii (chlorine followed by MCA). Yang and co-authors investigated the origin of nitrogen in nitrogen-containing DBPs (N-DBPs), such as HAN, during chloramination with labeled ^{15}N-monochloramine. They found that nitrogen in N-DBPs originates from both NH_2Cl and organic-N compounds (Yang *et al.*, 2010). We could then suggest that the increase observed on HAN formation when disinfecting water with chloramines prior to chlorine

could be due to the nitrogen addition to NOM that is further oxidised when adding the chlorine to generate HAN. Therefore, the final concentration of HAN generated increases because there is more nitrogenated organic matter.

Figure 3 shows the average formation of haloketones (HK) following disinfection strategies i-iv for the four source waters selected for this study. Bars represent the inter-day standard deviation of three sampling events.

Disinfection experiments of Capalaba source water taken during the second sampling generated a very high concentration (>100 µg/L) of 1,1,1-TCP in all instances except when using chloramines (data not included in the graph). Sample 2 also generated very high concentrations of THMs, in particular TCM (see Figure 1). More research is needed to understand the origin of those events.

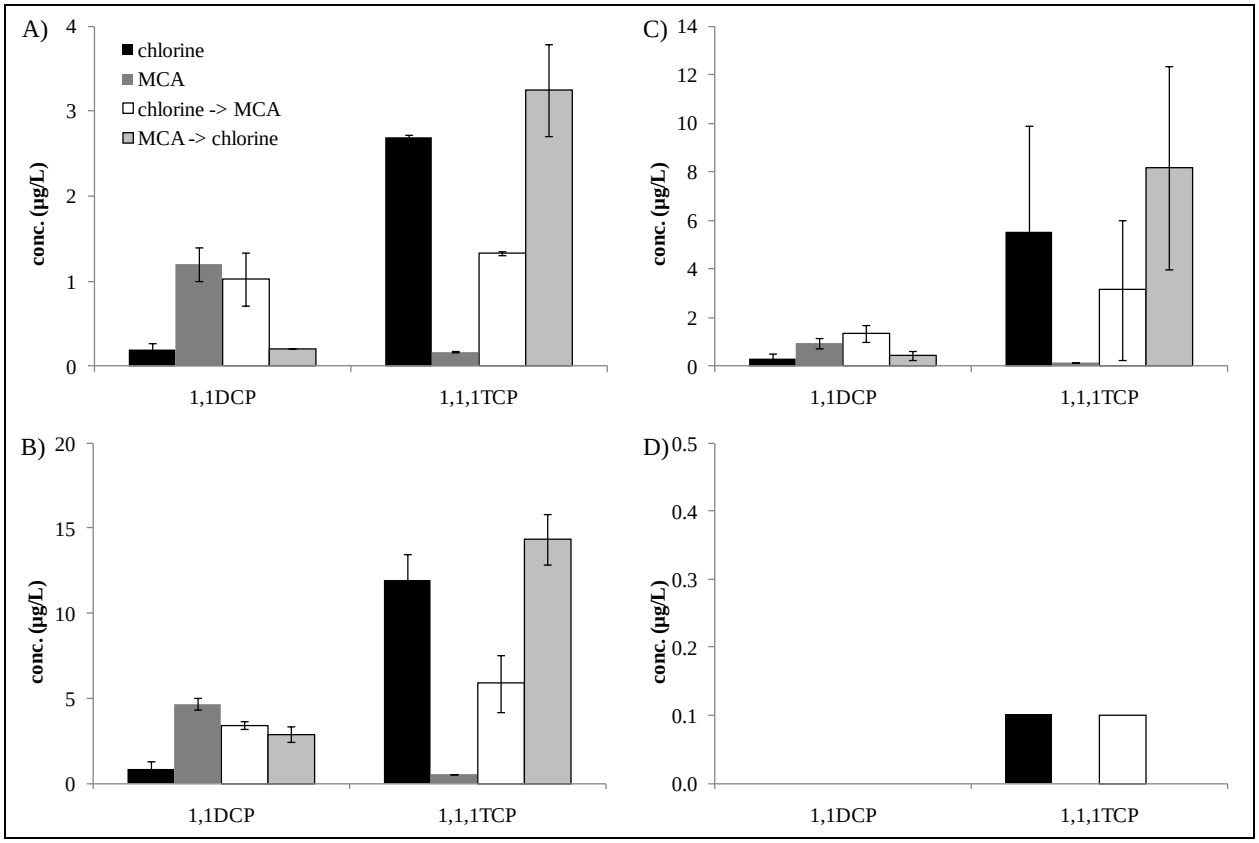


Figure 3. Haloketone formation potential of four different source waters for SEQ drinking water. pH=7, T=23±2°C. A=Molendinar, B=Capalaba, C=Mt Crosby, D=Tugun desalinated water. Disinfectant dose=disinfectant demand + 2 mg/L (chlorine) and/or 3.5 mg/L (monochloramine). n=3. Bars correspond to the standard deviation of the values of disinfection experiments of three samples from different sampling events.

There is no guideline value in the ADWG for any of the HK included in the study. However, 1,1,1-TCP is of particular interest for distribution systems since, even though it is not regulated, the hydrolysis of 1,1,1-TCP ($k=1.82x10^{-1}$ h^{-1} at pH7 (Nikolaou *et al.*, 2001)) can generate the same molar concentration of TCM and augment TCM concentration during transport. Hydrolysis may explain as well the lower concentration of 1,1,1-TCP in strategy iii (chlorine → MCA) compared to chlorine alone (i) and explain also the slight increases of TCM in Figure 1 in strategy iii compared to i. 1,1-DCP is the only DBP that is generated at higher concentration in the presence of MCA. These differences in the formation of 1,1-DCP and 1,1,1-TCP with chlorination and chloramination can be partially explained by a simplified model developed by Reckhow and Singer (1984). In this model, chlorination of fulvic acid solutions leads to the formation of intermediate by-products, such as 1,1-DCP ($CHCl_2–CO–CH_3$). 1,1-DCP can be further oxidised by chlorine and form 1,1,1-TCP ($CCl_3–CO–CH_3$). This model reveals that further chlorine attack and hydrolysis are essential for the formation of 1,1,1-TCP. However, MCA is not able to provide further chlorine substitution and hydrolysis, as 1,1-DCP is found to be very stable in the presence of MCA. Similarly, it is expected that other intermediate by-products ($CHCl_2–CO–R$, $R≠–OH$) may not easily be substituted by chlorine to form $CCl_3–CO–R$ in monochloramination. Thus, in a chloraminated solution, the concentration of 1,1-DCP is high but almost no 1,1,1-TCP is detected. However, from our experiments we observed that further 1,1-DCP could be formed even after prior chlorination (strategy iii). This could mean that maybe different precursors are involved in the generation of the two propanones. We are currently working to evaluate this hypothesis.

Chloral hydrate (CH) is included in the ADWG and its guideline value is 20 µg/L (ADWG, 2004 and 2010). The WHO recommends a guideline value of 10 µg/L (WHO, 2011). Besides THMs and HAAs, chloral hydrate (trichloroacetaldehyde, CH) is the next most prevalent DBP in chlorinated drinking water, formed as a result of the reaction between chlorine and NOM. Figure 4 shows the formation of CH and trichloronitromethane (TCNM) following disinfection strategies i-iv for the four source waters selected for this study. Bars represent the standard deviation of the values obtained from three samples from three independent sampling events.

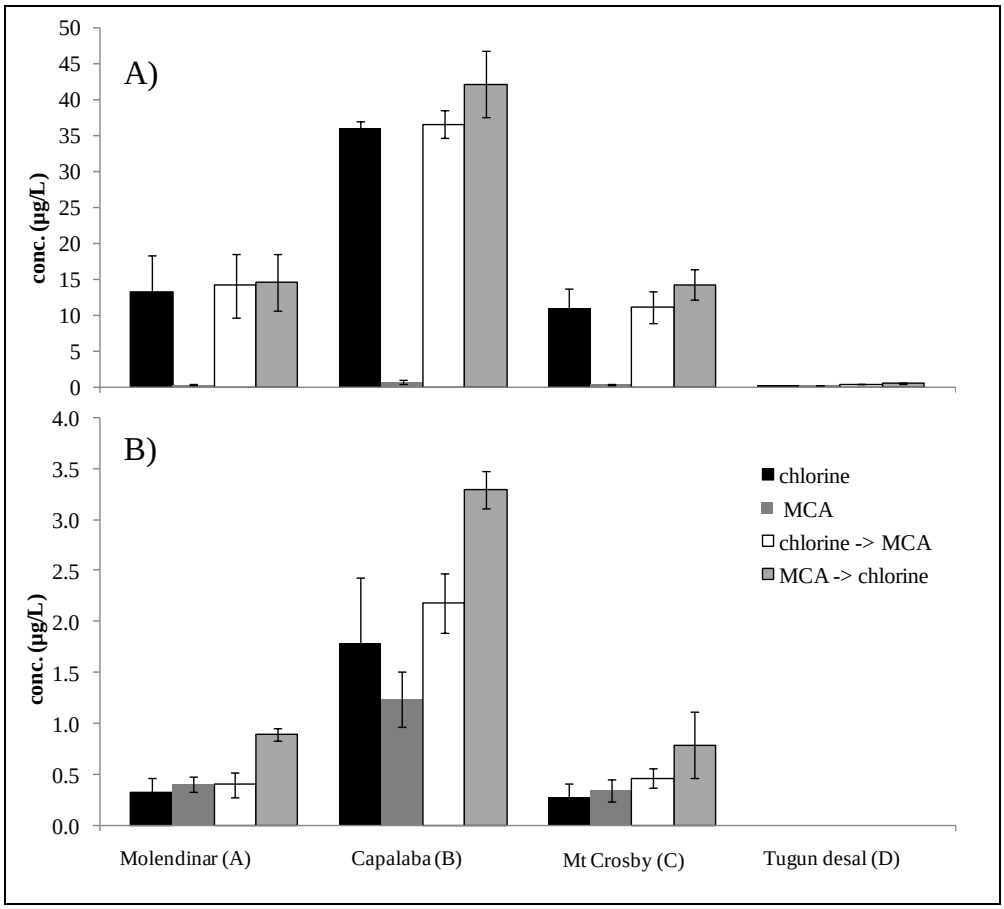


Figure 4. Chloral hydrate (A) and trichloromethane (B) formation potential of four different source waters for SEQ drinking water. pH=7, T=23±2°C. Disinfectant dose=disinfectant demand + 2 mg/L (chlorine) and/or 3.5 mg/L (monochloramine). n=3. Bars correspond to the standard deviation of the values of disinfection experiments of three samples from different sampling events.

As seen in Figure 4, MCA did not generate CH, while concentrations formed during chlorine disinfection, chlorine followed by MCA or MCA followed by chlorine where relatively high. Dabrowska and Nawrocki (2009) studied the effect of contact time on the formation of CH. They observed that the reaction of chlorine with organic matter takes place as long as chlorine is available in the water. The concentration of chloral hydrate may continuously increase in the water supply system. However, Koudjonou *et al.* (2008) observed upon conducting the controlled laboratory chlorination of acetaldehyde under typical drinking water conditions (pH 6.7, 7.6 and 8.8, and temperature 4 °C and 21 °C) the formation of chloral hydrate (CH) increased with contact time (0–10 days). However, at increased pH and temperature, CH reached maximum levels and subsequently broke down partially to chloroform and other unidentified compounds. TCNM seems to be the DBP that is most affected by the transformation of disinfectants since pre-chlorination followed by chlorine disinfection increased its formation by at least a factor of two in all instances. The increase observed for TCNM and other DBP formation could be explained by the alteration of NOM by pre-chloramination into a more reactive form.

Finally, the formation of I-THMs was also included in this study. I-THMs may be formed when disinfecting water with chloramines in the presence of iodine and these DBPs have been reported to be much more toxic than their brominated or chlorinated counterparts (Richardson *et al.*, 2008). Formation of DCIM was observed when disinfecting with MCA in Capalaba and Mt Crosby source waters at an average concentration of 0.61 ± 0.14 and 0.41 ± 0.06 µg/L, respectively. Theses concentration remained constant when transforming MCA to chlorine. Disinfection of Mt Crosby source water with MCA also generated BCIM and DBIM at 0.72 ± 0.54 µg/L and 0.17± 0.04 µg/L, respectively showing again the shift to brominated species observed in that WTP.

Conclusions

In general, the highest concentration of DBPs formed was observed when disinfecting source water from Capalaba WTP with chlorine. Molendinar and Mt Crosby WTPs generated lower and similar concentration of DBPs, though Mt Crosby showed a higher concentration of bromine containing DBPs. The Tugun desalinated water generated extremely low concentrations of DBPs as a result of the low TOC present in the water.

Compared to chlorine, MCA generally resulted in lower concentrations of DBPs with the exception of 1,1DCP and I-THMs.

Chloramines used after chlorine do not generate additional DBPs to the ones previously generated during chlorination, with the exception of 1,1DCP.

Chloramines used before chlorine alter NOM and affect the final DBPs formed compared to application of chlorine alone. Some DBPs such as THMs seem to be reduced when employing this strategy at Capalaba and Molendinar, while the final concentration of HAN, 1,1,1-TCP, CH and TCNM is increased in comparison to the use of chlorine alone.

Acknowledgements

The authors want to acknowledge Veolia Water Australia and Seqwater for giving access to the plants and helping with the sampling.

References

ADWG, 2004. Australian drinking water guidelines. in: Council, N.H.a.M.R. (Ed.), Camberra.

Bougeard, C.M.M., Goslan, E.H., Jefferson, B., Parsons, S.A., 2010. Comparison of the disinfection by-product formation potential of treated waters exposed to chlorine and monochloramine. Water Research 44, 729-740.

Clesceri, L.S., Greenberg, A.E., Eaton, A.D., 1998. Standard methods for the examination of water and wastewater. Method 5710 B. American Public Health Association, American Water Works Association and Water Environment Federation, Washington DC.

Croue, J.P., Reckhow, D.A., 1989. Destruction of chlorination byproducts with sulfite. ESandT Contents 23, 1412-1419.

Dabrowska, A., Nawrocki, J., 2009. Controversies about the occurrence of chloral hydrate in drinking water. Water Research 43, 2201-2208.

EPA, U.S., 1998. National primary drinking water regulations: disinfectants and disinfection byproducts; final rule, Fed. Reg. 63.

Hua, G., Reckhow, D.A., 2007. Comparison of disinfection byproduct formation from chlorine and alternative disinfectants. Water Research 41, 1667-1678.

Koudjonou, B., LeBel, G.L., Dabeka, L., 2008. Formation of halogenated acetaldehydes, and occurrence in Canadian drinking water. Chemosphere 72, 875-881.

Krasner, S.W., 2009. The formation and control of emerging disinfection by-products of health concern. Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences 367, 4077-4095.

Krasner, S.W., Sclimenti, M.J., Means, E.G., 1994. Quality degradation. Implications for DBP formation. Journal / American Water Works Association 86, 34-47.

Krasner, S.W., Weinberg, H.S., Richardson, S.D., Pastor, S.J., Chinn, R., Sclimenti, M.J., Onstad, G.D., Thruston Jr, A.D., 2006. Occurrence of a new generation of disinfection byproducts. Environmental Science and Technology 40, 7175-7185.

Nikolaou, A.D., Lekkas, T.D., Kostopoulou, M.N., Golfinopoulos, S.K., 2001. Investigation of the behaviour of haloketones in water samples. Chemosphere 44, 907-912.

Peters, R.J.B., De Leer, E.W.B., De Galan, L., 1990. Dihaloacetonitriles in Dutch drinking waters. Water Research 24, 797-800.

Reckhow, D.A., Singer, P.C., 1984. The removal of organic halide precursors by preozonation and alum coagulation. Journal / American Water Works Association 76, 151-157.

Richardson, S.D., Fasano, F., Ellington, J.J., Crumley, F.G., Buettner, K.M., Evans, J.J., Blount, B.C., Silva, L.K., Waite, T.J., Luther, G.W., McKague, A.B., Miltner, R.J., Wagner, E.D., Plewa, M.J., 2008. Occurrence and mammalian cell toxicity of iodinated disinfection byproducts in drinking water. Environmental Science and Technology 42, 8330-8338.

Richardson, S.D., Plewa, M.J., Wagner, E.D., Schoeny, R., DeMarini, D.M., 2007. Occurrence, genotoxicity, and carcinogenicity of regulated and emerging disinfection by-products in drinking water: A review and roadmap for research. Mutation Research - Reviews in Mutation Research 636, 178-242.

Weinberg, H., 1999. Disinfection byproducts in drinking water: The analytical challenge. Analytical Chemistry 71, 801A-808A.

WHO, 2011. Guidelines for drinking water quality. Fourth eddition. in: Organisation, W.H. (Ed.), Geneva.

Yang, X., Fan, C., Shang, C., Zhao, Q., 2010. Nitrogenous disinfection byproducts formation and nitrogen origin exploration during chloramination of nitrogenous organic compounds. Water Research 44, 2691-2702.

Yang, X., Shang, C., Westerhoff, P., 2007. Factors affecting formation of haloacetonitriles, haloketones, chloropicrin and cyanogen halides during chloramination. Water Research 41, 1193-1200.

How Does Natural Organic Matter Removal Impact on Disinfection By-Product Formation?

Knight, N.[1], Watson, K.[1], Farré, M.J.[2], King, H.[2], Shaw, G.[1], Leusch, F.D.L.[1], Sadler, R.[1], Birt, J.[3], Bartkow, M.[4], Keller, J.[2], Gernjak, W.[2] and Burrell, P.[5]

[1] Smart Water Research Centre (SWRC), Griffith University, Gold Coast, Queensland
[2] Advanced Water Management Centre (AWMC), The University of Queensland, St Lucia, Queensland
[3] Bond University, Gold Coast, Queensland
[4] Seqwater, Brisbane, Queensland
[5] SEQ Water Grid Manager, Brisbane, Queensland

Summary

Natural organic matter (NOM) and halides (Br^- and I^-) present in source waters react with disinfectants to form disinfection by-products (DBPs). Current coagulation and adsorption techniques used in drinking water treatment have good NOM removal capacity, but have low halide removal capacity. This is due primarily to the presence of competing anions in solution, such as bicarbonate, and components of NOM itself. This project aims to understand the relationship between the NOM:halide ratio and specific DBP formation, since removing NOM alone in a high bromide/iodide source water may lead to high brominated/iodinated DBP formation. In addition, this work will study the effectiveness of several adsorptive techniques on NOM and halide removal under controlled conditions. This will determine the conditions that are advantageous for use in DBP precursor removal and lowering subsequent DBP formation upon disinfection.

Keywords
DBP precursors, drinking water, enhanced coagulation, NOM.

Background

Disinfection of drinking water has played a significant role in the protection of public health against water-borne disease. Chemical disinfectants such as chlorine and chloramines, although effective in removing pathogenic organisms, can react with organic compounds and halides in source waters to form DBPs, which may be harmful after long-term exposure (Richardson, 2007). This long-term risk is small compared to the short-term risk of exposure to drinking water that has not been disinfected. NOM is present in all source waters and is comprised of a complex mixture of many individual chemicals, including humic acids, fulvic acids, carbohydrates and amino acids arising primarily from plant detritus (Chow, 2008). Different components of NOM may have a different propensity to form DBPs upon exposure to disinfectants, assuming the same availability of other precursors (halides, specifically Br^- and I^-). Halides appear at elevated concentrations in waters that have been subjected to salt-water intrusion (Richardson, 2007). Rivers with naturally low flow (eg, in low rainfall catchments) tend to have higher salinity than high-flow rivers (Nielson, 2003). Freshwater ecosystems in Australia are becoming increasingly threatened by salinity due to rising saline groundwater and reduced frequency of high-flow events, therefore, many Australian drinking water sources may be expected to have high halide concentrations (Nielson, 2003). Bromide and iodide, although present in much lower concentrations than chloride in a given source water, have the potential to form DBPs that are, in general, more problematic from a public health perspective than their chlorinated analogues (Richardson, 2007).

There are three general strategies for the control of DBPs in drinking water (Zwiener, 2006). These are: controlling disinfection parameters to minimise DBP formation (eg, disinfectant dose/disinfection method management); removing DBPs after disinfection but prior to distribution (eg, aeration to reduce trihalomethanes (THMs)); and removing DBP precursors from the source water prior to disinfection and distribution. This project is focussed on the third strategy – DBP precursor removal. Enhanced coagulation using alum is frequently employed in South East Queensland (SEQ) drinking water treatment plants in order to reduce NOM to acceptable concentrations, generally resulting in lower DBP formation upon disinfection. Although enhanced coagulation removes a significant amount of NOM present in source waters, it is most suitable for removal of hydrophobic, high molecular weight NOM, and this technique does not remove halides (Allpike, 2005). For source waters that have high concentrations of halides, or low molecular weight/high water solubility NOM, significant concentrations of DBP precursors may remain after enhanced coagulation and may form DBPs upon disinfection. A number of adsorptive techniques are commercially available for the removal of NOM from drinking waters, although fewer adsorptive techniques are available for the effective removal of halides in complex matrices such as drinking water sources. Membrane techniques such as reverse osmosis and electrochemical techniques such as capacitive deionisation are also available for the removal of NOM and halides from drinking water sources. Limited information on the effectiveness of these techniques using SEQ source waters is available. This project aims to understand the relationship between NOM concentration, halide concentration and DBP formation, as well characterising NOM from selected SEQ water sources. DBP precursor

(NOM, Br^- and I^-) removal by adsorptive techniques will also be investigated, and understood in terms of impact on regulated and emerging DBP formation.

Results and Discussion

Previous work at the Smart Water Research Centre (Knight, 2010) has shown that in a naturally high-bromide source water, lowering the NOM concentration by alum coagulation led to an increase in formation of brominated THMs and a decrease in concentration of chlorinated THMs. The overall result was an increase in total THM concentration, despite the removal of significant NOM precursors. This is important because the highly-brominated THMs are, in general, considered to pose a more significant public health risk than the highly-chlorinated THMs. The distribution of individual THMs formed ($CHCl_3$, $CHBrCl_2$, $CHBr_2Cl$, $CHBr_3$) is controlled by the NOM:bromide ratio, and preliminary results obtained during this project contribute toward understanding the ratios at which formation of the more highly brominated DBPs preferentially occur, rather than their chlorinated analogues. Early work studying the relationship between NOM concentration, halide concentration (Br^- and I^-) and pH has confirmed the general trend that, for a given bromide concentration, brominated DBP concentration increases with decreasing NOM concentration. The data obtained in this study will be modelled using multivariate regression analysis in order to quantify this relationship. In particular, we plan to relate the NOM:halide ratio to the specific DBPs formed, and their concentrations. An understanding of this relationship is important in controlling DBP formation through the use of enhanced coagulation, since NOM removal alone may be unable to sufficiently lower DBP formation under high halide conditions.

An extensive review of the current body of literature has revealed that, although there are excellent membrane and electrochemical techniques for halide and NOM removal from source waters, adsorptive techniques currently available for application in halide removal are limited in their effectiveness in raw waters. That is, halide removal by adsorption is difficult because naturally occurring anions (such as bicarbonate) and NOM compete for adsorption sites on many of the commercially available products. Although excellent NOM removals can be attained by using an adsorbent such as MIEX® resin or activated carbon after enhanced coagulation, the NOM and co-anion concentrations at which these techniques become ineffective for halide reduction is not well understood, and this is addressed within this project. This study examines the halide and NOM adsorption capacity of granular activated carbon, silver-impregnated activated carbon and MIEX® resin in synthetic water matrices which have undergone enhanced coagulation. The NOM concentration at which these techniques become ineffective for halide removal will be determined. The capacity of each technique to remove the fraction of NOM that is not amenable to enhanced coagulation will be concurrently assessed, and monitored by size-exclusion chromatography (SEC) and fluorescence spectroscopy (EEM). After each treatment process, the DBP formation potential will also be assessed and related to the NOM and halide concentrations, and the NOM:halide ratio. Adsorptive techniques for DBP precursor removal are the focus of this research project due to the relative ease of applicability and economy of these methods, relative to many other available techniques.

Natural Organic Matter Isolation Methodology

Understanding the relationship between NOM and halide concentration in terms of their impact on DBP formation requires the isolation of the NOM of the source waters of interest, so that the NOM concentration can be controlled in subsequent experiments using a synthetic water. This synthetic water has known concentrations of salts, alkalinity, conductivity, pH, and concentrations of DBP precursors (bromide, iodide and NOM). Also, one of the NOM characterisation techniques employed during this project is solid state ^{13}C-NMR, for which the NOM sample needs to be isolated as a solid. In order to isolate the water-soluble NOM as a solid sample from raw water, 1,000 L of raw water was collected from Molendinar water treatment plant (sourced from Hinze Dam), then filtered through a series of polypropylene particle filters (10 µm, 5 µm, 1 µm) (Figure 1). After this, the water was passed through a DOW Marathon-C cation exchange resin to remove hardness (Ca^{2+}, Mg^{2+}) and other cations. By running the water through the cation exchanger in the H^+ state (rather than the Na^+ state), essentially all cations are removed, rather than just Ca^{2+} and Mg^{2+}. After this point, the 1,000 L of filtered water was passed through a final polypropylene filter (0.5 µm) before being repeatedly cycled through two reverse osmosis (RO) membranes, that were plumbed such that the filtrate (pure water) was discarded and the retentate (NOM containing water) was retained and cycled through the RO system until the concentrate was reduced to approximately 40 L. This concentrate was then reduced further by reduced-pressure distillation at a maximum of 40 °C to avoid degradation of the sample, then the final volume (10 L) was freeze-dried to give a solid NOM isolate (several grams). The low dissolved organic carbon concentration of this source water (~3 mg/L) necessitated the use of a large volume of raw water in this case.

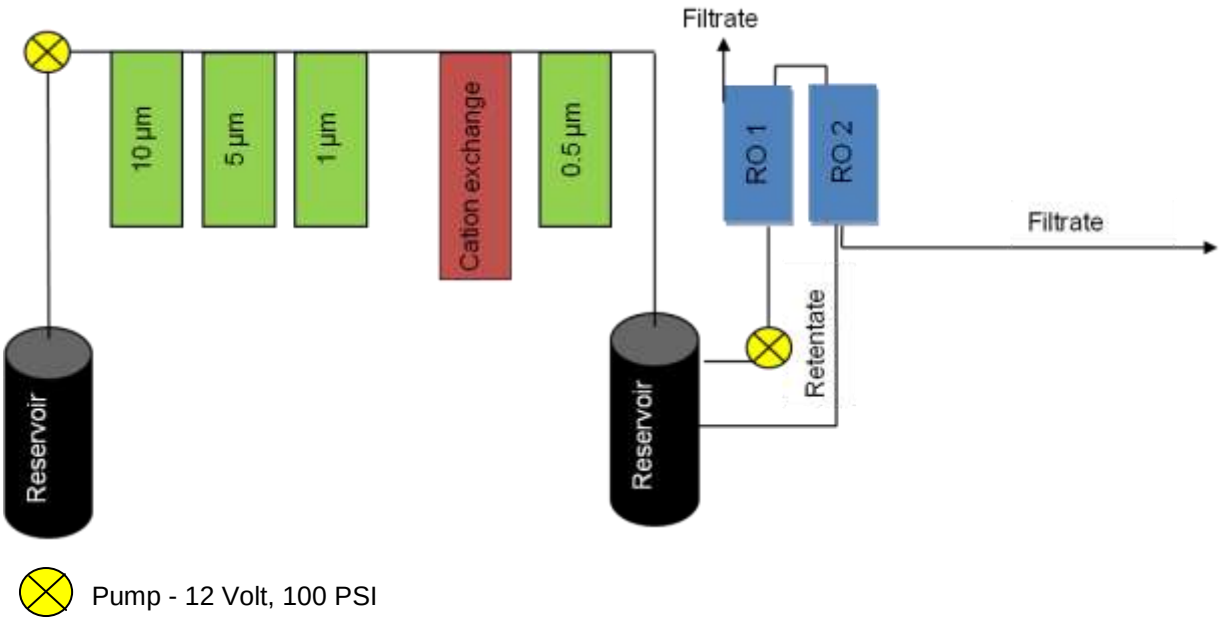


Figure 1. Schematic showing the NOM isolation process, starting with 1,000 L of raw water and resulting in 40 L of concentrate which is then further reduced by vacuum distillation, then freeze-dried to yield several grams of solid NOM.

Conclusions

Based on preliminary data, NOM removal from salinity-impacted source waters (containing high bromide and/or iodide) should be undertaken with an awareness that this may lead to high brominated DBP concentrations. Understanding the NOM:bromide ratio of a particular source water before and after coagulation may enable optimisation of water treatment to minimise DBP formation.

Halide adsorption methods are adversely impacted by the presence of competing anions (such as bicarbonate) and NOM occurring in natural waters. This is an important problem to be aware of in treating high halide source waters, due to the risk of high brominated and/or iodinated DBP formation. NOM isolation has been successfully achieved by a combination of filtration, cation exchange, reverse osmosis, vacuum distillation and freeze-drying, and this enables NOM characterisation in terms of determining its chemical 'fingerprint' and its relationship to DBP formation.

The significance of this study is in the development of an understanding of how the interrelationships between NOM, bromide and iodide impact on DBP formation upon disinfection, and how adsorptive methods and enhanced coagulation can be best used to give rise to drinking water with predictable, low DBP formation. This is in the interests of the continued protection of public health, in the provision of safe drinking water to SEQ residents.

Acknowledgements

The authors would like to acknowledge Prof. Howard Weinberg from the University of North Carolina for advice on the design of the NOM isolation system illustrated. The Urban Water Security Research Alliance and Water Quality Research Australia are also gratefully acknowledged for provision of research funding.

References

Allpike, B. P., A. Heitz, et al. (2005). "Size Exclusion Chromatography To Characterise DOC Removal in Drinking Water Treatment." Environmental Science and Technology **39**(7): 2334-2342.

Chow, C. W. K., R. Fabris, et al. (2008). "Assessing Natural Organic Matter Treatability Using High Performance Size Exclusion Chromatography." Environmental Science and Technology **42**(17): 6683-6689.

Knight, N., Shaw, G., Sadler, R., Wickramasinghe, W. (2010). Disinfection by-product formation and minimisation in South East Queensland drinking water, Urban Water Security Research Alliance.

Nielson, D. L., Brock, M. A., Rees, G. N., Baldwin, D. S. (2003). "Effects of increasing salinity on freshwater ecosystems in Australia." Australian Journal of Botany **51**: 655-665.

Richardson, S. D., M. J. Plewa, et al. (2007). "Occurrence, genotoxicity, and carcinogenicity of regulated and emerging disinfection by-products in drinking water: A review and roadmap for research." Mutation Research-Reviews in Mutation Research **636**(1-3): 178-242.

Zwiener, C. (2006). Trihalomethanes (THMs), Haloacetic Acids (HAAs), and Emerging Disinfection By-products in Drinking Water. Organic Pollutants in the Water Cycle, Wiley-VCH Verlag GmbH and Co. KGaA**:** 251-286.

Prudent and Efficient: The Use of Extended Cost-Effectiveness Analysis for Total Water Cycle Management Plans

Hall, M. R.
CSIRO Ecosystem Sciences. Ecosciences Precinct, Dutton Park, Queensland

Summary

Extended Cost Effective Analysis (CEA) is a method to integrate externality costs with project costs. This approach supports 'prudent and efficient' objectives of organisations such as the Queensland Competition Authority (QCA, 2010; Queensland Government) by identifying the most cost effective way to meet both service and environmental objectives. The method of Extended Cost Effectiveness has been proposed by the National Water Initiative for including externalities for full cost recovery (DEWHA, 2010) as well as in Queensland guidelines for Total Water Cycle Management Plans (TWCMP) for project evaluation (Hurikino, Lutton, and Eadie, 2010; WaterByDesign, 2010b). The paper presents the methodology and initial results for application to Moreton Bay Regional Council TWCMP.

Keywords
Extended Cost Effectiveness Analysis, Total Water Cycle Management, pollution mitigation costs, marginal cost curve, healthy waterways.

Introduction

The need to consider cost effectiveness in Total Water Cycle Management is highlighted by recent and anticipated expenditure to meet level of service and environmental objectives. In response to the recent drought, the Queensland Government has current and committed projects of $9 billion to ensure a level of service for urban water users in SEQ (QWC, 2008). In response to deteriorating aquatic ecosystems, the Government has spent approximately $400 million for wastewater treatment plant upgrades, $60 million for stormwater quality improvements and $18 million for waterway management and restoration (Bunn, Abal, Greenfield, Tarte, and Saxton, 2008). Yet despite this expenditure, the environmental conditions continue to degrade in Moreton Bay and estuaries (EHMP, 2009).

The expenditure required to meet environmental objectives is uncertain, as is the focus on point source or catchment based initiatives. For example, it was estimated in 2005 that the cost for meeting Environmental Values and Water Quality Objectives for a South East Queensland (SEQ) population in 2026 was $272 million for wastewater treatment plant upgrades over 5 years and $95 million for rural diffuse runoff management (Driml, O'Sullivan, and Hanna, 2005) - p5. In 2008, the expenditure on wastewater treatment plant upgrades was exceeded while only $18 million was spent on waterway management and restoration (QWC, 2008). In comparison, to meet the objectives of the *2007-2012 SEQ Healthy Waterways Strategy* (SEQHWP, 2007), it was estimated that approximately $350-500 million was required for riparian restoration, channel stabilisation and best practice land management in rural areas (Bunn et al., 2008). There is a large disparity between estimates and actual expenditure as well as differing focuses on point and diffuse source pollution. This raises the question of what is the most cost effective way to meet environmental objectives. To complicate matters further, an initial review of SEQ estimates of cost effectiveness for pollution mitigation suggest a wide range of estimates (EPA, 2007b) (EPA, 2008) (Rolfe, Donaghy, Alam, O'Dea, and Miles, 2005).

Cost effectiveness is an important measure of project viability and efficiency. However, the focus on traditional projects costs may result in project evaluation that is neither 'prudent' (needed) or efficient (cost effective) (QCA, 2010; Queensland Government). It is argued that environmental costs also need to be considered to provide a greater coverage of project costs. For example, a project that results in more pollution will require mitigation measures to meet environmental objectives and this may be the deciding factor between two competing options. Consideration of a range of pollution mitigation measures can also identify the most cost effective way to meet environmental targets across a range of projects. For example, it may be more cost effective to mitigate pollution in another part of the catchment than directly on the project under consideration. The use of a 'marginal cost pollution mitigation curve' can identify the most cost effective way to meet regional policy objectives.

Method

Extended Cost Effectiveness Analysis (CEA) was adopted as the framework for assessing the performance of options. This method was noted as being suitable for capturing sustainability issues of sub-regional total water cycle planning, such as water supply conservation and water supply augmentation (Hurikino et al., 2010) – p12. Extended CEA assumes previously agreed objectives, such as the level of service and environmental performance, and allows options to be ranked and compared on a least (net) cost basis (Fane, Blackburn, and Choong, 2010) – p12.

The Engineering Cost Method (IDC, 2003) - p40, (Standards Australia, 1999) - p14 was used to estimate the value of pollution. This method uses pollution mitigation costs, or engineering costs, to provide a measure of the pollutant value. This approach is considered valid for estimating environmental value if: (i) the engineered system provides functions that are equivalent in quality and magnitude to the natural function; (ii) the human-engineered system is the least cost alternative way of performing this function; and (iii) individuals in aggregate would in fact be willing to incur these costs if the natural function were no longer available (Bockstael, Freeman, Kopp, Portney, and Smith, 2000). The method satisfies the first condition by focusing on the mitigation of a physical quantity of pollution entering the catchment. The pollution is also considered in context of the sensitivity and health of the receiving environment. Conditions (ii) and (iii) are addressed by considering a marginal cost pollution mitigation curve. The cost curve identifies the least cost alternatives as well as explores the relationship of the cost to the objectives. As a starting point, the existing 'duty of care' for the environment (Bowers and Young, 2000) – p12 was adopted based upon Government water quality objectives, planning and legislation. Pollution mitigation cost curves provide a transparent method for costing pollution mitigation and were developed for the proposed nutrient trading scheme in Moreton Bay (BDA Group, 2005) – p87 as well as carbon trading schemes around the world (McKinsey and Co, 2008). Figure 1 illustrates how the least cost options are identified. It also illustrates that the value of pollution is responsive to pollution reduction targets and can be used to explore the willingness to incur these costs.

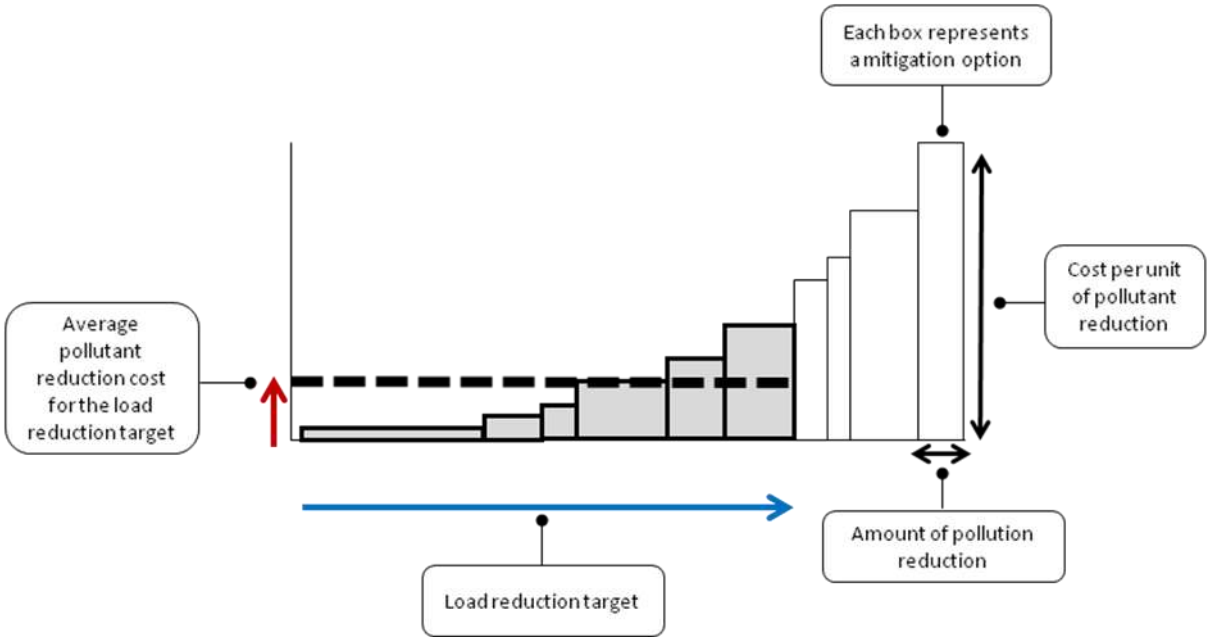
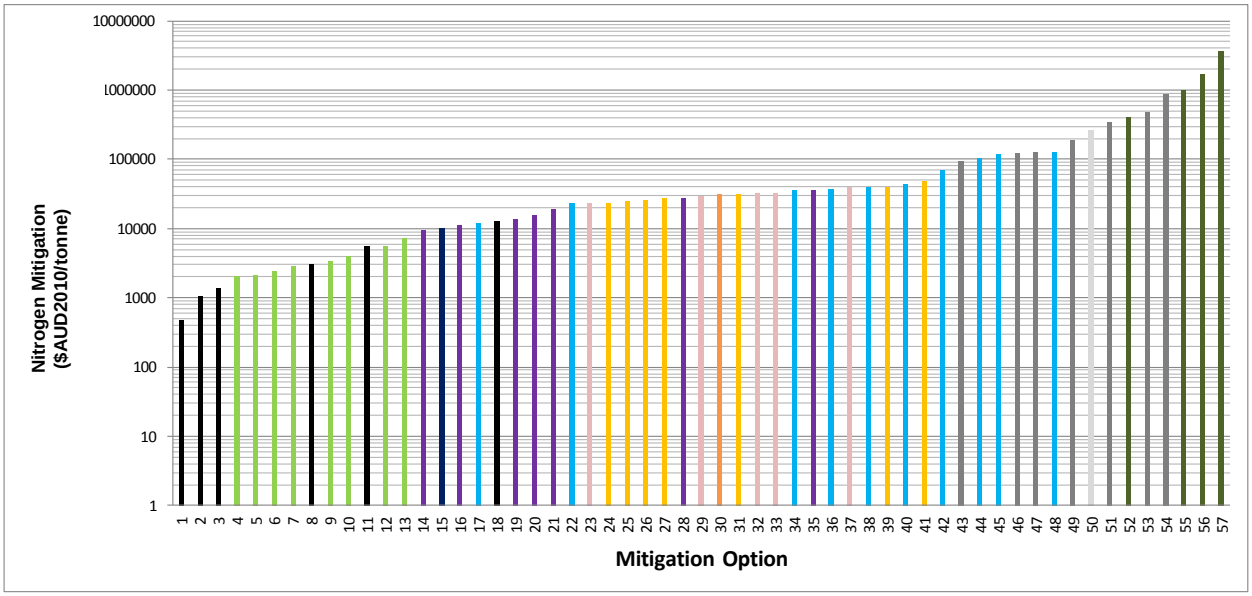


Figure 1. Pollutant Mitigation Cost Curve and the Average Pollutant Cost to Meet a Load Reduction Target.

Results

A selection of results is presented to illustrate the application of extended cost effectiveness to TWCMP and the use of the marginal cost curve to efficiently meet the sustainable loads targets. Cost curves have also been developed for other pollutants including total nitrogen, sediments and greenhouse gas emissions to capture a range of important pollutants associated with urban water services.

Figure 2 presents a SEQ regional cost curve for mitigation of nitrogen pollution. This curve presents a range of mitigation options for consideration in particular catchments. The mitigation options range from point source control, such as Biological Nutrient Removal (BNR) and tertiary filtration, through to the Water Sensitive Urban Design (WSUD) options of bioretention and stormwater harvesting. The range of options is not exhaustive but does cover the main pollution mitigation options and provides an indication of least cost pollutant mitigation. The mitigation options are presented on a logarithmic scale to capture the range in cost effectiveness. For example, Biological Nutrient Removal mitigation costs range from hundreds to thousands of dollars per tonne of nitrogen mitigation, depending on the scale of the treatment plant based upon US EPA data (EPA, 2007a). Bioretention systems range from one hundred thousand to a million dollars per tonne of nitrogen mitigation depending on the type of development based upon case studies (WaterByDesign, 2010a). The cost curve identifies the mitigation options that are likely to be orders of magnitudes more cost effective than others. Cost effectiveness was assessed over a 20-year period for capital and operating costs which were discounted to net present values. The value of water produced from rainwater tanks and stormwater harvesting has been subtracted from the operating costs.



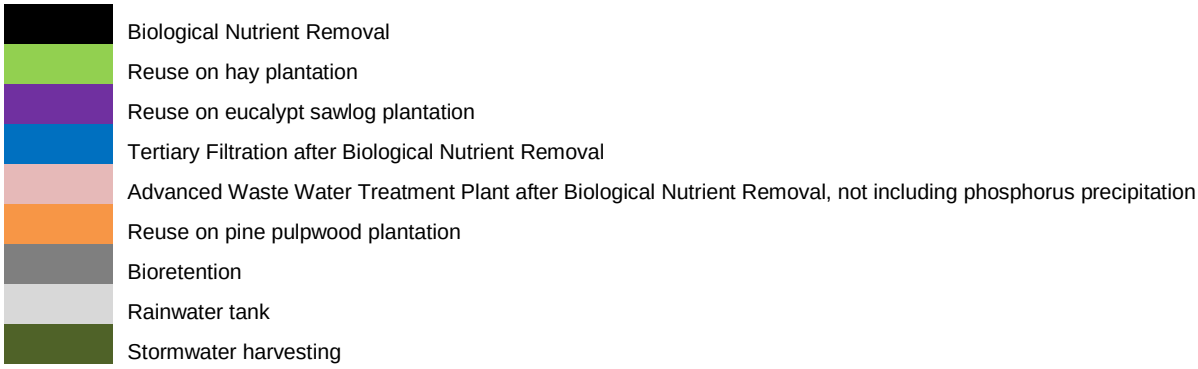


Figure 2. Nitrogen Pollutant Mitigation Cost Curve for a Range of Mitigation Options.

Application to Moreton Bay Regional Council TWCMP

The general cost curve presented in Figure 2 was further developed and applied to the Moreton Bay Regional Council (MBRC) Total Water Cycle Management Plans (TWCMP). In particular, the Caboolture Catchment with the Caboolture Identified Growth Area (CIGA) was considered as a case study. The demonstration illustrates the considerations required to develop a cost for pollutants and to identify cost effective measures. In summary, the following steps were considered:

1. Pollution status. Consideration of current catchment ecosystem health, current pollutant loads, future pollutant loads and sustainable pollutant load targets.
2. Mitigation options. Mitigation options available and approximate cost effectiveness and load reduction potential for the catchment.
3. Value of pollution and cost effective strategy. Development of a cost curve and illustrate the relationship of pollutant value to sustainable load targets and cost effective options to achieve targets.

1. Pollution Status

Figure 3 presents the key catchment characteristics and waterway health for Moreton Bay Regional Council catchments considered in the TWCMP. The figure is a combination of 2010 EHMP scores for waterway health with Key Catchment Characteristics from the TWCMP Strategy (where A= Excellent, B= Good, C = Fair, D= Poor and F = Fail) (BMT-WBM, 2010) - Fig 3-1, Table 4-7. High Environmental Value (HEV) areas such as Deception Bay and Bramble Bay coincide with the lowest EHMP scores. In general, the health of the waterways deteriorates from 'good' to 'fair' in the upper catchment to 'fair' in the lower catchment and becomes 'poor' in the Bay near the mainland coast. The health of the bay then improves to 'fair' further from the mainland coast.

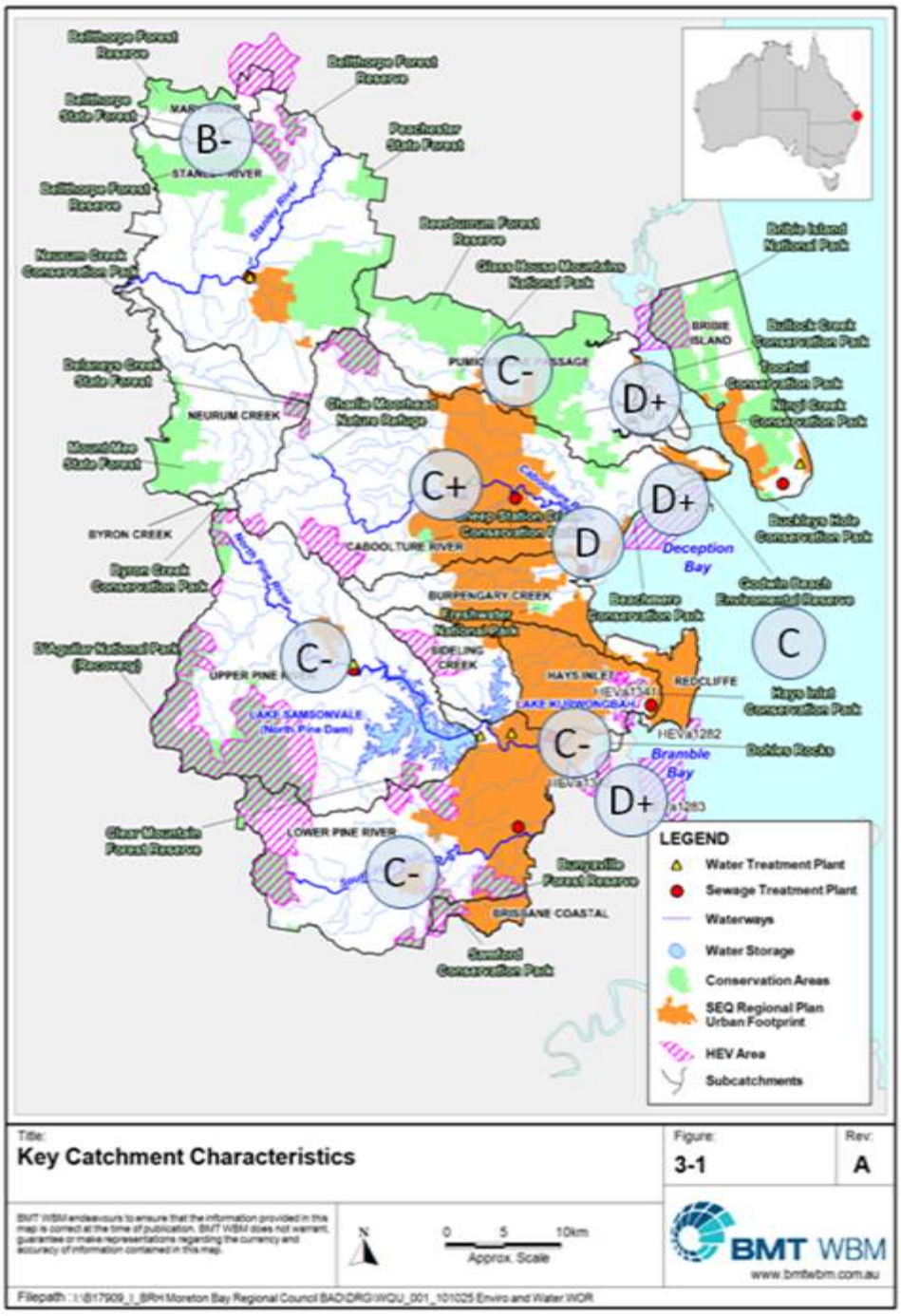


Figure 3. Key Catchment Characteristics and Waterway Health for Moreton Bay Regional Council Catchments.

Figure 4 presents the current and future loads and the sustainable load targets based upon loads presented in the TWCM Strategy (BMT-WBM, 2010) - Table 3-11, 3-12, 3-17, 3-20. Note that Burpengary East STP and Burpengary Creek were included in the loads as they discharge to the estuary and the loads do not include WSUD stormwater load reductions.

In summary, the sustainable load target for total nitrogen from point sources will be exceeded by 12 tonnes/year in 2031. However, the diffuse load is an order of magnitude greater than point source pollution and doubles over the next two decades. This suggests that the health of the river and estuary is likely to deteriorate in the future unless mitigation options are considered.

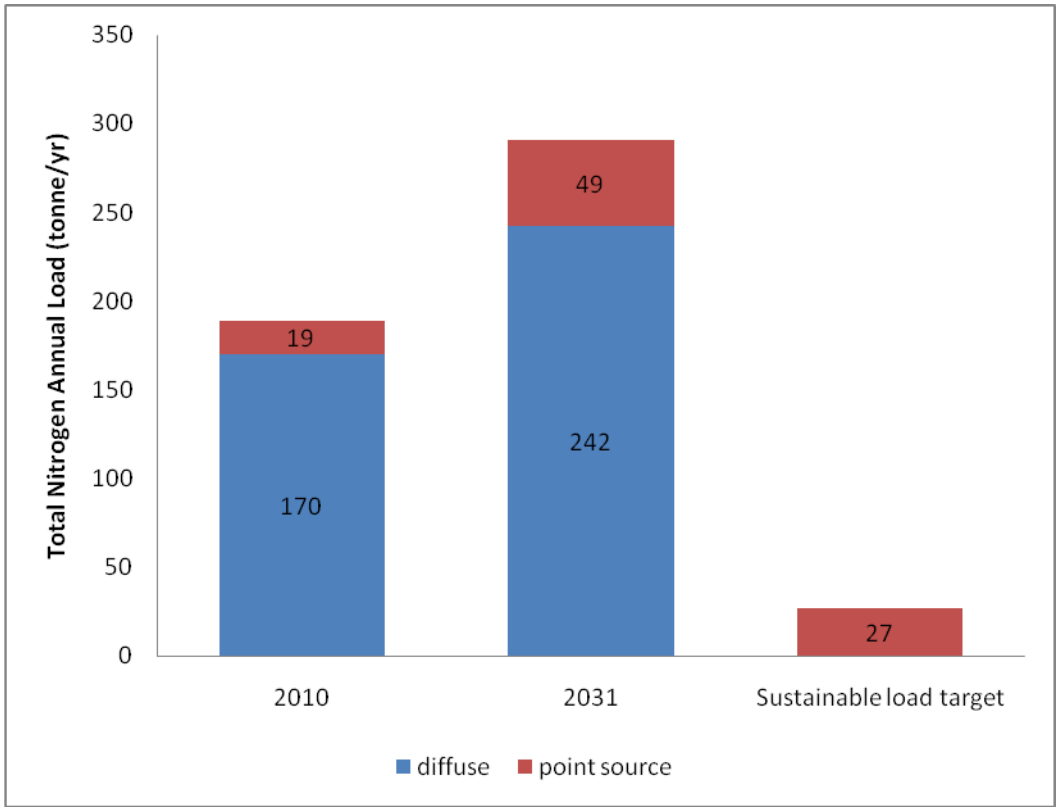


Figure 4. Current and Future Total Nitrogen Pollutant Loads in the Caboolture River Estuary and Comparison to current Sustainable Load Target.

2. Mitigation Options

The following discussion of mitigation options is for demonstration purposes and further discussions are required with local engineers and planners to determine the appropriate potential treatment trains and upgrade possibilities. Nonetheless, the calculations outline order of magnitude costs and options for meeting sustainable load targets. Many of the mitigation options correlate with the 'Catchment Solutions Set' outlined in the TWCMP (BMT-WBM, 2010) - Tables 1 and 5-10. The following discussion highlights some of the considerations required to tailor the cost curve to MBRC conditions.

STP Upgrades - Tertiary Filtration

Sewage treatment plants (STP) in the region already achieve low nutrient concentrations in the effluent discharged to the receiving environment. In 2010 the two STPs that discharge into the Caboolture catchment, namely Burpengary East STP and South Caboolture STP, had median total nitrogen concentrations of 3.9 mg/L and 1.6 mg/L (BMT-WBM, 2010) – Table 3-6, and total nitrogen loads of 13.9 tonnes/year and 4.7 tonnes/year respectively (BMT-WBM, 2010) – Table 3-7.

