

Urban Water Security Research Alliance

Science Forum and Stakeholder Engagement

Building Linkages, Collaboration and Science Quality

19-20 June 2012
Brisbane, Queensland

Program and Papers

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/>

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Begbie, D.K., Kenway, S.J., Biermann, S.M. and Wakem, S.L. (eds) (2012), *Science Forum and Stakeholder Engagement: Building Linkages, Collaboration and Science Quality*, Urban Water Security Research Alliance, 19-20 June, Brisbane, Queensland.

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Contents

Welcome from the Chair	5
The Alliance Research Program and Project Leaders.....	6
Alliance Research Projects 2011-2012 - Poster	7
Program.....	8
Guest Speaker Papers	13
Australia's Urban Water R&D – Needs and Capability Forum	14
Assessment of Alternative Water Options in Adelaide: The MARSUO and Optimal Water Resource Mix Projects	15
Rainwater Tank Study of New Homes.....	18
National Perspectives on Potable Recycling – Barriers and Costs	19
Water Efficiency – A Dry Topic.....	21
Characterisation of Disinfection By-Product Formation for Water Quality Management.....	25
Collaboration in Developing Total Water Cycle Management Planning – Critical for Success	27
Factors that Mitigate the Formation of Emerging Disinfection By-Products in Recycled Water.....	30
Re-Imagining Sustainability	33
The Significance of Large Scale Seawater Desalination in Achieving Urban Water Security.....	36
Papers.....	39
Can Stormwater Harvesting Restore Pre-Development Flows in Urban Catchments in South East Queensland?.....	40
Can Bugs Live Anywhere? Habitat, Hydrology and the Health of Urban Creek Ecosystems in South East Queensland	47
Potential Health Risks from Pathogens in Alternative Waters.....	53
Water Quality of Logan's Dam as a Function of Ecosystem Variables, Climate Fluctuations and Application of a Monolayer	58
Photodegradation of Dissolved Organic Matter: The Impact on Monolayers	64
The Impact of Artificial Monolayers on Water Quality: Data from Tank and Field Trials	70
Integration of Pathogen and Chemical Removal in Reservoirs with Hydrodynamic Modelling	76
Rainwater System Design for Urban Residential Households –the Role of Header Tanks and Pressure Vessels in Improving Energy Efficiency	82
Monitoring of Residential Rainwater Tanks in South East Queensland to Investigate Mains Water Savings and Volumetric Reliability	88
Enabling the Use of the Lockyer Valley Groundwater System as a Buffer in the South East Queensland Regional Water Grid – An Assessment Framework	95
Communication and Community Responses to Recycled Water.....	102
Towards the Quantification of Rainwater Tank Yield in South East Queensland by Considering the Spatial Variability of Tanks	108
Strategies for Managing the Condition of Rainwater Tanks in South East Queensland	114
Community Acceptance of Policy Options for Managing the Maintenance of Rain Water Tanks: a Qualitative Study.....	120
Identifying the Drivers of Water Consumption: a Summary of Results from the South East Queensland Residential End Use Study.....	126
Water End Use Feedback Produces Long-Term Reductions in Residential Water Demand.....	133
Case Study - Occurrence of Non-Regulated Disinfection By-Products from the Capalaba Region's Distribution System	139
Disinfection By-Product Minimisation by Organic and Inorganic Precursor Removal	146
Bioanalytical Assessment of the Formation Potential of Disinfection By-Products during Drinking Water Treatment	153
Cost of Pollution and Options Evaluation – Case Study of Moreton Bay Regional Council Total Water Cycle Management Plan.....	156
Life Cycle Perspectives for Total Water Cycle Planning	162
Assessing Urban Water Strategies for Total Water Cycle Management.....	168
Ozonation and Biological Activated Carbon Filtration of Wastewater Treatment Plant Effluents.....	174
Electrochemical Treatment of Reverse Osmosis Concentrate: Strategies to Minimise the Formation of Halogenated By-Products	184
Optimising Micropollutants Extraction for Analysis of Water Samples: Comparison of Different Solid Phase Materials and Liquid-Liquid Extraction	191
Implications of Resource-Efficient Technology on Peak Water Demand and Water-Related Energy Demand	196
Mathematical Material Flow Analysis of Water-Related Energy in a Brisbane Household Quantifies the System and Potential for Savings.....	205