In 2031, the TWCMP assumes that the median concentrations for Burpengary East STP and South Caboolture STP will be 3 mg/L and 2.5 mg/L respectively (BMT-WBM, 2010) – Table 3-19 and the pollutant loads will be 19 tonnes/yr and 30 tonnes/yr respectively. South Caboolture STP has an increase in nitrogen load of approximately 25 tonnes/yr and accounts for most of the point source load increase. However, the increase in load is largely due to the increase in wastewater flows. The treatment plant currently uses two Sequence Batch Reactors and most of the effluent is directed to a Water Reclamation Plant (WRP) for further processing prior to disposal to the Caboolture River (GHD, 2007). It appears that concentrations increase as relatively less of the STP effluent goes to the WRP. The water recycled from South Caboolture STP remains constant (BMT-WBM, 2010) Tables 3-8 and 3-21 while wastewater flows increase.

In this case, a mitigation option for the effluent not sent to the WRP may be tertiary filtration. In 2031, the wastewater flow not sent to the WRP is approximately 11,000 ML/yr (BMT-WBM, 2010) - Tables 3-18 and 3-21. In general, the treatment process of Activated Sludge with Biological Nutrient Removal reduces total nitrogen concentrations to 3-8 mg/L and the addition of filtration can further reduce concentrations to 2-5 mg/L (Asano, Burton, Leverenz, Tsuchihashi, and Tchobanoglous, 2007) - Table 3-14. The cost effectiveness for nitrogen filtration

after biological nutrient removal is about $30,000 -$35,000/tonne for plants of this scale - namely 19 ML/day and 33 ML/day for Burpengary East STP and South Caboolture STP (with CIGA) respectively (Asano *et al.*, 2007) – Table 3-18. In addition, a reduction in nitrogen effluent concentration from 3 mg/L to 2 mg/L would mitigate about 12 tonnes/year of total nitrogen in 2031 for the two STPs (multiplying Average Dry Weather Flow in 2031 with CIGA by the change in median concentration). This amount of mitigation would reduce the point source load to the sustainable load target for total nitrogen of 27 tonnes/year in the Caboolture River.

Bioretention

Water Sensitive Urban Design devices, such as bioretention basins, can potentially be applied to the CIGA as well as existing developments. In 2031, the increased total nitrogen load from stormwater flow from the CIGA is approximately 22 tonnes/year (BMT-WBM, 2010) – Table 3-17 (assuming the difference between Caboolture River and Caboolture with CIGA). The application of bioretention to achieve stormwater quality objectives of a 60% reduction in total nitrogen (DERM, 2009) – Table 2.1b would give a reduction of approximately 13 tonnes/yr. The cost effectiveness for nitrogen mitigation for bioretention in a greenfield residential site in Brisbane is approximately $90,000/tonne based upon Water by Design case studies (WaterByDesign, 2010a). The cost for a reduction of 13 tonnes/yr would be approximately $1.3 million/yr.

The cost effectiveness of nitrogen mitigation for bioretention in urban renewal is approximately $130,000/tonne based upon Water by Design case studies (WaterByDesign, 2010a). Current stormwater discharges to the Caboolture River and Burpengary Creek are approximately 170 tonnes/year (BMT-WBM, 2010) – Table 3-4. A potential 60% reduction in total nitrogen from existing urban development stormwater would be approximately 102 tonnes/year and would cost approximately $13 million/yr.

Rainwater Tanks

The estimated current rainwater tank yield in the Caboolture catchment is 0.25 GL/yr which is projected to increase to 1.2 GL/yr in 2031 due to urban development such as the CIGA (BMT-WBM, 2010) – Table 3-16. The increase in yield is approximately 0.95 GL/year, which can be used to estimate the pollutant reduction of approximately 1.5 tonnes/year (assuming a total nitrogen residential stormwater concentration of 1.6 mg/L (Wong, 2006) - Figure 3-4). The cost effectiveness for nitrogen mitigation for a greenfield residential site in Brisbane is approximately $260,000/tonne based upon Water by Design case studies (WaterByDesign, 2010a). The cost for a 1.5 tonnes/year reduction would be approximately $0.39 million/yr.

This order of magnitude calculation could be refined to reflect the modeling undertaken in the TWCMP study (BMT-WBM, 2010) such as the split between plumbed and non-plumbed tanks.

3. Value of Pollution and Cost Effective Strategy

The existing sustainable loads target could be met in 2031 (just) by using tertiary filtration and gives a pollutant value of approximately $35/kg. Lower cost options may be available by using controlled effluent disposal to irrigation and these options need to be explored further. However, despite upgrades to maintain the sustainable load target, the total nitrogen point source loads increase significantly from current loads and are likely to lead to further deterioration of aquatic ecosystems. This, and the fact that most of the nitrogen loads are from diffuse sources, suggests that a new sustainable load target is required that addresses both point and diffuse loads. Nonetheless, it appears to be a cost effective strategy to address point source loads as a priority.

It could be argued that, to avoid further deterioration of aquatic ecosystems, loads should not increase beyond current levels. In 2031, this would require an annual nitrogen load reduction of 30 tonnes and 172 tonnes for point source and diffuse source respectively. This would require exploration of more mitigation options than listed in the previous section. However, as a demonstration, just using tertiary filtration and bioretention could reduce the point and diffuse loads by 127 tonnes/yr. The use of these options gives a weighted average cost for pollution mitigation of approximately $116/tonne. Although this value is an order of magnitude higher than the estimate to meet the sustainable load target, it is not high compared to other estimates. For example, in 2005/2006, the cost of Victorian stormwater offsite treatment was calculated as $800/kg of nitrogen (MW, 2006). The difference, in part, reflects that the Victorian scheme only addresses nitrogen pollution and allocates all costs to this pollutant. The estimates presented in this paper allocate costs for WSUD devices to nitrogen, phosphorus and sediment reductions to reflect stormwater quality objectives.

Conclusions

By placing a price on pollution, it is possible to extend cost effectiveness to include environmental impacts. This supports options evaluation for Total Water Cycle Management. However, the price given to pollution using the engineering cost method depends upon existing sustainable load objectives. The sustainable load targets may need review to ensure that ecosystem do not continue to degrade.

An important next step in the research is to seek review for the cost effectiveness estimates. This includes review of the data used as well as the allocation of costs, discounting and other assumptions.

References

Asano, T., Burton, F. L., Leverenz, H. L., Tsuchihashi, R., and Tchobanoglous, G. (2007). *Wastewater Reuse: Issues, Technologies and Applications*: Metcalf and Eddy, McGraw Hill.

BDA Group. (2005). *Scoping Study on a Nutrient Trading Program to Improve Water Quality in Moreton Bay*: Environmental Protection Agency (A Queensland Government Document).

BMT-WBM. (2010). *Total Water Cycle Management Strategy for Moreton Bay Regional Council*. (A Moreton Bay Regional Council Document).

Bockstael, N. E., Freeman, A. M., Kopp, R. J., Portney, P. R., and Smith, V. K. (2000). On Measuring Economic Values for Nature. *Environmental Science and Technology, 34*(8), 1384-1389.

Bowers, J., and Young, M. (2000). Valuing Externalities: A Methodology for Urban Water Use. *Policy and Economic Research Unit, CSIRO*.

Bunn, S. E., Abal, E. G., Greenfield, P. F., Tarte, D. M., and Saxton, N. E. (2008). *Linking science, monitoring and management to improve the health of waterways in South East Queensland, Australia*. Paper presented at the 4th ECRR Conference on River Restoration.

DERM. (2009). *Draft Urban stormwater Queensland Best Practice Environmental Management Guidelines 2009*. (A Department of Environment and Resource Management Document).

DEWHA. (2010). *National Water Initiative pricing principles, Regulation Impact Statement*. In Department of the Environment, Water, Heritage and the Arts: 2010.

Driml, S., O'Sullivan, D., and Hanna, R. (2005). *River Economics - problems, progress and potential*. Paper presented at the 10th International River Symposium and Environmental Flows Conference.

EHMP. (2009). Report Card 2009 for the waterways and catchments of SEQ: Ecosystem Health Monitoring Program, South East Queensland Healthy Waterways Partnership, Brisbane.

EPA. (2007a). Biological nutrient removal processes and costs U.S. Environmental Protection Agency, Office of Water, http://www.epa.gov/waterscience/criteria/nutrient/files/bio-removal.pdf.

EPA. (2007b). Nutrient Offsets Scheme to Improve Water Quality in Moreton Bay. Stage 2: Progress Report 2. Attachment 6 Experimental Economics Report: Queensland Environment Protection Authority, Queensland Government, Australia.

EPA. (2008). *Moreton Bay water quality offsets scheme: Final Report*: Environment Protection Agency, (A Queensland Government Document).

Fane, S., Blackburn, N., and Choong, J. (2010). Sustainability Assessment in Urban Water Integrated Resource Planning: Prepared by the Institute of Sustainable Futures for The National Water Commission project: 'Integrated Resource Planning for Urban Water'.

GHD. (2007). *Caboolture Shire Council Report for South Caboolture STP Planning Study*. (A GHD document for Caboolture Shire Council).

Hurikino, P., Lutton, V., and Eadie, M. (2010). *Draft South East Queensland Sub-regional Total Water Cycle Management Planning Framework*. (A Queensland Water Commission Document).

IDC. (2003). Environmental Economic Valuation: An introductory guide for policy-makers and practitioners: Inter-Departmental Committee (IDC) on Environmental Economic Valuation for the Queensland Government.

McKinsey and Co. (2008). An Australian Cost Curve for Greenhouse Gas Reduction: McKinsey and Company, Australia.

MW. (2006). Stormwater Quality Offsets: A Guide for Developers: Melbourne Water.

QCA. (2010). *Final Report SEQ Interim Price Monitoring Framework.*

Queensland Competition Authority Act 1997.

QWC. (2008). *South East Queensland Water Strategy - Draft March 2008*.

Rolfe, J., Donaghy, P., Alam, K., O'Dea, G., and Miles, R. (2005). Considering the economic and social impacts of protecting environmental values in specific Moreton Bay/SEQ, Mary River Basin/Great Sandy Strait Region and Douglas Shire Waters.: Institute for Sustainable Regional Development, Central Queensland University, Rockhampton.

SEQHWP. (2007). South East Queensland Healthy Waterways Strategy 2007-2012: Strategy Overview: SEQ Healthy Waterways Partnership, Australian Government.

StandardsAustralia. (1999). *AS/NZS 4536:1999 Life cycle costing - an application guide.*

WaterByDesign. (2010a). *A Business Case for Best Practice Urban Stormwater Management: Casestudies Version 1.1 September 2010. A companion document to a Business Case for Best Practice Urban Stormwater Management* (B. South East Queensland Healthy Waterways Partnership, Queensland.

WaterByDesign. (2010b). *Total Water Cycle Management Planning Guideline for South East Queensland (Draft, September 2010).* Department of Environment and Natural Resource Management, Brisbane, Queensland.

Wong, T. H. F. (2006). *Australian Runoff Quality: A guide to Water Sensitive Urban Design*.

Externality Effects and their Monetary Values for Water Servicing Options in South East Queensland

Daniels, P. and Porter, M.
Griffith School of Environment, Griffith University, Nathan, Queensland

Summary

To comprehensively assess the consequences of resource management strategies across communities and over the long-term, it is necessary to consider both private financial costs and benefits, and other less direct, but often very significant impacts on human welfare. Effects that are not taken into account directly in market-place transactions are known as "externalities". This study has focused upon the compilation and presentation of the background information required to prepare basic externality analyses to supplement and strengthen the assessment of sustainable water management options. The study provided a comprehensive review of existing research and valuation data into the identification and economic assessment of externalities pertinent to the adoption of five different water supply options under consideration in South East Queensland (SEQ) including stormwater harvesting, rainwater tanks, centralised wastewater recycling, dams and desalination. This paper will present some results indicative of this methodology in the form of excerpts specifically focused on the externalities of rainwater tanks.

Keywords:
Externalities, valuation, water servicing option analysis.

Introduction

Any strategy adopted to provide water security and meet required levels of service will involve trade-offs. Trade-offs often incorporate external or indirect impacts, which pose real and significant impacts on well-being. These effects, when not taken into account directly in market-place transactions, are known as "externalities". Externalities typically involve welfare effects on others not involved in the transaction, and often affect long-term or unknown outcomes upon community well-being and are part of complex cause-effect chains (eg, the positive recreational gardening impacts of residential rainwater tanks).

This study has focused upon the compilation and presentation of the background information required to prepare basic externality analyses to supplement and strengthen the assessment of sustainable water management options. The study provided a comprehensive review of existing research and valuation data into the identification and economic assessment of externalities pertinent to the adoption of five different water supply options under consideration in SEQ including stormwater harvesting, rainwater tanks, centralised wastewater recycling, dams and desalination.

This paper presents indicative results only and focuses upon rainwater tanks as an example of detailed results for the seven main options analysed in the Urban Water Security Research Alliance's Technical Report No. 42 *A Compendium of Externality Effects and their Monetary Values for Water Servicing Options - A Study based on the South East Queensland Context,* expected to be published on-line at the UWSRA web site by September 2011.

Methodology

Firstly, an extensive review of relevant water externality research and related literature was conducted. This research provides the foundation for the identification and basic appraisal of the nature of the positive and negative externalities that tend to be associated with the implementation of water servicing options. While economic and environmental impacts are often the exclusive focus of impact research related to natural resources, the externality analysis information compiled here has also explicitly targeted more intractable, *social* externalities where possible. The second major set of results is a summary of the existing estimates of monetary values for each of the externalities identified in the preceding section. These valuations were gleaned from a diverse range of existing environmental economic research and values were converted to $AUD 2010 based on historical exchange rates and Gross Domestic Product (GDP) implicit price deflators.

The externality categories have been selected on the basis of their identified relevance and application throughout the existing research on water-related externalities (for example, see Young, 2000). The externality classification draws upon the popular "total economic value" (TEV) approach (Landell-Mills and Porras, 2002).

Whilst it is commonly acknowledged that it is important to incorporate the externalities associated with water service options into decision making, there is significant controversy surrounding the extent to which monetary values for externalities should be used as a basis for policy (Gardner, Hatton MacDonald and Chung, 2006; Rein, 1999). Thus, this article presents the externalities and the relevant valuations separately, leaving it up to the individual practitioner as to whether they wish to take the next step of compiling and combining the externalities and valuations in the decision process. The externalities identified and/or the valuations can be applied as inputs and background information for a variety of decision-making frameworks such as multiple criteria assessment or cost-benefit analysis.

As the foundation for subsequent economic assessment, the externality values are first assessed in "average" dollars per biophysical unit terms (eg, an economic cost of $100 per kg for nitrogen emissions from wastewater recycling). Per-unit values can be sourced on a benefit or costing transfer basis (that is, from other relevant studies), or they can be calculated, with much more time and expense, from primary research or proxy data (based on the study area and context-specific characteristics). Estimates from transfer methods may also be adapted or adjusted on the basis of the unique features and conditions of the study context. Appropriate monetary unit value ranges, or median, modal or other measures of central tendency, would need to be identified for multiplication by total flow, production or other water service changes. The valuation data was then converted to 2010 Australian dollars to facilitate comparison.

Results

The major externalities identified for rainwater tank systems are listed, with a short description of the nature of these impacts, in Table 1 below. The externalities have been allocated in accordance with the life-cycle and operational phases in which they are most likely to occur. A general category is also included for effects that are not neatly classified within these phases. The externalities are also identified in terms of more general externality "types" (eg, greenhouse gas emissions (GHGs), recreation (R), and so forth) as described in the methodology section. For each specific externality, symbols indicate the typical nature of the impact in terms of whether they tend to be a positive (+) or negative externality (-); and whether impacts tend to occur predominantly downstream (↓), upstream (↑), or within the immediate surrounds of the supply infrastructure (◙).

Currently, rainwater tanks are widely used in Australia, especially in rural areas. By 2001, around 16 - 17% of all households had rainwater tanks (Tam, Tam and Zeng, 2009, p.178). Governments have backed a funding push towards the use of rainwater tanks in response to the drought conditions occurring throughout Australia. In addition, given that 73% of houses in Australia are amenable to the installation of rainwater tank systems, the prevalence of rainwater tanks is likely to increase in the future (Tam, Tam and Zeng, 2009, p.179).

The supply of water from a rainwater tank involves collecting water directly from a non-trafficable roof into a rainwater tank (or equivalent holding device installed onsite). It is usually close to the roof of the dwelling used for water collection. Plastic tanks are becoming increasingly popular and the plastic high density polyethylene (HPDE) models have a life-span of approximately 25 yrs (Blue Scope Steel, 2010). A study of Sydney's new houses revealed that the most common size of tank was 2 to 3 kL followed by 4 to 5 kL (Retamal *et al.*, 2009, p.21). This statistic would be similar for SEQ urban areas. There are three common elements to all rainwater tank systems:

(i) Catchment surface off which rainwater is collected (eg, roof);
(ii) Storage reservoir (the tank itself); and
(iii) A delivery system which transports the water between the stages of collection and use (pumps and pipes) (Tam, Tam and Zeng, 2009, p.180).

Manufacture and Storage Impacts

The externalities associated with rainwater tank system manufacturing depend upon materials, energy sources and processes utilised. Manufacturing often requires the significant input of fossil fuels. The manufacturing process can result in releases of various noxious chemicals into the atmosphere – for example, the emission of SOx and NOx when manufacturing concrete tanks.

The full costs of rainwater tanks include the tank, installation and pump, as well as the occupied land area. In cases where garden space is small it may impinge on the aesthetics and useable space. Additionally, one of the most hazardous aspects of rainwater tanks is that they may pose a drowning risk (Byard, 2008). Another important potential health hazard is that rainwater tanks can provide breeding sites for disease vectors such as mosquitoes (Ritchie, Montgomery and Walsh, 2001, p.13).

Use and Distribution Impacts

Water quality collected in rainwater tanks is directly impacted by the characteristics of the roof catchments and run-off contamination (Evans, Coombes and Dunstan, 2006, p.37; Helmreich and Horn, 2009, p.121). The health impacts of rainwater tanks primarily arise due to either contamination of the rainwater or the provision of insect vector breeding grounds (Kahinda, Taigbenu and Boroto, 2007, p.1052; Goodall, 2000, p.126). The specific pathogens identified in tanks are diverse. Some species found in rainwater tanks include; *Salmonella, Shigella, Vibro, Clostridium, Legionella, Campylobacter, Cryptosporidium,* and *Giardia* spp (Evans, Coombes and Dunstan, 2006, p.37). In addition to airborne contaminants, the catchment surfaces themselves can be a source of heavy metals and organic substances (Helmreich and Horn, 2009, p.121; Lye, 2009, p.5430). Long-term exposure to some of the substances that may be present in some rainwater tank water may lead to biological disorders (Lye, 2009, p.5430). The health risks are particularly significant if chemical contaminants such as pesticides, herbicides, biocides, organhalogens, and petroleum hydrocarcons are present (Lye, 2009, p.5430). However there is a variation in scientific and medical opinion within the literature as to the severity of these risks (Evans, Coombes and Dunstan, 2006, p.37; Kahinda, Taigbenu and Boroto, 2007, p.1052).

Despite their recent portrayal as a 'green' water supply option, rainwater tanks can be relatively energy intensive. (Retamal *et al.*, 2009, p.61). However, the energy intensity of water tanks also varies greatly depending on the pump type and size, end-uses and the location and layout of the tank system (Retamal *et al.*, 2009, p.viii). The most common type of pump used in household water tanks is a fixed speed pump (Retamal *et al.*, 2009, p.23). Fixed speed pumps utilise equivalent amounts of power regardless of the volume of water being distributed (Retamal *et al.*, 2009, p.23). This is inefficient.

Decommissioning Impacts

Unfortunately, there is a pronounced lack of information on externalities related to rainwater tank disposal. Presumably, the impacts would differ with regards to the materials with which the tank was made. Recyclability of the component parts would be another important factor for consideration. It would be valuable to know where most tanks finish up at the end of their lifetime, ie, do owners actually recycle parts, or does the entire tank end up at landfill?

Positive Spill-Overs

There are a several positive externalities related to the use of rainwater tanks. Importantly, there is the potential to reduce reliance on mains water use by up to 50% (Hallmann, Grant and Alsop, 2003). The householder installing a rainwater tank may therefore find significant economic savings from reduced need to purchase mains water (Coombes and Kuczera, 2003, p.1). Furthermore, through this reduction, there can be a deferral of large scale water infrastructure, reduced peak stormwater flows, and sediment transport (Coombes and Kuczera, 2003, p.1; Blue Scope Steel, 2010; Mitchell and Rahman, 2007, p.41). Additionally, through capture of stormwater the nutrient loads to oceans and waterways may be significantly reduced with estimates for the reduction of phosphorus loads by 68% (Hallmann, Grant and Alsop, 2003). Increased use of rainwater tanks will also result in less frequent overloading of sewer systems and overflows, having health and economic benefits for the community (Vaes and Berlamont, 1999).

Social benefits can also occur. Householders and communities receive benefits relating to recreation and amenity. Rainwater tanks are able to mitigate the effects that water restrictions have on lifestyle, amenity and property values through enabling garden watering throughout drought periods (Marsden Jacob Associates, 2007b, p.4; Tam, Tam and Zeng, 2009, p.179). Decentralised sources help diversify the water supply system and create inbuilt resilience, thus assisting in lessening the water crisis on an individual and societal level (Tam, Tam and Zeng, 2009, p.179; Helmreich and Horn, 2009, p.121). Furthermore, decentralised water supplies create safety nets against terrorist attacks (Mitchell and Rahman, 2007, p.41), and enable people to continue with small scale productive activities such as brewing, food production, household construction (Helmreich and Horn, 2009, p.121; Kahinda, Taigbenu and Boroto, 2007, p.1051). It can also foster a sense of community-mindedness, acting as a form of environmental education promoting awareness of sustainable water use (Furumai, 2008, p.340; Marsden Jacob Associates, 2007b, p.4).

Table 1. Rainwater Tank Externalities.

Note: positive (+) or negative externality (-); whether impacts tend to occur predominantly downstream (↓), upstream (↑), or within the immediate surrounds of the supply infrastructure (◙). Reference label numbers are provided at the end of each externality listed.

RAINWATER TANKS	
LIFE CYCLE OR OPERATIONAL PHASE	**EXTERNALITY TYPE, DESCRIPTION (showing if positive or negative effect, main location and reference source number)** (see Appendix A for reference source code details) (see inset table to left for externality type codes)
General **Externality Type Codes =>** **GHG** Greenhouse gas emissions **En** Energy **WQ** Water Quality **N** Nutrients **P** Production values **R** Recreation values **A** Amenity values **H** Health values **E** Ecosystem values **B** Biodiversity values **NU** Non-use values **O** Other	**GHG:** GHG contributions are 18% higher than that of reticulated water supply (600L tank) (-) 224 *(= reference number – see bibliography for details)* **En:** Comparatively higher cumulative energy demand (67% higher than reticulated water for a 600L tank) (-) 224 **En:** typical energy intensity of water supply for the most common pump and rain switch system is approximately 1.5 kWh/kL compared to the energy intensity of mains water supply of less than 1 kWh/kL. (-) 206 **N**: Reduction in nutrients in the system, with greatest impacts when water drains to environmentally sensitive areas (+, ↓) 223, 205 **P**: Extends lifetime of existing large-scale water infrastructure by reducing peak loads and reduction in the cost of water and stormwater infrastructure (+, ↓) 201, 223 **R**: mitigating the effects of water restrictions on their lifestyle and enabling people to continue to grow food under water restrictions (+, ◙) 223, 80 **A**: Improving amenity and property values and improving the taste of water in areas of poor water quality (+, ◙) 223 **H**: Drowning hazard (-, ◙) 75 **H:** Safety net against terrorist attacks (+, ↑, ↓, ◙) 83 **H:** Increased use of rainwater tanks may heighten the risk of waterborne disease outbreaks unless appropriate preventative measures are undertaken. (-, ◙) 173, 266 **E:** Significant reductions in water use, up to 50% (+) 224, 205 **E:** Substantial reductions in nutrient loads to oceans and waterways (reduces phosphorus loads by 68% over reticulated water for a 600L tank) (+,↓) 224 **E:** Significant reductions in NO_x flowing to stormwater systems (+,↓) 224 **NU:** sense of community-mindedness (+, ◙) 223 **O:** it was shown that the use of rainwater tanks provided high levels of economic benefits to the community and individual savings on mains water bills (-) 78 **O:** Useful measure for water demand in emergency cases (+, ◙) 266 **O:** could reduce water and food crisis in some regions – increasing water security, especially important in the event of extreme climatic change (+, ◙) 80, 207, 84
Manufacture	**GHG:** Often tanks and associated plumbing requires high levels of transport throughout stages of manufacture to installation which increases GHG emission and degrades air quality. (-, ◙) 224, 205 **GHG and En:** energy and material impacts of water tanks much higher than water from reticulated water supply (-) (especially when pump is used) (-) 224 **GHG and En:** (PLASTIC) Materials used in the fabrication of plastic tanks is High Density Polyethylene (HDPE), derived from fossil fuels, currently 100% virgin plastic is used (-) 205 **WQ and P:** Reduces combined sewer system overflow and peak stormwater flow pressure (+ ↓) 86, 205 **WQ and H:** Catchment surfaces themselves may be a source of heavy metals and organic substances. Bacteria, viruses and protozoa may originate from faecal pollution by birds, mammals and reptiles that have access to catchments and rainwater storage tanks (-, ◙) 80, 82 **H:** Carcinogens released throughout manufacture (16% higher than reticulated water for 600L tank) (-, ◙) 224 **H and E:** Higher release of heavy metals through manufacture and disposal (40% higher than for reticulated water) (-, ◙) 224 **H and E:** High levels of photo oxidant chemicals stemming from plastic manufacture (4133% higher than reticulated water for a 600L tank) (-, ◙) 224 **O:** Avoided water storage infrastructure is significant, however, not large enough to offset the impacts of water tank construction and operation (-, ◙) 224
Storage	**P and O:** Reductions in combined sewer overflow (+, ↓, ◙) 86 **H**: Provision of mosquito breeding sites though risk is very low if tanks are well maintained (-, ◙) 53, 224, 99
Distribution	**En:** Consume energy associated with pumping for internal use though short distance tends to make more efficient than other sources (-) 39
Use (household)	**GHG and En:** Pumps are often oversized and inefficient (-) 224 **WQ and H**: Contamination from atmospheric pollutants including particles, microorganisms, heavy metals and organic substances (-, ◙) 80, 82 **P:** Reduces residential fresh water demands by up to 50% (+, ◙) 205 **P:** Annual cost savings for households ranged from $3 to $223 per year and from $0 to $240 per year for using rainwater both outdoor and indoor uses (+, ◙) 209 **R and A:** Enables watering of gardens under water restrictions (+, ◙) 83, 80 **A:** Water tanks may impact negatively upon aesthetics of garden space, taking up, on average, $4m^2$ of garden space (-, ◙) 224 **H**: risk of run-off contamination, via direct depositions by birds and small mammals, decay of accumulated organic debris, and atmospheric deposition of airborne micro-organisms, heavy metals from the roof surface and chemical pollutants (-, ↓, ◙) 93, 171, 80, 204 **O:** rainwater use can work as a kind of environmental education to make citizens aware of sustainable urban water use (+, ◙) 266
Disposal	**H and E:** Higher release of heavy metals through manufacture and disposal (40% higher than for reticulated water) (-, ◙) 224 **E:** Comparatively higher levels of solid waste to landfill due to electricity use and tank disposal , around 40% of tanks installed are recycled, 20% of copper used and 50% of the pumps (102% higher than reticulated water supply for a 600L) (-, ◙) 224

Valuation Estimates

Table 2. Selected Relevant Externality Valuations Relevant to Rainwater Tanks

Externality Type	$AUD (2010)	Original monetary value	Reference source[1]	Explanation/Comments
1. Pressure or state change indicators				
GREENHOUSE GAS EMISSIONS (CO_2 only, $/t CO_2)				
	24.15 – 33.6 – 42	$23 – 32 – 40 AU	294	CPRS prices between $23 t/ CO_2 (24.15) and $32t/ CO_2 (33.6) at the start of the scheme, '*the government considers a price cap of $40* t/ CO_2 *(42) is appropriate in balancing these requirements".*
GREENHOUSE GAS EMISSIONS (Social Cost of Carbon)				
Social Cost of CO_2 (SCC)	**18.59, 66.38**	$14 – 50t/ CO_2	70	In a recent working paper, Tol (2004) gathers 103 estimates from 28 published studies to form a probability density function. The mode is $2t/ CO_2 [2], the median $14t/ CO_2 (18.59), the mean $93t/ CO_2, and the 95 percentile $350t/ CO_2. The conclusion was that the marginal damage costs of carbon dioxide emissions are unlikely to exceed $50t/ CO_2 (66.38), under standard assumptions about discounting and aggregation, and probably much smaller.
ENERGY ($/MWh)				
Energy ($/MWh)	**19.75**	19.85 $/MWh AU	289	Current prices are on average, 19.85 $/MWh (peak periods, 7am – 10pm).
WATER QUALITY				
Water Quality (per Trip)	**0.89, 10.81**	£0.48 - £5.81/person EU	154	Hypothetical water quality improvements lead 1.3% increases in trip frequency. Consumer surplus due to water quality improvements was estimated to be £0.48 (0.89) per trip or £5.81/person (10.81).
NUTRIENTS				
Nutrients NITROGEN Diffuse Source	**8980.5**	$8,553/t	302	Average annual cost of diffuse source load reduction $8,553/tonne N (8980.65).
Waterway Health	**52.32 – 103.34 & 72.24 – 226.29 & 152.21 – 226.29**	26 – 51.35, 35.90 – 112.44, 75.63 – 112.44 US	95	Farber and Griner (2000), Pennsylvania; WTP for waterway health, stream improvements from: 'moderately polluted to unpolluted = $26 - 51.35 (52.32 – 103.34); 'severely polluted' to 'moderately polluted' = $35.90 - 112.44 (72.24 – 226.29); from 'severely polluted' to 'unpolluted' = $75.63 - $112.44 (152.21 – 226.29) (per household, p.a., over 5 yrs, US 2000 $).
ECOSYSTEM				
Ecosystem – Improvement to Flows	**57.54, 30.4**	$42, 22AU	56	AU$42 (57.54) annual increase in the household cost of water for an improvement in river flows from no to some rivers and from some to all rivers AU$22 (30.4).
Ecosystem - Wetlands	**67.12**	$27 AU	153	WTP $27/household/yr (67.12) (AU) to protect the Nadgee Nature Reserve (Aust.).
Ecosystem - Fisheries	**555.60**	$400 AU	95	KPMG, 1998 WTO per recreational angler in Moreton Bay for the preservation of a fishery threatened by pollution may be around $400 p.a. (555.60).
BIODIVERSITY				
Biodiversity – Endangered Species (unspecified)	**30.7, 6.4**	$24 AU	218	WTP $24 (30.7) (increase in household water prices) to reduce habitat loss for 5 species, or $5 (6.4) per uncommon species ($Au, 2001).
Biodiversity	**27.28 – 124.92**	€13.3 to €60.9 EU	262	Morrison et al. (1999) to estimate the non-use values of environmental, as well as social and economic attributes of the Macquarie Marshes wetland in Australia. They find that the Australian public is WTP substantial amounts (€13.3 to €60.9per household (27.28 – 124.92) depending on the model and management scenario employed) in order to increase the wetland area, to improve the biodiversity found in the wetland, and to increase irrigation related employment.

Table 2 (*cont'd*). Selected Relevant Externality Valuations Relevant to Rainwater Tanks

Externality Type	$AUD (2010)	Original monetary value	Reference source[1]	Explanation/Comments
2. Socioeconomic costs and benefits				
PRODUCTION				
Production - Commercial Fishing	**4.68mill**	$40 million AU	95	Taylor (2002), value of commercial fishing in Moreton Bay $40M p.a. (46800000).
RECREATION				
Recreation - Rivers	**29.38**	$23.69 AU	95	Walpole (1990), WRP average $23.69 (29.38) per household per visit for recreation at river sites in northern Victoria (2002, AU$).
Recreation - Swimming	**58.15**	$51.6	23	WTP $51.60 (58.15) per water quality increase to be fishable/swimmable.
AMENITY				
Amenity – WTP for Waterway Restoration	**0.46**	$0.37 AU	95	Robinson et al. (2002), residents of Bremer River in SEQ WTP $0.37 (0.46) per person p.a. (2002 AU $) to increase total length of the river with a 'very good' appearance by 1%.
NON-USE				
Non-use Values – Rivers	**16.26, 16.05**	$17.6, 17.38 NZ	51	Values associated with the Rakaia River (NZ): health of in-stream river flows. Option Price: $17.60 (16.26) and the Preservation Price: $17.38 (16.05) (NZ $ per household, p.a.).
Non-use Values	**12.79**	$10 AU	218	WTP $10 (12.79) (increase in household water prices) for a 10% reduction in household water use ($AU, 2001).
HEALTH				
Health	**488.52**	US $ 258	346	Cost of Illness: US $ 258 (1992) for an average emergency room visit.
	109.82	US $59	346	Cost of Illness: each restricted activity day due to illness is worth on average $58 (US, 1992).

[1] The reference sources for this table can be accessed in the full report (UWSRA Technical Report No. 42).

Discussion

Indicative monetary values can be very useful in decision-making if there is explicit treatment and awareness of the range of estimated values, the reliability, completeness, and accuracy of underlying data, and the need for specific contextual information on impacts. Economic valuations of externalities can be applied in many ways and to varying degrees in strategic water research, management and planning. It is quite rare to see the full-scaled adoption of private and external costs and benefits into economic decision-making frameworks such as cost-effectiveness (CEA) or cost-benefit analysis (CBA). Indeed, this would represent strong faith in the reliability and inclusiveness of market measures and monetarisation.

This article is unique in providing a systematic guide to externality analysis and its incorporation in strategic decision-making for natural resources – especially in terms of its focus upon water and the Australian context. However, it must be acknowledged that empirical data and research in the area are limited and direct application to the SEQ context is constrained by a dearth of local studies.

Specific data limitations affecting the efficacy of this study included:

- The general lack of knowledge about biodiversity and ecosystem impacts of the water servicing options.
- Externality and impact research is often weighted towards the political agendas at the time of publication.
- Lack of data on how water servicing options impact indigenous communities.
- There is a pronounced paucity of research and data on the decommissioning impacts of all supply options reviewed.
- General lack of published life cycle assessments (LCA) relating to water servicing options.

These data caveats are significant and should be acknowledged in the application of the research results.

Conclusions

Decision-making directed towards sustainable resource management and triple bottom line accounting requires the identification and incorporation (and ideally some idea of values or trade-offs) of the full range of private and external effects associated with major strategic options. Well-being effects of human actions that are not taken into account directly in market-place transactions are known as "externalities". However, externalities have very real and significant impacts on community well-being and hence the efficiency, effectiveness and desirability of alternative options for resource use in society. Externalities can no longer be treated as extraneous to water resource planning (Davis, 2006, p.7).

This paper presents some indicative results for rainwater tanks derived from a detailed survey and review of the externalities, and associated costs and benefits, based on an indicative range of water servicing options that are being implemented or considered for ensuring secure and sustainable water for the SEQ region. The first major data set presented is a systematic summary of all significant externalities associated with the selected option, rainwater tanks. The second data set presented in this paper demonstrates some indicative valuation estimates for externalities related to rainwater tanks. This constitutes a small selection of the valuations identified in the broader study. Commentary on the caveats for specific externality and group valuations, and gaps in data availability and coverage, is also included in this section.

This article provides water managers, scientists, and practitioners with a detailed reference to help incorporate the full range of costs and benefits into their option and scenario assessment and decision-making. Improved identification and information about the likely external effects of alternative water options and the magnitude of their economic impacts can assist in the efficient allocation of resources and avoid unexpected costs later in the project cycle (Siebert, Young and Young, 2000). It is important to incorporate both private (direct market) and external costs and benefits in water-related decision making activities, as their neglect can encourage investment in sub-optimal alternatives which may prove regrettable in the long-run (Bryan and Kandulu, 2009). Externality analysis provides a more systematic and comprehensive input to guide decision-making by identifying, quantifying and assessing the relative trade-off values that might be associated with the full range of long-term costs and benefits of proposed water options. Failure to factor in externalities such as those identified in this article may lead to the misallocation of resources and increased levels of damage to society and the natural environment in which it is embedded (Rein, 1999).

To the authors' knowledge, this article is unique in providing a systematic guide to externality analysis and its incorporation in strategic decision-making for natural resources – especially in terms of its focus upon water and the Australian context. However, it must be acknowledged that empirical data and research in the area is limited and direct application to the SEQ context is constrained by a dearth of local studies.

Major References

The numerous references cited in the externality and valuation tables can be accessed in the full report.

Bryan, B. and Kandulu, J. (2009). Cost-effective alternatives for mitigating Cryptosporidium risk in drinking water and enhancing ecosystem services. *Water Resources Research* 45(8).

Davis, C. K. (2006). *Ethical Dilemmas in Water Recycling*. Water Supply and Treatment Division, San Francisco Public Utilities Commission, San Francisco, US.

Gardner A, Hatton MacDonald, D. and Chung, V. (2006). Pricing water for environmental externalities in Western Australia. *Environmental and Planning Law Journal* 23 (4): 309-326.

Landell-Mill, N. and Porras, I. (2002). *Silver bullet or fools' gold? A global review of markets for forest environmental services and their impact on the poor.* International Institute for Environment and Development (IIED), London.

Marsden Jacob Associates (2007a). *The cost-effectiveness of rainwater tanks in urban Australia*, National Water Commission.

Marsden Jacob Associates (2007b). *The economics of rainwater tanks and alternative water supply options*, Camberwell, Victoria.

Rein, F. (1999). An Economic Analysis of Vegetative Buffer Strip Implementation Case Study: Elkhorn Slough, Monterey Bay, California. *Coastal Management* 27(4): 377-390.

Retamal, M., Glassmire, J., Abeysuriya, K., Turner, A., and White, S. (2009). *The Water-Energy Nexus - Investigation into the Energy Implications of Household Rainwater Systems, Final Report*. Institute for Sustainable Futures, University of Technology Sydney.

Young, M. (2000). Managing Externalities: Opportunities to Improve Urban Water Use, CSIRO Urban Water Program.

Understanding Uncertainty in Multi-Objective Decision Analysis: the Water Planning Context

Moglia, M., Sharma, A. and Maheepala, S.
CSIRO Land and Water, Highett, Victoria

Introduction

Planning of regional and urban water resources, and in particular Total Water Cycle Management or Integrated Urban Water Management approaches, often considers inter-relationships between human uses of water, the health of the natural environment as well as the cost of various management strategies. In other words, it has social, environmental and economic goals. The types of strategies employed can include water efficiency measures, water sensitive urban design, stormwater management, or catchment management. In other words, it involves choosing between a range of scenarios and evaluating them against a number of criteria. This type of problem has a discipline devoted to it, ie, Multi-Criteria Decision Analysis (MCDA), which has often been applied in water management contexts (Hajkowicz and Collins, 2007). The decision problem can also be approached using cost-benefit analysis methods, which is perceived by some to be on more solid theoretical foundation, especially because it does not inappropriately mix units in the analysis (Dobes and Bennett, 2009). Some researchers have approached the MCDA problem from an optimisation point of view, evaluating a range of strategies (S_i) using a mixture of assessment techniques for a range of outcomes (O_j) against a number of criteria (C_k). Analysis is often based on the concept of Pareto optimality and applying various types of linear programming algorithms. This is often challenging from the point of view of mathematical formulation, and uncertainty is not typically explicitly represented.

A critical concern of MCDA is how to prioritise different outcomes against each other. Priorities are introduced in different manners, such as by: assigning weights to different outcomes (as in weighted sum models, Analytic Hierarchy Process, goal programming etc); or pairwise comparisons between strategies for each outcome, as for example in applications using PROMETHEE (Behzadian *et al.*, 2010) or Analytic Network Process (Saaty, 2005). These tasks can be done either by the analyst, or via some method for eliciting judgments from groups (Proctor and Drechsler, 2006), which may be time consuming and costly. As this is a subjective task, it is absolutely critical to consider who is involved in setting priorities and assigning weights. Whilst there are some concerns with MCDA, there are also benefits in that it (Hajkowicz and Collins, 2007): helps structure a decision problem and provide transparency and accountability to decision procedures; may help in conflict resolution; may support multi-stakeholder engagement and community participation; and helps insure that ensure the analysis is logical and robust which can assist with auditability. It is sometimes also considered that MCDA has an advantage over cost-benefit analysis because strictly non-monetary issues can be considered.

This paper describes an application of a new method for undertaking MCDA using Bayesian Network (BN) analysis and Subjective Logic (SL) that hopefully resolves some of the concerns of MCDA.

Keywords
Multi-Criteria Decision Analysis, Bayesian Networks, subjective logic, water planning.

Theory

Bayesian Networks are based on what is called plausible reasoning which is in turn based on probability theory, and in particular the theory of conditional probabilities (Castelletti and Soncini-Sessa, 2007). Computationally, Bayesian Networks are made up of a series of nodes (representing factors) and directed arcs (representing causal links between factors) creating an acyclic graph. Causal links are represented by conditional probabilities, and the repeated application of Bayes theorem using such conditional probabilities generates a system of equations which can be solved to provide the probability distributions of all model factors. This study has applied Bayesian Networks within the software environment known as *Metafuze*. Unlike other BN software systems, this software is based on subjective logic for which all conditional probabilities are taken as "beliefs" or "opinions", dealing with the aspect of the human condition that nobody can truly tell whether a proposition about the world is true or false. It asserts that whenever the truth of a proposition is assessed, it is always done by an individual and no belief can be considered to represent a general or objective belief (Jøsang, 2001). Subjective logic is based on set theory applied to the set of possible outcomes of a random event. It is assumed that only one outcome/state can be occur at any one time and it assigns belief to different outcomes. Any given outcome in the state space has a belief, disbelief and a level of uncertainty. Intuitively, the belief represents the probability space for which we know a particular event is occurring, whilst the uncertainty represents our lack of knowledge of a situation. Judgements are combined using the *consensus* operator, taking as input two propositions. Each proposition is described mathematically with the belief, disbelief, uncertainty and the probability expectation value. The effect of the consensus operator is to reduce the uncertainty. Similarly, reliability of information is considered via an operator that considers the uncertainty of a proposition

provided by source A (which may be a person or a report, etc), on the basis of the opinion of person B on the reliability of source A. The advantage of the subjective logic approach is that it can consider the uncertainty of the evidence. This enables assessments that reflect data availability, highlighting the uncertainty created by lack of reliable data.

Method

The approach in this paper is a mix between MCDA and cost-benefit analysis. Thresholds are chosen that define the acceptable outcome space (ie, MCDA), and within this space the statistical expectation value of the monetary value of each strategy is calculated (ie, cost-benefit analysis and assigning monetary values to outcomes is in many ways equivalent to assigning weights). The approach explicitly models two types of uncertainty: uncertainty in variable outcomes; and varying levels of reliability of the underlying analysis. The mathematical formulation is done via model design using *Metafuze* software, which supports BN analysis with SL. This methodology is still under development, but when it is fully fledged and mature it will be comparable and complementary to existing approaches. This paper will describe the theory of the approach only in general terms.

An existing case study was used to define the scenario, indicators and measures (Sharma *et al.*, 2009). The uncertainty of the information was assessed and the effect on decision making considered using BN. Reliability of information was modelled using SL. The indicators are presented in Table 2. Values on indicators were evaluated using water balance modelling (WBM), lifecycle assessment (LCA) and lifecycle cost analysis (LCC) (Sharma *et al.*, 2009). A standard deviation was also assigned to each indicator (except the total cost), modelling these indicators as normal distributed stochastic variables. The total cost was modelled using a stochastic variable with uniform distribution on the basis of a specified range (ie, min, max). Next, these indicator variables were converted from continuous stochastic variables to discrete stochastic variables (meaning each indicator variable has a limited and discrete number of possible outcomes). These discrete steps represent ranges of indicator variable outcomes (for example one discrete state of the total cost may be a value between $700,000 and $710,000). Each indicator variable outcome was assigned a monetary value and measures were dollars. The translation of different measures into dollars also has uncertainty, but this is not yet modelled. This type of cost-benefit analysis provides a way for multiple agencies and stakeholders to find a consistent price on mitigation strategies (say for example regarding pollution of nitrogen into marine waters) across a broad spectrum of actions. As per Dobes and Bennett (2009), there is also a more solid theoretical foundation for cost-benefit analysis. The reliability of calculations was assigned on the basis of pre-defined rules. Reliability was assigned to scenarios as well as estimation methods (ie, WBM, LCA and LCC). Final reliability for a given indicator value distribution for a given scenario (ie, for example potable water use for scenario A) is calculated via multiplication of the reliability of WBM (the assessment technique used for calculating the distribution of the indicator value) multiplied with the reliability for scenario A.

Results

The methodology involves an adaptation of common steps in construction of BN models (Bromley, 2006). The study involves an evaluation of options for urban water services in a greenfield development in Kalkallo, 35 kilometres north of Melbourne CBD. The development is projected to have a population of 86,000 people, with commercial and industrial sites included as well. The study explored trying to find a more sustainable way to provide urban water services. Kalkallo was chosen for exploring a range of urban water servicing options considered by planners and decision makers as alternatives to traditional centralised approaches, namely: A) conventional centralised reticulated supply and wastewater (base case); B) water supplied via a third pipe system for reclaimed water for non-potable applications; C) water supplied via a third pipe system using stormwater for non-potable applications; and D) self-contained servicing configuration drawing upon on-site greywater reuse, rainwater tanks, and on-site wastewater treatment systems. These strategies/scenarios are also explored in this paper. Each strategy (A-D) is evaluated for its achievement of water planners' goals, via LCA, LCC and water balance analysis, as shown in Table 1.

Table 1. Decision variables in case study model.

Assessment approach	Variable	Development-specific objective	Mandated (M) or Benefit-adding (B)
Water balance	Potable water usage	Minimise potable water usage	B
	Wastewater flow	Minimise flows and maximise reuse	B
	Stormwater flow	Minimise flows and maximise reuse	B
	Availability of water supply	Reliability of water system	M
Lifecycle assessment	Energy and material usage	Minimise greenhouse gas emissions	B
	Nutrient flows	Eutrophication	B
	Solid waste	Minimise waste generation	B
Lifecycle cost	Total cost	Minimise LCC	M

Table 1 describes the variables in the model. These goals need to be categorised by the water planner or decision maker as either mandated or value adding (as they are done in the last column of Table 1). Mandated goals are those that need to be fulfilled in order for a strategy to be considered at all. In this case, there is a basic standard of service standard required (availability of water supply), and the system needs to be affordable (relating to total cost). The other variables are value-adding, in the sense that a monetary value can be given to each outcome, and the ultimate aim is to maximise the value of a strategy.

The next step in the methodology is to define the possible outcomes of the variables. For example, if a variable is representing an objective that is described in terms of a threshold (as is typically the case for mandated objectives), then the variable would have the following outcome space: A) objective has been achieved; and B) objective has not been achieved. However, in many cases there may also be a grey area for which it is difficult to establish whether the objective has been achieved. In such cases, the variable may have an additional possible outcome, ie, C) unclear whether objective has been achieved.

For goal variables that are considered to be "benefit-adding", a monetary value is being assigned for the achievement of different outcomes. For example, this may be done on the basis of cost analysis of various mitigation strategies for reducing nitrogen loads into marine waterways. Value-adding variables typically have more than two possible outcomes, but the computational complexity increases and up to 20 states per variable have been observed to considerably slow down the *Metafuze* software. The process to define possible outcomes needs to balance the need to minimise computational complexity against the need to ensure that models are meaningful.

Quantifying Model

Quantification of the models involves calculating the outcomes of strategies. In MCDA, as applied by Sharma *et al.* (2009), this is done by calculating a single value for each combination of scenario and outcome variable – hence representing outcomes in a purely deterministic sense. The reality however is that outcomes are uncertain for many scenarios due to various reasons. For example, cost estimates for planning purposes may range in the order of plus/minus 25% or more. Such uncertainty would be best represented using probability distributions and may for example be described using a normal distribution (or other distribution) with parameters, ie, mean, and a standard deviation chosen in order to keep 95% of the probability mass within the range of plus/minus 20% of the mean. Uncertainty in the variables would be provided by either sensitivity analysis or sensitivity assessment by judgment.

Sensitivity analysis: many types of assessment techniques used in water planning, including Lifecycle Analysis (Pamminger *et al.*, 2008) and water balance modelling (Mitchell *et al.*, 2001), are applied within a framework that explores sensitivity to input (such as assumptions and parameter values). Sensitivity analysis is likely to provide as output, mean values and standard deviations, which in conjunction with the assumption of normal distribution, mathematically specifies the discrete probability distribution required as input to the analysis (ie, calculating the probability mass of discrete states in the Bayesian Network model). Other types of distributions can also be accommodated.

Sensitivity assessment by judgment: it is often difficult to gauge the sensitivity of an assessment technique by means of mathematically formulated sensitivity analysis. In such cases, judgment has to be obtained in the most reliable way. Unfortunately, humans are in general poor at such judgments, but will nonetheless be likely to generate some ballpark estimates of variability. The person that makes such judgments needs to be chosen on the basis of his/her experience, insights and knowledge. It is suggested that in these cases judgments take the form of a range of values (a pair of numbers: maximum and minimum), in combination with a statement whether this is an absolute range, or whether it is at all feasible that values fall outside of this range. Ideally, the range is provided by a group of experts where each expert is providing a range. There is also a need to make a judgment about whether to use a uniform, normal or other distribution to describe uncertainty.

Table 2. Values calculated for each combination of variable and scenario.

	Scenarios			
Variables	**A**	**B**	**C**	**D**
Potable water usage (ML/year)	14,439	8,663	8,663	4,332
Wastewater flow (ML/year)	11,420	5,682	9,006	5,422
Stormwater flow (ML/year)	9,731	9,731	6,407	7,301
Availability of water supply	100%	98.52%	98.52%	95.36%
Greenhouse gas emissions (000's tonnes CO2eq per annum)	23	20	23	17
Nitrate to bay (Kg/year)	83,000	54,000	62,000	46,000
Solid waste (tonnes/year)	116,457	118,786	112,280	93,165
Total cost (000's $)	647,030	722,120	722,390	1,046,900

There is no information about uncertainty for the variables in the case study of this paper. So, for the purpose of illustration and in the absence of rigorous sensitivity analysis, the uncertainty of all variables in the case study model are based on 'sensitivity assessment by judgment'. Using the values provided by Sharma *et al.* (2009), all scenarios were evaluated for each of the criteria, as per Table 2. Furthermore, it is necessary to have some measure of the uncertainty in each estimate. For this purpose, standard deviations, or ranges of values, have been assigned arbitrarily to demonstrate the methodology to each variable as per Table 3. This information would normally come from either sensitivity analysis or by eliciting expert judgment (preferable multiple experts in a structured process).

Table 3. Standard deviations or ranges for each combination of variable and scenario.

	Scenarios			
Variables	**A**	**B**	**C**	**D**
Potable water usage (ML/year)	1,444	722	722	722
Wastewater flow (ML/year)	1,400	700	1,000	500
Stormwater flow (ML/year)	1,000	1,000	800	800
Availability of water supply	(99.9%; 100%)	(97.5%; 100%)	(97.5%; 100%)	(93%; 100%)
Greenhouse gas emissions (000's tonnes CO2eq per annum)	5	4	3	3
Nitrate to bay (Kg/year)	4,150	3,320	4,150	4,980
Solid waste (tonnes/year)	5,823	5,823	5,823	5,823
Total cost (000's $)	(600k; 700k)	(650k; 770k)	(650k; 770k)	(900k; 1,100k)

The information in Table 2 and Table 3 can also be translated into the probability mass in each discrete variable state. These calculations can be carried out in a regular Microsoft Excel spreadsheet with the guidance of any basic text book on probability theory.

Table 4. Values and assumptions used in estimation of monetary values.

Variable	**Unit cost**	**Assumptions**
Potable water usage	$1.35 per kL	Each person uses 155 litres Average 2.2 person per household 8 years payback* 30,000 households
Wastewater flow	$1.296 per kL	4 years payback time*
Nutrient load	$800/kg	Including from wastewater and stormwater Nitrogen load only 1 year payback time
Stormwater flow	$0.20 per kL	4 years return on investment
Energy and material usage	$50 per tonne of CO2eq	8 years payback time*
Solid waste	$40 per tonne of landfill	2 years payback period*

*** Note:** this is based on discounted cash flow analysis.

Assign Monetary Values to Variable States

The variables that are considered to "add benefit" are typically those that stakeholders would pay some price to either reduce or increase their values. For many of these, there is a range of strategies that can be employed. For example, in the case of eutrophication, each strategy may reduce nitrogen going into marine and freshwater environments. This reduction represents a value based on a specified cost per unit (for example a price on kilograms of nitrate going into waterways). A reasonable strategy for a policy maker (with the responsibility to reduce eutrophication) is then to select a portfolio of strategies that would achieve targets at lowest cost. The monetary values for the outcome space were calculated on the basis of the figures and assumptions in Table 4 (chosen on the basis of discussion with industry colleagues and review of grey literature). It is acknowledged that there may also be uncertainty in the assignment of monetary values, but this is not yet considered.

Reliability of Information

Uncertainty in outcomes is evaluated with the above method, but reliability of assessments is still to be evaluated. For example, to what extent can a given LCA analysis calculated for Scenario B be trusted? In subjective logic, all probabilities are seen as beliefs, and as such, each probability has an assigned level of certainty. Assigning the level of certainty to probabilities is a difficult task: imagine for example a doctor who wants to know the level of certainty of his judgment for a patient that he/she has most likely contracted a virus infection. With historical data, say a diary of judgments and descriptions of patients, he may be able to estimate what percentage of the time that he is correct. Therefore, whilst there are ways for objectively assigning certainty of judgments (where judgments are probabilities), there is typically not enough data to do this. Instead, one has to rely on rules for assigning the level of certainty to estimates. In the case study of this paper, such rules can be gleaned in the notes of Table 5 and Table 6. For example, assessment methods are assigned reliability on the basis of:

- Where it was published: A-grade journal (99%), B-grade journal (90%), C-grade journal (75%), Below C-grade journal (50%), and not published (25%);
- Extent of sensitivity analysis: rigorous (99%), limited (90%) and none (50%); and
- Level of scrutiny of data sources: only highly scrutinised data sources (99%), mostly highly scrutinised data sources (90%), well educated guesses by experts (75%), well educated guesses by non-experts (50%) and entirely hypothetical (25%).

Similarly, scenarios are evaluated on the basis of similar rules:

- Prior occurrence of scenarios: mainstream (99%), niche application (90%), mature pilot-testing (80%), early pilot testing (60%) and novel (40%); and
- Extent of description and definition of scenario: completely defined (99%), highly defined (90%), defined (80%), somewhat defined (60%) and vague (40%).

For the assessments in the case study application of this paper,Table 5 and Table 6 show assigned reliability and the last column shows the final reliability on the basis of multiplying the values of previous columns. In the analysis, certainty is assigned to probabilities and such numbers are used within a subjective logic framework. The inferred uncertainty (uncertainty + certainty = 1) is used in subjective logic calculus, which impacts on estimates of probabilities as described in the theory section.

Table 5. Reliability information for modelling components.

Model component	Publication and peer review	Scrutiny of data sources	Sensitivity analysis	Reliability value
Lifecycle Assessment	99% (A)*	90% (mostly)	99% (rigorous)	88%
Lifecycle Costing	90% (B)	75% (guesses)	75% (simple)	51%
Water balance modelling	90% (B)	90% (mostly)	90% (limited)	73%

*** Note**: This is based on rules to assign reliability as described above, with classifications abbreviated in brackets, next to numbers. These rules are soon to be published in the full length journal article.

Table 6. Reliability information for scenarios.

Scenario	Prior occurrence	Definition	Reliability value
A	99% (mainstream)*	99% (completely)	98%
B	80% (mature pilot testing)	80% (merely defined)	64%
C	60% (early pilot testing)	80% (merely defined)	48%
D	40% (novel)	80% (merely defined)	32%

*** Note**: This is based on rules to assign reliability as described above, with classifications abbreviated in brackets, next to numbers. These rules are soon to be published in the full length journal article.

Simulation Results

The case study has been structured within a software environment *Metafuze.* Table 7 shows the results of this analysis. The structure of the model includes two branches: 1) the probabilistic summation of (the $-value of) all those variables that are assigned a monetary value (for each possible outcome); and 2) calculation of a joint probability of achieving all thresholds for mandatory variables. The first column shows the joint probability of achieving all mandated goals, and the second column shows the probability of achieving half of all goals (these hence refer to the second branch of the model tree). The statistical expectation values of the benefits (in $'s) of strategies are shown in the fourth column (ie, referring to the first branch of the model tree). The reliability of assessments (ie, the extent to which you should trust them) is shown in the last column.

Table 7. Results of running Bayesian Network simulation model

Scenario	Prob(all goals)	Prob(>50% of goals)	Expectation	Reliability of assessments
A	54%	95.1%	$22.5M	71%
B	10%	68.8%	$63.6M	72%
C	25%	81.7%	$52.6M	53%
D	9%	66.7%	$64.1M	53%

Note: Fulfillment of goals is to be thought of in accordance with the following set of rules: 1) If goals are in the "grey-zone", then they are 50% fulfilled, if they are fully fulfilled, they are 100%, and if not fulfilled, then they are 0% fulfilled, and 2) the number of goals fulfilled in accordance with this is the "average" level of fulfillment. For example if there are two goals only, then 50% achievement of goals either means that one goals is completely fulfilled, and the other is not at all fulfilled; or that both goals are 50% fulfilled (ie, in the grey zone). Depending on how the model has been set up in this case, the software calculates the probabilities of 0%, 25%, 50%, 75% and 100% goal fulfillment.

Strategies ought to be ranked on the basis of firstly having an acceptable chance of achieving mandated goals, meaning that they are feasible, and secondly to rank on the basis of the expectation of the value of the strategy. For example, if one uses as a feasibility criteria such as "at least 70% chance of achieving more than half of the goals", then one would choose between A and C, and then select strategy C on the basis of a higher expectation value ($52.6M vs. $22.5M).

Summary and Discussion

This paper shows the application of a novel methodology to a type of decision problem that often occurs in urban water management, ie, choosing different strategies with different outcomes. In comparison to the original case study, the methodology that is described in this paper brings explicitly considers uncertainty and reliability of information within the analysis. This has been done on the basis of: 1) using existing assessment values for each decision criteria; 2) rule-based assessment of the reliability of assessments and scenarios; and 3) adding judgments of levels of uncertainty for each criterion. These are based on some understanding of level of uncertainty, but are still somewhat informed guesses and hence cannot be considered to be fully accurate. The decision problem has also been structured in a new way, in particular with the division of criteria in two categories: benefit-adding, and mandatory.

Does this type of analysis qualitatively change the recommendation for the decision (ie, in comparison to previous study)? Based on the fourth column in Table 7, Scenario D is the best in terms of expectation of added value, whilst the Scenario A is best in terms of achievement of mandatory goals. Interestingly however, Scenario B seems to have a better chance of achieving mandatory goals and also has a considerably higher expected benefit value (fourth column in Table 7) than the traditional scenario (A) and should perhaps be considered for implementation on the basis of being a compromise between the chance of added benefit and reliability. The original paper (Sharma *et al.*, 2009) did not specify a preferred solution but noted that scenario D was the preferred option for 6/9 criteria but may have essentially been disqualified on the basis of results in this paper, whilst scenario B appeared the best solution from a community cost perspective. So, the difference in the results is still dependent on the decision makers preferences, but it seems that the method described in this paper makes the risk (of not achieving mandated thresholds) more apparent. The output of the methodology in this paper allows decision makers to select an appropriate level of risk, and to balance the risk against an estimated likely benefit.

Uncertainty of Novel Technologies

Much of the uncertainty in the assessed reliability of decentralised systems in this paper comes from the fact that approaches are not commonly applied, which is represented by a lower reliability on numbers in the subjective logic framework. Conversely, one of the key reasons for knowing that traditional systems are reliable is that they have been tried and developed at a large scale. There is also a high level of uncertainty in assessments due to inadequate sensitivity analysis and sometimes inadequate scrutiny of data sources and limited availability. Refining assessment tools will lead to a higher level of certainty in assessment, and this in combination with further large scale uptake of decentralised systems is likely to reduce the uncertainty in the assessment of the reliability of decentralised systems.

It seems that the methodology of this paper "punishes" new technologies more harshly than does MCDA as was described by Sharma *et al.* (2009). This comes from the assumption that we don't know as much about the outcomes of novel technologies. There is an expectation that this higher level of uncertainty is also very much on the minds of decision makers, either consciously or subconsciously.

Being able to include and explicitly explore this type of uncertainty may help in making informed decisions in untested areas. In particular, the approach highlights the uncertainty involved with strategies, something which is usually not explored in MCDA. Humans are also not cognitively well equipped to make ad-hoc judgments of uncertainty. Well defined and well analysed uncertainty is much preferable to poorly defined uncertainty, because risk can then be handled and mitigated. Without formally considering uncertainty, decision makers are often risk averse. In the urban water sector, such risk aversion is particularly likely to lead to low uptake of promising by potentially risky new technologies.

Conclusions

The main contribution of this methodology is in the definition of a complementary approach to MCDA that incorporates uncertainty and judgment in an explicit manner. This approach can be particularly useful in water planning for scenarios assessment, including in Total Water Cycle Management and Integrated Urban Water Management. The extra information required in order to apply this methodology is in providing the probability distribution of values of each goal criteria, as well as judgments on reliability for each assessment and scenario. Uncertainty in goal criteria can be specified via specification of the parameters of a probability distribution, for example, mean and standard deviation in a normal distribution, or even directly the probabilities assigned to states in a discrete probability distribution. Reliability information is assigned on the basis of relatively simple but well considered rules and classifications. Finally, monetary values need to be assigned to possible discrete outcomes of each goal criteria.

It is completely plausible that this type of information is available to decision makers. In terms of results, this methodology highlights where there are information gaps, and risk involved in alternative strategies. This is especially important for mandated goals for which it is important that goal criteria are achieved with a high probability. In those cases, the methodology tends to be "harsher" on novel approaches than well tried approaches. Decision makers are hence faced with an estimated risk of not fulfilling mandated goals, at least in the short term. It is then up to the decision maker to make the informed but risky decision to choose a novel approach, probably in the case of urban water supply on the basis of long term public benefits.

References

Behzadian, M., Kazemzadeh, R.B., Albadvi, A. and Aghdasi, M. (2010). PROMETHEE: A comprehensive literature review on methodologies and applications. *European Journal of Operational Research*, 200 (1)**,** 198-215.

Bromley, J. (2006). Guidelines for the use of Bayesian networks as a participatory tool for Water Resource Management. Wallingford, UK: Centre for Ecology and Hydrology.

Castelletti, A. and Soncini-Sessa, R. (2007). Bayesian Networks and participatory modelling in water resource management. *Environmental Modelling and Software*, 22(8)**,** 1075-1088.

Dobes, L. and Bennett, J. (2009). Multi-Criteria Analysis: "Good Enough" for Government Work? *Agenda*, 16(3)**,** 7-29.

Hajkowicz, S. and Collins, K. (2007). A Review of Multiple Criteria Analysis for Water Resource Planning and Management. *Water Resource Management*, 21(9)**,** 1553–1566.

Jøsang, A. (2001). A Logic for Uncertain Probabilities. *International Journal of Uncertainty, Fuzziness and Knowledge-Based Systems* 9(3)**,** 279-311.

Mitchell, V.G., Mein, R.G. and McMahon, T.A. (2001). Modelling the urban water cycle. *Environmental Modelling and Software* 16(7)**,** 615–629.

Pamminger, F., Grant, T. and Sharma, A.K. (2008). Sustainability of Urban water infrastructure advanced by quantifying risk, environmental impacts and community costs. *Enviro08.* Melbourne, Australia: AWA.

Proctor, W. and Drechsler, M. (2006). Deliberative multicriteria evaluation. *Environment and Planning C: Government and Policy* 24(2)**,** 169 – 190.

Saaty, T.L. (2005). *Theory and Applications of the Analytic Network Process: Decision Making with Benefits, Opportunities, Costs, and Risks*, Pittsburgh, Pennsylvania, RWS Publications.

Sharma, A.K., Grant, A.L., Grant, T., Pamminger, F. and Opray, L. (2009). Environmental and Economic Assessment of Urban Water Services for a Greenfield Development. *Environmental Engineering Science*, 26(5)**,** 921-934.

Water-Related Energy and Greenhouse Gas Emissions in Cities - Key Emerging Messages

Kenway, S.[1], Lant, P.[1], Bader, H-P.[2], Larsen, T.[2], Scheidegger, R.[2], Priestley, A.[3], Keller, J.[1], Daniels, P.[4], Lenzen, M.[5] and Lane, J.[1]

[1] The University of Queensland, St Lucia, Queensland
[2] Swiss Federal Institute of Aquatic Science and Technology (Eawag), CH-8600 Dübendorf, Switzerland
[3] CSIRO Land and Water, Highett, Victoria
[4] Griffith School of Environment, Griffith University, Nathan, Queensland
[5] University of Sydney, Sydney, New South Wales

Background

The significance of urban water policy and management on energy use is rapidly clarifying. This document draws on recent PhD research for the Urban Water Security Research Alliance focused on water-energy connections within cities. It distills emerging messages. Further details are available through the referenced papers.

The Australian water sector has long been active in reducing operational (direct) energy use and greenhouse gas emissions. However, existing measures alone are unlikely to cope with the magnitude of forecast growth in energy use necessary to provide urban water services. In Australia this growth is largely driven through urban population growth, and the introduction of more energy-intensive and "climate-resilient" water supplies (Kenway et al. 2008). Greenhouse gas emissions reduction goals will provide an additional imperative to find more substantial improvements, as energy use is the major contributor to the greenhouse gas footprint of the urban water sector (Lane et al. 2011 (In Review)).

More broadly, planning for "water-sensitive" cities has become a priority for sustainable urban development in Australia. Recent Life Cycle Assessment (LCA) studies have helped to understand the extent of environmental issues associated with urban water systems. There are many more environmental externalities associated with urban water planning decisions (Lane and Gardner 2009; Lane et al. 2011 (In Review)) than energy use and greenhouse gas emissions alone. However water-related energy use makes a significant contribution to the environmental footprint across a number of environmental impact categories in LCA studies. Additionally, energy use is one of the most significant differences between the new technologies being considered through the current phase of urban water system diversification going on in Australia (Lane et al. 2011 (In Review)). Despite this significance, there has been little effort invested in quantifying, or even broadly articulating, how cities, together with their urban water systems, influence water- related energy and associated greenhouse gas emissions. This is particularly the case with regard to indirect influences of water on energy use.

Keywords
Water, energy, cities, greenhouse gas emissions, water sensitive cities, urban metabolism.

Methods

The methods used for the component papers summarised here include:

- Review and classification of available literature to extract emerging themes and identify gaps (Kenway et al. 2011);
- Systematic quantification of water-energy connections in an "average" or hypothetical city of one million people using best Australian data augmented with international results where gaps were evident (Kenway et al. 2011 (In press, accepted 6 June 2011));
- Preliminary detailed modelling of (i) water-energy connections in a South East Queensland (SEQ) household and (ii) construction of a multi-regional input-output model of the Queensland economy with the rest of Australia and the world.

Results

To date, key city-scale water-energy connections have not been systematically quantified. This contrasts with the energy use and related emissions of the water sector, where more analysis has been done. The potential of water policy and management to influence urban-scale greenhouse gas mitigation, including from households and industry, has not been evaluated to date.

Lack of knowledge of connections between water and energy in cities is s a major limitation to energy-sensitive urban water management and integrated urban design. Knowledge of water-energy links in the industrial and commercial sectors is a particularly large gap: a review of available literature found only two studies in this domain (Klein et al. 2005; Wolff and Wilkinson 2011). Very few studies consider economic, social or legal perspectives of the water-energy nexus. These are major omissions given Hightower *et al.* (Hightower and Pierce 2008) identify that water security underpins energy security and hence economic security.

Of the studies conducted, few identify key factors and sensitivity and uncertainty analysis have generally not been conducted (Kenway et al. in preparation). Not surprisingly, different factors are identified as "key", without clear data on assumptions or the circumstances under which the conclusions remain valid. Such information is critical for guiding scenario and policy options to have greatest effect. It is paramount that we understand which actions and policies increase emissions, and likewise, those that decrease it, by how much, and where? It is also critical to understand if results hold true for all household types and circumstances. There is a need to move on from the blanket claim that "saving water saves energy", to a detailed understanding of those water-conservation approaches and policy that reduce, and those that may exacerbate greenhouse gas, emissions (Kenway et al. in preparation). Understanding error and natural variability is necessary to guide future monitoring.

Water-related energy use in cities is estimated to account for 13% of total electricity and 18% of the natural gas used by the "average" Australian (Kenway et al. 2011 (In press, accepted 6 June 2011)). This includes provision and use of water in cities, and accounting for energy associated with related nutrient flows (eg, nitrogen lost in wastewater which needs to be resynthesised industrially to provide for agricultural demand) and the "water component" of the urban heat island effect (Kenway et al. Submitted). Collectively this represents 9% of total Australian primary energy demands or 8% of total national greenhouse gas emissions. Consequently significant opportunities exist for reduction of greenhouse gas emissions by addressing the indirect influence of water management on energy use. Residential, industrial and commercial water-related energy use comprises 86% of water-related greenhouse gas emissions. Analysis of the residential, industrial and commercial sectors is recommended as foci of water-related energy investigations.

Within homes, water-related energy includes water heating for showers, baths, taps, clothes-washers, dish-washers, hot water system losses, kettle water, as well as energy for swimming pool filters, rainwater tank and high-rise pumps. Collectively, water-related energy use could account for approximately 35-50% of the of the greenhouse gas footprint of a Queensland home (excluding transport emissions) that is relatively conservative in its water use (Kenway et al. in preparation). A limited number of parameters account for a high proportion of the uncertainty of water-related energy and associated greenhouse gas emissions in households. Consequently, improved monitoring of these parameters will greatly enhance our understanding.

The methodologies observed in use for understanding the water-energy nexus included deterministic, mechanistic modelling and systems-dynamic modelling, lifecycle analysis, input-output modelling, optimisation analysis, monitoring, systems dynamic modelling, scenario analysis, elasticity analysis and household expenditure analysis (Kenway et al. 2011). This diversity leads to many insights; however, it makes comparison between studies problematic. A unifying framework is also missing to help us understand the myriad connections between water and energy in cities, as well as the broader role that cities play in global environmental and resource problems.

A common approach to understanding water-energy links would help know whether management responses are optimising for both water and energy use, or simply shifting problems out of sight of urban water (or energy) managers. Outside the focus of the water-energy nexus, there are many other resource and environmental issues (including nutrient loss). Environmental problem-shifting analysis should take this wider scope into account. While recognising that water planning decisions should ultimately be considering a broader set of environmental externalities than energy use and greenhouse gas emissions, recent attempts to understand water-energy tradeoffs have been constrained by the lack of a systematic assessment protocol. Rectifying this would at least improve our capacity to understand one part of the environmental tradeoffs story, and enable us to build a much clearer picture of the problems that are emerging. Such an approach is also envisaged to better establish the role that cities could play in addressing natural resources problems such as water shortages and elevated greenhouse gas emissions.

The concept of urban metabolism offers high potential. Kenway *et al.* show how application of a systematic mass-balance framework to water can be used to quantify all anthropogenic and natural flows into and out of the urban environment. The approach is used for four major Australian cities to indicate that the full water mass-balance of most Australian cities is not well known. A full water balance includes all inputs of centralised and decentralised water and precipitation, and all outputs including flows to stormwater and groundwater. From such a balance, quantitative indicators of performance can be derived including: (i) degree of system centralisation; (ii) overall balance; potential of (iii) rainfall, (iv) stormwater and (v) wastewater to offset current demand; and (vi) water cycle rate (Kenway et al. 2011 (In press, accepted 16 November 2010)). It appears likely that such a concept could be used as a structure to further quantify the role and influence of cities on wider energy use and greenhouse gas emissions.

Material Flow Analysis (MFA) is a tool highly suited to the analysis of metabolism. The general approach is an extension of methods developed in the economic sector in the 1950s (Leontieff and Stroud 1963) to trace flows of money through the economy. This was later adapted to regional investigations (Baccini and Brunner 1991) and more recently been applied to problem solving for diverse environmental issues (Huang et al. 2007; Kwonpongsagoon et al. 2007; Schaffner et al. 2009). Building on this, Multi-Regional Input-Output (MRIO) modelling also offers much scope for advancing our understanding. Advantages of MRIO models include: (i) systematic tracking of all indirect (supply chain) interactions within a non-duplicative framework; and (ii) flexibility with regard to system boundary.

Conclusions

Urban water is a significant and overlooked lever that has major influence on urban energy consumption. This is particularly the case with regard to the influence of indirect effect water policy and management. Understanding connections between water and energy in cities will identify critical points of influence. Many research gaps exist. Development and application of systematic methodologies is anticipated to be necessary to fully understand water-related energy impacts of cities and the role of their component homes, industries and businesses in influencing water and energy consumption.

References

Baccini, P. and Brunner, P. H. (1991). *Metabolism of the anthroposphere*. Berlin, Springer Verlag. ISBN.

Hightower, M. and Pierce, S. A. (2008). The energy challenge. *Nature*, **452**(20 March).

Huang, D.-B., Bader, H.-P., Scheidegger, R., Schertenleib, R. and Gujer, W. (2007). Confronting limitations: New solutions required for urban water management in Kunming City. *Journal of Environmental Management*, **84**, 49-61.

Kenway, S. J., Gregory, A. and McMahon, J. (2011 (In press, accepted 16 November 2010)). Urban Water Mass Balance Analysis. *Journal of Industrial Ecology*.

Kenway, S. J., Lant, P. and Priestley, A. (2011 (In press, accepted 6 June 2011)). Quantifying the links between water and energy in cities. *Journal of Water and Climate.*

Kenway, S. J., Lant, P. and Priestley, A. (Submitted). Quantifying the links between water and energy in cities. *Journal of Water and Climate.*

Kenway, S. J., Lant, P., Priestley, A. and Daniels, P. (2011). The connection between water and energy in cities - a review. *Water Science and Technology*, **63**(9), 1983-1990.

Kenway, S. J., Priestley, A., Cook, S., Seo, S., Inman, M. and Gregory, A. (2008). *Energy use in the Provision and Consumption of Urban Water in Australia and New Zealand*, CSIRO and Water Services Association of Australia. ISBN 9780643096165. http://www.clw.csiro.au/publications/waterforahealthycountry/2008/wfhc-urban-water-energy.pdf

Kenway, S. J., Scheidegger, R., Bader, H. P., Larsen, T. A. and Lant, P. (in preparation). Understanding water-related energy in a South East Queensland Household.

Klein, G., Krebs, M., Hall, V., O'Brien, T. and Blevins, B. (2005). California's Water-Energy Relationship - Final Staff Report, California Energy Commission.

Kwonpongsagoon, S., Bader, H.-P. and Scheidegger, R. (2007). Modelling cadmium flows in Australia on the basis of a substance flow analysis. *Clean Techn Environ Policy*, **9**, 313-323.

Lane, J., DeHaas, D. and Lant, P. (2011 (In Review)). Life Cycle Assessment of the Gold Coast urban water system (Technical Report).

Lane, J. and Gardner, T. (2009). A life-cycle comparison of decentralised and centralised water cycle options for an SEQ Urban Development. *Ozwater09 Conference - From Challenges to Solutions*, Melbourne Convention and Exhibition Centre, Melbourne, Victoria, Australian Water Association.

Leontieff, W. W. and Stroud, A. A. (1963). Multiregional input-output analysis, in Structural Interdependence and Economic Development, T. Barna, Editor. Macmillan: London, UK. P. 119-149.

Schaffner, M., Bader, H.-P. and Scheidegger, R. (2009). Modeling the contribution of point sources and non-point sources to Thachin River water pollution. *Science of the Total Environment*, **407**, 4902-4915.

Wolff, G. and Wilkinson, R. (2011). Statewide Assessment of Water-Related Energy Use for the Year 2000. California Energy Commission, PIER Energy-Related Environmental Research Program. *CEC-500-2009-100*. Sacramento, Californian Energy Commission.

Methane Emissions from Two Reservoirs in a Steep, Sub-Tropical Rainforest Catchment

Sherman, B. and Ford, P.
CSIRO Land and Water, Canberra, ACT

Summary

The spatial and temporal variability of reservoir methane emissions were measured in Little Nerang Dam and Hinze Dam using floating chambers. A total of 202 flux chamber measurements and 36 vertical profiles of dissolved methane concentrations were collected covering 21 sites in Little Nerang Dam and 20 sites in Hinze Dam. The area-weighted average fluxes of methane were 148 – 341 mg-CH_4 m^{-2} d^{-1} at Little Nerang Dam and 37 – 64 mg-CH_4 m^{-2} d^{-1} at Hinze Dam. Bubbles supplied the majority of each reservoir's emission, with fluxes varying by a factor of 100 between sites and between measurements at a single site. In Little Nerang Dam, 80% of the total flux was emitted from 10% of the reservoir's surface area. In Hinze Dam 60% of the total flux came from 6% of the total area. Reservoir arms receiving fresh organic matter from the catchment had fluxes approximately 100 times greater than arms receiving water that had been previously intercepted by an upstream storage.

Keywords
Methane, reservoir, greenhouse gases, floating chamber.

Background

Greenhouse gas (GHG) emissions from reservoirs has been a topic of high interest to the hydropower industry since the 1990s (Anonymous, 1996; Fearnside, 1995), when the need to quantify the GHG emissions became apparent in order to more objectively evaluate the environmental cost of hydroelectric facilities. In the case of some Amazonian reservoirs, hydroelectric production has been estimated to emit more GHGs than equivalent fossil fuel generation (eg, Fearnside, 2005).

GHG emissions from Australian reservoirs are not well quanitified. Sherman *et al.* (2001) presented measurements from Dartmouth and Chaffey Dams that illustrated the wide range of potential emissions. Additional measurements have been made for some hydropower reservoirs in Tasmania (Bastien *et al.*, 2009) and the Snowy Hydro Scheme (Bastien, pers. comm.). The Tasmanian storages had amongst the lowest CO_2 and CH_4 emissions reported worldwide.

As part of the Life Cycle Analysis Project, Hall *et al.* (2009, 2011) estimated that GHG emissions from SEQ water supply reservoirs comprised nearly 45% of total current GHG emissions associated with the urban water supply cycle. This estimate assumed an emission rate of 80 mg-CH_4 m^{-2} d^{-1} based on observations of CH_4 emissions in some SEQ storages (Little Nerang, Wivenhoe, Borumba) by Grinham (pers. comm.). The range of estimates for SEQ storages varied from 1/2 to 10 times this amount. The apparent high relative contribution of reservoir methane emissions to the overall greenhouse footprint and the high uncertainty of the data upon which the estimate was based motivated the investigation presented here, which aims to provide a better quantitative assessment of: the average methane emissions from a reservoir; and the intrinsic variability of those emissions both spatially and temporally across seasons.

In this study, we focus on methane emissions because this represents the principal modification to the global warming potential arising from the construction of water reservoirs. Emissions of CO_2 and CH_4 from a reservoir occur as a result of the remineralisation of organic carbon (OC) by reservoir biota. The source of the OC is initially CO_2 which has been converted to plant matter by vegetation in the reservoir's catchment or by algal and plant photosynthesis within the reservoir itself. A portion of this plant matter is subsequently incorporated into animal matter. Catchment organic matter, primarily leaf litter and eroded soil, is deposited in the reservoir mainly by catchment run-off although some direct atmospheric deposition is possible. Within the reservoir, microbes consume the organic carbon and emit CO_2 and CH_4 as part of their normal metabolic processes.

A reservoir introduces two important changes to the aquatic transport and processing of organic carbon from a catchment: it increases the time available for terrestrial vegetation and soil carbon to be remineralised before being permanently buried in the ocean; and it provides an anaerobic pathway for remineralisation of OC into methane. The latter modification is relevant when the water column is deep enough to allow persistent seasonal thermal stratification (ubiquitous among Australian reservoirs > 15 m deep). Because the OC is formed initially from atmospheric CO_2, reservoir emissions of CO_2 do not necessarily represent a net greenhouse contribution, ie, atmospheric CO_2 may simply have been recycled. By comparison, the high global warming potential of methane (>20 times that of CO_2) means that if just 5% of the organic carbon load is outgassed as CH_4 rather than CO_2, then the global warming impact of the reservoir will double. Note also that some carbon is buried in reservoir sediments

and that we are assuming here that the total amount of carbon buried in ocean, river, and reservoir sediments is independent of the presence or absence of a reservoir, ie, if burial in a reservoir occurs then a corresponding decrease in carbon burial in ocean and river sediments balances it.

A simple conceptual model of methane flux pathways is shown in Figure 1. In this diagram we consider the reservoir as consisting principally of two layers: an oxygenated surface layer (epilimnion) and an anoxic bottom layer (hypolimnion) adjacent to the sediments. Methane is produced in the reservoir's sediments (even if the water column is saturated with oxygen, the sediments will be anoxic a short distance below their surface). A portion of the sediment methane flux, F_{sed}, is emitted directly to the atmosphere as bubbles, F_{bubble}, and the remainder dissolves into the anoxic bottom water where the concentration of dissolved methane $[CH_4]_{hypo}$ generally increases with time during seasonal stratification. This produces a concentration gradient (dCH_4/dz) and the methane diffuses along this gradient towards the surface layer of the reservoir with flux, $F_{gradient}$. Methane consuming bacteria are typically found near the top of the gradient where they receive oxygen from above and methane from below and consume (at a rate F_{biota}) a large proportion of the diffusive flux. Some of the gradient flux makes it to the surface layer, where the concentration $[CH_4]_{epi}$ changes with time to reflect the supply of methane from below and the diffusive flux of methane from the water to the atmosphere, F_{atm}.

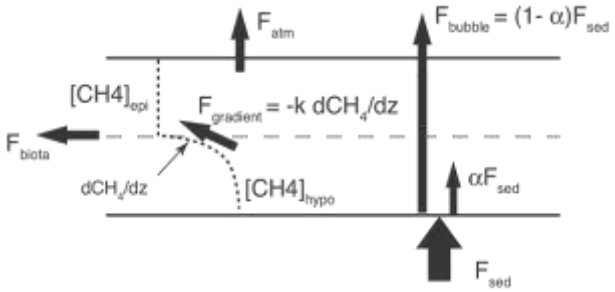


Figure 1. Conceptual diagram of methane flux pathways in a water column.

We refer to F_{atm} as the 'diffusive' flux because it is generally assumed to be controlled by diffusion across thin boundary layers on both sides of the air-water interface. Diffusion across the boundary layer on the water side of the interface has generally been observed to be the rate-limiting step for the diffusive flux and the strength of this flux depends on the concentration difference across the interface (the concentration of gas dissolved in the water - the concentration in the atmosphere) and the thickness of the boundary layer. The thickness of the boundary layer decreases, and so the flux increases, as turbulence in the water increases (Vachon *et al.*, 2010) either due to increasing wind stress or, in the case of rivers, increasing flow which introduces turbulence at the bottom of the water column.

Methods

Methane fluxes and dissolved gas concentrations were measured in Little Nerang and Hinze Dams at the sites shown in Figure 2. We measured the CO_2 and CH_4 fluxes by deploying a floating chamber (FC) on the water surface and recirculating the headspace (the 35-40 L of air contained within the chamber above the water) gas through a Picarro G1301 cavity ringdown spectrometer that continuously measured the concentration of CO_2 and CH_4 during each 15 minute FC deployment. The FC method measures the effect of both F_{atm} and F_{bubble} and we computed the net flux of methane at a site from the observed change in concentration over a defined time interval.

We also measured depth profiles of dissolved CH_4 by collecting samples from various depths in the water column for subsequent laboratory analysis by gas chromatography (GC). All GC analyses were performed by the Water Studies Centre at Monash University. These data provided the concentration of dissolved methane in the surface layer of the reservoirs and allowed the calculation of whole reservoir methane mass balances.

An example of a FC measurement with both diffusive and bubble fluxes is shown in Figure 3. Diffusive fluxes are indicated by the yellow bands and are characterised by an effectively linear change in concentration with time. Bubble fluxes are shown in the blue ovals and are characterised by rapid increases in concentration that produce a 'step' in the concentration time series. To calculate a diffusive flux, a linear regression is fitted to the gas analyser data and the resulting slope is multiplied by the height of the chamber head space and the density of the air (assumed constant at 1.2 kg m^{-3} which is accurate to within 2% for the conditions at the dam).

At sites with a lot of bubble activity, the methane concentration in the chamber could exceed the gas analyser's maximum limit of detection. In such circumstances, the gas analyser output updated irregularly once its limit of detection was exceeded, requiring us to terminate the deployment with the last normal output data point. This process meant that some of the deployments were shorter than the intended 15 minute duration and we had to make some assumptions regarding the likely flux of methane that occurred during the interval between the end of the

measurement (when the instrument went out of range) and the nominal 15 minute deployment. The flux data presented below are denoted as 'scaled' or 'instantaneous'. A 'scaled' flux assumes no methane was emitted after the instrument went out of range and is the most conservative estimate of the flux consistent with the data. An 'instantaneous' flux assumes that the flux computed from the shortened deployment continued at the same rate for the duration of the nominal 15 minute deployment. The true 15-minute flux cannot be lower than the 'scaled' flux but, in theory, it could be even higher than the 'instantaneous' flux.

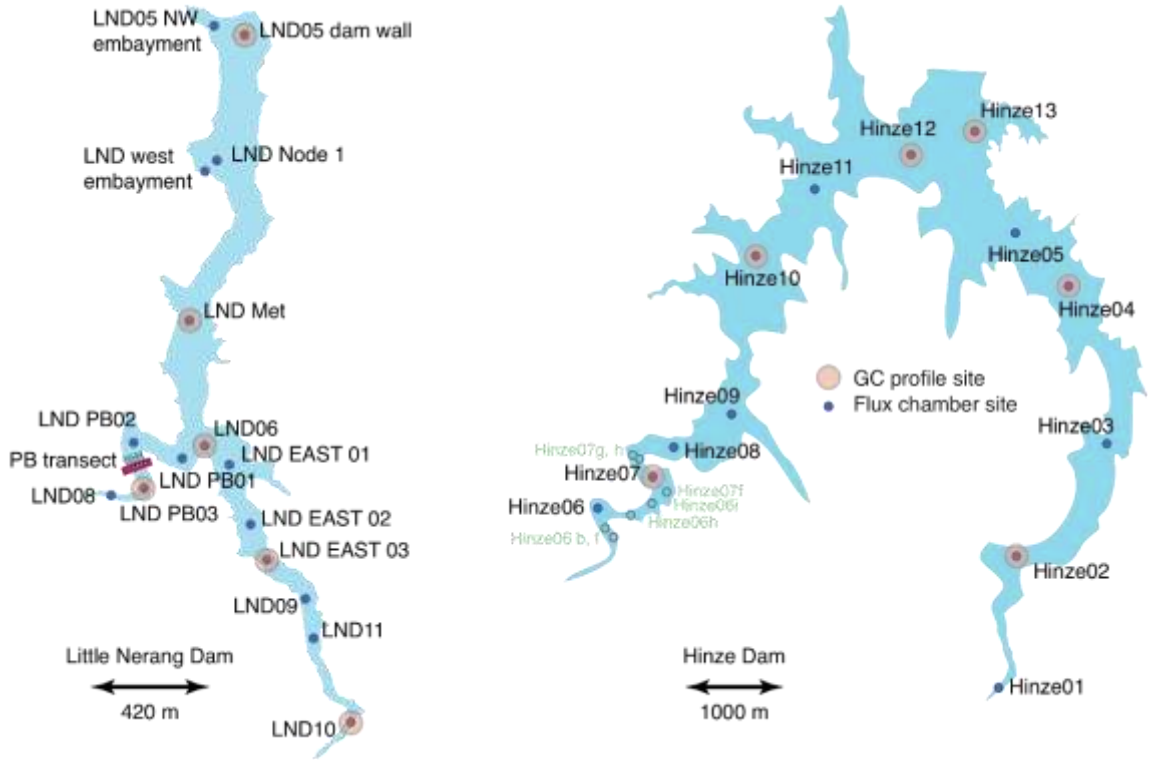


Figure 2. Measurement sites in Little Nerang Dam and Hinze Dam.

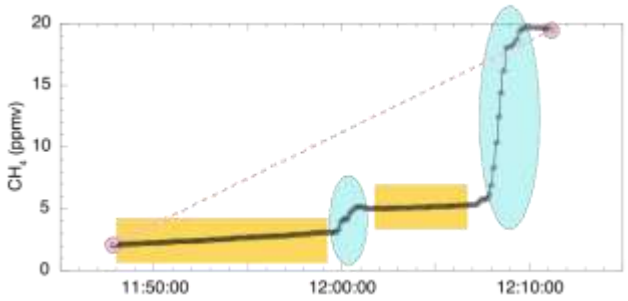


Figure 3. Example of bubbles in a flux chamber measurement. Gold shaded rectangles denote periods of diffusive flux only; blue shaded ellipses show entry of methane containing bubbles into flux chamber. Dashed line links end points used to compute net flux of methane.

Results

Spatial Variability in CH_4 Flux

Both reservoirs exhibited high reservoir-scale spatial variability in methane fluxes (Figure 4 and Figure 5) with fluxes ranging from < 10 to 2,000 mg m^{-2} d^{-1}. Longitudinal gradients in flux were a persistent feature in both reservoirs, with fluxes typically 100 times greater at the upstream ends of the reservoirs where most of the fresh organic matter arriving from the catchment would preferentially be deposited (sites LND08, LND10 and Hinze07). In contrast, the arm of Hinze Dam receiving water from Little Nerang Dam had relatively uniform emissions of roughly 30 mg m^{-2} d^{-1} (sites Hinze01 - Hinze05, Figure 5, 27-28 Feb). In all cases, emissions were lowest in the deep areas of the reservoirs closest to the dam walls (LND05, Hinze12, Hinze13).

Considerable variability was found at smaller spatial scales in both reservoirs as well. In Little Nerang Dam, medium-scale variability around both tributary arms between LND PB03 and LND EAST03 (Figure 6, top) revealed

a 100-fold increase in flux upstream of the confluence of the two tributary arms. Small-scale variability was measured at five sites roughly 20 m apart along a transect (Figure 2, PB Transect) across the western tributary arm. Along this transect, the largest fluxes being measured between the centre of the channel and the outside of the bend (Figure 6, bottom). In concert with the observed large-scale and medium scale variability, the small-scale variability is consistent with the hypothesis that fluxes are greatest in areas expected to experience higher deposition of catchment organic matter.

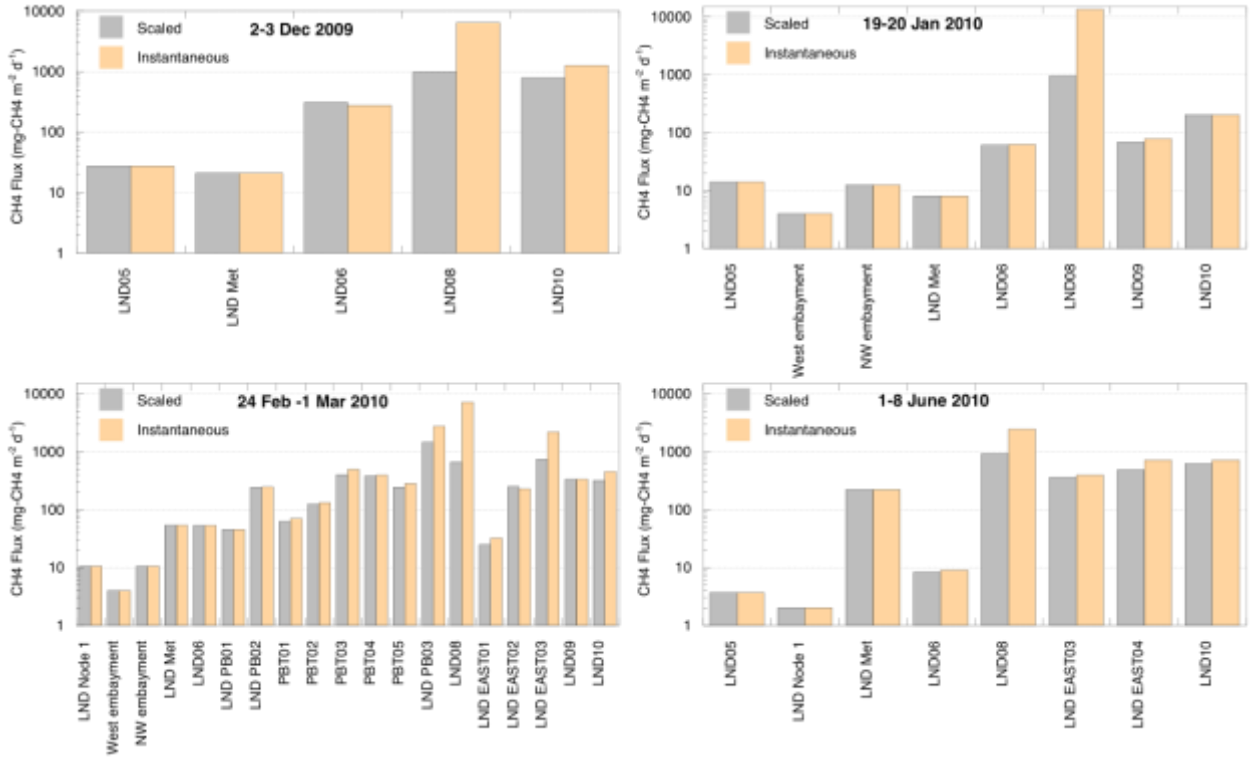


Figure 4. Reservoir scale spatial variability of floating chamber CH_4 flux measurements at Little Nerang Dam grouped by field trip (2-3 Dec 2009, 19-20 Jan 2010, 24 Feb - 1 Mar 2010, 1-8 Jun 2010). Grey bars denote fluxes scaled to 15 minutes. Tan bars denote the instantaneous measured flux. Dam wall is at the left side of each plot, the right side corresponds to the upstream ends of the reservoir.

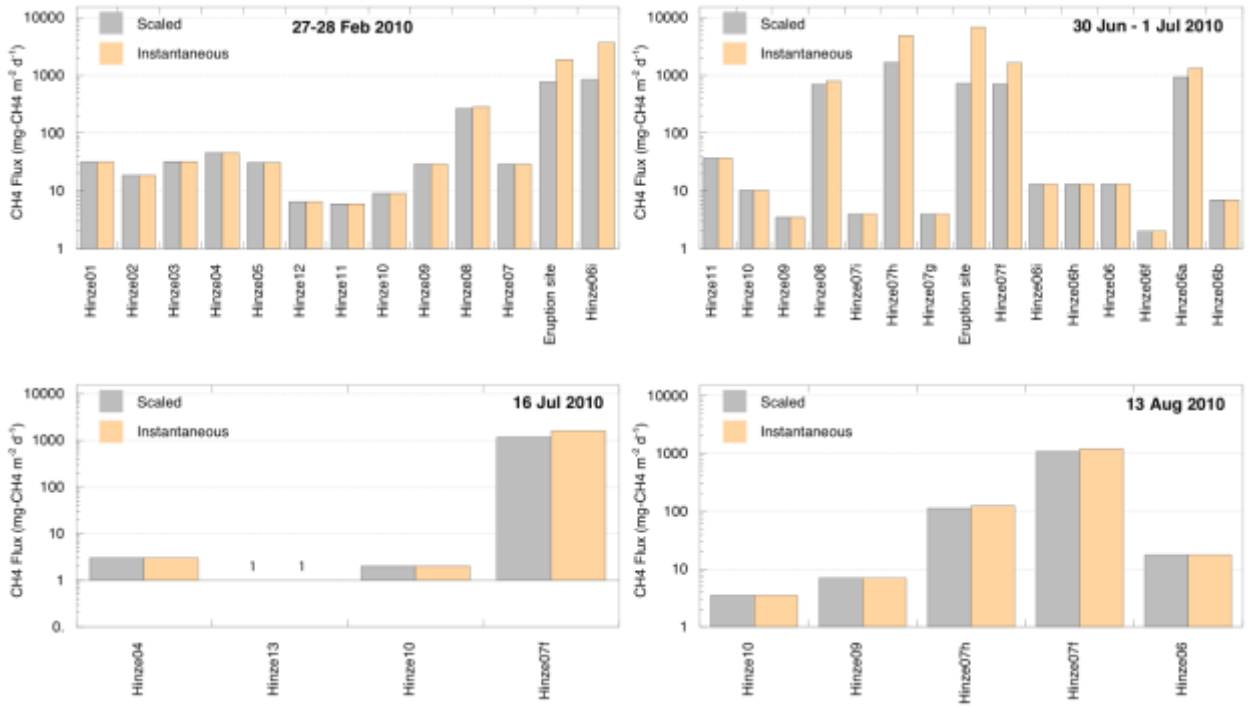


Figure 5. Reservoir scale spatial variability of methane flux measurements at Hinze Dam grouped by field trip (27-28 Feb 2010, 30 Jun - 1 Jul 2010, 16 Jul 2010, 13 Aug 2010). Grey bars denote fluxes scaled to 15 minutes. Tan bars denote the instantaneous measured flux. Dam wall is closest to site Hinze13 and Hinze12. Hinze01 is the upstream end that receives water from Little Nerang Dam. Hinze06 is the upstream end of the arm that receives fresh organic matter directly from the catchment.

Temporal Variability in CH_4 Flux

Short-term temporal variability in flux was measured by performing continuous consecutive chamber deployments at LND10 and LND06 for several hours at each site. Consecutive flux measurements were observed to vary by up to a factor of 10 depending on the level of bubble activity (data not shown) although changes of 10-60% were more common. Similar variability was observed at most sites in both reservoirs. The coefficient of variation (CV = standard deviation / mean) calculated on a daily basis for all sites ranged from 0 to 2.2 with mean 0.5 and median 0.35. The sites with the highest fluxes and persistent bubble activity typically had a CV of 0.6 - 0.85.

No significant seasonal trend in fluxes was detected at any of the sites in Little Nerang Dam, although the smallest fluxes measured at the dam wall (LND05) occurred in June 2010. The combination of episodic storm events causing Little Nerang Dam to spill followed by the rapid drawdown of the storage due to the failure of the Gold Coast desalination plant led to large changes in the volume and composition of the water in the reservoir. These factors may have confounded the detection of seasonal trends in the data. Data collected at Hinze Dam focused on winter conditions and were not suitable for seasonal trend detection.

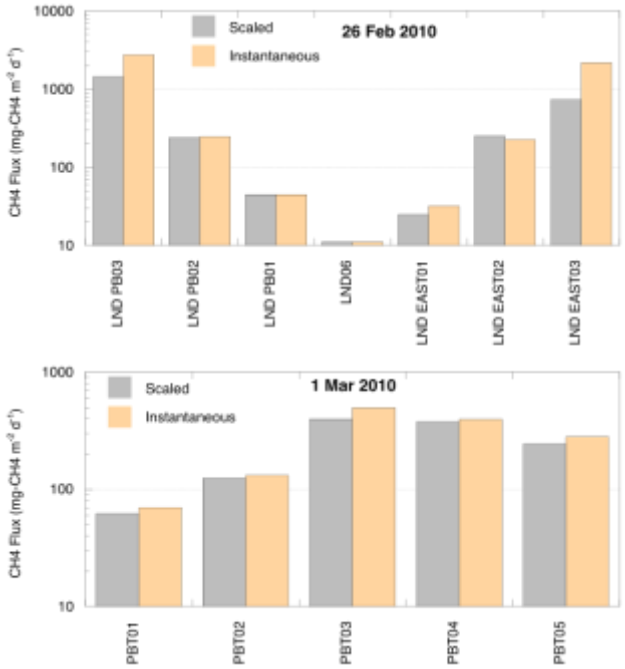


Figure 6. Small - medium scale spatial variability of methane flux in Little Nerang Dam along the two tributary arms (26 Feb 2010) and across the western tributary (1 Mar 2010). Grey bars denote fluxes scaled to 15 minutes.

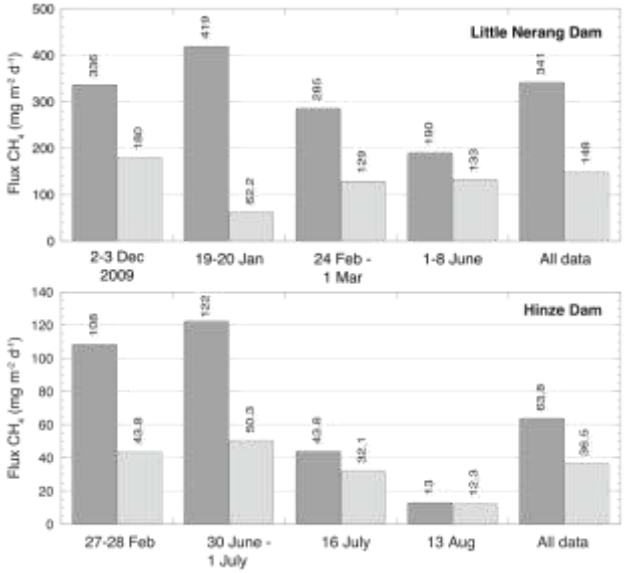


Figure 7. Areal mean CH_4 fluxes measured at Little Nerang Dam (top) and Hinze Dam (bottom). Estimated whole reservoir fluxes are shown for each field trip and for all data. Dark bars represent 'instantaneous' flux measurements whereas light bars represent the most conservative 'scaled' flux values.

Whole Reservoir Fluxes

The average reservoir-wide flux was computed for each trip and for all data by weighting each site's flux measurements by a corresponding fraction of the reservoir's total surface area. At Little Nerang Dam, the average methane flux over the four field trips was in the range 148 (scaled) – 341 (instantaneous) mg-CH_4 m^{-2} d^{-1}. There was little variation in the average flux between the four field trips (Figure 7). At Hinze Dam, the mean flux was in the range 37 – 64 mg-CH_4 m^{-2} d^{-1}. The Hinze Dam calculation is probably biased low for two reasons: a bias towards winter measurements and very limited spatial resolution necessitating extensive extrapolation. Three of the four field trips occurred in winter with much lower fluxes in the deep basin close to the dam (Hinze10 - Hinze13) during the final two field trips than were observed during late summer and early winter. Limited finances allowed only one day of flux chamber measurement and one day of water column sampling per field trip during this phase of the project. Given the need to resolve as best as possible the high flux regions in the western arm of Hinze Dam, no sampling was undertaken in the eastern (Little Nerang Ck) arm and flux values from the deeper sites were assumed to apply from Hinze11 through Hinze01.

Fate of Stored Methane

The fate of methane that accumulated in the hypolimnion of Hinze Dam during seasonal stratification, $[CH4]_{hypo}$, was estimated using a mass balance approach by multiplying the mean concentration at each depth by the corresponding volume of the reservoir for that depth band. Lateral gradients in dissolved methane concentration were negligible. The total mass of dissolved methane in Hinze Dam decreased from 37.4 tonnes to 4.4 tonnes between 1 July 2010 and 12 August 2010 as the surface layer of the reservoir progressively deepened during winter cooling. The corresponding areal loss rates were 166 mg-CH_4 m^{-2} d^{-1} during 1-15 July 2010 and 45 mg-CH_4 m^{-2} d^{-1} during 15 July - 12 August 2010. These fluxes are higher than the corresponding FC measurements during the final two field trips but within the range measured during the first two field trips (Figure 7).

The change in total mass of dissolved methane between field trips must be attributed to atmospheric outgassing and biological consumption. The epilimnion concentration of methane was less than half of the threshold concentration required for bacterial uptake in Lake Constance measured by Schmidt and Conrad (1993), which suggests that the loss of dissolved methane was predominantly through outgassing to the atmosphere. Schmidt and Conrad (1993) measured a residence time of 4-6 days for methane in the epilimnion and it is possible that the majority of outgassing occurred during the intervals between field trips if the most recent significant deepening of the surface mixed layer occurred several days before the field trips. This hypothesis is supported by automated water quality profiler data collected by Seqwater (data not shown) which show the major mixing events occurring on 9 July 2010 and 26 July 2010.

Conclusions

The mean reservoir flux of 148-341 mg-CH_4 m^{-2} d^{-1} and individual sites which consistently emit in the order of 1,000 mg-CH_4 m^{-2} d^{-1} ranks Little Nerang Dam amongst the highest methane emitting dams reported worldwide. Three years after filling, the tropical Petit Saut Reservoir emitted, on average, 117 mg-CH_4 m^{-2} d^{-1} (Galy Lacaux, 1999). Huttunen *et al.* (2002) report bubble fluxes up to 660 mg-CH_4 m^{-2} d^{-1} using bubble traps at Lokka Lake in Finland. Hinze Dam is also a relatively high emitter of methane, especially compared to Tasmanian storages (Bastien *et al.*, 2009). Our measured fluxes are consistent with the assumed fluxes used by Hall *et al.* (2009, 2011) and add confidence to their conclusion that methane emissions from water supply reservoirs in SEQ contribute approximately 40% of the total greenhouse gas footprint of the urban water cycle at present.

Persistent high spatial variability with fluxes 10-100 times greater at the upstream ends of the reservoirs resulted in 80% of the total flux emitted from 10% of the surface area of Little Nerang Dam and 60% of the total flux being emitted from 6% of the surface area of Hinze Dam. Qualitatively similar gradients have been reported by Schmidt and Conrad (1993) for Lake Constance. It is clearly a necessity to resolve the areas of high emissions due to bubbles as accurately as possible to provide the best estimate of the average flux from these reservoirs.

We believe the high methane emissions from Little Nerang and Hinze Dams reflect the much different climatic and catchment conditions from other reservoirs where methane fluxes have been measured. The Gold Coast hinterland has dense, subtropical and temperate rainforest vegetation, warmer temperatures and higher rainfall, which all contribute to relatively greater loading of labile organic matter to these storages. Kelly and Chynoweth (1981) demonstrated a linear relationship between organic input and methane production, and exponential temperature response (Q10 = 2.4), and an average return of 47% of organic input as methane in two Michigan lakes. They suggest that in environments with slowly varying temperature, organic input is the most important factor governing methane production and that the microbial community can be expected to adapt rapidly to suit the supply of organic matter. Our results are consistent with their findings.

Methane production is an anaerobic process and there is little practical scope to prevent it in a reservoir because anaerobic conditions will always be found a short distance below the sediment surface even with a well-oxygenated water column. Our observations of persistent very high fluxes (~ 1,000 mg m^{-2} d^{-1}) at site Hinze07f even when the overlying water column was fully saturated with oxygen suggest it will be extremely challenging to prevent bubble fluxes of methane in relatively shallow depths.

However, the accumulation of dissolved methane in the water column can be effectively prevented by maintaining oxic conditions in the water column (Huttunen, 2001). This displaces the oxic-anoxic boundary into the sediments, allowing the biological consumption to occur before the methane has an opportunity to dissolve into the overlying water. Reservoir oxygenation is frequently used to control sediment release of nutrients and reduced forms of dissolved iron and manganese, so the control of methane accumulation could be considered an ancillary benefit of a common water quality management strategy. The value of oxygenation will differ for each reservoir and we expect it to be greatest for deep reservoirs where water is discharged from below the surface layer as the dissolved methane in the discharged water is nearly completely outgassed to the atmosphere within a short distance downstream of the dam (Galy-Lacaux, 1999). At a carbon price of $23 per tonne of CO2-e, the elimination of 100 t of methane outgassing would be worth about $50,000.