Decentralised Wastewater Systems: Robustness and Carbon Footprint.....	210
Reduction of Pharmaceutical loads In Municipal Wastewater: Would Onsite Treatment of Hospital Wastewater Be Effective?	217
Antibiotic Resistant Bacteria in Hospital Wastewaters and Sewage Treatment Plants.....	225
Faecal Indicators and Zoonotic Pathogens in Household Drinking Water Taps Fed from Rainwater Tanks.....	230
Climate and Water in South East Queensland: Past and Future	236
A Research Agenda for Water Smart Tropical and Sub-Tropical Cities and Towns	242
Genetic Markers for the Detection of Sewage Pollution in Environmental Waters in Brisbane.....	248
Poster - Abstracts	255
Posters - Presented	259
Quantification of Evaporation from a Small Water Body using the Scintillometry, Eddy Covariance and Mass Transfer Techniques	260
Mechanisms of Micropollutants Removal by BAC Filtration.....	261
Water Governance Under Pressure: Demand Management Policy Influence on Water Management Practices in South East Queensland	262
Development of a Domestic Water End Use Consumption Forecasting Model for South East Queensland.....	263
Intelligent Sensors to Determine Water End-Use.....	264
Crop Mapping and Water Balance Modeling to Extrapolate Applied Irrigation in the Lockyer Valley, Queensland.....	265
N ₂ O and Ozone Layer Depletion - A New Consideration for Urban Water Planners...?.....	266
Human Factors in Urban Water System Safety	267
Awards	269
Delegates	271

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 4th Science Forum being held on Tuesday, 19 and Wednesday, 20 June 2012, in Brisbane.



Chris Davis

Chair, Urban Water Security Research Alliance

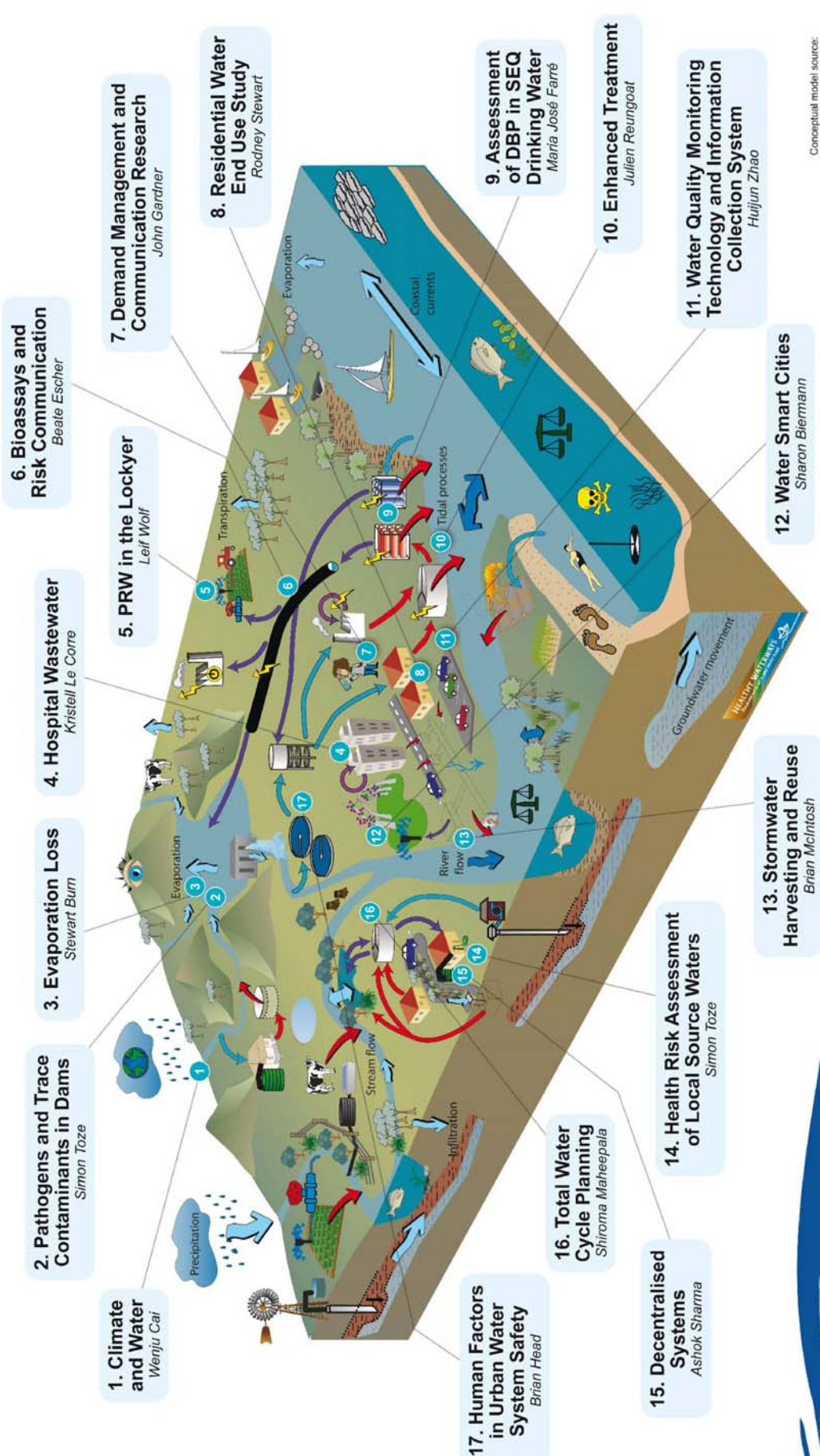
The Alliance Research Program and Project Leaders