Priorities for future research into reservoir methane emissions in Australia should include collecting data from reservoirs in a wider range of catchment and climatic conditions to better define the relationship between organic input and methane flux, and quantifying methane fluxes from natural flowing rivers and from rivers downstream of dams with deepwater discharges to allow more accurate quantification of the human-induced change to the methane cycle.

Acknowledgements

Financial support for this research was provided by UWSRA and the Water for a Healthy Country Flagship. Special thanks to: Laurie Fairall, Deb Gale, and James Udy (Seqwater) and for their valuable logistical support in the field; Danny Hunt and Chris Drury (CSIRO) provided logistical and technical support; Murray Hall, Shiroma Maheepala (CSIRO) for project leadership; Alistair Grinham (UQ) for many useful conversations and photographs; and Mike Grace, Todd Scicluna, Perran Cook (Monash University Water Studies Centre) for GC analysis of the water samples.

References

Anonymous. (1996). Trouble bubbles for hydropower. New Scientist 29.

Bastien, J., A. Tremblay, and A. Scanlon. (2009). CO2 and CH4 fluxes from Tasmanian aquatic systems, Australia. Verh. Internat. Verein. Limnol. **30**: 854–57.

Fearnside, P. M. (1995). Hydroelectric Dams in the Brazilian Amazon as Sources of `Greenhouse' Gases. Environmental Conservation 22: 7–19.

Fearnside, P. M. (2005). Do Hydroelectric Dams Mitigate Global Warming? The Case of Brazil's Curuá-una Dam. Mitigation and Adaptation Strategies for Global Change 10: 675 – 691.

Galy-Lacaux, C., R. Delmas, G. Kouadio, S. Richard, and P. Gosse. (1999). Long-Term Greenhouse Gas Emissions From Hydroelectric Reservoirs in Tropical Forest Regions. Global Biogeochem. Cycles **13**:

Hall, M., J. West, J. Lane, D. De Hass, and B. Sherman. (2009). Energy and Greenhouse Gas Emissions for the SEQ Water Strategy. CSIRO.

Hall, M. R., J. West, B. Sherman, J. Lane, and D. De Haas. (2011). Long-Term Trends and Opportunities for Managing Regional Water Supply and Wastewater Greenhouse Gas Emissions. Environmental Science and Technology 45: 5434–40.

Huttunen, J. T., Hammar, T., Alm, J., Silvola, J., and Martikainen, P. J. (2001). Greenhouse Gases in Non-Oxygenated and Artificially Oxygenated Eutrophied Lakes during Winter Stratification. *Journal of Environmental Quality*, *30*(2), 387-394.

Huttunen, J. T., Väisänen, T. S., Hellsten, S. K., Heikkinen, M., Nykänen, H., Jungner, H. *et al.* (2002). Fluxes of CH4, CO2, and N2O in hydroelectric reservoirs Lokka and Porttipahta in the northern boreal zone in Finland. Global Biogeochem. Cycles, 16.

Kelly, C. A., and D. P. Chynoweth. (1981). The Contributions of Temperature and of the Input of Organic-matter in Controlling Rates of Sediment Methanogenesis. Limnology and Oceanography **26**: 891–97.Schmidt, U., and R. Conrad. (1993). Hydrogen, carbon monoxide, and methane dynamics in Lake Constance. Limnol. Oceanogr **38**: 1214–26.

Schmidt, U., and R. Conrad. (1993). Hydrogen, carbon monoxide, and methane dynamics in Lake Constance. *Limnol. Oceanogr*, *38*(6), 1214-1226.

Sherman, B. S., P. Ford, A. Mitchell, and G. Hancock. (2001). Greenhouse gas emissions from reservoirs - is Australian hydropower environmentally friendly? p. 1–10. In Grimston, J. [ed.] Proc. NZSOLD/ANCOLD 2001 Conference on Dams: Dams - Development, Sustainability and Performance. ANCOLD.

Vachon, D., Y. Prairie, T., and J. Cole. (2010). The relationship between near-surface turbulence and gas transfer velocity in freshwater systems and its implications for floating chamber measurements of gas exchange. Limnology and Oceanography 55: 1723–32.

Environmental Risks and Benefits from the Supply of Purified Recycled Water in the Lockyer Valley

Wolf, L.[1], Moore, C.[1], Rassam, D.[1], Ellis, T.[1], Hartcher, M.[1], Hodgen, M.[1], Morrow, D.[1], Foley, J.[2], Kodur, S.[2], Robinson, B.[2], Silburn, M.[2], Bleakley, A.[3], Arunakumaren, J.[4] and Evans, P.[4]

[1] CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland
[2] Department of Environment and Resource Management, Toowoomba, Queensland
[3] Department of Environment and Resource Management, Water Services South East, Gatton, Queensland
[4] RPS Consultant, Fortitude Valley, Queensland

Summary

With a maximum combined production capacity of 232 million litres (0.232 GL) of Purified Recycled Water (PRW) a day, the South East Queensland (SEQ) Water Grid operates the third largest recycled water scheme in the world and the largest in the southern hemisphere. The additional water volumes are critical to provide supply security during drought conditions, but are under-utilised in wet periods. The option to supply a significant amount (ca. 15-25 GL/a) of the PRW to augment the over-utilised groundwater resources of the Lockyer Valley (approx. 80 km west of Brisbane) is currently being explored in detail by government agencies and the water grid managers. The environmental risks and benefits from the supply of purified recycled water (PRW) are the core subject of an ongoing research project by the Urban Water Security Research Alliance (UWSRA). Water and soil analysis show no risk of spontaneous soil dispersion with any of the soil-water combinations. The additional import of salt from increased irrigation with PRW would not have a significant impact on catchment salt balances but would lead to local reduction of salt in the soil profile if it substitutes more salty groundwater irrigation. A key hurdle to the uptake of PRW is currently the high cost compared to the traditional natural water resources. By employing a systems view perspective, a new understanding of the demand for PRW is proposed, which starts with the environmental requirements and the future cropping patterns. The supply of PRW into the Lockyer Valley provides the opportunity to actively steer the currently overused ecosystem to any desired state. This paper summarises the status quo as of June 2011 on the key environmental risks identified and introduces a framework for modelling the future demand for high quality water imports.

Keywords
Purified recycled water, irrigation, groundwater, modelling, deep drainage, crop type.

Introduction

PRW in SEQ

The water supply security of South East Queensland has been increased recently by the provision of advanced wastewater treatment plants for indirect potable reuse of effluent from urban areas. With a maximum combined production capacity of 232 million litres of purified recycled water (PRW) a day, it is the third largest recycled water scheme in the world and the largest in the southern hemisphere. To ensure drinking water quality a multiple barrier approach is used which involves source control, tertiary wastewater treatment followed by micro-or ultra-filtration (MF), reverse osmosis (RO), and H_2O_2/UV advanced oxidation (Figure. 1).

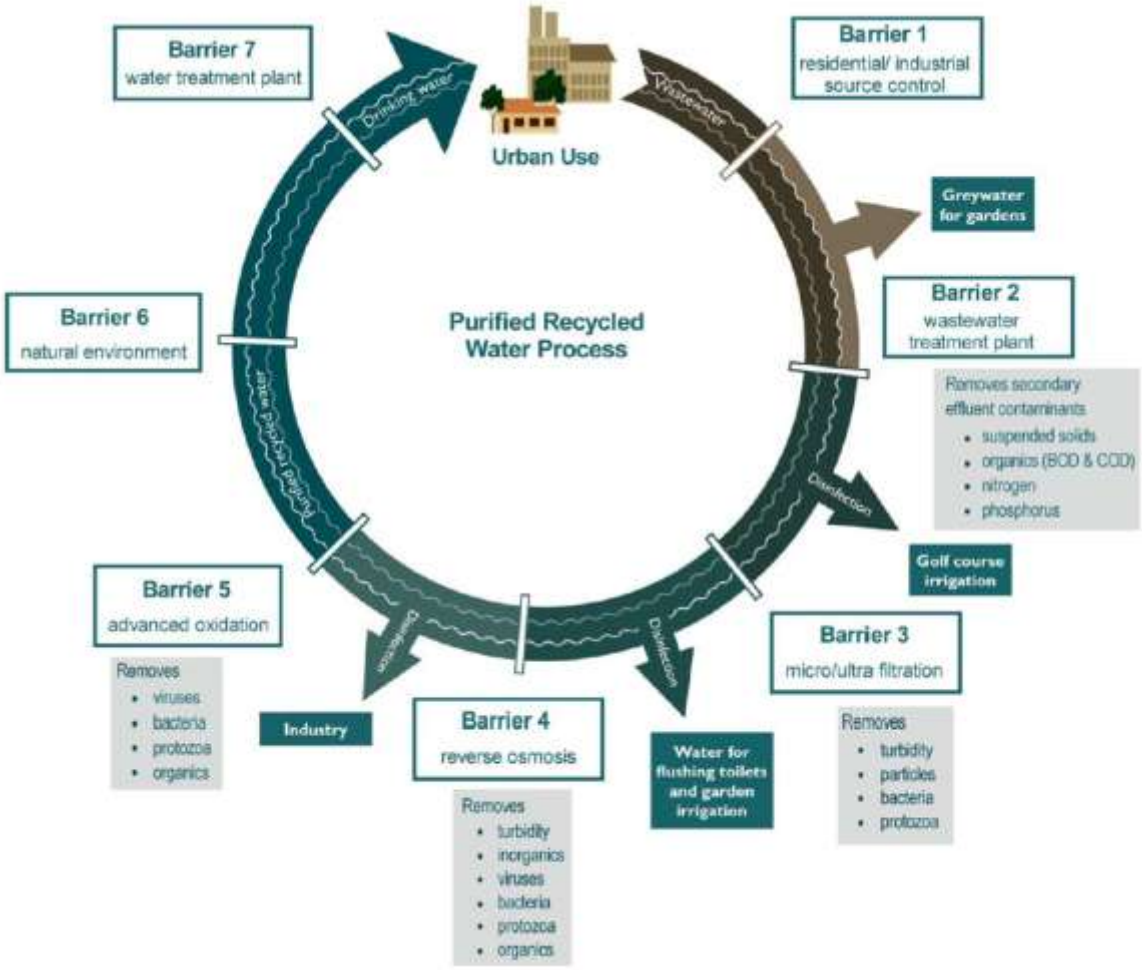


Figure 1. The multi-barrier concept for indirect potable reuse in Queensland (QWC in Traves *et al.*, 2008).

International Experiences with Desalinated Water in Agriculture

Supply of desalinated water to agriculture is a growing market despite high costs and the delivery and quality of this water presents many similar problems which may be encountered with PRW. One of the pioneers of such practices is Israel and the system used there is considered state of the art. The Israeli experiences are reported here in lieu of other global cases. Israel's state owned water provider MEKOROT uses its supply grid to supply various water qualities to farmers at different times. Supply for irrigation depends on the availability of surface water and the urban demand. Desalinated water is either supplied dominantly to agriculture or to urban users. The frequent changes however have upset farmers since some crops responded with growth deficits to the lower amounts of nutrients in the desalinated water. In 2007, Israeli researchers were calling for a reassessment of the use of desalinated water for irrigation, warning that desalinated water adversely affects some crops, such as tomatoes, basil and certain varieties of flowers (Ashkenazi, 2007).

When treated wastewater is used for irrigation purposes, the dominant absorbed cation (Na^+ vs. Ca^{2+}/Mg^{2+}) and the electrolyte concentration in the soil solution are the main factors that affect swelling and dispersion of clay particles (Lahav *et al.*, 2010). Under conditions of high Na^+ adsorption and low electrolyte concentration, significant swelling, dispersion and soil crust formation may occur, resulting in a drop in the soil infiltration rate to levels which mean that crops can no longer be supplied with an adequate water supply, proper drainage and oxygen. This phenomenon also results in runoff and erosion and both processes are enhanced by the negligible electrolyte concentration in the rain water falling on the high- sodium adsorption ratio (SAR) soil surface (Lahav *et al.*, 2010). The typical management response is to add magnesium, which both supports plant growth as well as well as lowering the sodium adsorption ratio.

The Lockyer Valley

The Lockyer Valley (approx. 80 km west of Brisbane) is an intensively used agricultural catchment and a major contributor to the vegetable supply of Brisbane. The Lockyer Valley has undergone significant change over the past

100 years, from a situation with perennial streamflow at the beginning of the 20th century to intermittent streamflow today. Irrigation bores have been drilled in increasing numbers since the early 1940s, with over 1,500 bores estimated to be currently servicing the Valley (DPI, 1994; DNRM, 2005). In selected areas, groundwater table drawdown occurred rapidly and, as a result, 21 weirs were installed in the 1950s for groundwater recharge, storage of water for direct surface water access and management of releases from the dams. During the 1980s, average annual groundwater withdrawal in the Lockyer Valley was estimated to be 46,500 ML, while safe annual yield is estimated to be 27,000 ML (DPI, 1994). Clearly, these extraction rates are not sustainable and additional water is thus required to sustain the farming practice in its current form. It is estimated that at least one-third of productive land in the valley has already has been withdrawn temporarily from cultivation due to lack of water. Over-appropriation of groundwater, particularly in 2007, has exacerbated salinity in some aquifers, as either saline water seeps in from adjacent sandstone areas to replace the higher quality water taken from alluvial aquifers or more salt is washed down from the unsaturated zone due to irrigation and dryland clearing. In short, the Lockyer suffers from the difficulties of managing a common pool resource (compare with "the tragedy of the commons"). Recently however, three consecutive wet years and the tragic flood events of January 2011 led to a major recharge pulse of the groundwater system. The current groundwater levels are close to long term maximum and demonstrate the ability to recover in most areas.

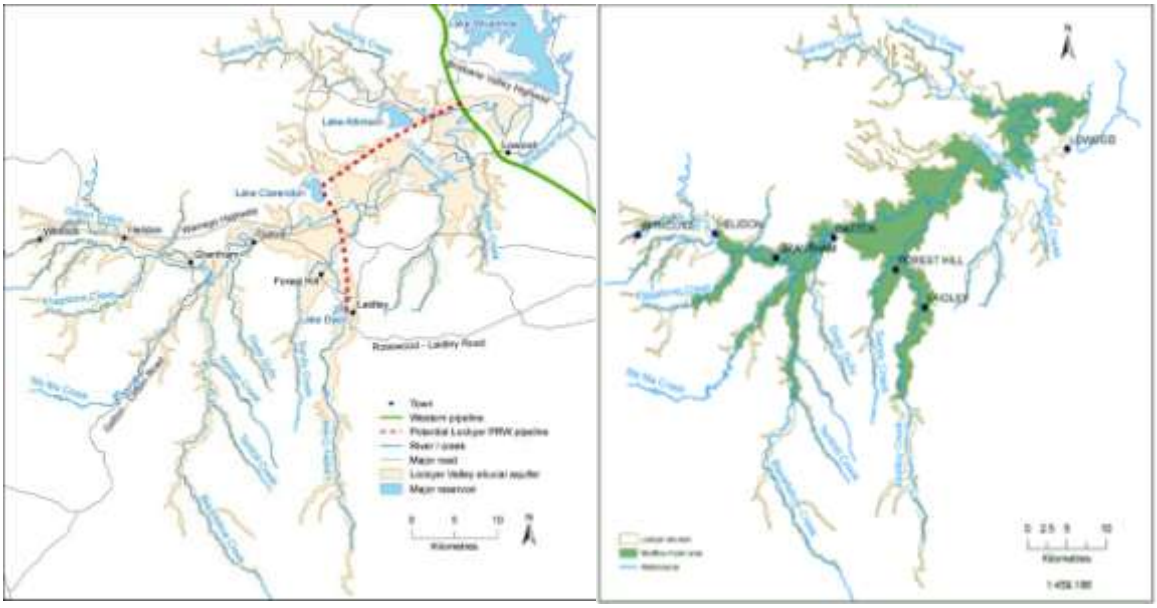


Figure 2. a) The potential common pool supply scenario for the Lockyer Valley; b) extent of the numerical groundwater model.

Methodology

For the integrated water resource management research on the Lockyer five methodological avenues are considered: (i) Stakeholder consultation in personal interviews and structured meetings; (ii) field and lab investigations on specific risk areas; (iii) physical process models for unsaturated and saturated water and salt transport; (iv) remote sensing for assessment of current, past and future land use; and (v) scenario analysis. The **stakeholder consultation** involved the SEQ Water Grid Manager, Queensland Water Commission (QWC), a number of different units and offices of the Department of Environment and Resource Management (DERM), different members of the Lockyer Valley Water Users Forum (LWUF), the Healthy Waterways Partnership, as well as local universities (UQ, QUT). **Field investigations** included deep coring of up to 20 m deep soil profiles at 5 sites (with analysis of soil parameters and salt content every ten centimetres), the installation of high time resolution groundwater level and salinity loggers after the flood event in January 2011, geophysical transects and groundwater sampling campaigns. **Lab investigations** covered spontaneous and mechanical soil dispersion under different water types including PRW, column experiments on different soil types using PRW, x-ray diffractometry of soils, analysis of water samples using ICP-MS. **Modelling** exercises start with the coupling of a soil water balance model (HOWLEAKY) with a numerical unsaturated zone flow model (HYDRUS) and a numerical groundwater flow model (MODFLOW). **Remote sensing** data classification of Landsat images aims at distinguishing individual crop types (eg, sorghum, lucerne, beetroot) with four ground-truthing campaigns in the last year. This will inform agro-economical modelling in an external collaboration. The **scenario analysis** is based on the stakeholder consultation and involves the key supply scenarios (farm gate or common pool augmentation), different irrigation patterns under PRW availability as well as climate change data from a related UWSRA project.

Results

Field and Laboratory Work

Laboratory tests using deionised water, PRW, local bore water and local surface water were undertaken on four representative Lockyer Valley soils. These tests showed that there was no risk of spontaneous dispersion with any of the soil-water combinations using the Emerson test (Ellis and Wolf, 2011). In addition, five cores of 30 cm depth were taken from the major soil types of the Lockyer Valley and were used for soil column experiments where PRW was applied. Most of the sediments had a quite low permeability and just produced pore drainage water, therefore they were not suitable, however, the experimental set up was successful for the Lawes soil type and this column could be operated with flow rates of up to 450 ml per hour. The Lawes soil was operated with groundwater taken from a bore at Gatton (EC 1090 uS/cm) until steady state conditions were reached. Then, the feed water quality was changed to PRW and within 8.5 hours, the outflow was dominated by PRW at a flow rate of +/-80 ml/h. In the following days, the pressure to the system declined while the flow rate was kept constant. The decreasing pressure at a constant flow rate indicates an increase in hydraulic conductivity which may result either from fine material leaving the column or rearrangement of clay particles in the column. Both processes are likely to be caused by clay dispersion (which can also support the transport of clay particles out of a sandy soil column).

Table 1 shows key water quality data for Purified Recycled Water, selected surface and groundwater water samples from the Lockyer Valley and the composition of drainage water from soil cores. From the literature, it is apparent that the most likely combinations leading to soil dispersion are those with high SAR values and low electrical conductivity (EC). From Table 1, it is apparent that the surface water samples taken have similar EC and higher SAR values than the PRW, thus comparing favourably for PRW. In summary, the water quality characteristics of surface water in the Lockyer are similar to PRW. It is suggested that the current management strategies of farmers are sufficient to deal with the amount of soil dispersion expected from additional PRW.

Table 1. Water quality data for Purified Recycled Water (PRW), selected surface water and groundwater water samples from the Lockyer valley and the composition of drainage water from soil cores.

	Collection date	El. Con. @ 25º C uS/cm	SAR	Calcium	Magne-sium	Sodium	Chloride as Cl	Phosphate phosphorus as P	Nitrate nitrogen as N	Manga-nese	Iron	Lead	Arsenic
				mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	µg/l	µg/l
Australian Drinking Water Guideline Standard (2006)							250		50	0.1	0.3	10	7
PRW Bundamba	20/04/2010	277	0.59	18.1	9.3	10	19.3	<0.05	0.51	<0.01	<0.04	<0.05	<0.35
Irrigation GW at UQ Gatton	9/06/2010	1060	2.74	74.4	83.7	104	182	0.1	1.29	<0.01	<0.04	0.15	0.81
Irrigation GW close to FH	10/06/2010	6590	13.59	142	304	807	**1770**	0.11	43.4	<0.01	<0.04	0.62	2.28
Logan Dam Surface Water	9/06/2010	208	3.50	6.3	5.3	37	33.5	0.18	0.55	<0.01	0.15	0.17	1.19
Lockyer Creek	11/05/2010	380	1.45	21.7	16	28	42.7	0.05	<0.50	<0.01	<0.04		
Alluvial Soil Core First flush	28/06/2010	1090	10.06	81.8	65.2	381	**424**	0.64	**105**	0.01	0.05	0.25	2.49
Blenheim Soil Core First flush	28/06/2010	4580	9.10	208	285	596	**1410**	0.27	29.5	0.05	<0.04	1.37	5.01
Creek Bed Sediment 1 pore water	22/04/2010	726	0.42	12.6	7.1	6	38.6	<0.05	<0.50	**3.78**	0.28	0.12	1.97
Creek Bed Sediment 2 pore water	4/06/2010	310	0.94	58	23.9	28	47.6	0.09	<0.50	<0.01	<0.04	0.1	0.55
Lawes Soil Core flushed with PRW	20/08/2010	221	1.28	14.6	9.4	20	18.1	1.24	<0.50	<0.01	<0.04	<0.05	<0.35

In 32 samples analysed, arsenic concentrations were below 5 ug/l, compared to the Australian Drinking Water Guideline value of 7 ug/l. Cadmium guideline values were exceeded for drainage from creek bed sediment 1, with values up to 4.72 ug/l compared to the guideline limit of 2 ug/l. Lead was below 1 ug/l in 20 samples, 1 sample with fresh drainage water from a creek bed sediment showed 3.24 ug/l, still significantly below the drinking water guideline of 10 ug/l . Manganese concentrations range between 2 – 5 mg/l for seepage water from the creek bed sediment, thus exceeding the guideline value for drinking water of 0.1 mg/l. A second sediment core from the creek however did only exhibit Manganese values up to 0.5 mg/l. Percolate from the alluvial soil showed manganese concentrations up to 0.16 mg/l.

Water Balance Considerations

The Central Lockyer Groundwater Model indicates that full supply of groundwater in the Central Lockyer aquifer is in the order of 115 GL (DERM, 2000). This compares to an entire volume of the Lockyer Valley alluvium of 4.66 billion m^3, with 0.9 billion m^3 of loam and silt, 30.3 billion m^3 of rather undifferentiated clayey sediments and 0.69 billion m^3 of coarse sand and gravel (Hawke *et al.*, 2010). Assuming typical effective porosities of 5%, 6.5% and 15% for each of the classes, results in a first order estimate of 347 GL of potential water storage. These numbers are subject to an uncertainty of at least +/-20% as a result of the poorly constrained information on the effective porosity. This compares well with estimates from Bleakley and Boreel (2011), who assume 345 GL groundwater storage when

the aquifer is full. Based on groundwater level evaluations, they estimated that in June 2007 the aquifer in the valley was some 29% full which would equate to 100,000 ML. This then rose to some 80% in January 2011 which would equate to 276,000 ML. Therefore the change in storage from June 2007 to January 2011 is 176,000 ML (Bleakley and Boreel, 2011). The figures will be constrained more in the following groundwater modeling exercises.

Salt Import from PRW

The introduction of an annual volume of 25 GL/a PRW into the Lockyer Valley, with a typical chloride concentration of 19.3 mg/l, would equal an import of 482 t of chloride per year. It is assumed that all of this water would be applied to the surface area of the alluvial aquifer, ie, the 30km^2 groundwater model area as depicted in Figure 2b. Using the average annual rainfall between 1978-2000 of 759 mm and a chloride concentration in rainfall of 4.76 mg/l (Biggs, 2006), the chloride input via rainfall amounts to 1084 t/a. Considering the entire catchment of the Lockyer creek upstream of the Rifle Range Road (DERM station number 143210B), which has a size of 2490 km^2, natural chloride input would amount to 8995 t/a. Other balance terms to consider are the redistribution of salt in the catchment due to irrigation with groundwater of higher salinity and the amount of salt leaving with streamflow. An application of 30 GL/a of groundwater from the Lockyer system, considering a median chloride concentration of 327 mg/l (recorded for 104 wells in the DERM groundwater database for 2009) would result in an application of 9810 t/a chloride to the soils of the Lockyer valley. With respect to the input from rainfall and redistribution of salt via groundwater irrigation, it can be concluded, that the additional import of salt from increased irrigation with PRW would not have a significant impact on catchment salt balances but should lead locally to lower salt concentrations in the soil, especially if salty groundwater is substituted with PRW in irrigation.

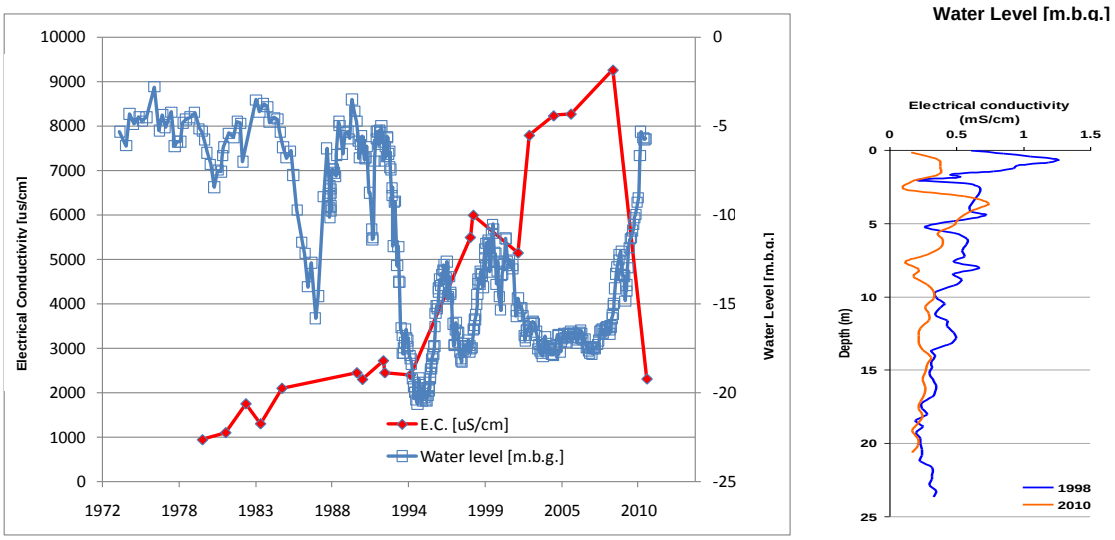


Figure 3. a) rising conductivity and recent water level rise in a well upstream Forest Hill (RN 14320277); b) repeated coring for E.C. in a paddock close to Forest Hill years 1998 and 2010.

This said, local problems can occur, especially if heavy irrigation or land clearing leads to rising water tables and redistribution of salt to the surface. It needs to be stated clearly, that this is already an issue today, with salinities rising over the last decade in selected bores. The most prominent example is a bore upstream of Forest Hill (Figure 3a), which has risen from ca. 1,000 uS/cm electrical conductivity in 1979 to about 9,000 uS/cm in 2009. Following the wet years since 2008 and the catastrophic flood event in January 2011, the electrical conductivity dropped to 2310 uS/cm at June 2011. This suggests that the increased recharge did not lead to a major mobilisation of salt from the unsaturated zone at this point, but rather to an improvement in groundwater quality. The data for EC of cored soil from 1998 and 2010 show a considerable loss of salinity (Figure 3b), which is expected to have moved downwards in the soil profile. As the salts lost from the profile are not stored in deeper soil layers, these salts must have moved below 20 m depth if the difference is not an artifact of small scale (eg, 5 m) lateral variations in soil chemistry

Modelling

The UWSRA research project employs the coupling of a soil water balance model (HOWLEAKY) with a numerical unsaturated zone flow model (HYDRUS) and a numerical groundwater flow model (MODFLOW) (Figure 2b). The sensitivity study using the soil water balance model HOWLEAKY demonstrated that crop type was one of the key parameters influencing the amount of deep drainage, for example modelled deep drainage rates were 1.5 mm/day for lettuce while only 0.2 mm/day for sorghum. This showed the need for updated data on crop type distributions in the Lockyer Valley.

Simulations on a specific farm plot with known cropping sequence and soil hydraulic parameters at Forest Hill also showed that the irrigation strategy (target amount and interval) has a major influence on deep drainage. Depending on the irrigation strategy, the deep drainage at study site ranged between 59 -262 mm/year. With moisture monitoring and judicious irrigation (targeting high transpiration), the simulated deep drainage at the study site would be ~84 -164 mm/year. The modelling of deep drainage is very sensitive to the amount of irrigation applied beyond the soil drained upper limit (DUL). The drainage rate simulations will now be considered alongside with the salt profile information. The surface water system of the Lockyer Valley is modelled using IQQM under consideration of the historic climate scenario as well as regional climate change scenarios. A numerical groundwater model is used to estimate the amount of water required to maintain agreed ecological targets. The modelling is subject to a predictive uncertainty analysis but work is still ongoing. More detail on the modeling can be sourced in (Moore et al. 2011, Wolf et al, 2011).

Management Strategies

The intended scheme is special in that it unites urban and rural water systems in one engineering solution to the expected benefit of both. Three major supply scenarios are currently discussed:

1 Delivery of PRW to the individual farm gates;
2 Delivery of PRW to major reservoirs, distribution via existing pipes to selected farmers and releases to the creek (see Figure 2 a); and
3 Direct injection of PRW into the groundwater via wells.

Recent developments show that the complexity of common pool resource management (options 2 and 3) is a barrier to farmer's acceptance of the PWR supply. As a consequence, a delivery to farm gates is favored by the local water users. While requiring a more costly pipe network, it provides farmers with full control of their PRW use.

A starting point for all models on environmental impact is the knowledge of future demand for PRW. In this case however, the future demand for PRW depends on finding a society consensus on ecosystem services and environmental flows.

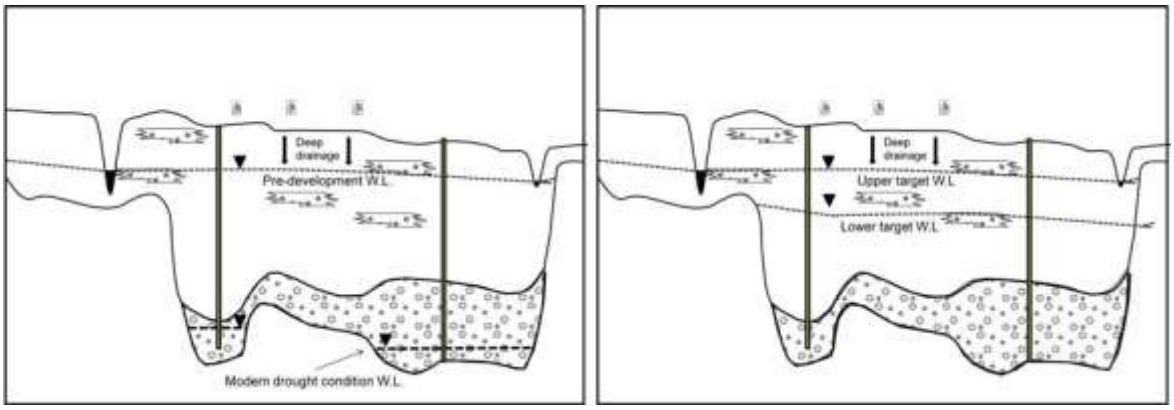


Figure 4. a) Schematic cross section through the Lockyer Valley including pre-development groundwater levels and modern drought conditions; b) Future management scenario with time variant PRW supply to keep groundwater levels between upper and lower target level. Sand and gravel dominated aquifer sequences shown with dot texture.

The supply of PRW into the Lockyer Valley provides the opportunity to actively steer the currently overused ecosystem to any desired state, ie, to achieve a more or less constant groundwater level (Figure 4) by importing variable amounts of PRW depending on the current climate. This can be achieved with both supply scenarios (farm gate/reservoir release). Due to the extensive treatment aimed at indirect potable reuse, the purified recycled water comes at a high production cost (>$AUD 500 per megalitre), compared to current water charges of a maximum of $AUD 30 per megalitre. For this reason, irrigators will go a long way to avoid using PRW at all. Only if creeks have dried up or groundwater wells run dry/turn salty will there be a business case for them to use PRW. From an environmental, regulator or water users perspective, clearly, the decision of where the target groundwater levels are set will govern future PRW demand. In certain years, such as the very wet 2010/2011 season with catastrophic flooding in the Lockyer, the demand for PRW will be close to zero. In the central Lockyer areas, target water levels were defined for the DERM groundwater modelling exercise. Generally this involved selecting the water level in July of 1990 and the water level at January 1994 for each bore and calculating the average of these two figures as the target water level.

Conclusion

The complexity, as well as the benefits, of introducing a new water source in an agricultural area has been demonstrated. Because of the increased availability of wastewater worldwide, the Lockyer Valley scheme is a parallel to many wastewater reuse schemes, but with the exception that drinking water quality could be available for irrigation. This may appear as a special case today but may become more widespread if concerns about pharmaceuticals, persistent organic trace substances or antibiotic resistance call for more strict quality standards in wastewater reuse. At the current stage of the study, environmental benefits from PRW import to the Lockyer Valley, along with the additional buffer capacity for the SEQ Water Grid, make the intended scheme appear very attractive. The preliminary results on the risk screening have not shown major risks from PRW application so far, except from a potential risk for water logging in selected areas if irrigation rates significantly increase. The economic viability of the scheme is defined by the demand for PRW in a situation of improving irrigation efficiency, changing crop types and climate change. The estimation of future PRW demand relies on: (a) prediction of future cropping patterns in the Lockyer (ie, to what extent will the farmers turn to crops with higher water demand or high value crops requiring high water supply security?); and (b) modelling of the spatially differentiated groundwater abstraction in the entire Lockyer Valley while meeting target groundwater levels in a variable climate. Within the UWSRA research project, more details on quantitative results of modeling and field studies are expected for the year 2012.

References and Further Reading

Ashkenazi, E. (2007). Desalinated water can harm crops, Hareetz, Israel, 9.11. 2007: http://www.haaretz.com/print-edition/news/desalinated-water-can-harm-crops-researchers-warn-1.232848

Biggs, A. J. W. (2006). Rainfall salt accessions in the Queensland Murray–Darling Basin, Australian Journal of Soil Research, 2006, 44, 637–645.

Bleakley, A. and Boreel, S. (2011). Groundwater Levels Lockyer Valley January 2011, internal report, DERM, 4.p.DNMR (2000): Central Lockyer Groundwater Model Report, 125p, Andrew Durick and Ashley Bleakley, Department of Natural Resources and Mines, QLD

DNRM (2005). Lockyer Valley Discussion Paper: Declaration of the Whole Lockyer Valley as a Subartesian Area. Department of Natural Resources and Mines, Queensland.

DPI (1994). Overview of the water resources of the Lockyer Valley.

Ellis, T. and Wolf, L. (2011). Impacts of applying purified recycled water in the Lockyer Valley, Queensland, Part 2: Soil physical assessment of PRW application to local soils. Published at UWSRA 2011, Brisbane.

Galletly, J.C. (2007). Baseflow in the Lockyer Creek, 356 p, PhD thesis, University of Queensland, Gatton, Australia.

Hartcher, M., Ellis, T. and Wolf, L. (2011). Spatiotemporal crop mapping in the Lockyer Valley using satellite imagery. UWSRA Alliance, Brisbane.

Hawke, A., Cox, M. and Allen, J. (2010). Application of Groundwater Visualisation System (GVS) to the Lockyer Valley Alluvial Aquifer, 23p., Report to the UWSRA Alliance, Brisbane, Australia.

Lahav, O., Kochva, M.. and Tarchitzky, J. (2010). Potential drawbacks associated with agricultural irrigation with treated wastewaters from delalinated water origin and possible remedies. *Water Science and Technology* 61(10): 2451-2460.

Moore . C., Wöhling, T. and Wolf L (accepted). Optimisation of monitoring data for increased predictive reliability of regional water allocation models, 8p, MODSIM2011, Perth.

Rassam, D., Robinson, B., Ellis, T. and Wolf, L. (2011). Impacts of applying purified recycled water in the Lockyer Valley, Queensland, Part 1: Sensitivity study on unsaturated zone modelling to assess deep drainage and recharge. Published at UWSRA 2011, Brisbane.

RPS Consult (2010). Lockyer Valley Groundwater Modelling Sensitivity Analysis, 61p, Report to the UWSRA Alliance, Brisbane, Australia.

W. H. Traves, E. A. Gardner, B. Dennien and D. Spiller (2008). Towards indirect potable reuse in South East Queensland, Water Science and Technology, vol 58.1, p.153-161.

WaterSecure (2010). Water Quality Report, 46 p., accessed 29.6.2011 at http://www.watersecure.com.au/images/stories/reports/20101019_Public_Confidence_Report_FINALWEB.pdf

Wolf, L., Moore, C. and Gallagher, M. (2011). Management of limited groundwater resources in Australia: new models and approaches for target water levels, sustainable yield, groundwater allocations and large scale reuse. Accepted for oral presentation at: International Conference on Integrated Water Resources Management: Management of Water in a Changing World: Lessons Learnt and Innovative Perspectives 12-13 October 2011, Dresden, Germany.

Assessment of New Chemical Film Monolayers for use as Long-Term Evaporation Reduction Systems

Schouten, P.W.[1], Putland, S.[1], Lemckert, C.J.[1], Solomon, D.[2], Leung, A.[2], Tran, D.[2], Talemi, P.[2] and Qiao, G.[2]

[1] School of Engineering, Griffith University, Gold Coast, Queensland

[2] Department of Chemical and Biomolecular Engineering, University of Melbourne, Parkville, Victoria

Summary

Up to this point in time, the vast majority of chemical film monolayers that have been tested in both real-world and laboratory environments have failed to reduce sizeable amounts of evaporation due to their incapacity to recover from various types of physical stressors such as wind and high frequency capillary wave action. New prototype monolayers are currently being synthesised at the University of Melbourne (in conjunction with the Cooperative Research Centre for Polymers) that each aim to eradicate the performance issues that plagued the previous types of monolayers. The ultimate objective of the research work is to calculate the influence of simultaneous wind and wave activity upon these prototype monolayers by performing tests in a large extended water basin (within an atmospherically controlled enclosure) using real-time environmental measurements. From this, relationships between the monolayer performance, atmospheric conditions, water wave properties and evaporation rates are being deduced. This information will allow for improved real-world deployment and reapplication strategies to be instigated in the future. The following manuscript will present some selected performance results measured for the prototype monolayers and directly compare these results to a control octadecanol suspension film.

Keywords

Monolayer, evaporation, film, solvent, solid.

Introduction

Before being used in a given application, water is usually stored in a tank, dam or reservoir. It has been calculated that up to half of this stored water can be lost via evaporation over a given period of time (Craig, 2005a). Alarmingly, the rate of evaporation in Australia has been greater than 2 metres every year, with mean rainfall across Australia less than only 500 millimetres every year. Specifically, Craig *et al.* (2007) has estimated that over 1,000 GL of water can be lost annually just from small dams. Also, it has been found that as much as 95% of all rainfall incident on our continent is eventually evaporated, leaving a severe hole in the water cycle (Craig, 2005a). Recent extreme drought conditions across Australia, especially across the majority of southern Queensland have created a psycho-social climate of unease and uncertainty amongst the general public, government and industry. Record-breaking rainfall received early in 2011 has helped to alleviate water savings and delivery pressures by greatly supplementing major water storage dams across the South East Queensland (SEQ) region. However, it is inevitable that drought conditions will again prevail over the coming years, and as such there is a requirement for the development and instigation of new cost-effective methods to save water and minimise evaporation.

There have been several techniques using different types of shade materials developed recently that have attempted to reduce the extreme influence of evaporative effects upon dams amongst varieties of natural and artificial water bodies. Some examples of this work has been described in CSIRO (2007) and Finn and Barnes (2007). Craig *et al.* (2005b) has delivered an extensive technical report detailing the ability of both shade coth/sail and chemical film monolayers in restricting the evaporative process. In this report, it was found that evaporation was reduced by roughly 75% for dams covered with shade and as much as 95% for dams covered by a floating cover. In addition, it was discovered that dams covered with an acetyl alcohol based chemical film monolayer had an overall reduction in evaporation of between 5% and 30%. From these results it might seem obvious to use shade cloth, shade sail or some kind of a floating cover to minimise evaporation. Unfortunately, the cost of covering and continuously maintaining entire water bodies with these materials is highly prohibitive, making chemical film monolayers the only economically viable choice in evaporation reduction applications. Also, the use of monolayers is advantageous as they only have to be applied when it is necessary to do so, for instance either when a water body is completely full or during dry spells and drought conditions when evaporation levels are at their peak (Craig, 2005a).

A chemical film monolayer is simply a long chain molecule with one hydrophilic end and one hydrophobic end (Craig *et al.*, 2007). This particular configuration allows the monolayer to spread out and sit on top of a body of water in a closely packed regime no more than 2 nm thick. The tight packing between the single molecules blocks the evaporating water below. Research detailing the chemical and physical properties of monolayers, their visualisation and their interaction with the natural environment has been ongoing since the 1950s. Some examples of monolayer related studies include Mansfield (1955), Barnes and La Mer (1962), Costin and Barnes (1975), Barnes (1986), Drummond *et al.* (1992), Barnes (1993), Gericke *et al.* (1993), Barnes (1997), Craig *et al.* (2005a), Craig *et*

al. (2005b), Craig *et al.* (2007), Pittaway and van den Ancker (2010), Schouten *et al.* (2011a), Schouten *et al.* (2011b) and Schouten *et al.* (2011c).

Many monolayer compounds possess a high equilibrium spreading pressure, which allows them to rapidly spread and reform across a water surface (Bikerman, 1963; Tsuji *et al.*, 2008). Monolayers can be deployed on water reserves as a solid (Mansfield, 1962), in a solvent (Schouten *et al.*, 2011c), as an emulsion (Herzig *et al.*, 2011), and also as a dispersion (Dressler and Guinat, 1973). These methods vary the equilibrium spreading pressures of the monolayer, with the solid deployment method generally having the lowest (Barnes and Gentle 2005; Barnes 2008; Herzig *et al.*, 2011). The greater the equilibrium spreading pressure, the greater potential a monolayer has of succeeding at suppressing evaporation over a large area (Barnes, 2008). The higher spreading capacities of the non-solid deployment methods make these highly desirable. However, solid monolayer compounds are more practical for commercial applications as they inherently have reduced transportation and handling costs. With these competing interests, finding a balance between these requirements of spreading performance and commercial viability leads to further research investigating monolayer capabilities and deployment techniques.

Monolayers employed in the past have been most commonly based on biodegradable fatty alcohol compounds. However, these monolayers have been found to be very susceptible to volatilisation, wind and wave stress, along with various types of biological degradation caused by natural microbial layer residue which has limited their useable life time to less than two days on average (Craig *et al.*, 2007). These particular monolayers were also restricted by their extremely slow spreading rate. Two relatively new chemical film products called Aquatain (www.aquatain.com.au) and WaterSavr (http://www.phoslock.com.au/watersavr.php) have been recently released to the public. Aquatain uses a thick silicone base that makes it less sensitive to the atmospheric stressors that plagued the use of the fatty alcohol based monolayers. WaterSavr combines lime, hexadecanol and octadecanol in a powdered form in order to achieve quicker spreading rates and more uniform wide-spread water surface distribution.

Despite the promise of Aquatain and WaterSavr, the Cooperative Research Centre for Polymers (CRCP) and the University of Melbourne, in collaboration with other industry and scientific partners such as Griffith University and the Urban Water Security Research Alliance, are presently working on producing a series of new, ultra-thin (near 1 nm), rapidly-spreading chemical film monolayers. The main goal in the development of these prototype monolayers is to make them even more resistant to physical and chemical stress and have even higher levels of performance longevity than any type of monolayer previously employed. It addition, they are being developed so that they have minimal environmental impact and are completely safe to use on drinking water supplies. The following sections of this paper detail precisely how this research has been carried out. Experimental data are provided showing how the monolayers perform when subjected to variable wind and wave action over extended time periods.

Experimental Environment

A series of trials were performed separately with potable water sourced from a tap over 24 hour intervals with 14 prototype monolayers synthesised by the CRCP, each with their own specific chemical formulation and with octadecanol film used as a control. These trials deduced the influence of wind upon these monolayers up to approximately 3 m/s (equivalent to 11 km/hr normalised to 2 m height by the wind profile power law assuming neutral stability conditions) incident at two different angles of attack (25^o and 0^o) with respect to the horizontal water surface plane. The winds approaching and hitting the water surface at these two inclinations allowed for different wave patterns with varying amplitudes and frequencies to be generated on the water surface, enabling a more detailed analysis of the performance of the prototype monolayers and octadecanol after being subjected to sustained capillary wave conditions. From visual inspection, the waves generated had amplitudes ranging from 0.2 to 0.5 cm and frequencies of approximately 0.1 s. A further series of trials were also performed by inflicting 3 m/s winds parallel to the water surface plane upon the prototype monolayers and octadecanol for 24 hours and then reducing the wind speed down to 0 m/s for a further 24 hours in order to determine the recovery efficiency and respreading characteristics for each of the films.

To maintain consistent temperatures and a stabilised humidity range throughout each experiment, a temperature controlled enclosure was designed and built around a rigid wave basin (dimensions: 15 m x 0.46 m x 0.85 m). Air conditioning was provided continuously by two split cycle long throw systems (MSZ-GA 7 kW Series, Mitsubishi Electric, Japan). Temperature was set to 23^oC for every experiment. This particular temperature was chosen as it matches closely to the average daily temperatures measured at typical SEQ agricultural dam sites during the majority of the year. A generic dehumidifier set to its highest possible setting was installed inside the enclosure in order to enhance the evaporative process and, in turn, to reduce the amount of time required to complete a single experiment. Winds controlled by a calibrated baffle/intake restriction system were delivered via a wind blower unit to the surface of the water. These winds were straightened by a cylindrical honeycomb unit made from tightly packed plastic cocktail straws before coming into contact with the water surface. The wave basin was covered by waterproof wooden boards in order to improve air flow and to maintain air mass conservation from the wind blower unit intake to the end of the basin. The wooden boards also provided a protective barrier between the water and diffuse infrared radiation helping to maintain consistent initial water temperatures between each trial. After the application of a

monolayer, evaporation was measured over the experimental time interval by measuring the change in water level using a pulse altimeter working at a collection frequency of 10 Hz (Senix Toughsonic Distance Sensor TSPC-30S1-232, United States). This altimeter has a manufacturer defined 0.1% characteristic repeatability. Additionally, the altimeter has an accuracy of ±0.086 mm, and is operated 600 mm from the water surface, falling between its optimum operating range of 102 mm to 4.1 m. Evaporation savings as detailed in this manuscript were calculated after each experiment using the following expression:

$$Savings = \left[1 - \left(\frac{E_{MONOLAYER}}{E_{BASELINE}}\right)\right] \times 100\%$$

where $E_{MONOLAYER}$ is the evaporation rate measured during a trial with a monolayer present on the water surface and $E_{BASELINE}$ is the evaporation rate measured during a baseline trial without any monolayer on the water surface. Atmospheric parameters such as relative humidity, air temperature and water temperature were also monitored for each trial at the same sampling frequency.

Monolayer Chemical Formulations

The monolayer compounds tested included 14 prototype solid-state monolayers and solvent delivered liquid monolayers. Figure 1 provides a picture of each of the particular monolayers before deployment. These monolayers are variations of the films detailed in the patent "Method for Controlling Water Evaporation" (WO 2010/071931 A1). The exact chemical composition of these monolayers has yet to be provided to the general public. However, each of these monolayers has slight differences in their molecular structure and/or composition, and as a result has different spreading and recovery characteristics. For each experiment, the solid monolayers were deployed in the wave basin by hand. The solids were delivered at a dosage rate of 0.051 g/m^2 for monolayer AAr and 0.065 g/m^2 for monolayer BAr, allowing for an 18 x molecular layer surface spread to be set across the water surface. All of the solvent monolayers were deployed via a syringe at a dosage rate of 4.6 mL/m^2, providing the same 18 x molecular layer surface spread as delivered by the solid monolayers. Each monolayer was delivered at four evenly spaced sections across the water basin. In order to ensure consistency between each trial, the water basin was emptied and rinsed out at the end of each experiment in order to ensure that no monolayer remnants were left on the basin walls or floor.

The trials were undertaken in order to determine the optimal balance between the amount of solvent required for the adequate delivery of a monolayer, monolayer compound concentration, monolayer dosage, and evaporation suppression capability. Once initial testing was completed, the best overall performing solvent monolayer was chosen for additional testing (solvent Ai). New formulations were then made of solvent Ai that contained increased monolayer compound concentrations (solvents Ah and AVh). Of these two compounds, solvent AVh contained the highest amount of active monolayer. These monolayers were then subjected to parallel and 25^o attacking wind trials at three different dosages. These dosages were; 0.464 mL/m^2, 0.155 mL/m^2, and 0.077 mL/m^2, which equated to 18x, 6x, and 3x molecular surface layers respectively. The evaporation reducing ability of these prototype monolayers was compared directly to a publically available thin film monolayer based on an octadecanol (CH_3 $(CH_2)_{17}$ OH) suspension formulation. Specifically, this mixture contained 5% octadecanol and was combined with 1% Brij 78 in distilled water. In order to allow for direct performance comparisons to be made, the spreading ratio for the octadecanol film was set to approximately 6 mL/m^2 allowing for an equivalent 18 x monolayer layer spread across the water surface.



Figure 1. Pictures of the prototype monolayers before deployment - A) Solvent monolayers A, B, C, D, E and F. B) Solvent monolayers Ai, G, H, and I. C) Solid monolayers AAr and BAr. D) Concentrated solvents Ah and AVh. E) Octadecanol suspension.

Results

Simultaneous Wind and Wave Experiments

Generally, the prototype solid and solvent monolayers from the CRCP offered a much better defence against the evaporative process in comparison to octadecanol. It can be seen from Figure 2 A) and Figure 2 B) that the monolayers provided a reduction in evaporation from 20% to 54% for wind speeds between 0 m/s and 3 m/s incoming from attack angles of 0^{o} and 25^{o} with respect to the horizontal water surface plane. These evaporative saving percentages represent approximately 6 times better evaporation resistance at wind speeds close to 11 km/hr in comparison to the control octadecanol film. Only solvent G and solvent H failed to consistently give savings greater than those delivered by octadecanol. From visual observations made in the water basin from both above and below the water surface, the prototype solvent monolayers spread much quicker than the octadecanol film, which reduced their application time and improved their usability. Additionally, the prototype monolayers appeared to be very buoyant and maintained a good level of surface coverage following the extended delivery of wind. As a result, they did not assemble in random thick conglomerations and did not appear to blend beneath the water surface with induced capillary wave action as was the case with the octadecanol film.

A reasonable correlation was seen to occur between wind incident angle and overall monolayer performance, as monolayer performance increased with increasing angle of attack, especially with solvents G, H and I. This result conflicted with what was seen to occur in the study by Schouten *et al.* (2011c), in which increased wind angle, and in turn capillary wave action was seen to decrease monolayer performance. The improved result obtained in this study may have been due to the solvent based monolayers minimising the average magnitude and frequency of the capillary waves, which would have decreased the movement and hence kinetic energy of water molecules attempting to escape from the water surface. The reduction of wave magnitude would have also assisted in decreasing the overall water surface area, further minimising the potential for evaporation to occur. These solvent monolayers could have also had improved water anchoring capabilities.

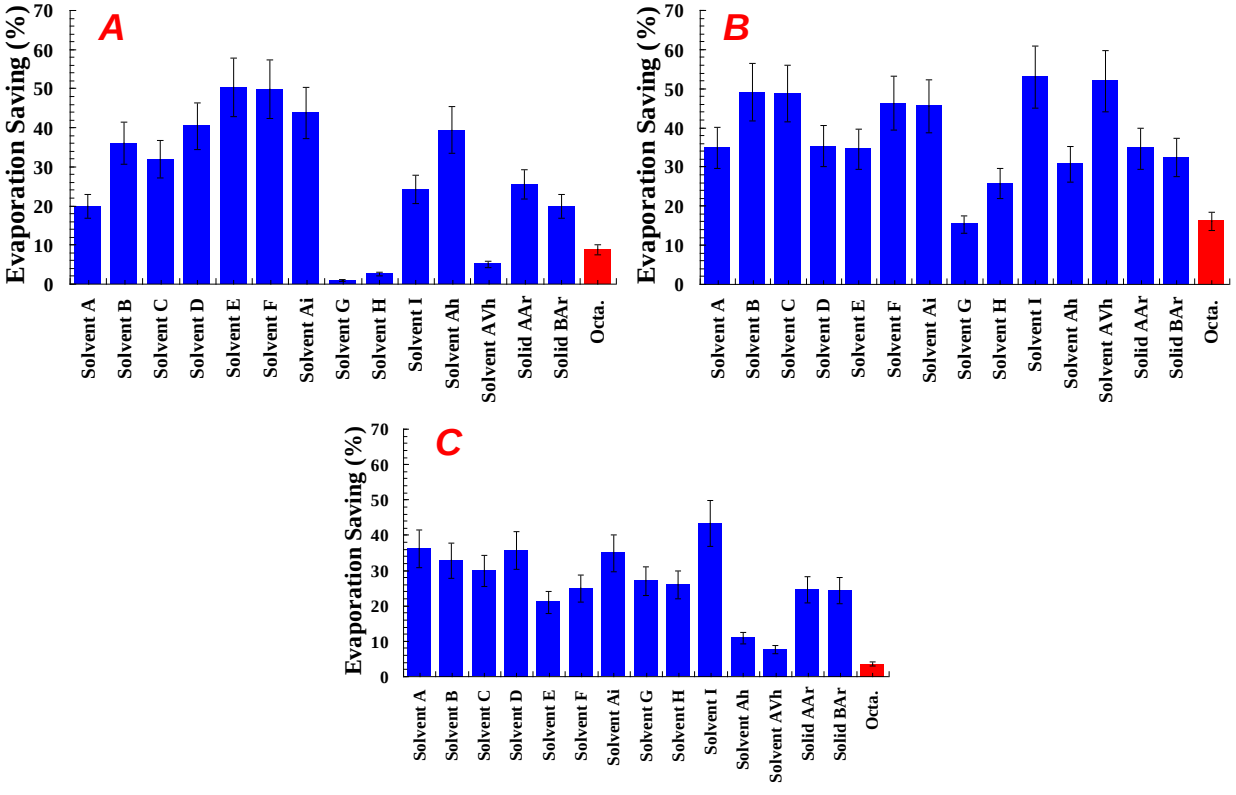


Figure 2. Evaporation savings measured with the solvent, solid and octadecanol monolayers – A) Parallel 3 m/s wind testing percentage savings. B) 25^{o} angle of attack testing percentage savings. C) Recovery testing percentage savings. The error bars in each graph represent the calculated range of uncertainty within each measured percentage saving.

Figure 2 C) shows that the 14 prototype monolayers were all capable of regrouping and respreading much more efficiently in comparison to octadecanol. After being subjected to a 48 hour trial interval with 24 hours of 3 m/s wind and a further 24 hours of no wind application, the prototype monolayers were still able to provide evaporative savings of between 11% and 43% compared to an extremely low 4% for octadecanol. This result suggests that when subjected to differential wind speeds in real-world water reserves, the prototype monolayers should be able to recover and spread over a given surface area and still provide substantial savings in evaporation as long as the

monolayer is contained within the water reserve (by a boom system or similar apparatus) and is not allowed to wash up on shore. The endurance, buoyancy and malleability of the prototype monolayers under harsh variable wind conditions also indicate that they will have a near negligible polluting effect upon the water column and the surrounding natural environment. In comparison, the tendency of the octadecanol film to accumulate on the water surface and sink through the water column may result in the film descending down to and settling on the benthic zone of a water reserve, especially after being subjected to substantial wind and waves. This could become a serious ecological issue over time.

The solid form monolayers provided slightly lower reductions in evaporation in comparison to the solvents (apart from solvent G and solvent H). These relatively smaller savings could be attributed to the slower and restricted spreading ability of the solid monolayers, and also their tendency to clump and amalgamate on the water surface. From visual observations, the solid monolayers would sometimes not completely disperse after deployment, primarily remaining as a randomly distributed collection of clusters. These clusters prevented the complete spreading of the monolayer over the entirety of the basin surface, and as a result, it could not achieve its total evaporation suppression potential. Therefore, it is possible that the variance in solid monolayer performances, especially solid AAr, is associated with their propensity to spread poorly in comparison to the solvent based monolayers. These issues have been previously identified by a study by Craig *et al.* (2005) where similar solid-state monolayers also failed to spread adequately, reducing monolayer performance. For real-world applications using solid monolayers, it would be advisable to implement a system to forcefully refine and disperse the monolayer before deployment, in order to maximise the total coverage of the monolayer across the water surface. A mixing hopper system such as those described in Craig *et al.* (2005) and Knights (2005) could possibly be used to achieve this requirement.

The solvent monolayers all exhibited excellent spreading characteristics. Specifically, in most trials with the solvents, an application at the initial deployment point was able to spread over half the water surface (approximately 7.5 m) before the next application began at the second point. This excellent spreading capacity does not just aid in the rapid deployment of the monolayer, but also in the rapid repair of breakages in the surface layer caused by wind and wave action, contaminants, or other disturbances. This ability to quickly spread over holes in the monolayer surface layer coupled with resilience to wind and wave action makes for an ideal monolayer compound for real-world deployment. As solvent A was determined to perform the best over the initial testing phase, two new concentrated formulations of solvent A were synthesised. These two formulations were labelled as solvents Ah and AVh, both having higher active monolayer chemical concentrations present in comparison to solvent A, with AVh having the higher active monolayer concentration of the two. These concentrated monolayer solvents were tested using varying doses, in an effort to minimise the amount of required raw monolayer material and to maximise their evaporation reduction capacity. From parallel and attacking wind trials, all dosages (18x, 6x and 3x molecular layers) provided reasonable to excellent water savings.

Figure 3 A) shows that solvent Ah achieved the highest evaporative saving during the parallel wind testing, using an 18x molecular layer equivalent dosage, which yielded a saving of 52%. In Figure 3 B) it can be seen that solvent AVh, also spread at an 18x molecular layer dosage, provided the best outcome in the 25^{o} attack angle trials, delivering an evaporative saving of 51%. The performance of these solvent monolayers deployed at the highest 18x molecular layer dosage indicates that superior spreading and surface distribution was achieved, thus resulting in optimal evaporation reductions. However, the savings given by these high dosages proved to be greatly inconsistent, as seen with the parallel wind trial performed on solvent AVh using an 18x molecular layer dosage giving a saving of only 5%. Contrary to common sense, this result illustrates that increased dosages still have the potential to fail and may require special delivery and /or pre-deployment mixing systems in order to enhance their spreading capabilities. For the lower dose trials, the savings obtained from the 6x and 3x molecular layers ranged from 10 to 49% across each of the parallel and attacking wind experiments. This shows that monolayers distributed at lower surface layer densities may provide similar levels of performance consistency in comparison to monolayers delivered at higher surface layer densities. Solvent Ah set to 3x molecular layers provided the most consistent results over both the parallel and attacking wind trials, giving evaporative savings of between 30 to 49%. It is thought that this compound provided good performance and consistency due to achieving a close to ideal balance between monolayer concentration in the delivery solvent and surface spread molecular density. This is an important result from an economic standpoint, as deploying monolayers at lower dosages over extended time periods will provide compounding monetary savings.

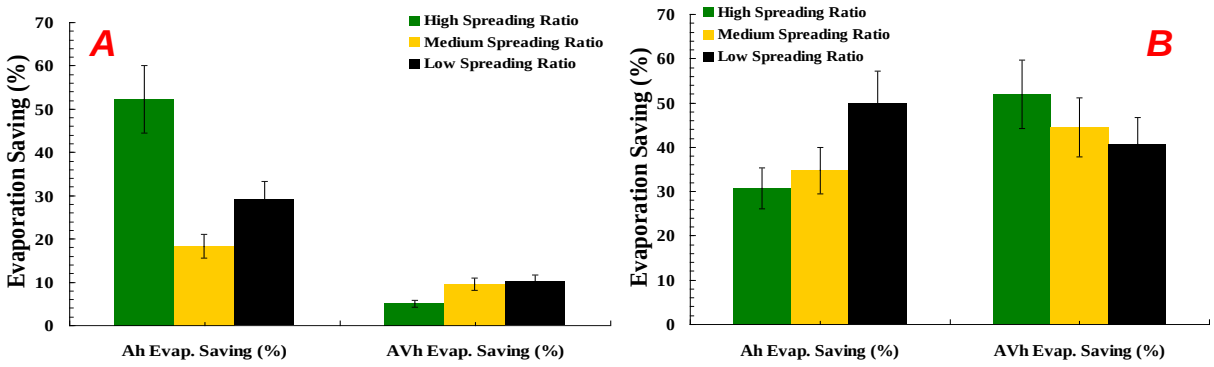


Figure 3. Dosage variation trial results for solvents Ah and AVh. A) Evaporation percentage savings for parallel wind testing with the monolayer spread set to 18x (high), 6x (medium), and 3x (low) molecular surface layers. B) Evaporation percentage savings for 25^{o} angle of attack wind testing with the monolayer spread set to 18x (high), 6x (medium), and 3x (low) molecular surface layers. The error bars in each graph represent the calculated range of uncertainty within each measured percentage saving.

Conclusions, Significance and Impact

From the extensive testing in the large water basin, both the solvent and solid form CRCP prototype monolayers provided greater levels of evaporative savings in comparison to octadecanol suspension. From this it can be concluded that they have the potential to be a much improved system for reducing evaporative losses at a wide variety of real-world water storages. Adequate wide-scale deployment systems are required to be developed in order to optimise the treatment of the monolayers across water bodies and to eliminate the requirement for an end user to be consistently on-site to make reapplications. The ultimate outcome from this project will be to develop an easily deployable and sustainable chemical film monolayer that can endure and adapt to the wildly varying atmospheric and environmental conditions prevalent across both SEQ and Australia in order to protect and save a large amount of water that would otherwise be lost in the industrial and agricultural sectors and also in government and council operated drinking water reserves. The wide-spread uptake of the monolayer, once completely developed and quantified, will greatly assist in reducing the need to enforce harsh water restrictions upon the general public along with limiting the need to extend current water supply infrastructure, such as dams, desalination plants and recycling systems. In turn, this will result in the large fiscal savings for both governments and the tax payer over both the short and long term.

Acknowledgements

The Authors of the manuscript would like to thank both the Urban Water Security Research Alliance and the Cooperative Research Centre for Polymers for funding this project. Also, the authors of this manuscript would also like to thank Ian Underhill, Geoff Turner, Malcolm Duncan and Dave Bellchambers from the School of Engineering Griffith University Gold Coast campus, for their technical guidance and assistance.

References

Barnes, G. and La Mer, V. (1962) The evaporative resistances of monolayers of long-chain acids and alcohols and their mixtures. In Retardation of Evaporation by Monolayers, ed. V. La Mer, Academic Press, London.

Barnes, G.T. (1986) The effects of monolayers on the evaporation of liquids. Adv. Colloid. Interfac. 25, 898-200.

Barnes, G.T. (1993) Optimum conditions for evaporation control by monolayers. J. Hydrol. 145, 165-73.

Barnes, G.T. (1997) Permeation through monolayers. Colloid Surface Physicochem. Eng. Aspect. 126, 149-158.

Barnes, G.T. and Gentle, I.R. (2005) Interfacial Science: An Introduction. Oxford University Press, Oxford, England.

Barnes, G.T. (2008), The potential for monolayers to reduce the evaporation of water from large water storages. Agr. Water Manage. 95, 339-352.

Bikerman, J.J. (1963) The nature of equilibrium spreading pressure. Colloid Polym. Sci. 191(1), 33-35.

Costin, I.S. and Barnes, G.T (1975) Two-component monolayers J. Colloid Interf. Sci. 51, 94-132.

Craig, I. (2005a), Loss of storage water due to evaporation. University of Southern Queensland, Toowoomba.

Craig, I., Green, A., Scobie, M. and Schmidt, E. (2005b) Controlling evaporation loss from water storages. University of Southern Queensland, Toowoomba.

Craig, I., Aravinthan, V., Baillie, C., Beswick, A., Barnes, G., Bradbury, R., Connell, L., Coop, P., Fellows, C., Fitzmaurice, L., Foley, J., Hancock, N., Lamb, D., Morrison, P., Misra, R., Mossad, R., Pittaway, P., Prime, E., Rees, S., Schmidt, E., Solomon, D., Symes, T. and Turnbull, D. (2007) Evaporation, seepage and water quality management in storage dams: A review of research methods. Environmental Health. 7 (3), 84-97.

CSIRO (2007) Environment: Cool savings. http://www.solve.csiro.au/0807/article9.htm.

Dressler, R.G. and Guinat, E. (1973) Evaporation control on reservoirs. Ind. Eng. Chem. Prod. Res. Develop. 12(1), 80-82.

Drummond, C.J. Elliott, P. Furlong, D.N. and Barnes, G.T. (1992) Water permeation through two component monolayers of polymerized surfactants and octadecanol. J. Colloid Interf. Sci. 88, 437-443.
Finn, N. and Barnes, S. (2007) The benefits of shade-cloth covers for potable water storages. CSIRO Textile and Fibre Technology.
Gericke, A., Simon-Kutscher, J. and Huhnerfuss, H. (1993) Comparison of different spreading techniques for monolayers at the air/water interface by external infrared reflection-absorption spectroscopy. Langmuir. 9, 3115-21.
Herzig, M.A., Barnes, G.T. and Gentle, I.R. (2011) Improved spreading rates for monolayers applied as emulsions to reduce water evaporation. J. Colloid Interf. Sci. 357(1), 239 -242.
Knights, S. (2005) Reducing evaporation with chemical monolayer technology, The Australian Cotton Grower. June-July 2005, 32-33.
Mansfield, W.W. (1955) The influence of monolayers on the natural rate of evaporation of water. Nature. 175, 247-249.
Pittaway, P. and van den Ancker, T. (2010), Properties of natural microlayers on Australian freshwater storages and their potential to interact with artificial monolayers. Mar. Freshwater Res. 61, 1083-1091.
Schouten, P.W., Lemckert, C., Parisi, A., Downs, N., Underhill, I. and Turner, G. (2011a), Chemical films and monolayers on the water surface and their interactions with ultraviolet radiation: a pilot study. Meas. Sci. Technol. 22, doi:10.1088/0957-0233/22/6/065703.
Schouten, P.W., Lemckert, C., Parisi, A., Downs, N., Underhill, I. and Turner, G. (2011b), Variable wind speed and evaporation rates: A practical and modelling exercise for high school physics and multi-strand science classes, 57(2), 47-51...
Schouten, P.W., Palada, C., Lemckert, C., Sunartio, D. and Solomon, D.H. (2011c), Analysis of chemical film monolayers under wind and wave conditions. Proceedings of the 34TH IAHR World Congress 2011, Brisbane, 26 June-1 July.
Tsuji, M., Nakahara, H., Moroi, Y.and Shibata, O. (2008) Water Evaporation Rates Across Hydrophobic Acid Monolayers at Equilibrium Spreading Pressure. J. Colloid Interf. Sci. 318(2), 322-330.

Application Systems for Optimal Management of Monolayer Films

Schmidt, E., Brink, G.N. and O'Shannassay, G.
National Centre for Engineering in Agriculture, University of Southern Queensland, Toowoomba, Queensland

Summary

The NCEA has developed a 'smart' autonomous monolayer application system (SAMAS) capable of adjusting the rate and location of monolayer placement, according to prevailing weather conditions. This system, when combined with new monolayer products currently under development, will allow optimum management of monolayer application. Laboratory studies have been undertaken to characterise monolayer distribution on a water surface under different wind conditions. A computer model has been developed to provide an indication of likely spreading of the monolayer product on a storage dam based on prevailing wind conditions, applicator placement and operating conditions. A monolayer application system was deployed on a 16 ha farm dam near Forrest Hill in the Lockyer Valley, South East Queensland (SEQ) to dose a new water-emulsion monolayer product. The trial has indicated the importance of modifying the monolayer application strategy in response to prevailing wind and the need to calibrate the system for future monolayer products. Loss of monolayer film by beaching on the lee shore was also found to be considerable and a floating containment grid will help mitigate future losses.

Keywords
Chemical monolayer, autonomous, electromechanical, low-impact, water, evaporation reduction.

Introduction

Drought and near-drought conditions over much of southern Australia in the last decade has generated renewed research interest into the use of chemical monolayer films ('monolayers') for evaporation suppression on reservoirs. Although monolayers are known to be less efficient and more variable in comparison to other evaporation mitigation techniques, they provide the only feasible option for large water reservoirs (ie, >10 ha in surface area). They require minimal up-front capital outlay, can easily be retrofitted and can be applied selectively during periods of high evaporation. Despite their potential, in-field performance has been shown to be highly variable, ranging between 0-43% efficiency (McJannet *et al.*, 2008). Many researchers have attributed this variability to wind-induced effects, such as drift across the water surface, beaching on the lee shore and waves which can break-up or submerge the film (Fitzgerald and Vines, 1963; Frenkiel, 1965; Crow, 1963; Reiser, 1969). These wind-induced effects act to reduce monolayer coverage of the water surface, which, in turn, reduces the ability of the monolayer to suppress evaporation. Therefore, the logical approach is to manage the application of monolayer in accordance with these wind conditions. This paper reports progress on deployment planning, installation, operation and performance of such a system at a trial site at Logans dam near Forrest Hill in the Lockyer Valley, South East Queensland (SEQ).

The National Centre for Engineering in Agriculture (NCEA) has developed a 'smart' autonomous monolayer application system (SAMAS), capable of adjusting the rate and location of monolayer application according to prevailing weather conditions on-site, to maintain monolayer coverage (Brink *et al.*, 2011). Currently, the system comprises an automatic weather station (AWS), coordinator and a number of applicators. The AWS measures the necessary on-site weather conditions, such as wind speed, wind direction and rainfall, and reports this information to the coordinator. The coordinator then uses this information to determine which applicators to apply from and the appropriate application rate for each. The coordinator relays this information to the appropriate applicators through a wireless communication network. Although the applicators are currently designed to dose monolayer in a liquid form, the rest of the system (ie, AWS, coordinator and the dosing algorithms used in the coordinator) would not need to be altered to cater to other forms of monolayer (eg, powder).

One of these SAMASs has been deployed on the 16 ha farm dam trial site in the Lockyer Valley, SEQ and has been dosing a new water-emulsion monolayer product (which is still in development) for evaporation mitigation.

Results

Deployment Planning

Successful deployment of a monolayer to reduce evaporation is influenced by a number of key considerations, specifically:

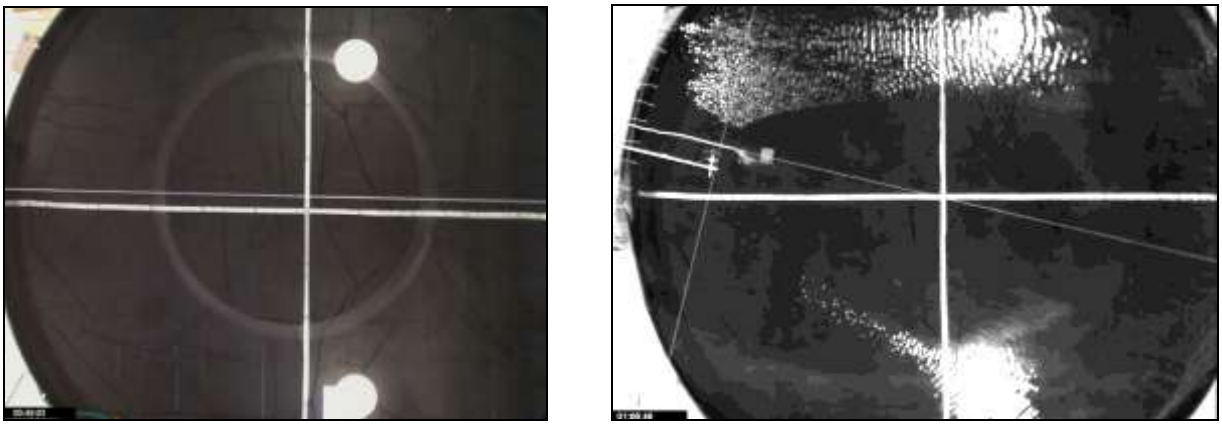
- Scale of storage dam: surface area and shape of the storage.
- Local topography and microclimate: exposure to wind, and orientation of dam including wind run across the water surface.

- Climate: average evaporation rate, rainfall, air temperature, wind speed frequency analysis and prevailing wind direction.
- Water quality and biological factors: water colour, turbidity and water chemistry (pH, electrical conductivity and UV absorbency) all of which are affected by the catchment supplying water.
- Monolayer product choice: it is likely that a range of monolayer compounds could be available that will suit different environmental characteristics of a particular water body.
- Water value: the value of the water and the cost that can be borne to effect savings.

The above considerations are addressed by application of a 'Universal Design Framework' for monolayer application (Brink *et al.*, 2010; Brink, 2011). In this study, the water quality and biological factors have been discussed by Pittaway *et al.* (2011) and the monolayer product used was specified by project collaborators. Hence the main focus in deployment planning was the number and location of applicators and management required to achieve the highest level of coverage according to the site-specific storage and wind conditions.

A key step was determination of the distribution characteristics of the particular monolayer when applied to the water surface under changing wind conditions. Laboratory studies have therefore been undertaken to characterise monolayer distribution characteristics on a water surface, specifically:

- spreading rate and spreading pattern under calm wind conditions (Figure 1a); and
- spreading angle, spreading pattern and drift rate under various wind velocities (Figure 1b).



(a) **(b)**

Figure 1. Images captured from video during large scale laboratory trials (performed on a 5.8m diameter test tank) to derive algorithms for monolayer spreading rate and dispersal angle: (a) Monolayer has been applied and is spreading under zero wind conditions in a linear circular shape; and (b) monolayer being applied continuously at 50 mL/min and dispersing in a wedge shape under an imposed uniform wind speed of 16.2 km/h.

Trends of monolayer drift velocity and spreading angle from the application point, as a function of reference wind velocity, are illustrated in Figure 2a and b for two experimental monolayer products. Similar empirical relations were generated for a range of product dosage rates.

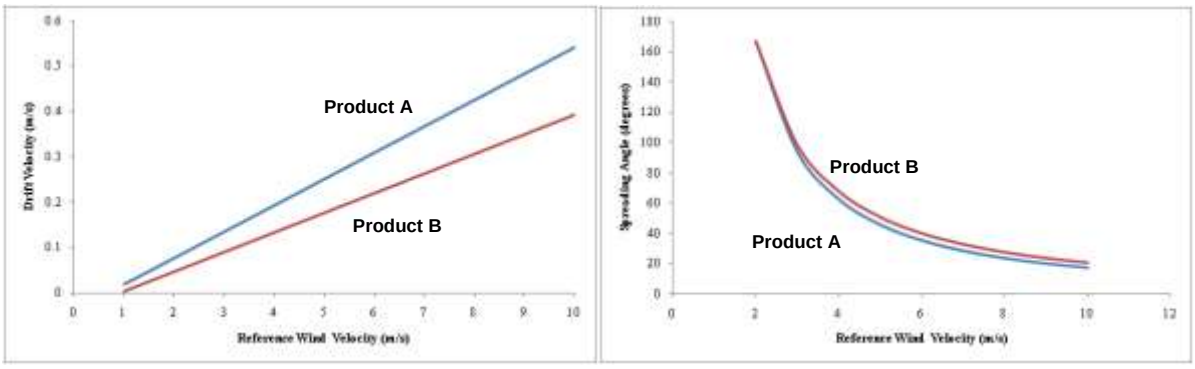


Figure 2a and b. Monolayer drift velocity and spreading angles for two experimental products as a function of reference wind velocity.

Data from the laboratory trials were used in a two-dimensional numerical simulation model to provide an indication of the spatial distribution of monolayer for different application strategies across a larger water body. The model allows different user scenarios to be modelled to determine the effect of different site conditions and user requirements on system layout, applicator numbers and application rate. While the model provides a useful tool to plan the placement and operation of monolayer applicators on Logans dam, simplicity required the following assumptions, each of which is open to question. Each would be reconsidered in further work.

1. The model is unable to account for dynamic wind conditions. Specifically:
    a. Windspeed and direction are assumed constant both across the storage and throughout the period of application for any individual simulation.
    b. Each wind speed and direction condition is considered separate and every simulation starts from zero coverage.
2. Any product that reaches a downwind shore is beached and permanently lost.
3. The effect of wave submergence is not considered.
4. Any biological degradation effects are not considered.
5. Application rate is constant and therefore can be considered analogous with loss rate (once steady-state has been reached). As product beaches, it is applied at the same rate.

A number of different application system arrangements for Logans dam were selected for simulation. The percentage of monolayer coverage for each applicator arrangement was simulated based on meteorological data of wind speed and direction occurrences from the local Gatton meteorological site. A representation of the results of the dispersion model superimposed on a 'Google Earth' aerial image of Logans dam is provided in Figure 3. For this contour plot, wind was from the north and dosage was from four applicators, two on each of NE and NW shores. Product contours represent 2 minute increments of product movement.



Figure 3. Overlay of a contour plot from the dispersion model on top of a 'Google Earth' aerial image of Logans dam.

It was found that nine applicators would provide effective coverage under local conditions and nine applicators were installed, one in the middle of the dam, one in each corner and one midway along each side (Figure 4).



Figure 4. (a) Autonomous applicator used to dose monolayer is floating, self-powered and communicates wirelessly with the coordinator for dosing instructions. **(b)** Illustration of the SAMAS layout on the 16ha farm dam at Forrest Hill in SEQ. Nine applicators were installed in total, one in each corner, one midway on each side and one in the middle of the dam. The coordinator and AWS were installed on the East corner of the dam.

The model was applied to Logans dam based on the nine applicator layout using experimental spreading data for the two monolayer products as well as information on wind speed and direction probabilities for the site. The expected percentage of time a minimum area would be covered, based on various application strategies, could be determined. Figure 5 illustrates that, for a six hour continuous application, Product B would provide a similar surface coverage using significantly less product.

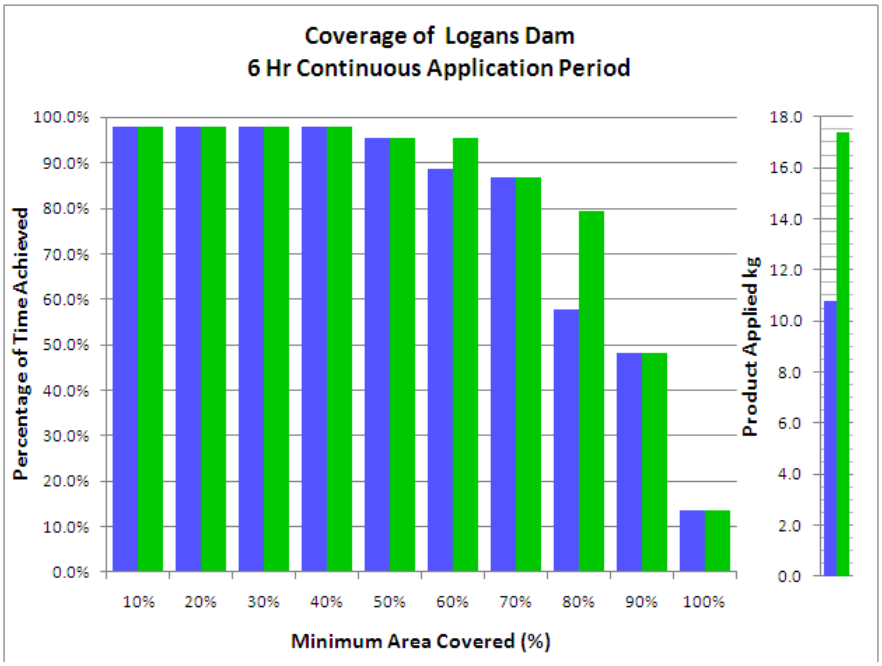


Figure 5. Predicted percentage of time Logans dam would be covered with monolayer based on installation of nine applicators and long-term wind direction and speed occurrences at the site. Two experimental monolayers were assessed based on six hour continuous dosage and recommended dosage rates. Product A Green and Product B Blue.

Installation

Nine floating applicators for dosing monolayer in a liquid form were constructed for deployment at Logans dam. Each applicator uses a 12 V DC peristaltic pump, powered by a sealed lead-acid (SLA) battery which is recharged by a 10 W polycrystalline solar panel. Therefore, the applicators are fully self-powered and portable for remote applications where alternative power sources may be limited, unavailable and/or unviable. The application system hardware is housed in a custom-built metal housing that has four closed-cell polyethylene foam floats for buoyancy (Figure 6a). A 20 L polypropylene reservoir for monolayer is attached to the bottom of the metal housing surrounded

by the foam floats (Figure 6b). Although the capacity of the monolayer reservoir is only 20 L for testing purposes, this would need to be increased for commercial applications, depending on the concentration of the liquid monolayer formulation. The reservoir also has a refill line inside the metal housing, which is accessed by removing the housing lid. The removable lid also allows easy access to application hardware for servicing, replacement, repair or modification (Figure 6c). The applicator is anchored in location via a 20 kg concrete deadweight. The mooring line attached to the applicator passes through an eye on the anchor and is tensioned by a float consisting of a sealed 20 L container as illustrated in Figure 6d.

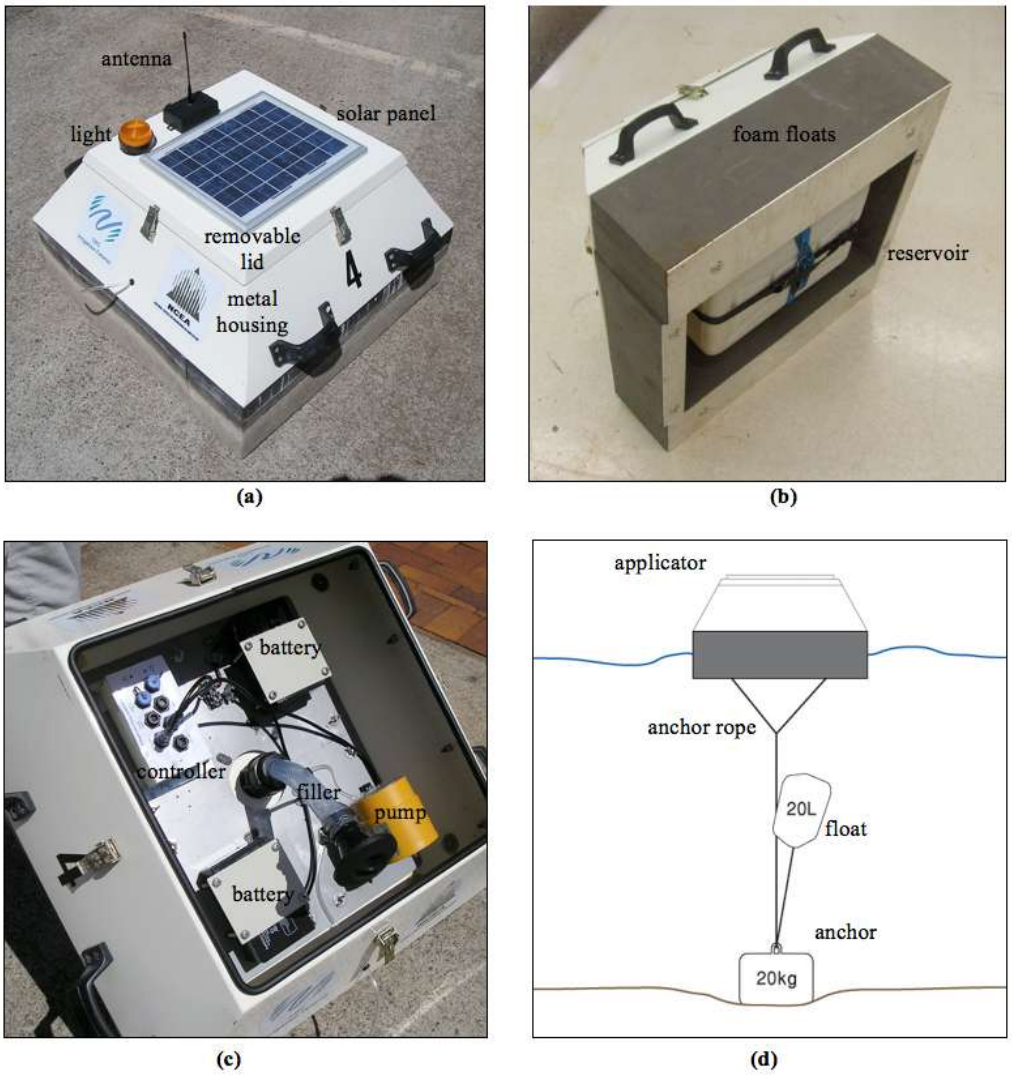


Figure 6. Prototype floating applicator; (a) three-quarter view of the applicator; (b) view underneath the applicator; (c) view inside the metal housing; and (d) schematic of the anchor and mooring set-up for the applicator on a farm dam.

Operation

Decision rules were built into the system to provide dynamic control by the coordinator of dosage rates, based on readings from the weather station. The coordinator unit allows transmission of calculated application rate to each applicator, verification of applications by logging each application timestamp and duration (which is also used for determining storage reservoir volume), communication with the AWS and use of data to perform real-time decisions of application amount. Decision support rules can be modified for each site and essentially determine:

1. Whether on-site conditions are suitable at this time for monolayer application.
2. The required monolayer replenishment rate under zero or low wind conditions.
3. Which applicators will be appropriate for monolayer application under prevailing wind conditions.
4. The monolayer application (or replenishment) rate for each applicator for the prevailing wind conditions.

Detailed information with regard to the prototype monolayer application system and decision support rules is presented in Brink *et al*. (2011).

Operation of the SAMAS was undertaken over the period 23 March to 29 March 2011 based on 15 minute application intervals with dosage adjustments based on prevailing meteorological conditions. Based on the data logs, the application system dosed as planned. However, significant problems were found with the stability of the monolayer product, which stratified in the applicator reservoirs causing inconsistency in the quality of the dosed product. The product eventually thickened to the point where pumping was compromised. On-site observations confirmed that nine applicators programmed for non-continuous dosing at the recommended dosage rate did not consistently achieve suitable levels of coverage of the monolayer product evaluated under high wind conditions.

Subsequent trials with the product were limited to manual dosages of well-mixed fresh product at designated times, with an aim of verifying the ability to measure product presence on the water surface through changes in the directly-measured evaporative flux (using both eddy covariance and large area scintillometry, as set out in McJannet *et al.*, 2011). Figure 7 illustrates the presence of monolayer on Logans dam following such an application: the presence of monolayer increased water surface tension, reducing capillary wave action and this is often directly visible. Further trials of monolayer performance are deferred to the summer of 2011 when new improved formulations should be available.

It was also observed that loss of monolayer by beaching on the lee shore due to wind was considerable (Figure 8a). A floating containment grid made of 100 mm diameter un-slotted drainpipe has subsequently been installed to mitigate future losses (Figure 8b).



Figure 7. Monolayer dosage indicated by surface calming (downwind) at Logans dam 1:44pm 29 March 2011. The floating eddy covariance equipment is just visible in the distance, RHS.





Figure 8a. Beaching of monolayer on the lee shore due to wind and wave action and **Figure 8b** containment grid subsequently installed on site.

Conclusions

Improved formulations and application systems that regulate product application in accordance with site conditions are pre-requisites for the commercialisation of monolayer technology. The NCEA has developed a reliable SAMAS capable of adjusting the rate and location of monolayer application according to prevailing weather conditions on-site. This system, when combined with new monolayer products currently under development, will allow optimised management of monolayer application. For the monolayer product evaluated, nine applicators applying the recommended dosage every 15 minutes, did not achieve suitable levels of monolayer coverage on the 16 ha dam. Revised application strategies, in particular the frequency and rate of dosage, will need to be calibrated for future monolayer products. Loss of monolayer by beaching on the lee shore was also found to be considerable but it has been demonstrated that losses can be mitigated by a floating containment grid adjacent to the shore.

References

Brink, G.N. (2011) *Universal Design Framework for optimal application of chemical monolayer to open water surfaces.* PhD thesis, University of Southern Queensland, Australia [*submitted*]

Brink, G., Wandel, A., Pittaway, P., Hancock, N. and Pather, S. (2010). A 'Universal Design Framework' for installation planning and operational management of evaporation-suppressing films on agricultural reservoirs. Presented at: *International Conference on Agricultural Engineering (Ag Eng 2010)*, Clermont-Ferrand, France, 6-8 September 2010. Book of Abstracts: REF062

Brink, G.N., Symes, T.W. and Hancock, N.H. (2011). Development of a 'smart' monolayer application system for reducing evaporation from farm dams – introductory paper, *Australian Journal of Multi-Disciplinary Engineering* [*In press*].

Crow, F.R. (1963). The effect of wind on evaporation suppressing films and methods of modification. In: Proceedings of the International Union of Geodesy and Geophysics, International association of scientific hydrology. General Assembly of Berkeley, California, p. 26-37.

Fitzgerald, L. and Vines, R. (1963). Retardation of evaporation by monolayers: Practical aspects of the treatment of large storages. *Australian Journal of Applied Science*, **14**, 340-346.

Frenkiel, J. (1965). *Evaporation reduction: physical and chemical principles and review of experiments*, UNESCO, Paris, France.

McJannet, D., Cook, F., Knight, J. and Burn, S. (2008). Evaporation reduction by monolayers: overview, modelling and effectiveness. CSIRO: Water for a Healthy Country National Research Flagship. Urban Water Security Research Alliance Technical Report No. 6.

McJannet, D.L., Cook, F.J., McGloin, R.P., McGowan, H.A, and Burn, S. (2011). Estimation of evaporation and sensible heat flux from open water using a large aperture scintillometer. *Journal Water Resources Research* **47**, W05545, doi:10.1029/2010WR010155

Pittaway, P., Matveev, V. and Stuckey, N. (2011). Effect of a monolayer on water quality at the air water interface. *Urban Water Security Research Alliance Science Forum,* Australian 14-15 September 2011. Book of Proceedings.

Reiser, C.O. (1969). A system for controlling water evaporation. *Industrial and Engineering Chemical Research*, **8**, 63-69.

Effect of a Monolayer on Water Quality at the Air-Water Interface

Pittaway, P.[1], Matveev, V.[2] and Stuckey, N.[1]
[1] National Centre for Engineering in Agriculture, University of Southern Queensland, Toowoomba, Queensland
[2] CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland

Summary

Water quality in an irrigation storage in South East Queensland (SEQ) was monitored on a fortnightly basis, for 18 months prior to monolayer application. Monitoring continued during the eight weeks of intermittent application, when 840 L of an experimental product was applied to the surface of the 16 ha storage. Indicators of water chemistry and biology were monitored on surface (microlayer), immediate subsurface and water column samples. Application of the monolayer compound was immediately detected as an increase in surface tension across the storage. There was no detectable increase above pre-application variation in dissolved oxygen, temperature, pH, conductivity, COD or BOD. Populations of phenol-degrading bacteria and phytoplankton peaked during the application period, indicating that the monolayer compound and/or climatic conditions prevailing during the selected application dates stimulated growth. Monolayer application to this storage appears relatively benign, but results should only be extrapolated to other less agriculturally impacted water storages with caution.

Keywords
Monolayer, air/water interface, microlayer, limnology, water quality.

Introduction

The potential for artificial monolayers to adversely affect the ecology of reservoirs supplying potable water to urban populations is a key issue that must be resolved before the technology can be deployed by water managers. At the molecular level, monolayer application increases the transport resistance in the gaseous and the liquid thermal boundary layers of the air/water interface (microlayer). Increasing transport resistance reduces evaporative loss, and therefore latent heat loss. Minor changes in water temperature, gaseous exchange and microbial processes induced by monolayer application have the potential to adversely affect aquatic biology and water quality (Parker and Barsom, 1970; McJannet *et al.*, 2008).

However, surface films (microlayers) varying in thickness from 100 to 500 μm develop naturally on lakes and ponds (Munster *et al.*, 1998). At the mm scale, these natural surface films also modify surface tension, gaseous exchange and temperature dynamics at the air/water interface (Gladyshev, 2002). The chemically diverse organic constituents of natural films (aquatic humic substances) selectively enhance microbial activity within the microlayer, but lack the molecular homogeneity required to effectively reduce evaporative loss. In this study, seasonal changes in microlayer and immediate subsurface water (taken at a depth of 5 to 10 cm below the surface) sampled from a farm irrigation storage (Logans dam, Forest Hill in SEQ), were monitored for eighteen months prior to the application of an artificial monolayer. Surface tension, pH, electrical conductivity (EC), temperature, UV absorbance (253.7 nm, an index of humified organic carbon), and dissolved oxygen (DO) were monitored every two weeks within one hour after dawn, to establish a base line for natural microlayer variation. The differential between microlayer and subsurface temperature and DO records was used as an index of variation in transport resistance within the gaseous and liquid thermal boundary layers of the air/water interface.

The water column was also monitored every two weeks for pH, electrical conductivity, turbidity, dissolved oxygen, total, nitrate and ammonia (NH_4) nitrogen, total and filterable reactive phosphorus, zooplankton and phytoplankton. The mean weighted crustacean length (a component of the zooplankton) was used as an index of water quality. The biochemical oxygen demand (BOD), chemical oxygen demand (COD), phenol-degrading bacteria (capable of degrading artificial monolayer compounds) and the concentration of non-ionic surfactants were monitored after key seasonal events most likely to change the composition and dynamics of the microlayer. The photoreactivity and total organic carbon content of water sampled from Logans dam was compared with samples from three other storages within the region, to determine how the chemical properties of aquatic humic substances may affect the persistence of artificial monolayer compounds.

Results

Derivation and Properties of Aquatic Humic Substances in Logans Dam

Logans dam was constructed on black soil within the last 10 years, receiving indirect run-off from a cleared pasture catchment. The water is highly turbid, containing a high concentration of dissolved organic carbon that is chemically resilient. Seasonal maxima for COD and BOD occurred in March 2010 (223 mg/L and 150 mg/L respectively), after overland flow from summer storms was pumped into the storage. Seasonal minima in UV absorbance (Figure 1) and

EC were recorded after summer storms, with the highest concentrations coinciding with the lowest water depth. Aquatic humic substances derived from the decomposition of plants are brown in colour, and readily absorb light in the ultraviolet range (Mrkva 1983; Knuutinen *et al.*, 1988). Amphiphilic aquatic humic substances (containing hydrophobic and hydrophilic parts) concentrate at the air/water interface, to produce a natural surface film or microlayer. Compression of the microlayer on the lee shore at Logans dam caused temporary increases in surface pressure (Figure 1) in the absence of monolayer application. However, change in surface pressure over time was not associated with change in the concentration of aquatic humic substances (UV absorbance).

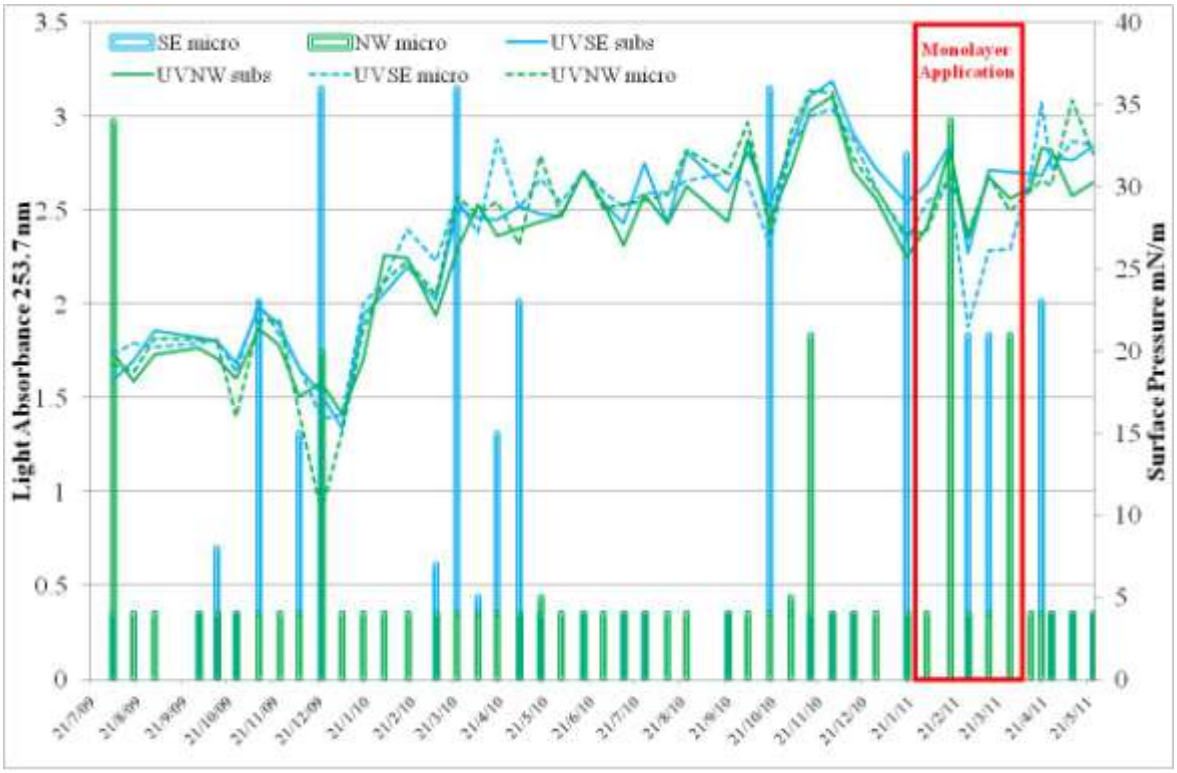


Figure 1. Seasonal change in surface pressure (columns, measured using indicator oils) and absorbance of ultraviolet light (lines, 253.7 nm wavelength) by microlayer and subsurface water from Logans dam (Lockyer Valley, SEQ), sampled fortnightly within the first hour after dawn from July 2009 to June 2011. The red rectangle indicates the duration of monolayer application from 25/1/2011 to 18/04/2011. Water samples were glass fibre filtered prior to analysis in a spectrophotometer. NW refers to water sampled within 2 m of the Northwestern shore of the storage, and SE within 2 m of the Southeastern shore. Sampling was restricted to 4 to 6 m of the lee shore during the peak of algal blooms, to avoid clogging the sampling plate with algal cells.

Logans dam is raised above the water catchment, with water pumped in after overland flow from the catchment fills the table drain. The contribution of fresh, allochthonous inputs (leaf and bark litter) is therefore low, with fresh autochthonous inputs (photosynthetic algal metabolites) proportionally greater (Norkrans, 1980). During the sampling period, the algal population in Logans dam exceeded the Queensland algal bloom alert level of 4 mm^3/L eleven times.

Long -chain alkanes lacking chromophores that absorb UV light are the predominant chemical class of metabolites produced by algae (Gladyshev, 2002). The peaks in surface pressure observed before monolayer application (Figure 1) may be associated with the release of algal metabolites, and/or with chemical changes in older, more structurally complex aquatic humic substances. The pigment of the black vertosol soil within the catchment and from which Logans dam was constructed is derived from very old allochthonous inputs such as charcoal. The complexity of allochthonous organic compounds makes them highly resilient to microbial degradation, but highly susceptible to photodegradation (Moran and Zepp, 1997). However, the rate of photodegradation of water sampled from Logans dam is lower than rates recorded for storages receiving higher rates of fresh, allochthonous inputs (Figure 3). The half-life of the chemical standard Pentachlorophenol for Logans dam under sunny, summer SEQ conditions was 190 seconds, whilst that of the more reactive temporary pond was 100 seconds. Water from the temporary pond had a higher total organic carbon content, but a lower UV absorbance reading (31.63 and 11.05 mg/L, and 1.21 and 3.41 absorbance units respectively). These results suggest that photodegradation may be a key determinant of the half-life of artificial monolayer compounds applied to water storages in SEQ.

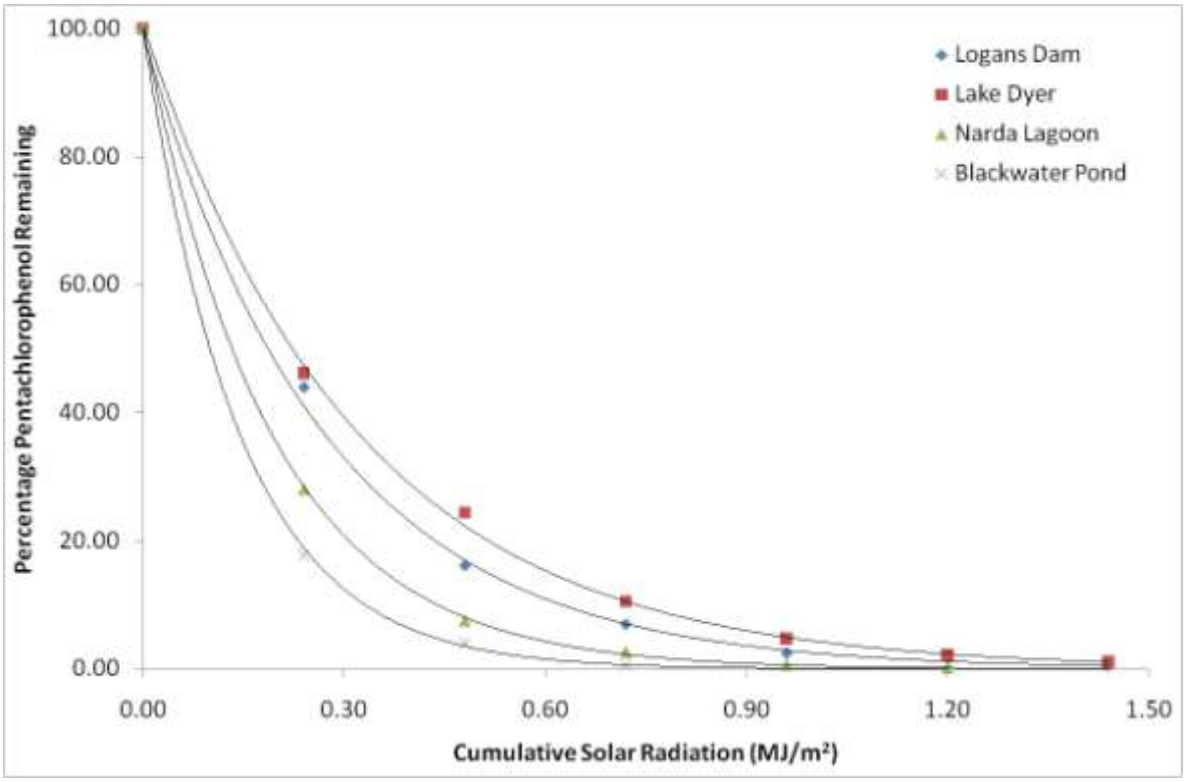


Figure 2. Rates of photodegradation for the chemical standard Pentachlorophenol applied to water sampled from four storages within the Lockyer Valley (SEQ). The highest rate of degradation occurred in a temporary pool containing a high concentration of fresh eucalypt leaf and bark litter. The rate for Logans dam was closer to that of Lake Dyer, receiving water from a cleared, grassy woodland (autochthonous inputs). The more reactive Narda Lagoon storage is fringed by eucalypts and acacias, receiving additional allochthonous inputs of hardwood sawdust from a timber mill located on the bank.

Seasonal Dynamics of Phytoplankton and Zooplankton

Total phytoplankton biovolume was highest with the increase in water temperature over summer, with populations of cyanobacteria growing rapidly as water temperature increased. In summer, the total phytoplankton biovolume repeatedly exceeded the Queensland algal bloom alert level, with the largest bloom of 107 mm^3/L recorded during the monolayer application period (Figure 3). Phenol-degrading bacteria were selected as an index of aquatic heterotrophic bacterial populations most likely to utilise artificial monolayers as a metabolic substrate (Pittaway and van den Ancker, 2010). The population fluctuations of phenol-degraders over the sampling period was significantly correlated with UV absorbance (Pearson Correlation P <0.00, N = 36), but not with either COD or BOD. The population increased after monolayer was first applied, and remained consistently high during the application period of eight weeks.

Water temperature and the availability of ammonia appeared to regulate the abundance of phytoplankton. During the pre-treatment period, about 38% of variance in the total phytoplankton biovolume was explained by the concentration of ammonia in the water column (r^2 = 0.38, P < 0.001). During periods of high total abundance over summer, the cyanobacteria *Microcystis aeruginosa* and *Anabaena* sp were dominant (Figure 3). The peak in abundance occurred during the monolayer application period (Figure 4), suggesting that an increase in the concentration of ammonia may have occurred during this time. No change in ammonium was detected in the fortnightly chemistry monitoring, but phytoplankton are known to sequester free ammonia very rapidly.

Phytoplankton biovolume was not significantly affected by zooplankton grazing. Zooplankton abundance showed a delayed response to phytoplankton abundance (r^2 = 0.30, P <0.01 with a lag of 14 days), but there was no corresponding lag phase in phytoplankton biovolume. The mean zooplankton biomass of 0.42 g/m^2 was relatively low compared to other storages in SEQ, which usually range from 1-2 g/m^2. Prior to monolayer application, the Mean Crustacean Length was significantly lower than for relatively unimpacted storages such as Wivenhoe (0.67 mm and 0.83 mm respectively), suggesting that insecticide residues or some other factor is limiting zooplankton populations. Mean Crustacean Length increased sharply after the start of monolayer application (mean 0.93 in April-May) indicating a shift in size structure towards larger species, but total zooplankton biomass remained relatively stable (Figure 3).

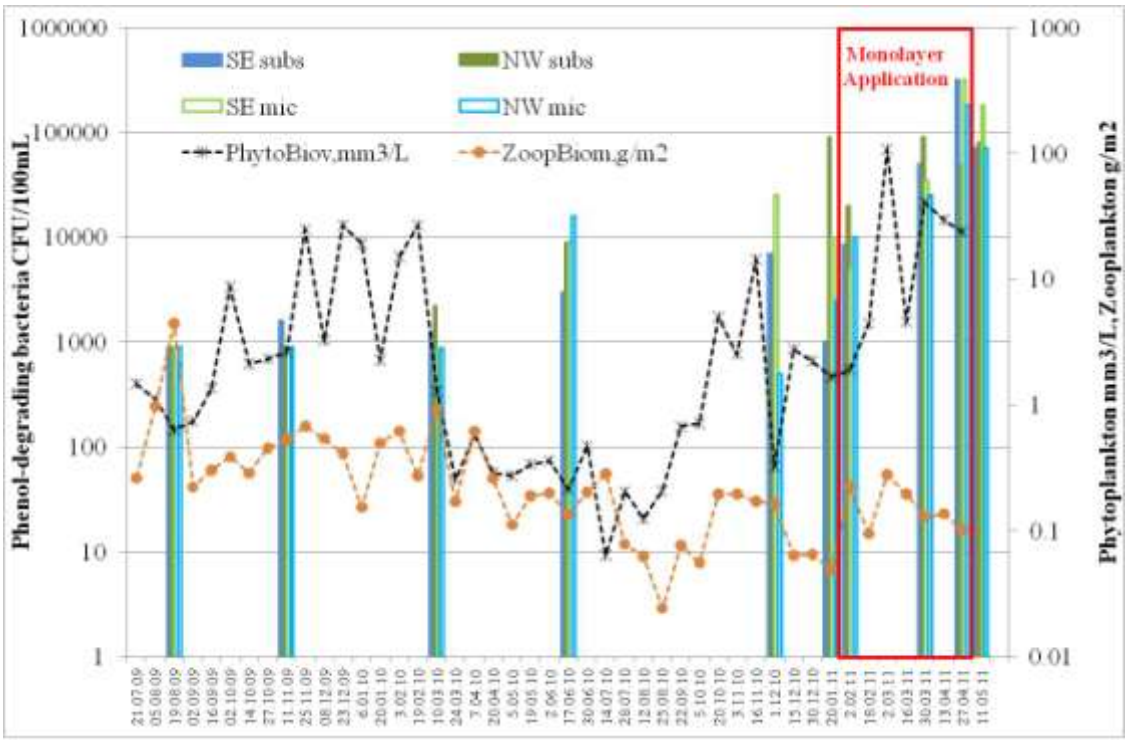


Figure 3. Seasonal change in phytoplankton biovolume and zooplankton biomass in the water column of Logans dam from July 2009 to June 2011, and the activity of phenol-degrading bacteria monitored after key seasonal events and before, during and after artificial monolayer application (red rectangle). The population (Colony Forming Units (cfu) per 100mL) of Phenol-degraders was estimated using the indicator dye remazol blue and a most probable number method.

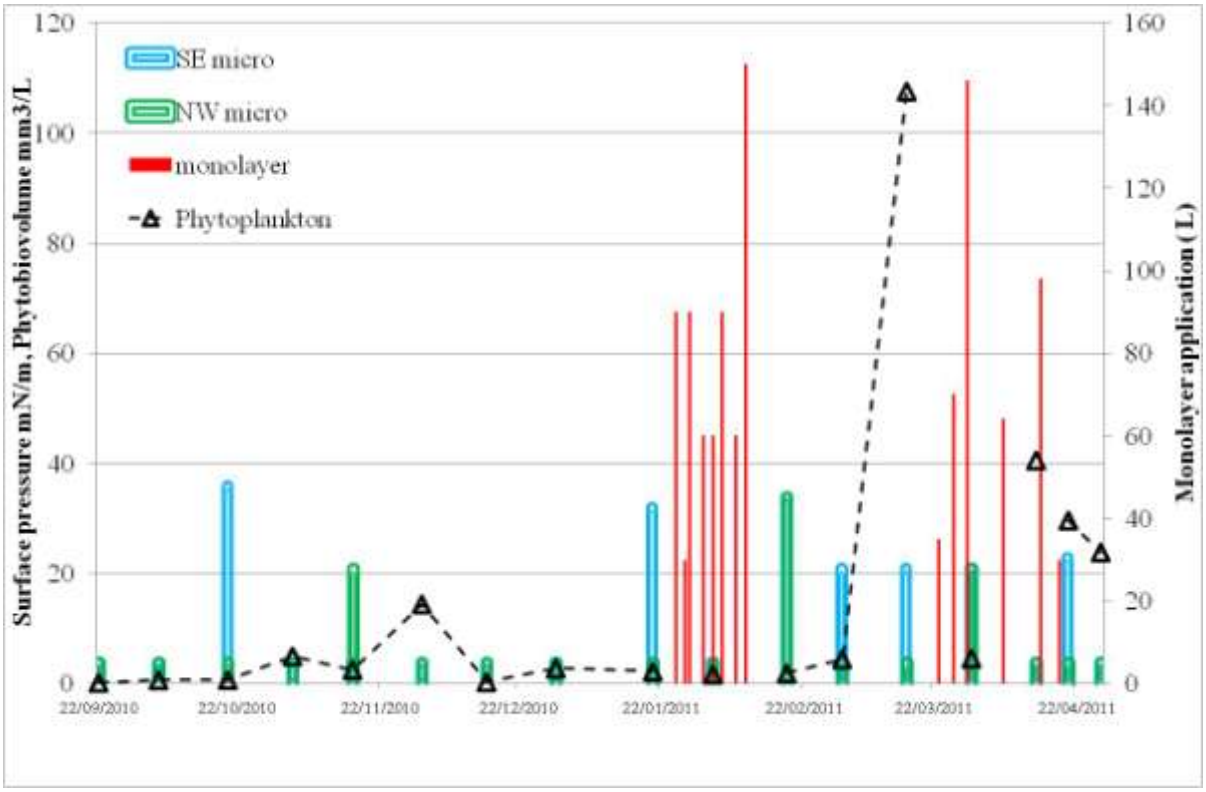


Figure 4. Seasonal change in surface pressure and phytoplankton biovolume of water in Logans dam sampled fortnightly within one hour after dawn from September 2010 to June 2011. Monolayer application commenced 25/1/2011 and concluded 18/4/2011. Application occurred manually from the windward shores, and automatically from floating applicators located around the perimeter and centre of the storage, and was restricted to days with no rain, and wind less than 6m/sec. Monolayer application (solid columns) refers to the total amount of monolayer applied per selected day, to the surface of the 16 ha surface.

Impact of Artificial Monolayer Application on Water Quality

The first evidence of monolayer application was the increase in surface pressure across the storage, which declined to background levels within 24 hours after application ceased (Figure 1). Repeated applications of monolayer increased the duration and uniformity of high surface pressure across the centre and shore-line, relative to pre-monolayer application conditions. The highest surface pressure recorded during application ($\geq$ 35 mN/m), was equivalent to the highest surface pressure recorded for the natural microlayer prior to monolayer application. Due to the limited volume available, monolayer application was restricted to days forecast with no rain and only light winds, conditions under which monolayers exert the greatest potential impact on evaporative loss. Temperature records over this period indicate the water was regularly stratifying in the early afternoon, which may in part explain the peak in phytoplankton abundance (Bormans *et al.*, 2004). Cyanobacteria are capable of exploiting stratification by using gas vacuoles to enter the upper, warmer layer, minimising competition and predation. Phenol-degrading bacteria also increased in population during monolayer application, but in contrast to phytoplankton, populations remained consistently high until after application ceased (Figure 4).

There was no evidence of heat gain within the liquid thermal boundary layer, and the temperature differential recorded during monolayer application remained within the range recorded for the year prior to monolayer application (Figure 5). The impact of a monolayer retarding evaporative loss results in an increase in the temperature of the immediate subsurface water (liquid thermal boundary layer), due to the reduction in latent heat loss. However, on a diurnal basis, the temperature gradient in the liquid thermal boundary layer changes from heat gain during the day to heat loss during the night, compensating for any temperature increase associated with the drop in latent heat loss.

There was also no evidence of a reduction in the dissolved oxygen concentration during monolayer application, with the oxygen differential (microlayer concentration minus subsurface concentration) remaining within the range recorded for the year prior to monolayer application (Figure 5). No increase in the COD or the BOD was observed during monolayer application, and the compound was not detected using the CTAS test for non-ionic surfactants. With the exception of the observed increase in phenol-degrading bacteria, there is no conclusive evidence that application of a total of approximately 1,073 L of monolayer to the 16 ha storage over a period of eight weeks had any adverse impact on water quality. The rise in phenol-degrading bacteria may be interpreted as beneficial, confirming that the compound is biodegradable, and does not produce any potentially hazardous byproducts.

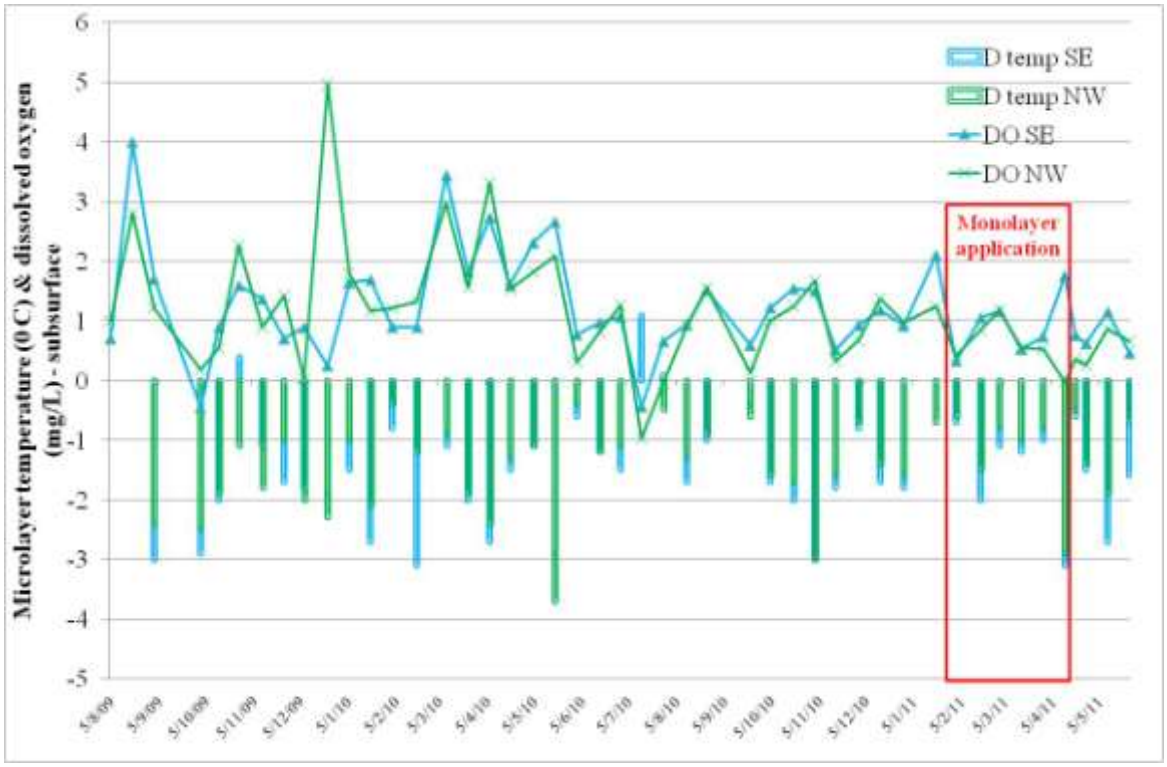


Figure 5. Seasonal change in the difference in temperature (columns) and dissolved oxygen concentration (lines) of the microlayer and subsurface water, sampled fortnightly from Logans dam within the first hour after dawn from August 2009 to June 2011. The red rectangle indicates the duration of monolayer application from 25/01 to 18/04/2011. Water was sampled within 1-2 m of the Southeastern (SE) and Northwestern (NW) shore lines. Sampling was restricted to 4 to 6 m of the lee shore during the peak of algal blooms, to avoid clogging the sampling plate with algal cells.

Conclusions

The reactivity and concentration of aquatic humic substances in Logans dam is representative of other medium-sized agricultural and/or recreational water storages in SEQ. However, relative to larger potable water storages that are not as affected by agricultural land use practices (eg, Wivenhoe dam), the frequency of algal blooms in the absence of monolayer application is much greater. Phytoplankton biovolume peaked during the eight-week monolayer application period, which could be stimulated by the combination of monolayer application and natural causes such as water column stratification. The Mean Crustacean Length increased during the application period, suggesting a compositional shift towards larger species of zooplankton associated with monolayer application, but total zooplankton biomass did not increase.

The population of bacteria most likely to degrade monolayer compounds (biosurfactant-producing phenol-degraders) peaked and remained consistently high during the application period. The high rate of photodegradation recorded for Logans dam suggests that the synergy between photo- and microbial degradation renders this monolayer compound readily biodegradable. The application of approximately 1,073 L of monolayer to the 16 ha storage over a period of eight weeks did not induce any detectable change in the pH, conductivity, BOD or COD of the water, and there was no evidence of temperature deviation or change in the concentration of dissolved oxygen beyond pre-application levels. These results suggest that the application of this specific monolayer formulation to the highly turbid, agriculturally impacted water of Logans dam is relatively benign. However, results from this study should only be extrapolated to water storages with different water chemistry and biology, or with the application of a different monolayer formulation, with caution.

References

Bormans, M., Ford, P. W., Fabbro, L. and Hankock, G. (2004). Onset and persistence of cyanobacterial blooms in a large impounded tropical river, Australia. Marine and Freshwater Research 55, 1-15.

Gladyshev, M. (2002). Biophysics of the Surface Microlayer of Aquatic Ecosystems. IWA Publishing London, United Kingdom.

Knuutinen, J., Virkki, L., Mannila, P., Mikkelson P., Paasivirta, J. and Herve, S. (1988). High-performance liquid chromatographic study of dissolved organic matter in natural waters. Water Research 22, 985-90.

McJannet, D., Cook, F., Knight, J. and Burn, S. (2008). Evaporation reduction by monolayers: overview, modelling and effectiveness. CSIRO: Water for a Healthy Country National Research Flagship. Urban Water Security Research Alliance Technical Report No. 6.

Mrkva, M. (1983). Evaluation of correlations between absorbance at 254nm and COD of river waters. Water Research 17, 231-35.

Moran, M. A. and Zepp, R. G. (1997). Role of photodegradation in the formation of biologically labile compounds from dissolved organic matter, Limnology and Oceanography 42, 1307-1316.

Munster, U., Heikkinen, E. and Knulst, J. (1998). Nutrient composition, microbial biomass and activity at the air-water interface of small boreal forest lakes. Hydrobiologia 363, 261-70.

Norkrans, B. (1980). Surface microlayers in aquatic environments. Advances in Microbial Ecology 4, 51-85.

Parker, B. and Barsom, G. (1970). Biological and chemical significance of surface microlayers in aquatic ecosystems. Bioscience 20, 87-93.

Pittaway, P., and van den Ancker, T. (2010). Properties of natural microlayers on Australian freshwater storages and their potential to interact with artificial monolayers. Marine and Freshwater Research 61, 1083-1091.

How Long Will a Rainwater Tank Last? Do You Know?

Moglia, M., Tjandraatmadja, G. and Sharma, A.
CSIRO Land and Water, Highett, Victoria

Introduction

In Australia, rainwater tanks in urban areas were initially promoted and subsidised in order to reduce the use of mains water for garden irrigation. Their application was subsequently extended to indoor toilet and/or laundry uses through legislation. The expected water conservation benefits from rainwater tanks as part of the overall water planning strategy and the ongoing long-term effectiveness of these tanks are of critical concern to water planners. In South East Queensland (SEQ), similar to other parts of Australia, rainwater tanks are owned and operated by individual householders. Maintenance of rainwater tanks requires the observance of simple and uncomplicated tasks such as: maintenance of gutters; regular cleaning of first flush devices; strainer and/or filter clean-up to prevent blockage of the tank inlet; ensuring adequate protection of tank openings to prevent entry of vectors or breeding of mosquitoes; the upkeep of structural integrity of the tank; and operation of pump systems. In rural areas, rainwater tanks have been maintained and have served as a reliable supply of water for many decades as being the only potable water source. In urban areas, however, and particularly in areas were tanks are mandated, it is difficult to assess if tank owners are providing the same level of care to ensure their long-term on-going operation. The question is: will rainwater tanks in SEQ continue to operate effectively for years to come? And if this is not the case, what should be done about it? And whose responsibility is it? This study explores some of these questions through a series of surveys to provide some initial insights on the topic.

Keywords
Rainwater tanks, asset management, governance.

Theoretical Framing

This study has been planned and undertaken on the basis of participatory research and modelling principles. The aim is to help a group of stakeholders to identify and adopt some kind of strategy to address an identified problem, ie, rainwater tank governance. The stakeholders are all those that have a direct or indirect stake in the issue as per definition by Nandalal and Simonovic (2003). From the outset, it was also recognised that rainwater tank governance, because of the inter-mingling of public and private interests, is a highly contentious topic; and this has subsequently been confirmed via interviews and the web based survey of professionals. This has considerable implications for the research methodology in the sense that governance implementation decisions are outside of the scope of research. These implications are described in Table 1.

Table 1. Methodological implications of dealing with a contentious issue.

Stakeholders	- Need to identify and involve all stakeholders in the process. - Allow stakeholders to have some input into the identification of strategy alternatives.
Transparency	- Strive for communication and engagement with all stakeholders. - Strive to increase the awareness of the interest and preferences of all stakeholders.
Protect Core Values	- Contentious topics involve issues that can create real strife or moral outrage amongst stakeholders, and these therefore need to be identified. - Handle and attempt to resolve conflict in a way that aims to find compromise or consensus solutions.
Substance	- Being clear about the role of researchers as providing information and to facilitate discussion. - Ensuring research process provides multiple alternatives from which decision makers can choose.

Note: These points have been adapted from HarmoniCOP (2005) which is a key output of a major EU project into improving participation in water management.

On the basis of a growing literature of experience in participatory research (Jones *et al.,* 1999), these participatory considerations have:

1. Normative functions by increasing the legitimacy of the process of knowledge generation;
2. Substantive functions by allowing greater integration of more sources of knowledge and information, hence improving the capacity for problem solving; and
3. Instrumental functions by trying to help build collaborative relationships to assist with implementation of strategies and for reducing conflict.

In light of these considerations, a key initial research step is to identify the stakeholders, and their roles and preferences in relation to rainwater tank governance. Furthermore, it is necessary to explore the nature of the rainwater tank governance problem, and the extent of the problem. Thirdly, it is necessary to identify, with the help of stakeholders, a range of possible strategies to deal with rainwater tank governance.

The stakeholders were identified on the basis of a semi-structured and convergent interview process. Semi-structured because a strict script was not followed, but diagrams, tactics and concepts were used to facilitate questions, discussion and conversation (as per Cohene and Easterbrook 2005). This method was chosen in recognition of an exploratory research topic where not much is known at the outset (ie, demanding less structure), but where there is still a need to somewhat steer the direction of interviews in order to achieve the goal of developing a stakeholder map. Convergent because the representation of the list/map of stakeholders develops and changes as more people are interviewed; and one may stop when interviews no longer provide changes or additions (as per Dick 2002).

Another issue of concern was that the project was motivated by so far unsubstantiated claims that rainwater tanks are not very well managed and that they have a tendency to fail if not maintained. There is hardly any literature with hard numbers available on condition of rainwater tanks and the benefits of maintenance. Hence, the project is largely based on hypothesis of possible would be worlds. As a survey of rainwater tank condition as part of the project was out of the question, a survey of judgments was devised in order to get some baseline information.

The survey was also used to explore attitudes, and values relating to rainwater tank governance, and to elicit possible strategies that may be employed in rainwater tank governance. However, whilst the survey reached out to many key stakeholder groups (ie, government, council, service enterprises and others) it was still largely incomplete as it did not explicitly target a key category of stakeholders, ie, householders. This survey is hence complementary to that carried out with householders by Tucker *et al.* (2011).

Finally, it was recognised that in order to evaluate strategies and to make sense of complex condition and failure rate data, it was necessary to devise a basic computer simulation model. This model which employs a Monte-Carlo simulation framework and assuming the Markov property in tank condition (ie, only the current condition is important and the history is unimportant for making predictions) allows for exploring the benefits of various maintenance schedules. Whilst it takes as input failure modes and related failure frequencies (as identified in the professional survey), the output is estimates of failure frequencies that are likely to occur in a stationary population of rainwater tanks.

The outputs of the above research steps are briefly described in this paper. The key outputs are:

1. Stakeholder mapping;
2. Judgments about the level of maintenance and condition of rainwater tanks;
3. Failure simulation results; and
4. Identification of some possible strategies for consideration.

Brief Results of Stakeholder and Issues Mapping

In simplified terms the key stakeholder groups that were identified are shown in Table 2.

Table 2. Stakeholder groups, roles and interests in rainwater tank governance.

Stakeholder Group	Role	Interest
Householders	Householders have rainwater tanks for a range of reasons, including having: • Purchased a tank using the government rebate scheme. • A newly developed house with a mandated tank. • A tank to provide water for gardening during times of restriction.	Paid and own rainwater tank and its water. Interests differ across the group: • Rainwater in tank is a personal discretionary supply of water. • Rainwater tank is part of water supply infrastructure (mandated tanks). • Want water supply to work. • "Feel good", helping the environment".
Council	Councils are responsible for implementation of State legislation at local level. This includes: • Administration of rainwater tanks rebates claims during their validity period. • Approvals of new dwelling plans to ensure compliance with local laws and compliance with Queensland Development Code (QDC) MP4.2. • Inspection and certification of rainwater tanks in new dwellings if contracted to do so. • Handling public health (mosquitoes), contaminants and amenity complaints and prosecution. • In some instances testing of water quality.	• Manage certification requirements, but cannot ensure that all properties certified by third parties are inspected to their standards. • Have no role after implementation unless following up on the investigation of complaints on public health, contaminant or amenity. • Are concerned that tanks may become a public health problem if not properly maintained. • Want systems to comply with accreditation and safety requirements for their life and rainwater tanks to operate smoothly so that minimum intervention is required. • Resourcing if to extend beyond current roles.

Stakeholder Group	Role	Interest
State government	Develops goals, strategy and legislation for achieving water demand management as part of the overall water planning. Examples include: • Queensland Development Code MP4.2-water savings targets. • Public Health Act: protection of public Health. • Plumbing Act: ensure minimum standards for systems and installation.	• Achieving long-term development goals and sustainable development.
Private enterprises supply (excl. plumbers)	This includes a fairly broad group including: • Developers: develop new areas and build houses for sale to the public. They are required to comply with local and State legislation. Typically sell house packages with pre-determined and standardised rainwater system options. • Contractors (builders, building certifiers): hired by developers for their services. • Rainwater tank/system retailers: stock and sell rainwater tanks and system components to private and corporate customers. Often sell rainwater and tank bundle to homeowners.	• Maximisation of profit upon sale/service. • Contractors: maintain service contracts and fulfill service requirements. Prefer clear guidelines and requirements standards for construction, installation and certification of dwellings and rainwater systems. • Retailers: provision of goods and services as requested by buyer. Benefit from products with well defined specifications and standards.
Plumbers/Service enterprises	• Service providers qualified to install, inspect and maintain rainwater tanks and their components. They are also required to comply with the Plumbing and Drainage Act and to provide indemnity and liability for services provided. Some plumbing companies specialise on rainwater tank installation and maintenance. • They can advice householders on rainwater system needs and maintenance, but householders decide if they want to implement their recommendations.	• Believe rainwater tanks are an asset for future sustainability. • Have observed rainwater systems with issues such as wear and tear, incorrect plumbing and material. • Want more rigors in rainwater installation standards and minimum quality requirements for system components. • Want improvement in quality of rainwater tank installations and its standards. • Want clarity in legislative requirements (roles and liabilities) regarding rainwater tanks.
Academics	• Explore knowledge gaps and develop recommendations/strategies/tools in a range of disciplines. • Often provide advice/recommendations to government/industry.	• Want legislation and regulations to be developed based on facts (scientific basis).
Consultants	• This includes a broad range of professionals that may be involved in design, implementation, project management, as well as research and development of O&M guidelines and transfer of developments where rainwater tanks are adopted among other technologies.	• Believe rainwater tanks contribute to sustainability. • Want well defined design standards and guidelines, clarity in legislative requirements (roles and liabilities) regarding rainwater tanks design, O&M.

Brief Results of Professional Survey

The stakeholder groups (council, state government, service enterprises etc) were surveyed about a range of topics related to the governance of rainwater tanks. This was done using a questionnaire distributed via SurveyMonkey to a number of email distribution lists. This yielded 175 responses from Queensland, and 253 responses nation-wide. 57% of respondents were owners of a rainwater tank, and the breakdown of participants into stakeholder categories is: work for council (13.4%); government (17.5%); service enterprises (46.3%); and other (6.1%). Approximately 17% of respondents did not state their affiliation. The questions were related to some broad topics: 1) adequacy of operation and maintenance (O&M) of the existing rainwater tank stock; 2) types and causes of problems; 3) frequencies of problems occurring; 4) responsibilities and roles in governance of tanks; and 5) possible strategies in governance of tanks.

The types of problems that people considered to be the most problematic were: 1) breakdown of pumps/parts (36%); 2) water quality issues (25%); 3) mosquitoes (23%); and 4) structural integrity (8%). Furthermore, only 8% of respondents thought that other issues ought to be included in the list, and the types of issues that were suggested were: filters/mesh getting clogged; sludge build-up; blocked inlet screens; failure of tank support; discoloration of water; and noise from pumps. There were some differences in the focus between different stakeholder groups. Councils had a higher rate of respondents focusing on water quality, and state government had a higher rate of respondents focusing on mosquito issues; whilst non-affiliated respondents suggesting at a disproportionally high rate that there are no problems with the rainwater tanks.

When asked about the cause of problems, 41% reported that they thought it was *Lack of Operation and Maintenance* that was the culprit, followed by *Poor installation* (17%), *Incorrect O&M* (15%), *Design failure* (15%), and *Other* causes (13%). Differences in judgments exist between different stakeholder groups but the trend is clearly consistent amongst all groups that the level of maintenance is the main cause for concern. Subsequently, when asked whether tanks are being adequately maintained, there is a clear difference in judgments between different stakeholder groups (see Figure 1). For example 90% of government respondents believe that 25% or less of tanks are adequately maintained, in contrast to nearly half of respondents that have not indicated an affiliation (ie, N/A), who believe that 99% of all tanks are adequately maintained. It is also notable that owners are considerably more pessimistic than non-owners. When asked about the frequency of failure modes, results are as shown in Table 3.

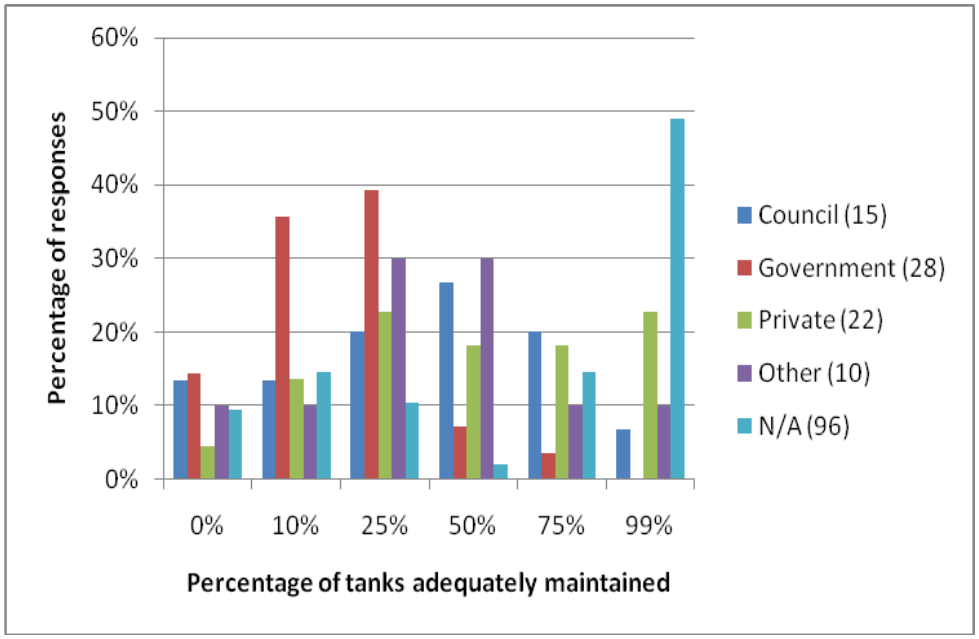


Figure 1. Judgments on the level of maintenance of rainwater tanks in SEQ.
Note: this is the responses to the question: What level of maintenance do you believe is currently carried out for mandated rainwater tanks in South East Queensland?

Table 3. Survey respondents' expected time until failure, per failure mode.

Frequency	Pumps/Parts	Gutters (no overhanging trees)	Gutters (overhanging trees)	Structural Integrity of Tank and Foundation	Mosquito Meshing
Less than a week	-	1%	7%	-	-
Less than a month	3%	1%	23%	-	2%
About two months	2%	1%	26%	-	-
About six months	9%	16%	25%	1%	10%
About a year	9%	18%	7%	1%	20%
More than a year	-	64%	13%	-	-
About two years	28%	-	-	-	16%
More than two years	50%	-	-	-	50%
About four years	-	-	-	10%	-
About ten years	-	-	-	84%	-

Note: There are also considerable differences in the judgments by different stakeholder groups on the frequency of occurrence of these types of failures. As a general principle, owners are more pessimistic than non-owners; and council and state government are more pessimistic than those in service enterprises or those who are non-affiliated (N/A).

The data in Table 3 are obviously not real failure data, but are based on perceptions and judgments, some of which will be based on personal experience. It is recognised that it is not easy to make these judgments (even when there is personal experience), but that this is the second best option in the absence of real data. This can be seen as the equivalent type of data collected about local practices or environmental concerns (for example fishers or farmers tend to have considerable local and personal experience and they use heuristic rules for decision making) that need to be used in natural resource management dilemmas where strategic decision makers are often poor on (scientific and technical) information (for an example, see Foale 2006).

When asked about views on which type of stakeholder ought to have responsibilities to inspect, maintain or pay for O&M of rainwater tanks, there were some consistent and clear responses. Not surprisingly, a vast majority (78%) believes that households ought to maintain tanks. A smaller majority (56%) believes that households also ought to be responsible for inspections, whilst 17% believe that councils ought to be responsible, whilst some argue that a third party (such as a service enterprise) ought to take responsibility for inspections. It is notable that council and state government respondents have a higher rate of believing that councils ought to be responsible for inspections, but neither council nor state government are showing any desire to be responsible for maintenance of tanks. The overall attitude towards greater involvement of private enterprises (PEs) is somewhat ambiguous. 41% of responses are completely against this notion, whilst 14% believe that this should be the case; and 41% of the respondents believe that PEs may play a role in certain circumstances. Many are very skeptical about the quality of service that PEs might be able to provide and are worried that they may be overly expensive. The overall view however seems to be that their role is to provide a service on a voluntary basis to those that require it.

In the survey, there was a question on the issue of a potential payment on an annual basis for regular upkeep of the tank. This question raised a fair amount of angry responses (13 out of 175, ie, 7%). The type of concerns raised in relation to these "angry" responses is: outrage against government interference in service matters; 2) concerns over "punishing" tank owners for contributing to a public good; and 3) concerns that the community is already being charged too much for water. Despite the level of outrage at this question, many still volunteered an estimated monetary value that they thought householders may pay for upkeep on an annual basis, and the cumulative distribution of these responses is shown in Figure 2. The median estimate is $50 and the mean estimate is $100 (although 80% of participants think that it may be less than $100).

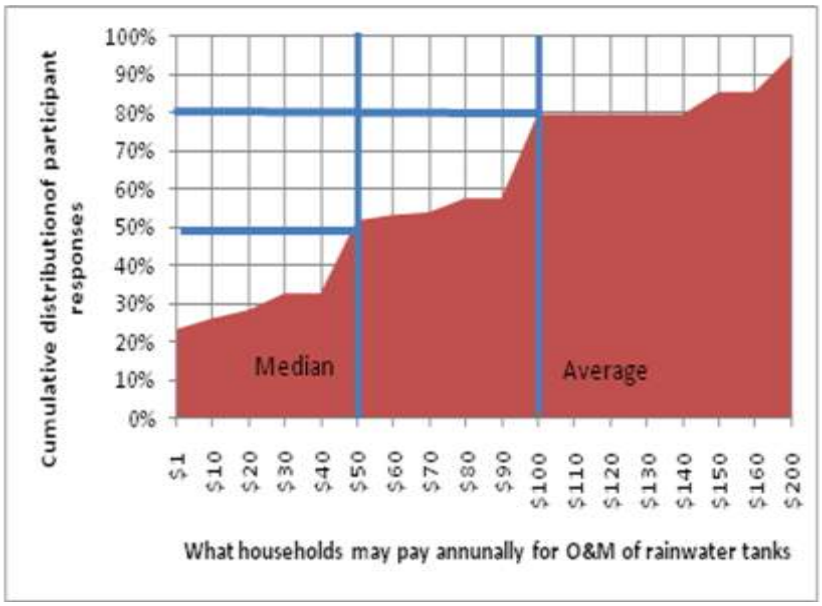


Figure 2. Cumulative distribution of estimates of how much households may be willing to pay for upkeep of tanks.

Failures Simulations

To explore the benefits of inspection and maintenance schedules in terms of reduced rates of failure and a healthier population of rainwater tanks, Monte Carlo simulation was undertaken (scripts available in the forthcoming project report). The simulation is essentially an experiment of what happens with a population of tanks with known probability of failure, day by day. The rates of failure have been estimated on the basis of the survey responses that are described in Table 3, and converted into cumulative probability functions, survival functions and hazard rates (for definitions of these see Crowder *et al.,* 1991). The simulation algorithm was then repeated for different hazard rate tables (one for each failure mode) and different inspection times to arrive at the Table 4 below. These two tables show the relationship between frequency of inspections, assuming maintenance occurs in conjunction of inspections when failures are found, and the occurrence of failures in a population of rainwater tanks.

Table 4. Relationship between frequency of inspections and rates of failures in a population of tanks.

Frequency of Inspections	Proportion of Tanks with Blocked Gutters (overhanging trees)	Proportion of Tanks with Broken Pumps	Proportion of Tanks with Broken Meshing	Proportion of Tanks that are Structurally Broken	Proportion of Tanks in Good Condition (excl. gutters)
1 month	17%	1%	1%	0%	98%
3 months	39%	4%	4%	0%	92%
6 months	54%	8%	10%	1%	82%
1 year	67%	19%	21%	2%	63%
2 years	75%	37%	38%	5%	37%

Discussion

Whilst the research in this project has been exploratory, some take home messages have already been identified.

- There is inadequate information regarding the condition of the existing asset stock and its impact.
- Based on judgments of expected time to occurrence of various types of tank failures, tanks would need to be inspected and fixed regularly or the likelihood of failure increases rapidly (see Table 4).
- There is a considerable level of resistance and outrage in the community in response to any notion of greater government control of what is perceived as a private matter.
- Most believe that O&M, as well as inspection of tanks, is a matter for householders themselves.
- It is unlikely that households will want to pay much more than $50 for annum for upkeep of tanks, especially if it is a mandatory charge.

These points are, however, troubling in the sense that most would agree that tanks are currently not adequately managed, but few seem to want to challenge the status quo of full householder responsibility (although Councils do undertake spot checks). Householders are, perhaps on valid grounds, also not likely to want to pay for the kind of regular inspections that appear necessary (see Table 4). For example, if inspections would cost $150, then the householders' contribution of $50 per year would mean only an average of one inspection every third year. If these were the only inspections, then less than a third of all tanks would be fully functional. Obviously, one will have to assume that householders do indeed take some responsibility for inspection and maintenance of tanks.

It seems clear that the key to this dilemma is in the motivation of householders to undertake the O&M of their tanks, and this is explored in a parallel project (Tucker *et al.*, 2011). The gut instinct by policy makers would perhaps be to grasp for either the carrot or the stick, but this is probably a mistake given the inherent power relations of such strategies and what appears to be a low level of trust between government and citizens. The better option would seem to be to work together with the community in a respectful manner, and in a way that helps further motivate and empower householders to maintain tanks. The difficult part however is that this may be a costly exercise and no-one so far seems willing to contribute the necessary funds to ensure that tanks are adequately maintained. At the end of the day, strategic water planners will need to assess the value for money of maintaining an asset stock of rainwater tanks, in comparison to other supply investment strategies. It is however difficult to evaluate the cost benefit of inspections in the absence of hard data on failure frequencies of rainwater tanks.

If funds are found to tackle this problem, then a number of different strategies may be considered. A program of spot checks to assess the condition of tanks is an obvious component of any strategy. The extent of this program will have to be based on strict cost-benefit analysis, which will need to be based on a stock take of tanks, and quality reliability data. Furthermore, it appears that there is a real need to provide more information to householders about O&M of tanks, and the basic tasks that need to be performed. This may be distributed in many different ways, such as with the water bill, annual rate reminder, via email or in the mail. Educational programs in media may help raise awareness and create an environment where householders are more motivated. However, there is also a considerable level of skepticism about such media campaigns and about their real value. If there was better monitoring of tank condition, it may be possible to assess the efficiency of such campaigns with the use of comparative studies.

Finally, the quality of tanks needs to be regulated, and there is a need to continue the program of monitoring the installation and design practices. Despite the extensive program that currently exists, design and installation is still raised as a major cause of failure, and one may ask how this program can be improved.

Conclusions

Rainwater tanks are one way of achieving the water savings targets for SEQ mandated under QDC MP4.2. It is essential to ensure that the rainwater tanks are functional over a long period of time as being the part of the overall water planning. Very little is known about their existing conditions and predictions into the future. Thus, there is a real need to collect reliable information via condition surveys of household rainwater tanks to initiate discussion on the level of awareness of tank owners, adequate management as well as operation and maintenance programs. Private and public agencies can play an important role in ensuring on-going operation of rainwater tanks, which can lead to various business models. However, detailed economic analysis is required towards the cost benefit analysis of possible business models considering community willingness to pay.

References

Tilbrook, R. (2009). Rainwater tanks in new housing development: a survey of the issues, knowledge and uses of residents in the Northlakes Estate, A study for Lake Macquarie City Council, NSW; University of Newcastle (available from bhodges@lakemac.nsw.gov.au).

Nandalal, K.D.W. and Simonovic S.P. (2003). *State-of-the-art report on systems analysis methods for resolution of conflicts in water resources management*. A Report Prepared for Division of Water Sciences UNESCO, UNESCO-IHP, 127.

Cohene T. and Easterbrook S. (2005). Contextual risk analysis for interview design, in: *Proceedings of the 13th IEEE International Conference on Requirements Engineering (RE'05)*, IEEE Computer Society, Washington DC.

Crowder, M.J, Kimber, A.C., Smith, R.L. and Sweeting, T.J. (1991). Statistical analysis of reliability data, London: Chapman and Hall.

Dick, B. (2002). *Convergent interviewing, Session 8 of Areol - action research and evaluation on line*, Available from: http://www.uq.net.au/action_research/areol/areol-session08.html, Accessed: 25th March 2010.

Foale, S. (2006). The intersection of scientific and indigenous ecological knowledge in coastal Melanesia: implications for contemporary marine resource management, International Social Science Journal 58(187), pp. 129-137.

Tucker, D., Mankad, A. and Greenhill, M. (2011). *Rainwater Tank Adoption in South East Queensland – Factors Influencing Maintenance and Management*, Urban Water Security Research Alliance 3rd Science Forum, Brisbane 14–15 September 2011.

Rainwater Tank Adoption in South East Queensland – Factors Influencing Maintenance and Management

Tucker, D.I.[1], Mankad, A.[2] and Greenhill, M.[1]
[1] CSIRO Ecosystem Sciences, Floreat, Western Australia
[2] CSIRO Ecosystem Sciences, Ecosciences Precinct, Dutton Park, Queensland

Summary

Water shortages in South East Queensland have necessitated investigations of ways to reduce household water use. Rainwater tanks are just one of a suite of possible decentralised systems available to householders. In the past, financial incentives have been offered for rainwater tank installation, and since 2007, the Qld State Government introduced legislation requiring new homes to meet water savings targets, most often achieved through the installation of rainwater tanks. While the number of household rainwater tanks in SEQ continues to grow, past research on decentralised systems adoption suggests a need to understand potential differences between residents who have voluntarily installed tanks and those who install due to legislation. It remains unclear which tank management and maintenance behaviours people engage in, if any, once a rainwater tank is installed and whether there are differences between people who have voluntarily installed tanks versus those who have installed them due to legislation. The research hypothesis is that those who voluntarily install rainwater tanks are likely to be more motivated to engage in maintenance behaviours than those installing due to government legislation. Self-Determination Theory provides the framework for interpreting results. Results have implications for the way in which household level alternative water supply systems are encouraged in the future in order to maximise efficient use.

Keywords
Decentralised systems, behaviour, community adoption, self determination theory, rainwater tanks, maintenance behaviours.

Introduction

In South East Queensland (SEQ), inconsistent rainfall and unprecedented population growth have resulted in a water strategy reliant on permanent water conservation measures (Queensland Water Commission, 2010). There is also a greater emphasis on household installation and use of decentralised water systems such as rainwater tanks. Individual household tanks allow residents to collect and use rainwater in a localised way, reducing reliance on the central water grid. As part of the Queensland Development Code MP4.2- Water savings targets (QDC 4.2), all homes built in SEQ from 1 January 2007 are required to save 70 kL of mains water per year. One acceptable solution for achieving the water savings is through the installation of a 5 kL rainwater tank connected to 100 m^2 roof area and plumbed to the washing machine cold water tap, toilets and at least one external tap (DLGP, 2008). The mandate further includes regulations for the maintenance of tanks, including mosquito proofing and devices such as 'first flush' devices that keep contaminants out of the tank. Many existing homes in SEQ are also being encouraged to retrofit such systems within the home's existing infrastructure. The State and Federal governments, at various times, have offered short-term rebates designed to encourage and compensate individuals who choose to install a 3,000-5,000 L rainwater tank on their property and later on, rebates were available for those households who chose to plumb their retrofitted tanks into some of the home's water fixtures (Beal, *et al.*, 2011).

Given the high proportion of homes in SEQ that have a rainwater tank (mandated or retrofitted), it is important for water stakeholders to understand the types of rainwater tank use and management behaviours that residents are engaging in. The primary research question in the present study examines how people use their rainwater, and, how homeowners are maintaining their tanks (if at all) for long-term capability and performance. It is believed that factors associated with the choice to install a tank will be influential in attitudes and behaviours associated with rainwater use and tank maintenance and affect one's motivation to engage in certain behaviours. The objective for this study is to understand the key motivational descriptors that vary among residents who have voluntarily installed tanks on their property (ie, *retrofitted*) and those who have installed a tank based on the QDC 4.2 (ie, *mandated*).

The framework for this psycho-social analysis is Self-Determination Theory (SDT) (Deci and Ryan, 1991; 1995). This theory is concerned with understanding the motivational drivers of choice (eg, why do people choose to get a rainwater tank, what motivates someone to maintain their tank) and focuses on the degree to which an individual's behaviour is self-motivated (ie, self-determined). The application of SDT in the environmental context has demonstrated the utility of the theory in explaining pro-environmental behavioural choices, as well as how different types of motivation can regulate behaviour. According to the theory, there are three innate psychological needs that must be fulfilled in order to self-motivate individuals to initiate behaviour (Deci and Ryan, 2000; Deci and Ryan, 2002):

- **Competence:** belief that one is effective in dealing with the situation they are experiencing;
- **Autonomy:** the universal urge to be independent; and
- **Relatedness:** the need to interact and be connected with others.

SDT also describes that human motives can be explained in terms of three key categories of motivation, defined by the amount of self-determination associated with engagement in the target behaviour(s):

- **Intrinsic motivation:** engaging in activities for the inherent pleasure and satisfaction derived from the practice of the activity itself;
- **Extrinsic motivation:** engaging in an activity because the goal of the behaviour is to bring about positive consequences or to avoid negative consequences; and
- **Amotivation:** individuals experience a lack of control from the activity, making the actions meaningless and almost mechanical.

It was hypothesised that higher perceptions of competence and autonomy will predict more frequent engagement in tank-related behaviours and that the level of self-determined motivation (ie, intrinsic, extrinsic, amotivated) will also influence these behaviours.

It is believed that results from this study will help researchers in understanding differences in tank maintenance and rainwater use behaviours among residents in SEQ based on a greater understanding of motivational influences. This information can inform policy and water researchers as to the facilitating and inhibiting variables that can impact public engagement with rainwater tanks, as well as residents' willingness to look after their tanks, once installed.

Methodology

The study comprised a mail out survey, containing a range of questions designed to collect descriptive information about participants' rainwater tank set-up, as well as psycho-social questions (ie, self-determination theory variables) to provide greater insight into attitudes and behaviours of rainwater tank owners. Specifically, along with demographic data, the survey contained scales adapted from the Motivation Towards the Environment Scale (MTES; Pelletier *et al.*, 1998), the Amotivation Towards the Environment Scale (AMTES; Pelletier *et al.*, 1999), and items relating to tank ownership and perceived water rights (developed by the authors). The dependent variable in this study was maintenance of rainwater tanks, including engagement in rainwater tank maintenance behaviours.

Participants were 1,984 SEQ residents who had a rainwater tank installed on their property. These participants were randomly selected from a database of households known to have rainwater tanks. In order to compare differences between mandated and retrofitted tank owners, selection of participants was defined by two criteria:

(1) Households with *retrofitted rainwater* tanks - these households applied for government rebates to install rainwater tanks at home. They were not required by legislation to have rainwater tanks at home but chose to have them (in conjunction with government rebates).

(2) Households with *mandated* rainwater tanks - these households applied for new water accounts from 1 January 2007 under QDC4.2 include the mandated rainwater tanks.

A total of 10,000 households were approached to take part in the study, via a mail out of the survey. This included 5,000 households considered to have retrofitted tanks, and 5,000 thought to have mandated tanks. A total of 570 mandated and 1,443 retrofitted groups returned their surveys. This discrepancy in sample numbers between groups is likely due to the fact that many people who had applied for licenses to build homes since 2007 (ie, those in the mandated group), had not yet finished building their homes and so valid postal locations for these properties did not exist. Approximately 5% of the mandated database was deemed unreachable due to this reason.

A three-staged mail-out strategy was employed in the present study. The first stage commenced in late March 2011 and involved mailing out an introductory postcard, to notify potential participants of the upcoming survey on rainwater tanks. One week later, potential participants were sent a questionnaire, along with a return paid envelope and an eco-friendly shopping bag with CSIRO logo. The eco-friendly shopping bag was intended to serve as a minor incentive to encourage participants to complete their surveys (Dillman, 2007). Two weeks after mailing the survey, the final stage was implemented and comprised a reminder postcard sent to participants, encouraging them to return their completed surveys. Participants were given approximately six weeks to return completed surveys.

Results

Comparing Mandated and Retrofitted Adopters with Self-Determination Theory

Motivation Towards the Environment Scale (MTES)

Independent samples *t*-tests were used to compare the average scores between mandated and retrofitted groups on each of the MTES subscales, as well as the total weighted SDI score for individuals in each group. *T*-tests comparing means for each subscale are presented in Table 1 and it can be seen that all of the self-determination subscales were significantly different between mandated and retrofitted groups, except for external regulation. However, from visual inspection of the means and calculation of effect sizes (η^2), the magnitude of differences between the groups was small, as all effect sizes were around the .01 value. According to interpretation guidelines proposed by Cohen (1988), 0.01 is considered a small effect.

Table 1. *T*-test results comparing mandated and retrofitted groups on the six MTES subscales.

	TYPE	Mean	SD	t	df	Sig.
AMOTIV	Mandated	10.66	5.71	3.78	1986	.000
	Retrofitted	9.57	5.83			
EXTER	Mandated	14.54	5.39	-.74	1986	.462
	Retrofitted	14.74	5.42			
INTRO	Mandated	14.34	4.13	-2.55	1067.65	.011
	Retrofitted	14.87	4.29			
IDENT	Mandated	21.80	4.69	-6.72	943.42	.000
	Retrofitted	23.33	4.24			
INTEG	Mandated	17.02	5.83	-5.01	1074.78	.000
	Retrofitted	18.52	6.12			
INTRI	Mandated	18.20	5.10	-3.60	1071.34	.000
	Retrofitted	19.12	5.32			

Note: amotivation (AMOTIV), external regulation (EXTER), introjected regulation (INTRO), identified regulation (IDENT), integrated regulation (IDENT) and intrinsic motivation (INTRI).

The summary of *t*-tests in Table 1 indicates that:

- Mandated participants reported experiencing greater amotivation than those in the retrofitted group;
- Participants with retrofitted rainwater tanks reported significantly greater introjected regulation, identified regulation, integrated regulation and intrinsic motivation than mandated participants; and
- Overall, individuals within the retrofitted group were more motivated to engage in tank maintenance behaviours than those in the mandated group.

Another independent samples *t*-test was also conducted to compare the SDI between mandated and retrofitted groups. Results showed that there was a significant difference between SDI scores for mandated (M = 35.04, SD = 34.38) and retrofitted (M = 44.68, SD = 34.79) groups, t (1986) = -5.58, p = .000. The magnitude of the differences in means was relatively small (η^2 = .02). These results demonstrated that participants who had retrofitted their domestic rainwater tanks recorded significantly higher self-determined motivation than those with mandated rainwater tanks.

Amotivation Towards the Environment Scale (AMTES)

Independent samples *t*-tests were used to compare the average scores between mandated and retrofitted groups on each of the two AMTES subscale composites. These *t*-tests indicated that mandated participants scored significantly higher on the effort/capacity scale (see Table 2), indicating a greater level of amotivation within these subscales. However, no significant differences were identified between mandated and retrofitted groups on the help/strategy scores. Analysis of eta squared values was undertaken to assess the magnitude of the differences in the effort/capacity scores, which showed the effect to be small (η^2 = 0.02).

Table 2. Comparison of mandated and retrofitted scores on AMTES composite scales: effort/capacity and helplessness/strategy.

	TYPE	Mean	SD	t	df	Sig.
Effort/Capacity	Mandated	2.80	1.25	5.75	1986	.000
	Retrofitted	2.43	1.31			
Help/Strategy	Mandated	2.21	1.25	-1.17	1986	.24
	Retrofitted	2.28	1.33			

Engagement in Maintenance Behaviours

MTES

A hierarchical multiple regression (HMR) was used to assess the ability of the six MTES subscales to predict engagement in rainwater tank maintenance behaviours (DV), after controlling for the influence of tank installation type (ie, mandated vs. retrofitted). Preliminary analyses were carried out to ensure no violation of the assumptions of normality, linearity, multicollinearity and homoscedasticity. The type of tank installation (mandated/retrofitted) was entered at Step 1, explaining 5% of the variance in maintenance behaviours. After entry of the six MTES subscales at Step 2, the total variance explained by the model as a whole was 27%, F (7, 1980) = 102.43, $p < .001$. The MTES subscales explained an additional 21% of the variance in maintenance behaviours, after controlling for the type of installation, $R^2_{change} = .21$, F_{change} (6, 1980) = 95.29, $p < .001$. In the final model, installation type and two of the MTES subscales were statistically significant contributors, with the amotivation subscale recording the highest beta value ($\beta = .26$, $p < .001$), indicating that amotivation was the most important predictor of maintenance behaviours within the model. Integrated regulation recorded a higher beta value ($\beta = .23$, $p < .001$) than installation type ($\beta = .18$, $p < .001$), showing their relative contribution to the prediction of tank maintenance behaviours.

The results of the HMR indicate that while the type of tank installation is important in predicting tank maintenance, motivational factors – in particular, amotivation and integrated regulation – play a significant role in maintenance behaviours above and beyond that explained by the circumstances under which one has installed a tank.

AMTES

A HMR was used to assess the ability of the two AMTES subscales to predict engagement in rainwater tank maintenance behaviours (DV), after controlling for the influence of tank installation type (ie, mandated vs. retrofitted). Preliminary analyses were carried out to ensure no violation of the assumptions of normality, linearity, multicollinearity and homoscedasticity. The type of tank installation (mandated/retrofitted) was entered at Step 1, explaining approximately 5% of the variance in maintenance behaviours. After entry of the two composite AMTES variables at Step 2, the total variance explained by the model as a whole was 21.4%, F (3, 1984) = 113.05, $p < .001$. The AMTES subscales explained an additional 16% of the variance in maintenance behaviours, after controlling for the type of installation, $R^2_{change} = .16$, F_{change} (2, 1984) = 202.36, $p < .001$. In the final model, installation type the two adjusted AMTES subscales were statistically significant contributors, with the effort/capacity scale recording the highest beta value ($\beta = -0.43$, $p < .001$), indicating that effort/capacity was the most important predictor of maintenance behaviours within the model. Instillation type was also a significant predictor of maintenance behaviour ($\beta = .18$, $p < .001$) whereas help/strategy was not ($p > .01$).

The results indicate that people who felt they didn't have the capacity to understand or know how to maintain their tanks, and those who mentally felt they couldn't make the effort to maintain their tanks, were less likely to engage in tank maintenance due to a lack of directed, or apathetic, motivation.

Conclusions

Results relating to the first two hypotheses (H1 and H2) showed that, despite small effect sizes, retrofitters reported more self-motivated behaviour towards rainwater tank maintenance. In contrast, participants with mandated rainwater tanks were found to have higher levels of amotivation than retrofitters, suggesting that individuals with mandated rainwater tanks may experience a lack of control and independence when relating to their rainwater tank and, subsequently, their drive to engage in maintenance behaviour may lack self-directed motivation. As a result, relating to one's tank could be meaningless and perhaps mechanical, leading to a reduction or cessation in the behaviour (ie, tank engagement). This indicates potentially poor maintenance outcomes among the mandated sample, suggesting that motivation driven by perceptions of choice and autonomy are important in consistent tank maintenance behaviours.

In terms of influential motivational attributes encouraging individuals to engage in rainwater tank maintenance, in spite of the type of tank installation (ie, retrofitted or mandated) one may have, the results consistently highlighted *integrated regulation*, *amotivation*, and *effort/capacity* as important predictors of whether one will carry out maintenance on their tank and how often. *Integrated regulation* was found to be high among retrofitters and this indicates that although retrofitters are not engaging in tank maintenance for purely intrinsic reasons, they are engaging in maintenance because it helps them to achieve a goal (eg, supplementary water) while also serving as an expression of one's personal values (eg, sustainability). Therefore, individuals with high integrated motivation have the drive to look after their tank and learn more about tank maintenance so as to ensure the longevity of their tank.

Conversely, *amotivation* can lead to least self-determined types of behaviour and amotivation was found to be significantly higher among the mandated sample. As an influential predictor of tank maintenance, the results highlighted that having a sense of apathy with respect to owning a rainwater tank can negatively impact future engagement in tank maintenance. Individuals high in amotivation also have little drive to learn more about their tank or to use the rainwater collected for appropriate applications. The amotivational subscale composite *effort/capacity* was also found to be an important negative predictor of maintenance, with mandated individuals once again reporting higher scores on this factor.

The message from this finding is that among the mandated sample, people felt as though they did not know enough about rainwater tank maintenance and they were not willing to put in the effort to find out more or to engage in many of the maintenance behaviours required. This suggests that greater education is needed among those who install rainwater tanks or other decentralised systems on their property as part of a government mandate, rather than as an individual choice to do so. The subject of "choice" seems an important one when dealing with psychological motivation that will ultimately drive householders' behaviour.

Overall, this research demonstrates that the use of Self-Determination Theory in understanding and explaining rainwater tank maintenance behaviour is appropriate and effective. Findings from this research have highlighted that engaging in maintenance behaviour is most likely if people are self-directed in their adoption of decentralised technology, rather than if they feel a lack of motivation towards the process. That is, allowing individuals to make their own informed choice with respect to owning and using a rainwater tank is likely to predict more frequent and correct engagement in tank maintenance. This will also promote a greater sense of tank ownership among householders, which is further likely to predict increased maintenance behaviours, as demonstrated by this research.

In conclusion, there is an indication that there are important difference in the tank maintenance behaviours of those with retrofitted tanks and those with mandated tanks. It is recommended that future research further examine this discrepancy in tank maintenance behaviours between the two tank ownership groups and potential educational interventions to determine the effect of hands-on tank maintenance knowledge. This type of strategy is likely to positively impact perceptions of competency, autonomy and relatedness, which is likely to lead to greater self-determined behaviour with respect to engagement in tank maintenance. If maintenance issues are not addressed at the psycho-social level, there is a real danger that in as little as 10 years, SEQ may be dealing with significantly greater tank maintenance issues, such as disrepair and abandonment. Encouraging frequent and correct maintenance among all types of decentralised system users at this early adoption stage will help to minimise problems in the future.

References

Beal, C., Gardner, T., Sharma, A. and Barton, R. (2011). A desktop analysis of potable water reductions from internally plumbed rainwater tanks in South East Queensland. *Urban Water Security Research Alliance Technical Report No. 26*. http://www.urbanwateralliance.org.au/publications.html. Accessed 6 June 2011.

Deci, E. L. and Ryan, R. M. (1991). A motivational approach to self: Integration in personality. In R. Dienstbier (Ed.), Nebraska Symposium on Motivation: Vol. 38. *Perspectives on motivation* (pp. 237-288). Lincoln: University of Nebraska Press.

Deci, E. L. and Ryan, R. M. (1995). Human autonomy: The basis for true self-esteem. In M. Kernis (Ed.), *Efficacy, agency, and self-esteem* (pp. 31-49). New York: Plenum.

Deci, E. L. and Ryan, R. M. (2000). The "what" and "why" of goal pursuits: Human needs and the self-determination of behavior. *Psychological Inquiry, 11*, 227-268.

Deci, E. L. and Ryan, R. M. (2002). *Handbook of self-determination research*. Rochester, NY: University of Rochester Press.

Department of Local Government and Planning (2008). Queensland Development Code, *MP 4.2-Water Savings Targets*, Queensland Government, September 2008.

Dillman, D. A. (2007). *Mail and internet surveys: The tailored design method* (2nd ed.). Hoboken, NJ: Wiley.

Pelletier, L. G., Dion, S., Tuson, K. and Green-Demers, I. (1999). Why do people fail to adopt environmental protective behaviors? Toward a taxonomy of environmental amotivation. *Journal of Applied Social Psychology, 29*, 2481-2504.

Pelletier, L. G., Tuson, K. M., Green-Demers, I., Noels, K. and Beaton, A. M. (1998). Why are you doing things for the environment? The Motivation Toward the Environmental Scale (MTES). *Journal of Applied Social Psychology*, 28, 437-468

Queensland Health (2007). Rainwater tanks: A guide to keeping your tank safe. Queensland Government, Queensland Health. http://www.health.qld.gov.au/ph/Documents/ehu/32922.pdf. Accessed 7 December 2010.

Queensland Water Commission (QWC) (2010). *South East Queensland Water Strategy*. The State of Queensland (Queensland Water Commission) 2010. http://www.qwc.qld.gov.au/planning/pdf/seqws-full.pdf. Accessed 6 June 2011.

Electrochemical Treatment of Problematic Reverse Osmosis Membrane Concentrates

Bagastyo, A.Y.[1], Radjenovic, J.[1], Kristiana, I.[2], Mu, Y.[1], Batstone, D.J.[1], Gernjak, W.[1], Joll, C.[2], and Rabaey, K[1].

[1] Advanced Water Management Centre (AWMC), The University of Queensland, St Lucia, Queensland

[2] Curtin Water Quality Research Centre, Curtin University, Bentley, Western Australia

Summary

In this study, the performance of five anode materials (ie, Ti/IrO_2-Ta_2O_5, Ti/RuO_2-IrO_2, Ti/Pt-IrO_2, Ti/PbO_2, and Ti/SnO_2-Sb) in oxidising reverse osmosis concentrates (ROC) from Bundamba advanced water treatment plant (AWTP) was evaluated. The best performance in terms of dissolved organic carbon (DOC), dissolved organic nitrogen (DON), and chemical oxygen demand (COD) removal was observed for Ti/Pt-IrO_2 and Ti/SnO_2-Sb anodes with an accumulated current (Q) = 0.55 Ah L^{-1}. However, the enhanced oxidation was accompanied by the formation of oxidation by-products, eg, trihalomethanes (THMs) and haloacetic acids (HAAs) due to the reaction of organic materials with the electrogenerated oxidants, eg, HOCl/HOBr, with a consequent increase in baseline toxicity (determined in *Vibrio fischeri* bioassays). Electrochemical reduction using a carbon block cathode was therefore evaluated as post-treatment. A decrease of adsorbable organic halogen (AOX) during electrochemical reduction suggested the removal of toxic by-products through dehalogenation/adsorption mechanisms, providing opportunities for coupling electrochemical oxidation and reduction.

Keywords

Coupled electrochemical oxidation and reduction, adsorbable organic halogen, membrane concentrate.

Introduction

During recent years, there has been an increasing interest in applying electrochemical methods for the treatment of refractory waste streams containing high concentrations of Cl^- ions, including landfill leachate (Deng and Englehardt, 2007), and in particular, reverse osmosis concentrates (ROC) from water treatment processes (Dialynas *et al.*, 2008; Perez *et al.*, 2010; Van Hege *et al.*, 2004). This is because the high conductivity in these streams allows for efficient electrochemical oxidation and reduction. Disposal of these streams represents a concern due to the large enrichment of organic contaminants such as pharmaceuticals, pesticides, endocrine disrupting compounds and others. Biological treatment processes, in many cases, are inefficient due to recalcitrance of the compounds of interest, as well as the presence of saline and other inhibitors. Apart from alternative physico-chemical and chemical oxidation processes evaluated for the treatment of ROC (Bagastyo *et al.*, 2011; Dialynas *et al.*, 2008; Westerhoff *et al.*, 2009), electrochemical oxidation has been investigated due to its versatility and in-situ generation of various reactive oxygen species (ROS) (eg, $OH^{\bullet}$, O_2, H_2O_2, O_3) (Anglada *et al.*, 2009). Furthermore, this method is beneficial since the high concentrations of Cl^- ions would not only promote indirect oxidation via reactive halogen species (RHS) (eg, Cl_2/ClO^-, ClO_2), but they also lower the ohmic resistance of the system, making it more energy-efficient. The indirect oxidation would include chlorination and chloramination during the electrochemical oxidation of ROC. Chloramination refers to the reaction of ammonia and organic amines present in ROC with chlorine species (ie, Cl_2, HOCl, and OCl^-) forming combined chlorine, which represents the organic and inorganic chloramines formed.

Among the wide range of electrode materials evaluated to date, mixed metal oxide (MMO) coated electrodes are readily available in practical mesh geometries with improved electrocatalytic activity and stability, making them attractive for the treatment of various waste streams (Panizza, 2010). As classified by Comninellis (1994), the so-called "active" anodes (eg, Ti/IrO_2-Ta_2O_5, Ti/RuO_2-IrO_2, and Ti/Pt-IrO_2) have a strong interaction with electrogenerated $OH^{\bullet}$, forming higher oxides which are effective catalysts for oxygen evolution reactions (OER) but mainly lead to partial oxidation of organics. In the case of "non-active" anodes (eg, Ti/PbO_2, and Ti/SnO_2-Sb) this interaction is weak, resulting in a poor activity for OER and thus favoring complete oxidation. Therefore, the type of anode coating is expected to influence the electrochemical reaction pathway and by-products distribution, caused by the different concentration of electrogenerated ROS and RHS (Comninellis, 1994). The main issue raised among environmental electrochemists is the detrimental effect of by-products on the outcome of oxidation (Richardson *et al.*, 2002).

The application of biological treatments for detoxification may not only be limited by the relatively long reaction time and strict reaction conditions (McRae *et al.*, 2004), but also by the toxicity of organic halides and their dehalogenated products (Paul *et al.*, 2005). Since adsorption on activated carbon may not perform well for compounds which have higher hydrophilicity, eg, haloacetic acids (HAAs), electro-reductive processes is an alternative method to detoxify and dehalogenate the by-products formed during electrochemical oxidation (Korshin and Jensen, 2001; Li *et al.*, 2007). Presently, there are extensive studies of electrochemical reduction of halogenated organic compounds at various electrode materials (Rondinini and Vertova, 2010), including the application of

carbon-based electrodes which are known for their robustness, low cost, mechanical stability, and reasonably high over-potential for the competing hydrogen evolution reaction (Durante *et al.*, 2009; Zou *et al.*, 2008).

In the present work, we have investigated the performance of MMO anodes represented by five different coating materials (ie, Ti mesh coated with IrO_2-Ta_2O_5, RuO_2-IrO_2, Pt-IrO_2, PbO_2, and SnO_2-Sb) in electrochemical oxidation of ROC. The efficiency of electrochemical oxidation on each anode was evaluated in terms of dissolved organic carbon (DOC), dissolved organic nitrogen (DON), and chemical oxygen demand (COD) removal, as well as generation of hazardous oxidation by-products, eg, THMs and HAAs. The four measured THMs were chloroform ($CHCl_3$), bromodichloromethane ($CHCl_2Br$), dibromochloromethane ($CHClBr_2$), and bromoform ($CHBr_3$). The six measured HAAs were monochloroacetic acid (MCAA), dichloroacetic acid (DCAA), trichloroacetic acid (TCAA), bromochloroacetic acid (BCAA), monobromoacetic acid (MBAA), and dibromoacetic acid (DBAA).

The second part of the work was focused on the cathodic reduction of halogenated by-products generated during oxidation of ROC. Most previously reported reductive approaches have focused only on cleavage of halogen atoms from aromatic structures, leading to total or partially dehalogenated products which have been individually analysed by gas chromatography-mass spectrometry (GC-MS) (Ji and Peters, 2001; Chao *et al.*, 2008) or UV-vis spectroscopy techniques (Li *et al.*, 2007). Since a wide category of halogenated organic compounds in ROC can be generated from chlorination and/or chloramination during electrochemical oxidation, analysis of halogen-specific adsorbable organic halogen (AOX) was employed as a more appropriate measure for evaluating removal of all halogenated by-products during the electro-reduction of oxidised ROC. Separate analysis of adsorbable organic chlorine (AOCl), adsorbable organic bromine (AOBr) and adsorbable organic iodine (AOI) was performed in the AOX method. These analytes (AOCl, AOBr and AOI), represent the amount of each halogen present as organohalogen compounds in the fraction of organic matter within the sample which is adsorbable on an activated carbon. This is the first study utilising the halogen-specific AOX technique to study the removal of halogenated by-products during electro-reduction of oxidised ROC. In addition, THMs and HAAs were also analysed, by-products which represent small fractions of AOX but are currently regulated. The toxicity of ROC at various points was determined using *Vibrio fischeri* bioassays (Microtox) (Escher *et al.*, 2008).

Experimental Methods

The ROC used in the experiments was obtained from three-stage RO units at an advanced water treatment plant (AWTP) in Bundamba (Queensland, Australia). After collection from four wastewater treatment plants, this AWTP is processing a mixture of secondary effluents via coagulation and solid precipitation prior to membrane filtrations, ie, microfiltration and reverse osmosis membranes. The characteristics of the ROC are given in Table 1.

Table 1. Characteristics of the ROC used in the experiments indicates high conductivity with low concentrations of organics.

Measures, unit	DOC, mg L^{-1}	DON, mg L^{-1}	COD, mg L^{-1}	pH	Conductivity, mS cm^{-1}	[Br^-], mg L^{-1}	[Cl^-], mg L^{-1}
Value	58 $\pm$ 3	6.5 $\pm$ 0.3	173 $\pm$ 8	8.1 $\pm$ 0.1	5.0 $\pm$ 0.1	1.5 $\pm$ 0.2	1229 $\pm$ 25

For the first part of the study investigating the performance of electrochemical oxidation of ROC on five anode materials, 10 L of each electrolyte, ROC as anolyte and 0.1 M HCl as catholyte, was continuously recirculated in batch mode through the anodic and cathodic half-cell, respectively, during 22 h with flow rate of 162 mL min^{-1}, to maintain well-mixed conditions and increase the contribution of formed short-lived oxidants. The anodes used were IrO_2-Ta_2O_5 (12 g m^{-2} of Ir:Ta = 65:35), RuO_2-IrO_2 (12 g m^{-2} of RuO_2:IrO_2 = 70:30), Pt-IrO_2 (12.5 g m^{-2} of Pt:IrO_2 = 70:30), PbO_2 (3.75 g m^{-2} PbO_2), and SnO_2-Sb (15 g m^{-2} Sn/Sb_2O_5 doped) electrodes coating on Ti mesh (dimensions: 4.8 × 5 cm; thickness: 1 mm), supplied by Magneto Special Anodes (The Netherlands). The cathode used was a stainless steel woven wire mesh (dimensions: 4.8 × 5 cm; 80 μm × 0.050 mm wire diameter). All electrodes had a projected electrode surface area of 24 cm^2. Performed in duplicates, 70 mL and 130 mL samples of ROC were taken at 0, 1, 3, 7, and 22 h during oxidation for the analysis of organics and oxidation by-products, respectively. The experiments for analysing oxidation by-products were done separately to minimise the impact of feed volume change caused by sampling.

The second set of experiments evaluating electro-reductive dehalogenation was conducted using 4 L of pre-oxidised ROC (as abovementioned procedure) as catholyte and 4 L of 0.1 M H_2SO_4 as anolyte medium, recirculated in batch mode during 72 h with flow rate of 162 mL min^{-1}. The cathode used was an impregnated carbon block (Morgan AM and T, Australia) with dimension of 25 × 9 × 0.6 cm (active surface area: 20 × 5 cm^2), while the anode material used was Ti/Pt-IrO_2. 130 mL samples were taken at 0, 24, 48, and 72 h for THMs and HAAs analyses, while 100 mL and 50 mL samples were taken for *V. fischeri* bioassays and AOX, respectively at 0 and 72 h during reduction process.

Both electrochemical oxidation and reduction experiments were performed using divided electrochemical cells to understand the role of each process with galvanostatically controlled conditions. A constant current at 240 mA (current density, J = 100 and 24 A m^{-2} in the electrochemical oxidation and reduction experiments, respectively) was supplied through a potentiostat/galvanostat (VMP3, Princeton Applied Research, USA). This applied current density J = 100 A m^{-2} was chosen based on our previous study (Radjenovic *et al.*, 2011) as an optimum value. The electrochemical cell consists of two equal rectangular polycarbonate frames (inner dimensions of 20 × 5 × 1.2 cm, effective volume was 114 mL) bolted together by two polycarbonate rectangular plates. The anode cell was separated from the cathode cell by a cation exchange membrane (CEM) Ultrex CMI-7000 (Membranes International, U.S.A.). In all experiments, the half cell potentials were measured against an Ag/AgCl reference electrode (RE-5B, Bio Analytical, USA), assumed at +0.205 V versus the Standard Hydrogen Electrode (SHE). In addition, 'control' experiments for the electro-reductive process were conducted by recirculating oxidised ROC in an open circuit.

Routine analyses of DOC, COD, and DON, as well as free available chlorine (FAC) and total chlorine (sum of FAC and combined chlorine) measured with the *N*,*N*-diethyl-*p*-phenylenediamine (DPD) ferrous titrimetric method 409E, were conducted as described previously (Bagastyo *et al.*, 2011; Radjenovic *et al.*, 2011). Oxidants other than FAC may react similarly with DPD, and hence may interfere with the FAC measurements. THMs and HAAs were analysed in Queensland Health Forensic and Scientific Service Laboratory using GC-MS (purge and trap) for THMs and GC-electron capture detection (ECD) confirmed by GC-MS for HAAs. The Microtox (*V. fischeri* bioassays) was performed as described previously (Escher *et al.*, 2008) with sample pre-adjustment to pH 7 by adding an appropriate amount of 0.1M NaOH or 0.1M HNO_3. The results of the Microtox test were converted to baseline toxicity equivalent concentration (baseline-TEQ) relative to a virtual reference compound with an EC_{50} of 12.2 mg L^{-1}. Halogen-specific AOX was analysed in a modified version of our previously published method (Kristiana *et al.*, 2009). The method involved acidification of the sample to pH 2, followed by adsorption onto activated carbon, which was then combusted, and the hydrogen halide gases produced were trapped in MilliQ water in an absorber chamber, and the dissolved halide ions were then analysed by ion chromatography with conductivity detection. Where the concentration of AOX is given (mg Cl^- L^{-1}), it refers to the sum of the molar concentration of AOCl, AOBr, and AOI, multiplied by the atomic mass of Cl.

Results

Electrochemical Oxidation of ROC on Selected Materials

Removal of DOC, DON, and COD

The observed removals of DOC, DON, and COD, as well as the measured free and total chlorine during electrochemical oxidation of ROC on each anode are illustrated in Figure 1. Limited mineralisation of organic compounds was observed in all experiments, with removal of the overall DOC being <16% (Figure 1a), suggesting partial oxidation had occurred. After the supplied charge reached 0.55 Ah L^{-1} (Q = 0.55 Ah L^{-1}), both Ti/Pt-IrO_2 and Ti/SnO_2-Sb achieved 16 ± 1% DOC removal, while the other three electrodes removed only 9 ± 2% of DOC. In this study, mineralisation via direct oxidation was unlikely to have occurred due to the high current densities applied, low specific anode area (ie, anode surface area per active volume, 0.21 cm^{-1}) and low ratio of active volume (V_{ACT}) per total volume (V_{TOT}), ie, V_{ACT}/ V_{TOT} = 0.011. When using "non active" anodes, removal of DOC can be achieved via direct and indirect oxidation (Szpyrkowicz *et al.*, 2005: Van Hege *et al.*, 2004). Nevertheless, indirect oxidation mediated by FAC is a dominant mechanism particularly in a high chloride-containing waste stream.

As found for DOC removal, Ti/Pt-IrO_2 and Ti/SnO_2-Sb were also superior in removing DON, with 57 ± 16% and 58 ± 3% removal, respectively, measured at Q = 0.55 Ah L^{-1} (Figure 1b). Only ~10% of DON removal was obtained by the other electrode materials (Ti/IrO_2-Ta_2O_5, Ti/RuO_2-IrO_2 and Ti/PbO_2). Figure 1c shows that effective oxidation based on COD removal was obtained by Ti/Pt-IrO_2 and Ti/SnO_2-Sb (30-40%), while Ti/PbO_2, Ti/RuO_2-IrO_2, and Ti/IrO_2-Ta_2O_5 were considerably less effective (10-20%). The instantaneous current efficiency (ICE) calculated in relation to COD removal confirms that after Q = 0.17 Ah L^{-1}, Ti/SnO_2-Sb, Ti/PbO_2, and Ti/Pt-IrO_2 yielded values ranging from 50 to 60%, while Ti/IrO_2-Ta_2O_5 and Ti/RuO_2-IrO_2 yielded 28 ± 5%.

It can be assumed that the presence of Cl^- ions affected the performance of each anode as reported previously (Scialdone *et al.*, 2009), enhancing the removal of COD at the Ti/Pt-IrO_2 electrode. The higher oxidising power of the Ti/Pt-IrO_2 anode and increased generation of FAC than in the case of the other two DSA electrodes (ie, Ti/IrO_2-Ta_2O_5 and Ti/RuO_2-IrO_2) is particularly interesting considering that the anode potential (E_{AN}) of Ti/Pt-IrO_2 electrode was 1.55±0.14 V, lower than in the case of Ti/RuO_2-IrO_2 electrode (ie, E_{AN}=1.74±0.20 V). Although for all three anodes participation of active chloro-species is undoubted, additional oxidation mechanisms and/or surface reactions of organic matter in the case of Ti/Pt-IrO_2 were likely responsible for the enhanced oxidising power of this electrode. The "non-active" anodes Ti/SnO_2-Sb and Ti/PbO_2 have lower selectivity toward chlorine evolution with respect to water oxidation, which leads to a decreased contribution of chloro-species to the mediated electrolysis of ROC. Enhanced oxidation performance of the Ti/Pt-IrO_2 electrode in our study may be due to the increased generation of FAC as determined by the DPD method (Figure 1d), and enhanced chlorine-mediated electrolysis.

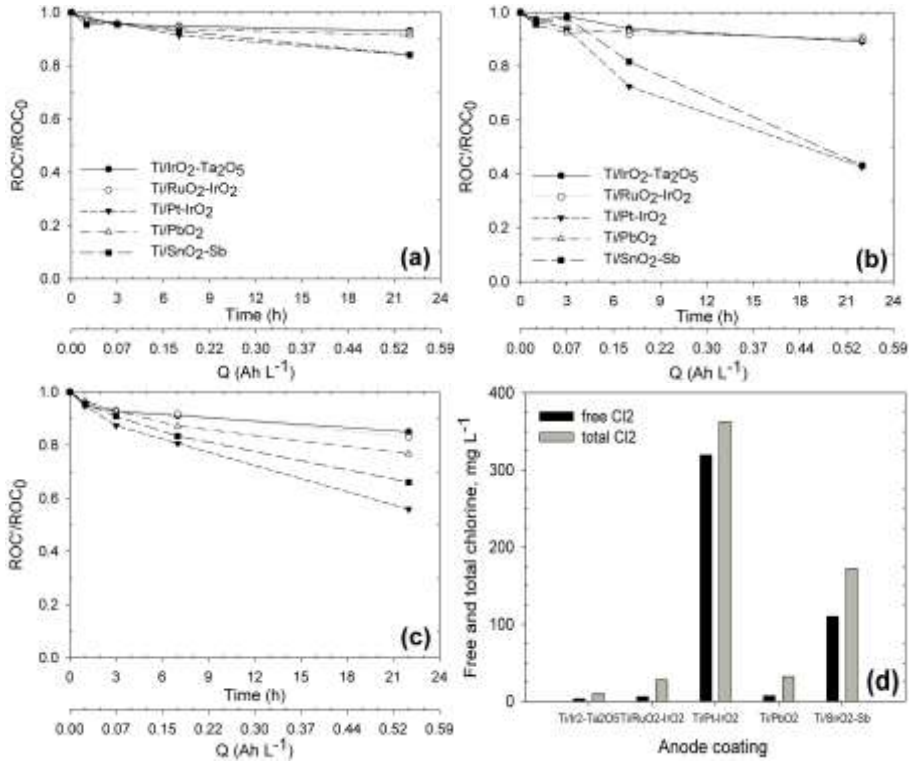


Figure 1. Normalised removal of: (a) DOC, (b) DON, (c) COD on each anode vs supplied charge Q (Ah L^{-1}), and (d) generation of free and total chlorine measured at Q = 0.55 Ah L^{-1}.

After Q = 0.55 Ah L^{-1}, the largest concentration of FAC was measured for Ti/Pt-IrO_2 (319±22 mg L^{-1}), followed by Ti/SnO_2-Sb (110 ± 7 mg L^{-1}). The total chlorine measured by the DPD method for Ti/Pt-IrO_2 and Ti/SnO_2-Sb anodes was 365 ± 30 mg L^{-1} and 170 ± 6 mg L^{-1}, respectively. The amount of total chlorine formed was already substantial at low Q, eg, after Q = 0.05 Ah L^{-1}, the total chlorine concentration measured for the Ti/Pt-IrO_2 and Ti/SnO_2-Sb anodes was 32 ± 9 mg L^{-1} and 25 ± 7 mg L^{-1}, respectively. Considerably lower concentrations (< 25 mg L^{-1} of FAC) were found for Ti/PbO_2, Ti/RuO_2-IrO_2, and Ti/IrO_2-Ta_2O_5. The Ti/Pt-IrO_2 anodes have higher electrocatalytic activity toward chlorine evolution (Szpyrkowicz *et al.*, 2005), while the lower FAC generation on Ti/IrO_2-Ta_2O_5 and Ti/RuO_2-IrO_2 can be explained by the competition with the oxygen evolution in acid media (Morimitsu *et al.*, 2000). The measured FAC for "non-active" anodes, ie, Ti/SnO_2-Sb and Ti/PbO_2, was likely affected by the other long-lived oxidants (eg, O_3, H_2O_2) accumulated in the bulk solution, apart from active chlorine (Panizza, 2010). Lower FAC formation on the Ti/PbO_2 anode compared to the Ti/SnO_2-Sb anode in our study may suggest that the former anode material exhibited a poorer electrocatalytic activity for the generation of oxidants in the chloride-containing waste stream (Szpyrkowicz *et al.*, 2005).

Generation of THMs and HAAs

Figure 2 below illustrates the total THMs (tTHMs) and total HAAs (tHAAs) generated during the electrochemical oxidation of ROC on each anode due to the oxidised Cl^- and Br^- forming HOCl and HOBr, respectively, which then react with organic compounds in ROC. The graphs of tTHMs and tHAAs represent the sum of four and six measured individual species of THMs and HAAs, respectively. The amount of tTHMs formed during electrochemical oxidation of ROC decreased in the order Ti/Pt-IrO_2 > Ti/RuO_2-IrO_2 ≥ Ti/SnO_2-Sb ≈ Ti/PbO_2 > Ti/IrO_2-Ta_2O_5 (Figure 2a). In the untreated ROC, THMs were already observed at concentration of 38 ± 12 µg L^{-1}, and were further increased with the supplied Q in all anodes over time, reaching 82-312 µg L^{-1} of tTHMs at Q = 0.55 Ah L^{-1}. The highest increase of total THMs per supplied Q was measured in Ti/Pt-IrO_2, ie, 10 mg A^{-1} h^{-1} at Q = 0.55 Ah L^{-1}. The higher applied current density would increase the concentration of tTHMs, eg, as reported by Perez *et al.* (2010). At J = 200 and 100 A m^{-2}, tTHMs achieved was near 200 and 80 µg L^{-1}, respectively, in the case of electrochemical oxidation of ROC using BDD anode.

As illustrated in Figure 2b, already at Q = 0.17 Ah L^{-1}, substantial amounts of tHAAs were formed by Ti/SnO_2-Sb and Ti/Pt-IrO_2 anodes (0.67 and 0.56 mg L^{-1}, respectively), followed by the Ti/RuO_2-IrO_2 and Ti/PbO_2 (0.41 and 0.38 mg L^{-1}, respectively), with the lowest tHAAs was noted for the Ti/IrO_2-Ta_2O_5 anode (0.23 mg L^{-1}). As observed at higher supplied Q, the concentration of tHAAs decreased in the order: Ti/SnO_2-Sb > Ti/Pt-IrO_2 > Ti/RuO_2-IrO_2 ≈ Ti/PbO_2 > Ti/IrO_2-Ta_2O_5. The formation of HAAs increased with electrolysis time, and was between 0.6 and 2.7 mg L^{-1} at the end of experiments. In the case of Ti/Pt-IrO_2 and Ti/SnO_2-Sb electrodes, higher generation of TCAA than

DCAA and MCAA was found, indicating that at these anodes, formed dichlorinated HAAs undergo consecutive attack by free chlorine and/or chloro-radicals to form trichlorinated derivatives (data not shown). This is likely due to the high FAC generated not only at the surface of the Ti/Pt-IrO_2 and Ti/SnO_2-Sb anodes, but also in the bulk by $OH^{\bullet}$ and other oxidants.

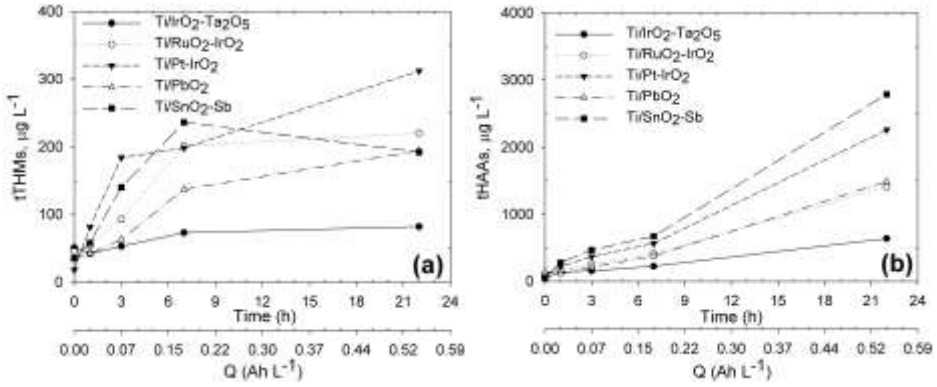


Figure 2. Concentration vs. time profiles of: (a) total THMs (tTHMs) and (b) total HAAs (tHAAs) generated on each anode during electrochemical oxidation of ROC at J = 100 A m^{-2}.

Electrochemical Reduction of Pre-Oxidised ROC

Changes in Concentrations of the THMs and HAAs

In the second part of the study, electrochemical reduction at a resin-impregnated carbon cathode was used as an approach to dehalogenate the chlorinated and brominated by-products formed during electro-oxidation. As shown in Figure 3a, the results confirmed that the tTHMs were reduced from 715 µg L^{-1} to 59 µg L^{-1}, while the tHAAs were reduced from 2092 µg L^{-1} to 1501 µg L^{-1} (Figure 3b), after Q = 4.8 Ah L^{-1}. Already at Q = 1.4 Ah L^{-1}, all species of THMs were decreased substantially, including the large fraction of chloroform ($CHCl_3$).

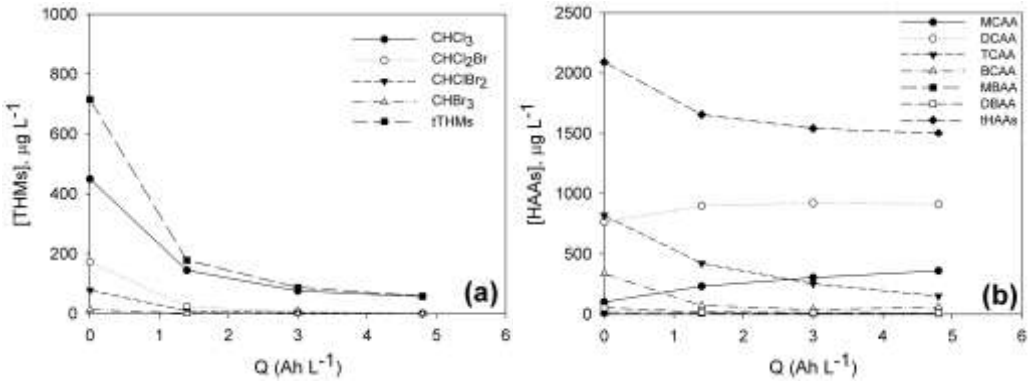


Figure 3. Concentration vs. specific electrical charge (Q, Ah L^{-1}) profiles of: (a) THMs, and (b) HAAs observed during electrochemical reduction of oxidised ROC vs using a carbon cathode.

The degradation of polyhalogenated HAAs was clearly observed. The reduction of brominated compounds is much easier than chlorinated compounds since the C-Br bond is weaker than the C-Cl bond, hence brominated HAAs can be completely dehalogenated in electrochemical reduction (Li *et al.*, 2007). Conversely, the reduction of chlorinated HAAs was always accompanied by the formation of MCAA, which was not further degraded and thus accumulated. Reduction of tri- and di-chlorinated HAAs to MCAA, which does not undergo further cleavage of the C-Cl bond and hence accumulates, has been previously observed (Korshin and Jensen, 2001; Li *et al.*, 2007). By controlling the current at 240 mA, the observed cathode potential (E_{CA}) ranged from -2.1 to -1.9V vs SHE, where hydrogen evolution reaction takes place, decreasing the current efficiency. On the other hand, for many polychlorinated contaminants very negative reduction potentials have been observed, eg, the reduction potential of $CHCl_3$ at a carbon cathode was determined to be -2.1 V vs SCE in an organic solvent (Durante *et al.*, 2009). In the case of HAAs, brominated HAAs could be readily degraded at -0.2 V vs SCE, while more negative reduction potential, ie, at -0.4 V vs SCE was reported for the chlorinated HAAs using carbon nanotube electrodes (Li *et al.*, 2007). Further investigation needs to be performed on the optimisation of the process to allow more successful application of potentiostatically controlled electrolysis.

Reduction in AOX Concentration and Toxicity

As illustrated in Figure 4, the concentration of AOX and TEQ increased during oxidation of ROC from 1.08 to 7.89 mg Cl^- L^{-1} and from 3.21 to 165 mg L^{-1}, respectively. However, a substantial decrease of these values was observed during the cathodic reduction when applying a cathodic current of 240 mA, resulting in 4.53 and 11.6 mg L^{-1} of AOX and TEQ, respectively, after 72 h of reduction.

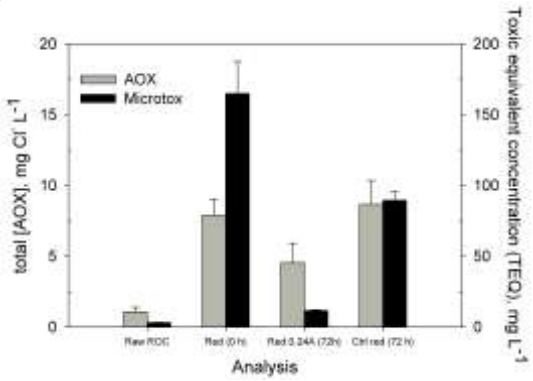


Figure 4. Concentration of AOX and baseline toxicity measured on untreated ROC, electrochemically oxidised ROC as initial electrochemical reduction, 72 h after electrochemical reduction (240mA), and 72 h control without supplied current.

TEQ represents the non-specific toxicity of complex mixtures of compounds, where the contribution of each chemical is weighted by its hydrophobicity. Since the samples were extracted by the previously optimised solid phase extraction (SPE) procedure (Radjenovic *et al.*, 2011), generated oxidants, salts, and particulates were removed by the SPE sample pre-treatment. The chlorinated organic compounds will have increased hydrophobicity and at the same time are expected to be more toxic, and thus they were the likely the cause of elevated TEQ during oxidation. These hydrophobic fractions which contribute to the toxicity are well retained in the cartridge, while the more hydrophilic part of molecules remained in the solution. Therefore, the decrease of toxicity of the electrochemically oxidised ROC after the open circuit experiment demonstrated a likely partial removal of the toxic organic compounds by sorption onto the cathode surface which were not measured in AOX since the concentration of AOX was relatively constant, although the comparative decrease was greatly enhanced in the case of a charged electrode (Figure 4). In the closed circuit experiments, AOCl and AOBr decreased 44 and 52%, respectively after 72 h reduction, while the DOC concentration remained constant (data not shown). Furthermore, after 72 h reduction experiments, the release of Br ions was observed (ie, 1.2 mg L^{-1}), however, the concentration of Cl ions released after the reduction was only 2 mg L^{-1} lower than we expected since these ions may diffuse even through a CEM to the anode and possibly get oxidised. Thus, the results of overall reduction are currently inconclusive. AOCl, AOBr and AOI were possibly formed in the open circuit experiments (data not shown). Any formation would have been due to the presence of residual oxidants (eg, FAC), but these oxidants would have been quickly quenched at the beginning of the reduction process. Further experiments on the optimisation of this electrochemical reduction process, ie, longer electrolysis times and controlled-potential electrolysis, are under way.

Conclusions

Electrochemical treatment is chemical-free, and operates at ambient temperature and pressure. For electrochemical oxidation, the best overall performance was observed for the Ti/Pt-IrO_2 and Ti/SnO_2-Sb anodes. Although the presence of Cl ions in ROC promotes indirect oxidation mechanisms, they may lead to the formation of hazardous halogenated by-products. Combined electrochemical oxidation and reduction, as applied here, could enable a complete degradation of organic compounds, including their halogenated by-products. Electrochemical reduction using a carbon block cathode suggested the removal of toxic by-products through dehalogenation and/or adsorption mechanisms. At present, the reduction stage does not bring sufficient removal of halogenated by-products to warrant application. Future work will couple the electrochemical treatment in one cell, which may improve the electrical yield and lower the operating cost of the treatment. Furthermore, improved reduction is necessary, requiring better cathode catalysis and fluid/electrode contact.

Acknowledgements

This research was supported by the Australian Research Council (grant LP0989159), Veolia Water Australia, Water Secure, Queensland Health Forensic and Scientific Service, Magneto Special Anodes and the Urban Water Security Research Alliance. The authors would like to thank Prof. Beate Escher and Dr Eva Glenn of National Research Centre for Environmental Toxicology, The University of Queensland, for performing the bioassays.

References

Anglada, A., Urtiaga, A. and Ortiz, I. (2009) Contributions of electrochemical oxidation to waste-water treatment: fundamentals and review of applications. Journal of Chemical Technology and Biotechnology 84(12), 1747-1755.

Bagastyo, A.Y., Keller, J., Poussade, Y. and Batstone, D.J. (2011) Characterisation and removal of recalcitrants in reverse osmosis concentrates from water reclamation plants. Water research 45, 2415-2427.

Comninellis, C. (1994) Electrocatalysis in the electrochemical conversion/combustion of organic pollutants for waste water treatment. Electrochimica Acta 39, 1857-1862.

Deng, Y. and Englehardt, J.D. (2007) Electrochemical oxidation for landfill leachate treatment. Waste Management 27, 380–388.

Dialynas, E., Mantzavinos, D. and Diamadopoulos, E. (2008) Advanced treatment of the reverse osmosis concentrate produced during reclamation of municipal wastewater. Water research 42, 4603-4608.

Durante, C., Isse, A.A., Sandona, G. and Gennaro, A. (2009) Electrochemical hydrodehalogenation of polychloromethanes at silver and carbon electrodes. Applied Catalysis B: Environmental 88, 479–489.

Escher, B.I., Bramaz, N., Mueller, J.F., Quayle, P., Rutishauser, S. and Vermeirssen, E.L.M. (2008) Toxic equivalent concentrations (TEQs) for baseline toxicity and specific modes of action as a tool to improve interpretation of ecotoxicity testing of environmental samples. Journal Environmental Monitoring 10, 612-621.

Ji, C. and Peters, D.G. (2001) Electrochemical reduction of benzyl iodide, benzal bromide, and benzal chlorobromide at carbon cathodes in the presence of nitri oxide in acetonitrile. Journal of Electroanalytical Chemistry 516, 39-49.

Korshin, G.V. and Jensen, M.D. (2001) Electrochemical reduction of haloacetic acids and exploration on their removal by electrochemical treatment. Electrochimica Acta 47, 747–751.

Kristiana, I., Gallard, H., Joll, C. and Croue, J.-P. (2009) The formation of halogen-specific TOX from chlorination and chloramination of natural organic matter isolates. Water research 43, 4177-4186.

Li, Y.-P., Cao, H.-B. and Zhanga, Y. (2007) Reductive dehalogenation of haloacetic acids by hemoglobin-loaded carbon nanotube electrode. Water research 41, 197-205.

McRae, B.M., LaPara, T.M. and Hozalski, R.M. (2004) Biodegradation of haloacetic acids by bacterial enrichment cultures. Chemosphere 55(6), 915–925.

Panizza, M. (2010) Electrochemistry for the Environment. Comninellis, C. and Chen, G. (eds), Springer Science+Business Media.

Paul, D., Pandey, G., Pandey, J. and Jain, R.K. (2005) Accessing microbial diversity for bioremediation and environmental restoration. Trends in BIotechnology 23(3), 135-142.

Perez, G., Fernandez-Alba, A.R., Urtiaga, A.M. and Ortiz, I. (2010) Electro-oxidation of reverse osmosis concentrates generated in tertiary water treatment. Water research 44, 2763-2772.

Radjenovic, J., Bagastyo, A., Rozendal, R.A., Mu, Y., Keller, J. and Rabaey, K. (2011) Electrochemical oxidation of trace organic contaminants in reverse osmosis concentrate using RuO_2/IrO_2 coated titanium anodes. Water research 45(4), 1579-1586.

Richardson, S.D., Simmons, J.E. and Rice, G. (2002) Disinfection by-products: the next generation. Environmental science and technology 36, 198A-205A.

Rondinini, S. and Vertova, A. (2010) Electrochemistry for The Environment. Comninellis, C. and Chen, G. (eds), Springer Science+Business Media.

Scialdone, O., Randazzo, S., Galia, A. and Silvestri, G. (2009) Electrochemical oxidation of organics in water: role of operative parameters in the absence and in the presence of NaCl. Water research 43, 2260-2272.

Szpyrkowicz, L., Radaelli, M. and Daniele, S. (2005) Electrocatalysis of chlorine evolution on different materials and its influence on the performance of an electrochemical reactor for indirect oxidation of pollutants. Catalysis Today 100, 425-429.

Van Hege, K., Verhaege, M. and Verstraete, W. (2004) Electro-oxidative abatement of low-salinity reverse osmosis membrane concentrates. Water research 38, 1550-1558.

Westerhoff, P., Moon, H., Minakata, D. and Crittenden, J. (2009) Oxidation of organics in retentates from reverse osmosis wastewater reuse facilities. Water research 43(16), 3992-3998.

Zou, L., Morris, G. and Qi, D. (2008) Using activated carbon electrode in electrosorptive deionisation of brackish water. Desalination 225, 329-340.

Poster - Abstracts

Unpacking Residential Water End Use Diurnal Patterns to Understand Capital Efficiency Opportunities

Carragher, B.J.[1], Stewart, R.A.[2], Huang, T-T.[1] and Beal, C.D.[1]
[1] Smart Water Research Centre, Griffith University, Gold Coast, Queensland
[2] Centre for Infrastructure Engineering and Management, Griffith University, Gold Coast, Queensland

The purpose of the project is to assess the efficiency of household water fixtures/appliances (eg, taps, shower heads, clothes washers) and the impact of this on average day peak demand diurnal patterns. Investigations aimed to observe residential water end use consumption and understand how more efficient stock (appliances/fixtures) could reduce peak demand and subsequently impact supply infrastructure.

To help accomplish these outcomes, the project has produced a weighted stock efficiency scheme which categorises residential households depending on their level of stock efficiency and allows homes to be clustered for efficiency comparisons. Preliminary results suggest that there is a clear reduction in average daily water consumption from homes with more efficient water fixture/appliances. Not only does this provide evidence for implementation of wide scale efficient appliances for bulk water savings but also shows that these efficient appliances create a reduction in demand at peak times of use. This is important, as reductions in peak demand would potentially offset supply network upgrade costs to local and state authorities made necessary by increasing population growth and other service requirements.

Development of a Domestic Water End Use Consumption Forecasting Model for South East Queensland

Makki, A.[1], Stewart, R.A.[2], Panuwatwanich, K.[2] and Beal, C.D.[1]
[1] Smart Water Research Centre, Griffith University, Gold Coast, Queensland
[2] Centre for Infrastructure Engineering and Management, Griffith University, Gold Coast, Queensland

The purpose of this study is to explore the predominant determinants of various residential end use consumptions and the overarching approach to building a residential demand forecasting model using aligned socio-demographic and natural science data sets collected from 200 households fitted with smart meters in South East Queensland. ANOVA, as well as multiple regression analysis statistical techniques, were used to reveal the determinants (eg, household makeup, stock efficiency, income, etc.) of household end use consumptions (eg, shower, tap and clothes washer). The study provides evidence that both household makeup and appliance efficiency are the most significant determinants of usage among other factors. For example, the generated multiple regression model shows that these significant determinants explained 90.2% of the variation in household shower consumption. The model also shows that the number of teenagers in the household is the most important household makeup characteristic in terms of shower consumption. The study discusses the significant end use determinants and how this statistical approach will be followed to predict overall household residential consumption. The implications of the research on conservation strategies and policy design are discussed, along with future research directions.

Influence of Organic Carbon in Bioassays: Role of Assimilable Organic Carbon

Neale, P. A. and Escher, B.I.
National Research Centre for Environmental Toxicology (Entox), The University of Queensland, Brisbane, Queensland

Bioassays are increasingly used in organic matter rich samples, such as wastewater, to assess toxicity. However, the influence of this complex matrix remains relatively unexplored. While larger fractions of dissolved organic carbon (DOC) can reduce the bioavailability of organic micropollutants, low molecular weight DOC, including assimilable organic carbon (AOC), may actually negatively impair bioassays as they are likely to cause a similar non-specific toxic effect as organic micropollutants. In this study, the influence of wastewater derived DOC on bioassays was studied by measuring AOC content in DOC fractions and relating this parameter to bioassay results (Goal 1B of the 'Bioassays and Risk Communication' project). AOC is typically measured using pure bacterial cultures, but in this study AOC was measured using natural microbial communities to provide a more accurate measurement of AOC. Wastewater derived DOC samples were fractionated into different molecular weight fractions using size exclusion chromatography, with AOC measured in each fraction. In addition, baseline toxicity of each fraction was assessed using a non-specific bioluminescence inhibition test (Microtox) and DOC was also characterised using liquid chromatography-organic carbon detection (LC-OCD). Preliminary results indicate increased baseline toxicity for the low molecular weight fractions, which are expected to be rich in AOC.

Disinfection By-Products (DBPs) in South East Queensland: Characterisation of Natural Organic Matter and Adsorptive Removal Strategies for Halide and Organic DBP Precursors

Watson, K.[1], Knight, N.[1], Farré, M.J.[2], King, H.[2], Shaw, G.[1], Leusch, F.D.L.[1], Sadler, R.[1], Birt, J.[3], Bartkow, M.[4], Keller, J.[2], Gernjak, W.[2] and Burrell, P.[5]

[1] Smart Water Research Centre (SWRC), Griffith University, Gold Coast, Queensland
[2]Advanced Water Management Centre (AWMC), The University of Queensland, St Lucia, Queensland
[3] Bond University, Queensland
[4] Seqwater, Brisbane, Queensland
[5] SEQ Water Grid Manager, Brisbane, Queensland

Understanding the formation and minimisation of disinfection by-products (DBPs) in drinking water is important for the continued protection of public health, as although water disinfection is essential for preserving the safety of drinking water supplies, DBPs may be associated with detrimental health effects after long-term exposure. Although the risk associated with DBP exposure is small compared to the risk associated with drinking water that has not been disinfected, DBP minimisation is still an important priority for water authorities. In addition to regulated DBPs, analysis of iodinated and brominated emerging DBPs is included in this research project, since the prevalence of these is not well understood in the South East Queensland (SEQ) context and they are potentially more problematic than their chlorinated analogues. This project includes characterisation of the natural organic matter (NOM) of several SEQ source waters, and identification of fractions relevant to DBP formation. Additionally, this project investigates targeted adsorptive strategies for the removal of NOM and halide (bromide and iodide) DBP precursors from source waters prior to disinfection and examines the effect of these removal strategies on DBP formation. These pre-treatment strategies could enable lower concentrations of disinfectants to be used, as well as lowering the formation of many DBP species, thus ensuring the continued protection of the public health of SEQ residents.

This is a component of the "DBP Formation in Alternate Source Waters" project, lead by Dr Maria Farré (AWMC) and Dr Nicole Knight (SWRC).

Comparison of Full Scale Ozone/Biological Activated Carbon Treatment of WWTP Effluents: Removal of Micropollutants and Reduction of Toxicity

Reungoat, J.[1], Escher, B.I.[2], Macova, M.[2], Rattier, M.[1], Gernjak, W.[1] and Keller, J.[1]

[1] Advanced Water Management Center (AWMC), The University of Queensland, St Lucia, Queensland
[2] National Research Centre for Environmental Toxicology (Entox), The University of Queensland, Brisbane, Queensland

The poster presents an update on the work that was presented last year at the 2010 Science Forum. It includes additional data from the three full scale ozone/biological activated carbon (BAC) plants that have been sampled for micropollutant quantification and bioassay analysis. The full set of data now allows the comparison of the performance of the three plants. The results confirm that the combination of ozonation with biological activated carbon can achieve high removal of organic micropollutants (>90%) and reduction of non-specific toxicity (50-70%) and of estrogenicity (>95%). Moreover, the results clearly show that the ozone dose and the filtration time in the BAC influence the removal of micropollutants; both increase the degree of removal when increased. Finally, it is important to note that in the plant using the lower ozone dose and filtration time, some micropollutants were removed less than 90%. It is therefore important to design plants with a sufficient ozone dose and filtration time to ensure high removal of a wide range of compounds.

Tracking the Source of Pathogens in Reservoirs

Stratton, H.[1], Christie, M.[2], Toze, S.[2], Katouli, M.[3], Ahmed, W.[2] and Watkinson, A.[4]
[1] Smart Water Research Centre, Griffith University, Gold Coast, Queensland
[2] CSIRO Water for a Healthy Country, Ecosciences Precinct, Dutton Park, Queensland
[3] University of Sunshine Coast, Maroochydore, Queensland
[4] Seqwater, Brisbane, Queensland

Pathogenic microorganisms are the single greatest acute health risk within a community's water supply. Understanding the sources of these organisms and the risks they present is paramount to effectively managing the water supply. The standard measurement for faecal contamination involves determining the presence of *Escherichia coli*. However, the detection of *E. coli* does not provide any information about the source of contamination and therefore is not always an effective indicator of risk to humans. Microbial source tracking is an emerging field that has shown promise in being able to assess the source of faecal contamination in water sources. The β-glucuronidase gene found in *E. coli* is one gene that has shown promise as a possible genetic marker to distinguish sources of isolates. The aim of the current work was to collect a high number of *E. coli* isolates from two catchments in South East Queensland and compare their β-glucuronidase gene to determine whether they originated from human or animal faeces. A 500bp segment of the gene was sequenced in 382 isolates from both catchments. Early analysis and comparison of the sequenced 500bp gene segment suggest that the isolates from the two catchments originate from both animal and human sources.

Outcome of the Assessment of Factors Influencing the Decay of Pathogens in South East Queensland Reservoirs

Toze, S., Hodgers, L. and Palmer, A.
CSIRO Water for a Healthy Country, Ecosciences Precinct, Dutton Park, Queensland

Research has been undertaken to investigate decay rates for a variety of enteric pathogens in Wivenhoe Dam. This poster summarises the results obtained so far. The results for the first five experiments demonstrate that *Cryptosporidium* oocysts have the longest survival times of 13-17 days for a 1 log removal. The bacteria and MS2 had faster rates of decay with overall 1 log removal times of eight days or less, while adenovirus had decay rates slower than for the bacteria but faster than those observed for *Cryptosporidium*. The results from the non-sunlight experiments also showed that there was minimal variation in the decay of individual microorganisms regardless of the experiment and variable being tested. The exception was for *Campylobacter* at depths below 15m, where a 10 day increase in the 1 log decay time was observed (14 days at 15m opposed to four days at the surface). The outcomes of the sunlight experiment indicate that sunlight has an impact on pathogen decay at the reservoir surface but has no major impact at 5m depth.

Wind and Wave Testing of a New Evaporation Reduction - Monolayer System

Putland, S.[1], Schouten, P.W.[1], Lemckert, C.J.[1], Solomon, D.[2], Leung, A.[2], Tran, D.[2], Talemi, P.[2] and Qiao, G.[2]
[1] School of Engineering, Griffith University, Gold Coast, Queensland
[2] Department of Chemical and Biomolecular Engineering, University of Melbourne, Parkville, Victoria

Water loss generated by evaporation is an ever present threat to the world's increasingly scarce surface water storages. The use of chemical monolayers as evaporation suppressing agents has been tested extensively both recently and in the past with varying results. However, the main physical factor that inhibits the evaporation reducing ability of monolayers has been shown to be simultaneous wind and wave action that acts to break up the monolayer film, leading to a reduction in its surface coverage. New prototype solid-state and solution monolayers that have been developed by the Cooperative Research Centre for Polymers are formulated to better cope with the harsh and unpredictable environmental and hydrodynamic conditions that can be present within agricultural dams and reservoirs. Utilising the Griffith School of Engineering wind and wave tank, the project will ultimately determine the effectiveness of the new prototype chemical monolayers when subjected to varying winds and wave action. Their capacity to recover from wind and wave damage will also be tested. In addition, monolayer dosages will be varied across the water surface in order to determine an optimal deployment surface spreading ratio for application in real-world water reserves.

This poster will cover laboratory testing of prototype monolayers as part of the UWSRA Water Loss project.

Spatiotemporal Crop Mapping in the Lockyer Valley using Satellite Imagery

Ellis, T., Hartcher M. and Wolf, L.
CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland

The urban-rural water exchange project is evaluating the possible supply of ~25 GL/y of Purified Recycled Water (PRW) for irrigation in the Lockyer Valley, Queensland. Options could include delivery to: 1) farm gate; 2) surface water storages; 3) aquifer injection; 4) stream channels, which interact with groundwater. To help evaluate these, and the effects of PRW additions, regional groundwater models are being updated. A significant component of the water balance is groundwater abstraction. Although smaller, equally important is deep drainage from crops. Crop water balance modeling is one method for quantifying these. Crop rotations in the Lockyer Valley are complex in space and time, and present a staggering modelling task. Here we show early developmental stages of a method for simplifying and mapping crop rotations to inform water balance modelling. Cloud-free Landsat Thematic Mapper imagery was obtained; images were clipped to the Lockyer Valley alluvium polygon. On-ground crop identification was used as 'training' data for supervised, parallelepiped, classification of images from September 2010. The classification will be applied to historic Landsat images to map crop rotations in time. Updated and more accurate crop maps will help improve abstraction estimates in unmetered areas. Smaller, metered, areas will be used for evaluation.

Deep Drainage Estimates Under Irrigated Cropping in the Central Lockyer Valley - a Case Study at Forest Hill using Historic Farming Records and Soil Water Balance Modelling

Kodur, S.[1], Robinson, J.B.[1], Foley, J.L.[1], Wolf, L.[2] and Ellis, T.[2]
[1] Department of Environment and Resource Management, Toowoomba, Queensland
[2] CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland

The Lockyer Valley in South East Queensland is an important area for crop production, supplying substantial quantities of Queensland's irrigated vegetables. Changes to deep drainage are of concern as they may lead to increased ground water recharge and migration of existing salt loads within the regolith (flushing to groundwater). Thus, as a part of Lockyer Valley Purified Recycled Water Project, we modelled deep drainage rates using the Howleaky model, at Forest Hill in the Central Lockyer. At this site, soil hydraulic parameters were measured and used in the model along with 12 years of paddock management history from a local farmer. Crop sequences for two irrigation scenarios were modelled. The scenarios differed in application amounts (that were based on soil moisture characteristics). Annual average deep drainage ranged from 94 mm/year to 55 mm/year, depending on irrigation amount and strategy. In contrast, transpiration remained a constant amount (653 mm) for both scenarios. As the irrigation amount increased, contribution of irrigation to deep drainage increased (from 5% to 8% of the total water balance), while transpiration changed little as a proportion of the water balance (from 54% to 52%). The study highlights the importance of adopting suitable irrigation strategies to minimise deep drainage.

Salt Loads and Water Storage Changes in the Regolith in the Lockyer Valley 15 Years on - the Ellis Sites Revisited

Foley, J.L.[1], Wolf, L.[2], Harris, M.[1], Ellis, T.[2], Kodur, S.[1] and Manders, J.[3]
[1] Department of Environment and Resource Management, Toowoomba, Queensland
[2] CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland
[3] Department of Environment and Resource Management, Ecosciences Precinct, Dutton Park, Queensland

The Lockyer Valley is a highly productive region, providing Queensland with around 35% of its irrigated vegetables. Groundwater aquifers are used for irrigation in this region. Water use currently exceeds sustainable levels, resulting in declining groundwater levels and diminished creek flows. The potential to introduce additional water to the region in the form of purified recycled water (PRW) is being investigated as part of the Lockyer Valley Purified Recycled Water project, in an effort to create a more sustainable irrigation practice.

In this component of the project, we looked at potential impacts of introducing PRW on the regional water and salt balances. Specifically, we have assessed the potential salt mobilisation/flushing (of existing high loads in the regolith built up by irrigating with salty groundwater) and export from the catchment by mechanisms of increased runoff, deep drainage and rises in stream flow (at least until a new equilibrium is obtained). Soil water and salt/chloride profiles were measured at five soils/sites to assess current landscape salt loads. These sites had been cored and analysed for chemistry to depth (20 m) by Ellis in the late 1990s. Revisiting the sites enabled us to look at changes in soil salinity for the 15-year period, and also to model and estimate deep drainage. This was accompanied by geophysical surveying of the whole paddock to qualify the uniformity and patterns of water and salt stored in the regolith.

Posters - Presented

Unpacking Residential Water End Use Diurnal Patterns to Understand Capital Efficiency Opportunities

Carragher, B., Stewart, R. and Beal, C.

Smart Water Research Centre, Griffith University

Research Background

- Society requires continued water security and more sustainable use of current potable water resources.
- This study investigated the impact of efficient household water stock (toilets, showers, taps, etc) on average day demand patterns to assess the link between stock efficiency and peak demand.
- The study utilised consumption data from 191 homes of the SEQREUS winter 2010 baseline analysis (14 June – 28 June).

Results and Analysis

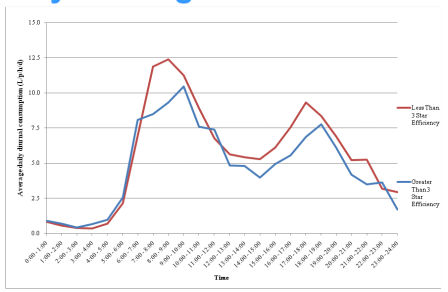
Table 1. Household Efficiency Cluster Information

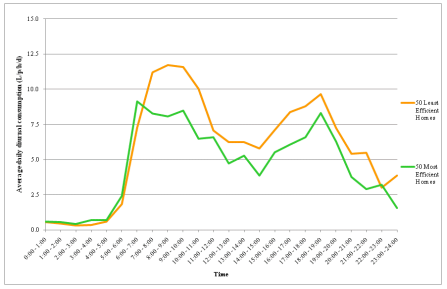
	All Homes	< 3 Star Cluster	≥ 3 Star Cluster	Least Efficient Cluster	Most Efficient Cluster
No. Homes	191	102	89	50	50
Average Rating	2.88 Stars	2.40 Stars	3.43 Stars	2.03 Stars	3.64 Stars

Table 2. Percentage Reduction From Water Efficient Stock

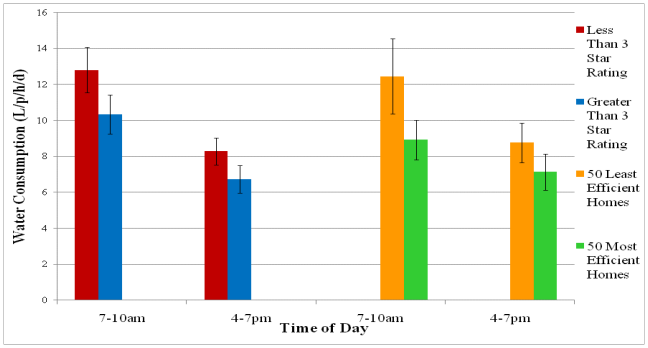
	Peak Morning Hour	Peak Afternoon Hour	Morning Peak Period (7am-10am)	Afternoon Peak Period (4pm-7pm)
≥ 3 Star Homes vs. < 3 Star Homes	15%	16%	19%	18%
50 Most Efficient Homes vs. 50 Least Efficient Homes	22%	14%	28%	18%

Key Findings





- Household water efficient technology significantly reduces residential consumption flow rates especially in peak periods of demand.
- Reductions demonstrate the potential for smaller infrastructure in greenfield developments or deferral of upgrades for existing systems.



Future Study Direction

- Utilise summer read data from SEQREUS to reiterate the impact of efficient stock on peak demand flow rates in higher demand months.
- Future modelling of networks should consider peak flow reductions as a result of water efficient stock as well as quantify cost of savings.

Queensland Government
National Research FLAGSHIPS Water for a Healthy Country CSIRO
Griffith UNIVERSITY
The University Of Queensland AUSTRALIA

Development of a Domestic Water End Use Consumption Forecasting Model for South East Queensland

Makki, A.A., Stewart, R.A., Panuwatwanich, K. and Beal, C.D.

Smart Water Research Centre, Griffith University

Project Summary

The purpose of this study is to explore the predominant determinants of various residential water end use consumptions and the overarching approach to building a residential demand forecasting model using aligned household makeup, stock efficiency, socio-demographic and natural science data sets collected from 200 households fitted with smart meters in SEQ.

Dummy Coding, ANOVA, as well as multiple regression analysis statistical techniques, were utilised to reveal the determinants (e.g. household makeup, stock efficiency, income, etc) of household end use consumptions (e.g. shower, tap and clothes washer). Figure 1 illustrates an example of the general shower end use forecasting modeling approach.

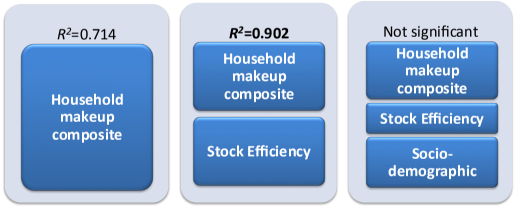


Figure 1 Block-wise entry multiple regression approach

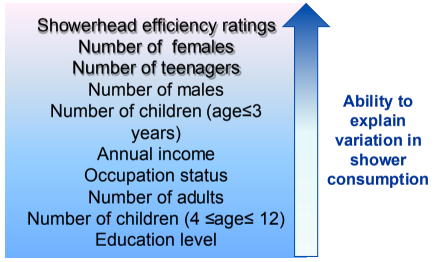


Figure 2 Determinants of daily per household shower end use consumption

Results to Date

A pilot study on shower end use revealed that showerhead efficiency ratings, numbers of females and teenagers in households are the most significant determinants of shower consumption. Other determinants are shown in Figure 2.

The preliminary shower end use forecasting model uses household size (represented by a household age makeup characteristics composite), and showerhead efficiency ratings, as the most significant predictors.

Average L/hh/d shower consumption = 99.1** – 10.4* (One Adult) + 76.2** (Three Adults) + 68.0** (One or More Teenagers) + 16.3** (One or More Children [4≤Age≤12 years]) + 42.0** (One or More Children [Age<3 years]) – 67.3** (AAA or 3 Stars) – 67.0** (AA or 2 Stars) – 44.1** (A or 1 Star) – 27.8** (Standard) ± 16.7

**p< 0.01, **p<0.001*

To generate a prediction, assign the membership of household(s) under prediction to its /their group characteristics by substituting a value of 1 in case of membership and a value of 0 if not, to calculate the equation.

Key Findings from Shower End Use Model

- *Showerhead efficiency ratings* is the most significant determinant of daily per household shower consumption.
- From a gender perspective, *Number of females* in households is the most significant determinant of daily per household shower consumption.
- From an age perspective, *Number of teenagers* in households is the most significant determinant of daily per household shower consumption when compared to other age categories.
- The developed shower end use forecasting model is explaining ***90.2%*** of the variation in households shower consumption with a standard error of 16.7 L/hh/d.

Future Direction

- Revealing significant determinants of other indoor water end uses (eg, toilet, tap, bathtub, clothes washing, dish washing, etc).
- Building a forecasting model for each indoor water end use.
- Development of a web-based forecasting tool that uses the summation of all end uses predictions to provide an evidence-based forecast of domestic household indoor demand.

Influence of Organic Carbon in Bioassays: Role of Assimilable Organic Carbon

Neale, P.[1], Hammes, F.[2] and Escher, B.[1]

[1]Entox, The University of Queensland; [2]Swiss Federal Institute of Aquatic Science and Technology (Eawag)

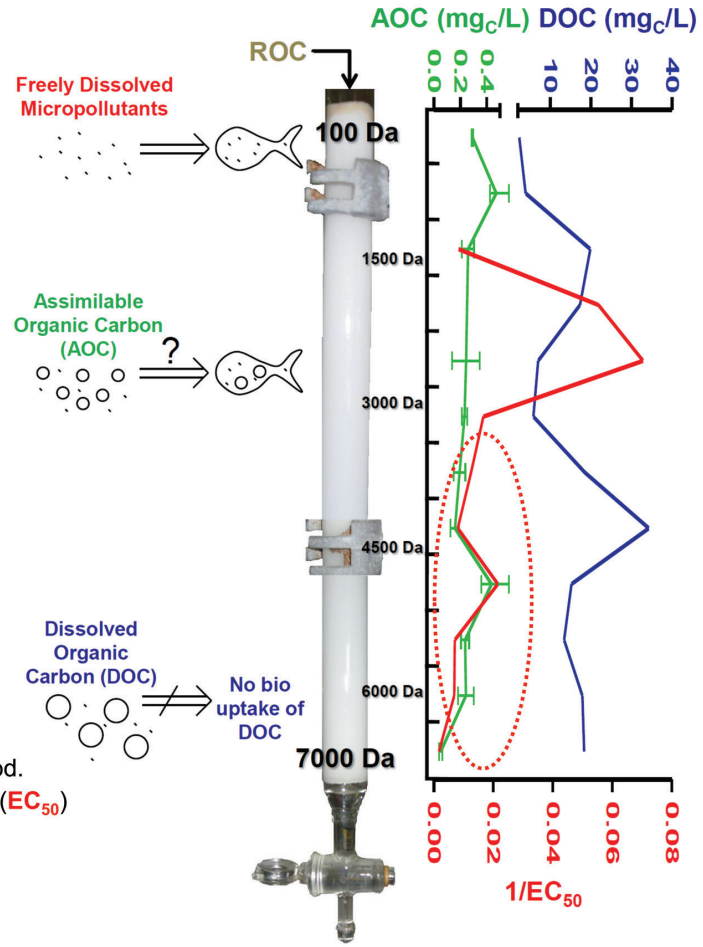
Project Summary

- Bioassays are used for samples rich in organic carbon, such as wastewater.
- This project aims to understand the influence of wastewater derived organic carbon on bioassays.
- Dissolved organic carbon **(DOC)**: < 0.45 µm, contains natural organic matter, soluble microbial products and breakdown products.
- Assimilable organic carbon **(AOC)**: Low molecular weight DOC, that can be readily utilised by microorganisms.

Hypothesis: AOC may negatively impair bioassays as it may cause similar non-specific bioactive effect as micropollutants.

Methodology

- Reverse osmosis concentrate (**ROC**) fractionated using size exclusion chromatography (100-7,000 Da).
- For each fraction the following parameters were measured:
  - **DOC**: Measured using total organic carbon analyser.
  - **AOC**: Measured using adenosine triphosphate (ATP) method.
  - **Baseline toxicity**: Effect concentration causing 50% effect (EC_{50}) using non-specific Microtox assay.
- Method for **AOC** assay establishment at Entox.



Preliminary Results

- Good correlations between **AOC** and EC_{50} from 3,000 - 7,000 Da.
- Suggests **AOC**, rather than total **DOC**, influences the studied non-specific assay.
- No effect (EC_{50}) for lowest molecular weight fractions despite high **AOC** values:
  - Low bioaccumulation?
- Peak toxicity ~2,700 Da.
- Further research required.

Future Direction

- Validate toxicity results using other non-specific assays:
  - Caco2 (human colon cancer cell line).
  - Algae.
- Expand to other types of wastewater derived **DOC**.

Queensland Government
National Research FLAGSHIPS Water for a Healthy Country CSIRO
Griffith UNIVERSITY
The University Of Queensland AUSTRALIA

Disinfection By-Products (DBPs) in Southeast Queensland: Natural Organic Matter (NOM) Characterisation and Adsorptive DBPs Precursor Removal Strategies

Watson, K.[1], Knight, N.[1], Farré, M.J.[2], King, H.[2], Shaw, G.[1], Leusch, F.D.L.[1], Sadler, R.[1], Birt, J.[3], Bartkow, M.[4], Keller, J.[2], Gernjak, W.[2] and Burrell, P.[5]

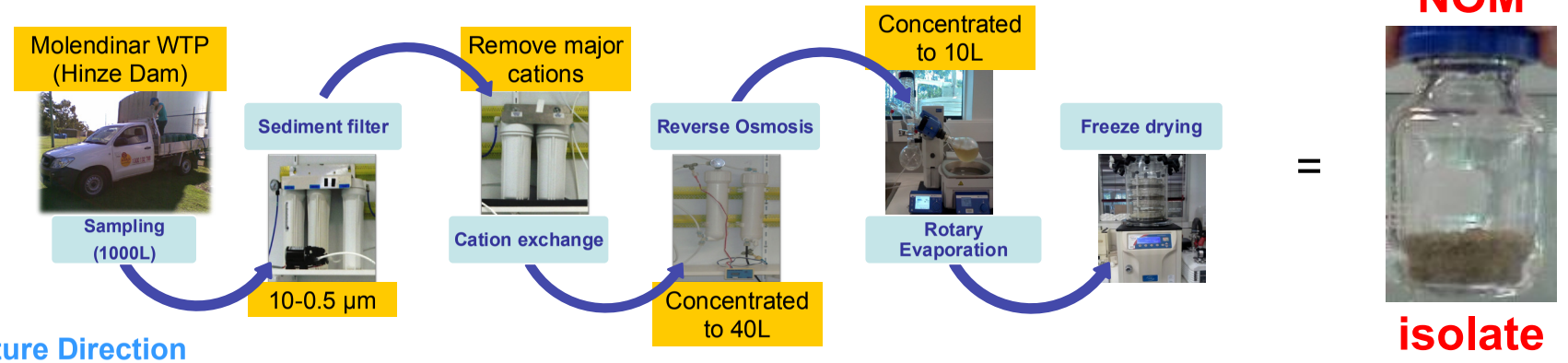
[1]Smart Water Research Centre, Griffith University, [2]Advanced Water Management Centre, The University of Queensland, [3]Bond University, [4]Seqwater, [5]SEQ Water Grid Manager

Project Summary

This project will develop an understanding of the occurrence of regulated and emerging DBPs in SEQ drinking water, with an emphasis on iodinated and brominated DBPs. This will include the characterisation of natural organic matter (NOM) of three major SEQ source waters. It will also explore strategies for halide and organic matter DBPs precursor removal from water, together with examining the effect of these removal strategies on DBPs formation.

Figure 1. NOM Isolation Process^

^Adapted from a design provided by Prof Howard Weinberg, University of North Carolina



Results

The procedure detailed in Figure 1 (below) has been developed and successfully used to isolate NOM from raw water collected at the Molendinar Water Treatment Plant, sourced from Hinze Dam. NOM isolates will be characterised by size exclusion chromatography (SEC) and fluorescence spectroscopy (EEM). The change in DBPs formation potential after enhanced coagulation and other adsorptive NOM removal strategies will be assessed to determine the NOM components most relevant to DBP formation in SEQ source waters, and how they can best be removed.

Future Direction

- Assess emerging and regulated DBPs in SEQ drinking waters.
- Isolate and characterise NOM from other sources (Mt Crosby and Capalaba water treatment plants).
- Characterise NOM and determine the reactive components relevant to DBP formation in SEQ.
- Provide strategies for lowering formation of regulated and emerging/ unregulated DBPs by increased removal of organic precursors.
- Develop practical strategies for halide (bromide and iodide) removal, thereby reducing halogenated DBPs formation.

Ozonation and Biological Activated Carbon Filtration of Wastewater Treatment Plant Effluents

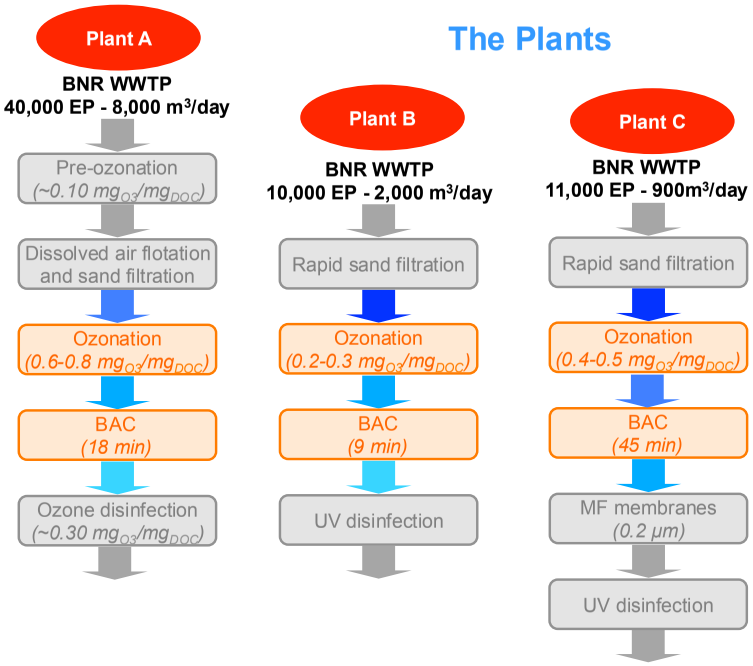
Reungoat, J.[1], Escher, B.I.[2], Macova, M.[2], Rattier, M.[1], Gernjak, W.[1] and Keller, J.[1]

[1] The University of Queensland, Advanced Water Management Center (AWMC), Qld 4072
[2] The University of Queensland, National Research Centre for Environmental Toxicology (Entox), QLD 4108

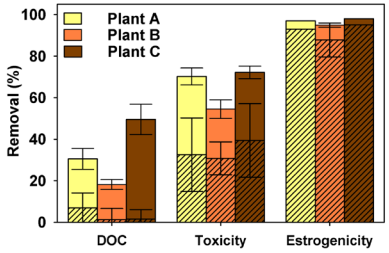
Project Summary

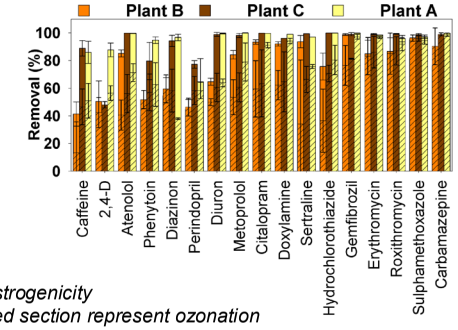
The enhanced treatment project assessed the combination of ozonation followed by biological activated carbon filtration for the reclamation of wastewater. The project focused on the removal of organic micropollutants to produce high quality recycled water. Three full-scale reclamation plants were sampled to determine their efficiency for micropollutant removal and to identify operating parameters influencing their performance.

The Plants



Results





Removal of dissolved organic carbon, toxicity, estrogenicity (left) and selected micropollutants (right). Hatched section represent ozonation alone, the entire bar represent ozonation + BAC.

The results confirm that the combination of ozonation with biological activated carbon can achieve high removal of organic micropollutants (>90%) and reduction of non-specific toxicity (50-70%) and of estrogenicity (>95%). Moreover, the results clearly showed that the ozone dose and the BAC filtration time influence the removal of micropollutants; both increase the removal when increased. Finally, it is important to note that in the plant using the lower ozone dose and filtration time, some micropollutants were removed less than 90%. Therefore, it is important to design plants with a sufficient ozone dose and filtration time to ensure high removal of a wide range of compounds.

Future Direction

In the last year, the project will aim to disseminate these results to a broader community, particularly to water utilities and city councils in regional areas. Indeed, in these areas the O_3/BAC process could be an alternative to membrane filtration for the production of high quality recycled water as membrane treatments pose the issue of brine disposal.

Queensland Government
National Research FLAGSHIPS Water for a Healthy Country CSIRO
Griffith UNIVERSITY
The University Of Queensland AUSTRALIA

Tracking the Source of *Escherichia coli* in SEQ Catchments

Stratton, H.[1] (Christie, M.[2], Toze, S.[2], Katouli, M.[3], Ahmed, W.[2] and Watkinson, A.[4])

[1]Smart Water Research Centre, Griffith University, [2]CSIRO Water for a Healthy Country, Ecosciences Precinct, Dutton Park, [3]University of Sunshine Coast, [4]Seqwater

Project Summary

Pathogenic microorganisms are the single greatest acute health risk within a community's water supply. Understanding the sources of these organisms and the risks they present is paramount to effectively managing the water supply. The standard measurement for faecal contamination involves determining the presence of *Escherichia coli*. However, the detection of *E. coli* does not provide any information about the source of contamination and therefore is not always an effective indicator of risk to humans.

The aim of this work was to collect a high number of *E. coli* isolates from two catchments in SEQ and compare their β-glucuronidase gene to determine whether they originated from human or animal faeces.

Results and Discussion

Early analysis of the sequenced 500bp gene segment suggests that the isolates from two catchments originate from both animal and human sources, with the majority aligning with animals.

- Of 311 *E. coli* isolates from two dams, 254 (81.7%) appeared to originate from animals and they aligned mostly with cattle and kangaroos (Table 1).
- Fifty seven (18.3%) of the 311 isolates aligned with the known human cluster. Although this is lower than the animal isolates, any human input warrants further investigation.
- The database requires additional known isolates from humans to reduce the number of rogue isolates (ie, humans (30.2%) aligning with animals) and give more confidence in the analyses.

Isolates	Alignment with Animals	Alignment with Humans	Total
Somerset Dam	69 (84%)	13 (16%)	82
Baroon Pocket Dam	185 (80.8%)	44 (19.2%)	229
Total	254 (81.7%)	57 (18.3%)	311
Knowns	Alignment with Animals	Alignment with Humans	Total
Animals	97 (90.6%)	10 (9.4%)	107
Humans	16 (30.2%)	37 (69.8%)	53
Total	113	47	160

Table 1. Information from neighbour-joining analysis (1,000 boostrap resampling of data) of the 500 bp segment of the β-glucuronidase gene from *E. coli* isolates from two catchments.

Key Findings



- Animals, most likely cattle and kangaroos, are the greatest source of *E. coli* in the catchments studied.
- Human input was less but still significant in terms of a human health risk.

Future Direction

- Confirm, improve and test the robustness of this method by:
  - Expanding the number of known isolates from humans and animals;
  - Include multiple methods and other bacteria in the study.
- Continue to sample catchments in SEQ and storing isolates.
- Investigate the pathogenecity of isolates.

Outcome of the Assessment of Factors Influencing the Decay of Pathogens in South East Queensland Reservoirs

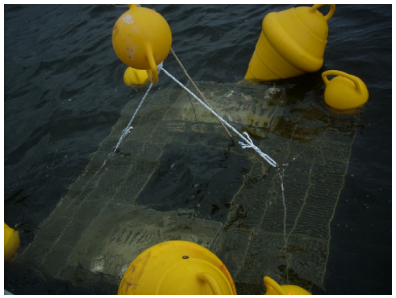
Toze, S., Hodgers, L. and Palmer, A.
CSIRO Water for a Healthy Country, Ecosciences Precinct, Dutton Park, QLD

Project Summary

The research depicted here is part of the Fate of Pathogens and Chemicals in Dams project. The aim is to investigate the factors within SEQ reservoirs that impact on the removal of biological and chemical contaminants. The aim is to determine if catchment health can be used as an indicator of the water quality reaching the treatment plant.

These results are a summary of the first five experiments undertaken on Lake Wivenhoe.





Results and Discussion

The results for the first five experiments are given in Table 1.

- *Cryptosporidium* oocysts have the longest survival times of 13-17 days for a 1 log removal.
- The bacteria and MS2 had faster rates of decay, with overall 1 log removal times of eight days or less.
- Adenovirus had decay rates slower than for the bacteria but faster than those observed for *Cryptosporidium*.

Table 1. Log decay times[1] for target microorganisms in Lake Wivenhoe testing different variables

Experiment	Specifics	Month	*E. coli*	*Salmonella*	*Campy*	*Enterococci*	MS2	Adenovirus	*Crypto*
General Decay Assessment		Jun 2009	NR	7	8	2	5	11	13
Repeat of Exp 1		Aug 2009	2	4	6			RP	
Indigenous Microbes	Filtered	Oct 2009	3	NR	NR		4	RP	
	Unfiltered		1.5	NR	NR		3	RP	
Depth	Surface	Sept 2010	3	5	4	5		RP	11
	15m		3	5	14	ND		RP	
Sunlight	Surface – Sun	Mar 2010		18			27		93
	Surface – Dark			27			122		263
	5m Light			29			250		270
	5m Dark			24.5			122		588

[1] decay of days for a 1 log reduction for all experiments except for sunlight which was hours NR = no result obtained due to culture issues; RP = Results Pending.

Key Findings

- The results from the non-sunlight experiments demonstrated that there was minimal variation in the decay of individual microorganisms regardless the experiment and variable being tested. The exception was for *Campylobacter* at depths below 15m.
- Sunlight has an impact on pathogen decay at the reservoir surface but has no major impact at 5m depth.

Future Directions

Further testing is being undertaken to test:

- Seasonal impacts.
- Impacts of flood conditions.
- Influence of sunlight in different seasons.
- Role of the mid-Brisbane River on decay.

Queensland Government
National Research FLAGSHIPS Water for a Healthy Country CSIRO
Griffith UNIVERSITY
The University Of Queensland AUSTRALIA

Wind and Wave Testing of a New Evaporation Reduction Monolayer System

Putland, S.G.[1], Schouten, P.W.[1], Lemckert, C.J.[1], Solomon, D.[2], Leung, A.[2], Tran, D.[2], Talemi, P.[2] and Qiao, G.[2]

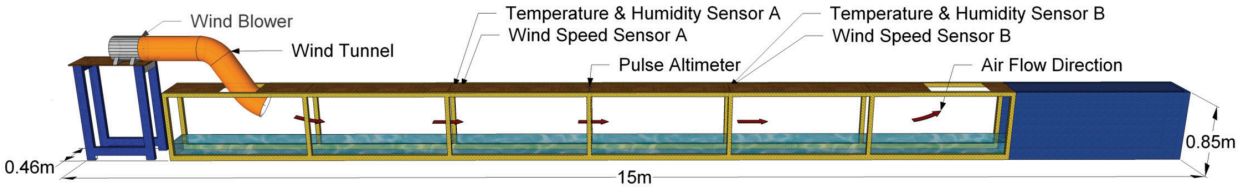
[1] School of Engineering, Griffith University, Parklands Drive, Gold Coast, 4215; [2] Department of Chemical and Biomolecular Engineering, University of Melbourne, Parkville, 3010

Project Summary

This project examined the evaporative reduction capacity of several new prototype monolayers developed by the Cooperative Research Centre for Polymers (CRCP) under varying wind and wave conditions, such as those found in real-world water reserves.

Methods

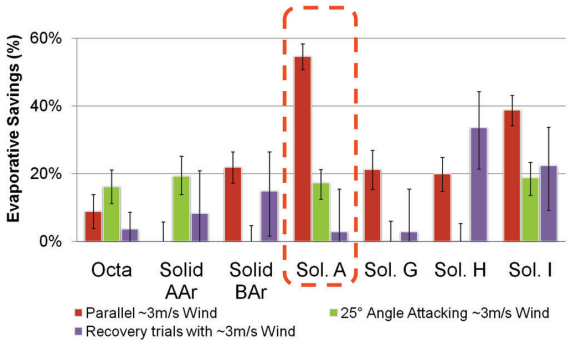
- A rigid water basin (shown below) confined in an atmospherically controlled enclosure was used to ensure testing conditions were regulated and user defined;
- Monolayer formulations were tested under various wind and wave conditions for 24 and 48 hours; and
- Monolayer deployment dosage was optimised by testing concentrated formulations of monolayers under varying molecular layer spreading ratios.



Schematic of the Monolayer Testing Wave Basin Setup

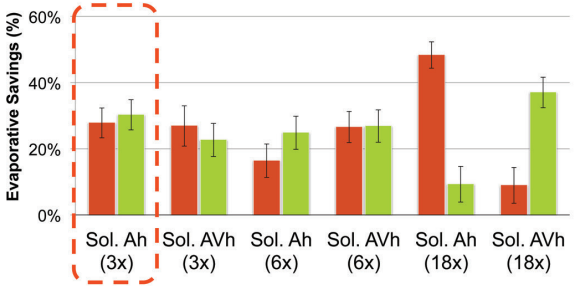
Results

Evaporative Savings for CRCP Monolayers



- In the majority of experiments, the CRCP monolayers outperformed the previously tested octadecanol suspension control film;
- Solution monolayer A was chosen for further trials to optimise dosages.

Evaporative Savings for Concentrated CRCP Monolayers



Values in brackets indicate number of molecule layers (i.e. dosage quantities)

- Concentrated monolayer solution Ah at a 3 x surface layer spread performed the most consistently and was also determined to be the most economically viable.

Conclusions

- Disparity in the evaporation savings occurring in the trials was a result of variations between the chemical formulations of each monolayer; and
- On average, the prototype monolayers were able to reduce evaporation by 10 – 50% in the controlled environment and are now ready to be tested in real-world water reserves.

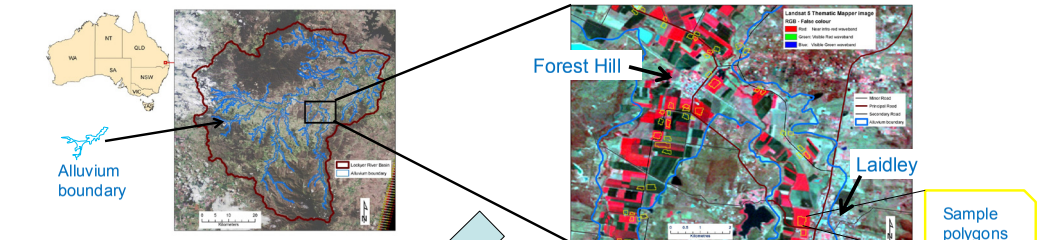
Crop Mapping in the Lockyer Valley using Satellite Imagery

Ellis, T.W., Hartcher, M., Hodgen, M. and Wolf, L.

CSIRO Land and Water, Ecosciences Precinct, GPO Box 2583, Brisbane, Queensland 4001

Project Summary

Purified Recycled (urban) Wastewater (PRW) may be available to augment irrigation in the Lockyer Valley (a). This could significantly affect groundwater levels, ecosystem health and agricultural production. Existing groundwater models are being updated to better understand this, but they require new estimates of groundwater abstractions and recharge. Land use mapping from satellite imagery has previously been used to predict water balance components eg, Castaño *et al.* (2010). We used Landsat Thematic Mapper imagery to produce detailed crop maps for this purpose.



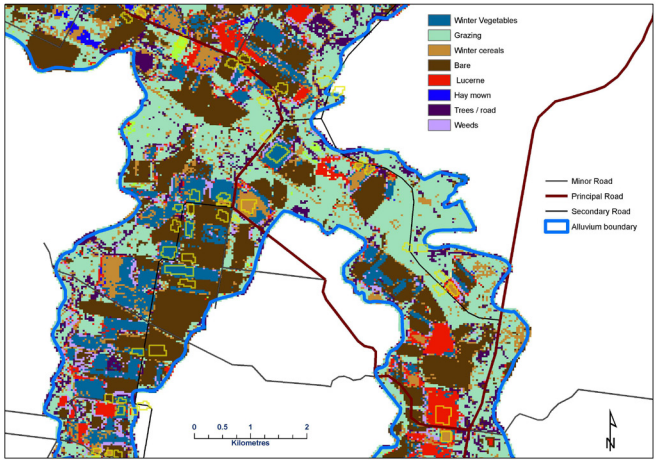
a) Lockyer Valley Catchment

b) False Colour Image

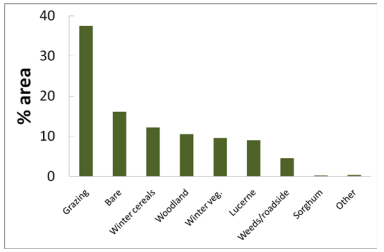
Results and Discussion

September 2010 cloud-free Landsat Imagery (30m resolution) was sampled at 54 ground-truthing sites within the alluvium boundary (b). A supervised classification of the image is shown in (c). The distribution of crop types in September is shown in (d). The September classification was applied to an October 2010 image and evaluated with concurrent ground-truthing data (e). Four subsequent cloud-free Landsat images with coordinated ground-truthing observations are being used to refine the method for identifying crops in any season.

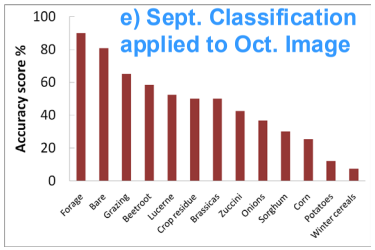
c) Classified Image



d) Crop Distribution Sept. 2010



e) Sept. Classification applied to Oct. Image



Key Findings

- We developed a detailed crop map 'snapshot' of the Lockyer alluvium (c);
- The preliminary evaluation was encouraging (e); further calibration is required for most crops, particularly for winter cereals.

Future Directions

- On-going ground-truthing and evaluation of methods;
- Identification of representative crop rotations in time and space, to simplify water balance modelling;
- Interrogation of previous Landsat imagery to understand historical cropping patterns and water use.

Reference

Castano, S., Sanz, D. and Gomez-Alday, J. J. (2010). *Methodology for Quantifying Groundwater Abstractions for Agriculture via Remote Sensing and GIS*. Water Resources Management 24(4): 795-814.

Queensland Government
National Research FLAGSHIPS Water for a Healthy Country
CSIRO
Griffith UNIVERSITY
The University Of Queensland AUSTRALIA

Deep Drainage Estimates Under Irrigated Cropping in the Lockyer Valley

Kodur, S.[1], Robinson, J.B.[1], Foley, J.L.[1], Ellis, T.[2] and Wolf, L.[2]

[1]Department of Environment and Resource Management, PO Box 318, Toowoomba, Queensland 4350, [2] CSIRO Land and Water, Ecosciences Precinct, GPO Box 2583, Brisbane, Queensland 4001

Project Summary

- Deep drainage and diffuse groundwater recharge make up significant parts of the water balance in irrigated areas. They define the need for water imports in overused aquifer systems and are the basis for salt transport prediction.
- As a part of the Lockyer Valley Purified Recycled Water Project, deep drainage rates were modelled using the Howleaky model at Forest Hill in the central Lockyer Valley, for likely irrigation scenarios (that differed in target amount and interval).
- Soil hydraulic parameters were measured and used in the model along with 12 years of real paddock management history from a local farmer.

Results and Discussion

- Deep drainage ranged from 59 mm/year (scenario 1) to 262 mm/year (scenario 3). The deep drainage estimates from forward modelling are similar with estimates from measured chloride profiles using SODICS model [eg, Thorburn *et al.* 1990, ~100-300 mm/year; and others (unpublished data), ~91-218 mm/year].
- As the irrigation amount increased (from 482 mm/year to 717 mm/year), the contribution of irrigation (or rainfall) to deep drainage increased (from 5% to 19% of the total water balance).
- Irrigation beyond Drained Upper Limit (DUL) (for a fixed irrigation interval based on water deficit) contributed little to transpiration but mostly to deep drainage. Excess irrigation contributes to ground water recharge in the long term.
- Modern agricultural practice in the Lockyer, which is optimised by using online measurements, was simulated in the model assuming irrigation occurs if soil water potential >25 kPa. Results show significant deep drainage, similar to irrigation of 25% above DUL.

Table 1: Soil water balances (mm/year) for three irrigation scenarios at Forest Hill over a period of 12 years (1997-2010).

Irrigation Scenario	Irrigation	Deep drainage	Transpiration
1: irrigated to DUL	482[a] - 530[b]	59[a] - 84[b]	661[a] - 673[b]
2: irrigated 25% above DUL	527[a] - 614[b]	103[a] - 164[b]	664[a] - 677[b]
3. Irrigated 50% above DUL	717[b]	262[b]	676[b]

Note: Irrigation interval (all scenarios) is based on soil water deficit; [a] = 50 mm for high value vegetables; 60-75 mm for other crops; [b] = 25 mm for high value vegetables; 50-60 mm for other crops.

1 = DUL for all crops; 2 = 25% above DUL for all high value vegetables; DUL for other crops; 3 = 50% above DUL for all high value vegetables; 25% above DUL for other crops.



Key Findings

- Depending on irrigation strategy (target amount and interval), deep drainage at the study site ranged 59-262 mm/year. Even with modern soil moisture monitoring and judicious irrigation (targeting high transpiration), the simulated deep drainage at the study site would be ~84-164 mm/year.
- The modelling of deep drainage is very sensitive to the amount of irrigation applied beyond the DUL. Knowledge of farming practice is essential.
- The drainage rate simulations will now be considered alongside the salt profile information.

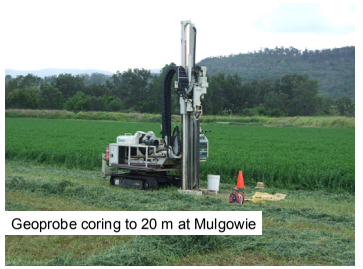
Reference:

Thorburn, P.J., Rose, C.W., Shaw, R.J. and Yule, D.F. (1990) *Interpretation of solute profile dynamics in irrigated soils. I. Mass balance approaches.* Irrigation Science, (11), 199-207.

Salt Loads and Water Storage Changes in the Regolith in the Lockyer Valley 15 Years on - the Ellis Sites Revisited

Foley, J.L.[1], Wolf, L.[2], Harris, M.[1], Ellis, T.[2], Kodur, S.[1] and Manders, J.[3]

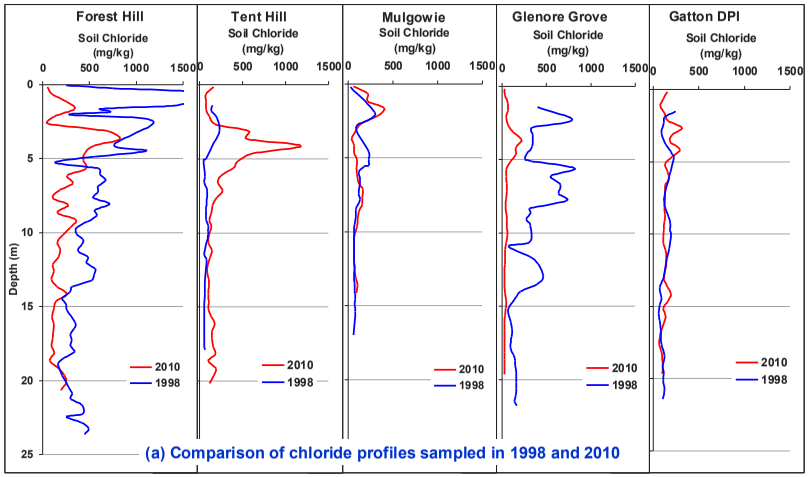
1 Department of Environment and Resource Management, Toowoomba Qld, 2 CSIRO Land and Water, Ecosciences Precinct, Brisbane, Qld,
3 Department of Environment and Resource Management, Brisbane Qld



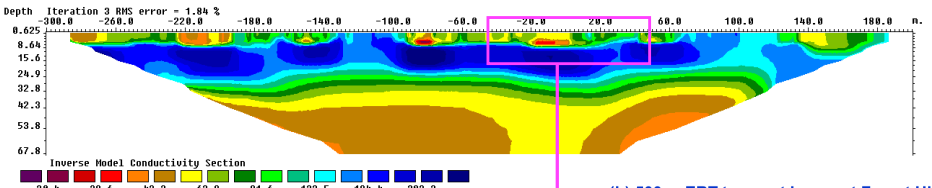
Geoprobe coring to 20 m at Mulgowie

Project Summary

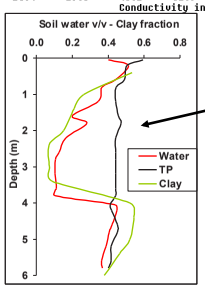
Lockyer Valley Purified Recycled Water project – In this component of the project potential impacts of introducing purified recycled water (PRW) on regional water and salt balances in the Lockyer Valley were determined to assess the potential mobilisation/flushing of existing salt in the regolith and export from the catchment resulting from potential increased runoff, deep drainage and rises in stream flow. Soil water and salt/chloride profiles were measured at five sites (previously cored in 1998[I]) to look at decadal changes in soil salinity and estimate deep drainage. Electrical resistivity tomography (ERT) was used to map spatial variability of water and salt stored in the regolith.

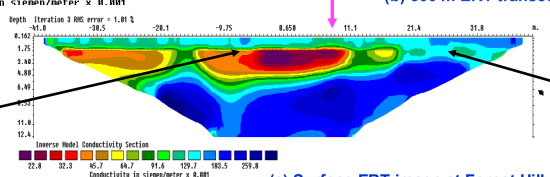


(a) Comparison of chloride profiles sampled in 1998 and 2010

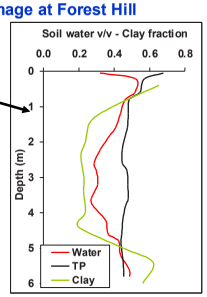


(b) 500 m ERT transect image at Forest Hill





(c) Surface ERT image at Forest Hill



Results and Discussion

2010 chloride (Cl) profiles from geoprobe cores were compared with Cl in cores taken by Ellis in 1998[I] (a). 45 t/ha of Cl moved down the regolith at Forest Hill over 12 years, coinciding with a change in irrigation water use from 'salty' groundwater to high quality surface water. Estimates of deep drainage (DD) at this site (using the SODICS method[II]) range from 120 to 590 mm/yr. Significant loss of Cl also occurred at Glenore Grove (12 t/ha) with a DD estimate of 290 mm/yr. Cl stores at other sites were stable or showed slight increase. ERT imaging and coring/chemistry along transects at Glenore Grove and Forest Hill indicate highly variable water/salt/clay both spatially and with depth (b) (c).

Key Findings

Water and salt stores were found to be highly spatially variable across the landscape, indicating complex and detailed modelling is required to predict environmental effects of increasing the landscape water balance and potentially flushing present salt loads.

I Ellis, R. and Bajracharya, K. 1999, Water Quality Deterioration in Alluvial Aquifers, LWRRDC Project QPI30-Final report, August 1999
II Thorburn, P.J., Rose, C. W., Shaw, R. L., and Yule, D. F. 1990, Interpretation of solute profile dynamics in irrigated soils I. Mass balance approaches. Irrigated Science 11:199-207

Queensland Government
National Research FLAGSHIPS Water for a Healthy Country CSIRO
Griffith UNIVERSITY
The University Of Queensland AUSTRALIA

Awards

Best Presentation

Grace Tjandraatmadja – Dissecting Rainwater Pump Energy Use in Urban Households

Best Paper

Matthew Gooda – Climate Change Impact on Water Availability in the South East Queensland Region

Best Poster

Jenny Foley – Salt Loads and Water Storage Changes in the Regolith in the Lockyer Valley 15 Years on - the Ellis Sites Revisited

Recipients were presented with a bottle of wine.

Delegates

Official Opening

Stephen Robertson, MP, Minister for Energy and Water Utilities, QLD

Keynote Speaker

Paul Doherty, Manager Water Security Planning, Strategy Planning and Regulation, SA Water, Adelaide, SA

Guest Speakers

Rolf Altenburger, Head, Department of Bioanalytical Ecotoxicology, Helmholtz Centre for Environmental Research UFZ, Leipzig, Germany
Alistair Boxall, Reader, Environmental Science, Environment Department, University of York, Heslington, England
Greg Ryan, Manager, Environment and Technology, South East Water Limited, Melbourne, VIC

UWSRA Management Board

Christine Williams, Department of Environment and Resource Management (DERM), QLD
Scott Keyworth, CSIRO, ACT
Alan Gregory, CSIRO, NSW
John Mott, University of Queensland, QLD
Karen Waldman, Queensland Water Commission, QLD

UWSRA Research Advisory Committee

Tad Bagdon, Queensland Water Commission, QLD
Ted Gardner, Department of Environment and Resource Management, QLD
Mark Pascoe, International Water Centre, QLD

Attendees

Aisling Redmond, Office of the Water Regulator (DERM), QLD
Alan Hoban, Healthy Waterways Partnership Limited, QLD
Alan Tyrresturp, Queensland Water Commission, QLD
Alex Coe, Marchment Hill Consulting, QLD
Amy Lomax, Gold Coast City Council, QLD
Anas Makki, Griffith University, QLD
Andrea Walton, CSIRO, QLD
Andrew Huang, Griffith University, QLD
Anna West, KBR (Kellogg Brown and Root Pty Ltd), QLD
Anne Simi, Brisbane City Council, QLD
Anthony Armstrong, Queensland Urban Utilities, QLD
Arran Canning, Seqwater, QLD
Ashok Sharma, CSIRO, VIC
Barnaby Resch, Logan City Council, QLD
Barry Holcroft, Unitywater, QLD
Barry Spencer, Seqwater, QLD
Beate Escher, University of Queensland, QLD
Bradford Sherman, CSIRO, ACT
Brian Davis, Office of the Water Regulator (DERM), QLD
Brian McIntosh, CSIRO, QLD
Bruce Gaydon, Toowoomba City Council, QLD
Byron Carragher, Griffith University, QLD
Cara Beal, Griffith University, QLD
Chris Owens, SEQ Water Grid Manager, QLD
Chris Tanner, Bligh Tanner Pty Ltd, QLD
Craig Walton, Department of Environment and Resource Management, QLD
Dana Hallett, Queensland Urban Utilities, QLD
Daniel Parker, Gold Coast City Council, QLD
Dave Tucker, CSIRO, WA
David Cameron, Queensland Water Directorate, QLD
David Haug, Queensland Urban Utilities, QLD
David Kent, CSIRO, VIC
David Merheb, Queensland Urban Utilities, QLD
David Robinson, Queensland Climate Change Centre of Excellence, QLD
Declan Page, CSIRO, QLD

Donald Mackenzie, Logan City Council, QLD
Emma Anastasi, University of the Sunshine Coast, QLD
Emma Brewis, Queensland Water Commission, QLD
Erik Schmidt, University of Southern Queensland, QLD
Fiona Henderson, CSIRO, QLD
Fiona Irwin, Brisbane City Council, QLD
Freyja van Woerden, Moreton Bay Regional Council, QLD
Gary Soutar, Allconnex Water, QLD
Grace Tjandraatmadja, CSIRO, VIC
Greg Jackson, Qld Health, QLD
Greg Laves, Griffith University, QLD
Greg Tucker, Brisbane City Council, QLD
Greta Clogan, SEQ Water Grid Manager, QLD
Helen Stratton, Griffith University, QLD
Hollie King, University of Queensland, QLD
Huijun Zhao, Griffith University, QLD
James Udy, Healthy Waterways Partnership Limited, QLD
Jane-Louise Lampard, Griffith University, QLD
Janet Tang, University of Queensland, QLD
Jatinder Sidhu, CSIRO, QLD
Jawad Youssef, Linkwater, QLD
Jenny Foley, Department of Environment and Resource Management, QLD
Jim Fear, SEQ Water Grid Manager, QLD
Joan Meecham, Department of Environment and Resource Management, QLD
Jodieann Dawe, Water Quality Research Australia, SA
Joe Lane, University of Queensland, QLD
John Gardner, CSIRO, QLD
John Ruffini, Department of Environment and Resource Management, QLD
Judith Jensen, Department of Environment and Resource Management, QLD
Judy Blackbeard, Melbourne Water, VIC
Julien Reungoat, University of Queensland, QLD
Kalinda Watson, Griffith University, QLD
Kate Lanskey, Seqwater, QLD
Kathy Baker, Allconnex Water, QLD
Katrin Doederer, University of Queensland, QLD
Kelly Fielding, University of Queensland, QLD
Kevin Chu, Griffith University, QLD
Kevin Kodur, Department of Environment and Resource Management, QLD
Kimberley Wildbore, Linkwater, QLD
Kirk Underwood, Queensland Urban Utilities, QLD
Kristell Le Corre, University of Queensland, QLD
Lars Pelzer, International Water Centre, Germany
Lavanya Susarla, Moreton Bay Regional Council, QLD
Leif Wolf, CSIRO, QLD
Lesley Rogers, Department of Environment and Resource Management, QLD
Linda Sheehan, Earth Law Centre, USA
Madeleine Porter, Griffith University, QLD
Magnus Moglia, CSIRO, QLD
Malcolm Hodgen, CSIRO, QLD
Malika Fernando, Department of Environment and Resource Management, QLD
Maria José Farré, University of Queensland, QLD
Mark Angles, Sydney Water, NSW
Mark Crabtree, Link Water, QLD
Mark Gibbs, AECOM, QLD
Mark Quayle, Irrigation Australia Ltd, QLD
Mark Sherson, Queensland Urban Utilities, QLD
Mary Boydell, Queensland Water Commission Board, QLD
Matthew Ferguson, Sydney Water, NSW
Matthew Gooda, Department of Environment and Resource Management, QLD
Matthew Ingerman, Allconnex Water, QLD
Matthew Potter, Melbourne Water, VIC
Maxime Rattier, University of Queensland, QLD
Melita Stevens, Melbourne Water, VIC
Meng Chong, CSIRO, QLD
Michael Bartkow, Seqwater, QLD
Michael Lukin, Unitywater, QLD
Michael Moore, Water Quality Research Australia, QLD
Mike Taylor, Seqwater, QLD
Murray Hall, CSIRO, QLD
Naomi Soustal, Healthy Waterways Partnership Limited, QLD
Nathaniel Fox, Department of Environment and Resource Management, QLD

Nicholas Stuckey, University of Southern Queensland, QLD
Nick Morgan, Brisbane City Council, QLD
Nicole Knight, Griffith University, QLD
Nigel Garson, Office of the Water Regulator (DERM), QLD
Oz Sahin, Griffith University, QLD
Padma Cuppala, Office of the Water Regulator (DERM), QLD
Pam Pittaway, University of Southern Queensland, QLD
Pat Bourke, Brisbane City Council, QLD
Paul Burrell, SEQ Water Grid Manager, QLD
Paul Heaton, Allconnex Water, QLD
Peter Daniels, CSIRO, QLD
Peter Dillon, CSIRO, QLD
Peter Schouten, Griffith University, QLD
Peter Sommer, Linkwater, QLD
Reza Talebpour, Griffith University, QLD
Rob Fearon, Queensland Water Directorate, QLD
Rob Townsley, Seqwater, QLD
Rodney Stewart, Griffith University, QLD
Roger O'Halloran, CSIRO, VIC
Rolf Rose, Queensland Water Commission, QLD
Sam Putland, Griffith University, QLD
Sanja Oldridge, Aurecon, QLD
Sanne Voogt, Department of Environment and Resource Management, QLD
Scott Eade, Allconnex Water, QLD
Shannon McBride, Unitywater, QLD
Shannon Murphy, Policy Advisor, Water Utilities, Office of the Minister for Energy and Water Utilities, QLD
Shelley Luxton, Department of Environment and Resource Management, QLD
Sheri Fitton, Seqwater, QLD
Shiroma Maheepala, CSIRO, VIC
Shivanita Umapathi, CSIRO, QLD
Simon Cowland-Cooper, Irrigation Australia Ltd, QLD
Simon Toze, CSIRO, QLD
Sriya Fernando, Queensland Urban Utilities, QLD
Stephanie Ashbolt, CSIRO, QLD
Stephen Cook, CSIRO, QLD
Steve Spencer, CSIRO, QLD
Steve Storer, Department of Local Government and Planning, QLD
Steven McVeigh, Gold Coast City Council, QLD
Stewart Burn, CSIRO, VIC
Sudhara Anandakumar, University of Queensland, QLD
Sue Larsen, Department of Environment and Resource Management, QLD
Tamara Stevenson, Allconnex Water, QLD
Thuwaragan Sritharan, CSIRO, QLD
Tim Cowan, CSIRO, VIC
Tim Ellis, CSIRO, QLD
Tom Moore, MWH Australia, QLD
Vicki Ross, Griffith University, QLD
Vourn Lutton, Queensland Water Commission, QLD
Wai-Tong Wong, Queensland Water Commission, QLD
Warish Ahmed, CSIRO, QLD
Wenju Cai, CSIRO, VIC
Yan Du, CSIRO, VIC
Yvan Poussade, Veolia Water Australia, QLD

UWSRA Executive

Don Begbie, Director, QLD
Sharon Biermann, Program Manager, QLD
Steven Kenway, Consultant, QLD
Sharon Wakem, Executive Officer, QLD



RECYCLED CONTENT



FOREST MANAGEMENT



MANAGEMENT SYSTEMS



BLEACHING PROCESS

Urban Water Security Research Alliance



www.urbanwateralliance.org.au