Stormwater Harvesting and Reuse	<i>Project Leader:</i> Dr Brian McIntosh, CSIRO	07-3123 7766 b.mcintosh@watercentre.org
This project researches the innovative capture and storage of stormwater for additional water supply in SEQ and the impact of harvesting stormwater on creek and ecosystem health.		
Decentralised Systems	<i>Project Leader:</i> Dr Ashok Sharma, CSIRO	03-9252 6151 Ashok.Sharma@csiro.au
This project will validate the contribution rainwater tanks can make to water savings targets in SEQ. It also researches tank maintenance approaches and energy costs associated with tanks and decentralised wastewater treatment.		
Demand Management and Communication Research	<i>Project Leader:</i> Dr John Gardner, CSIRO	07-3833 5552 John.Gardner@csiro.au
This project is researching household water conserving behaviour with demand management interventions.		
Residential Water End Use Study	<i>Project Leader:</i> Dr Rodney Stewart, GU	07-5552 8778 r.stewart@griffith.edu.au
This project is quantifying residential water end uses and the impact of urban water demand management strategies in SEQ.		
Hospital Wastewater	<i>Project Leader:</i> Dr Kristell Le Corre, UQ	07-334 67207 k.lecorre@awmc.uq.edu.au
This project will quantify the relative contribution of hospital wastewater to sewage treatment plants.		
Pathogens and Trace Contaminants in Dams	<i>Project Leader:</i> Dr Simon Toze, CSIRO	07-3214 2698 Simon.Toze@csiro.au
This project investigates pathogen and trace contaminant attenuation within reservoirs and develops source tracking methods.		
Bioassays and Risk Communication	<i>Project Leader:</i> Dr Beate Escher, UQ	07-3274 9180 b.escher@uq.edu.au
This project further develops the scientific, technical and communication basis for implementation of bioanalytical tools in Queensland's water quality monitoring programs.		
Health Risk Assessment of Local Source Waters	<i>Project Leader:</i> Dr Simon Toze, CSIRO	07-3214 2698 Simon.Toze@csiro.au
This project quantifies the health risks associated with exposure to rainwater and storm water.		
Enhanced Treatment	<i>Project Leader:</i> Dr Julien Reungoat, UQ	07-3346 3235 j.reungoat@awmc.uq.edu.au
The project researches the effectiveness of biological activated carbon in non MF/RO options to achieve potable water quality.		
Assessment of Regulated and Emerging Disinfection By-Products in SEQ Drinking Water	<i>Project Leader:</i> Dr Maria José Farré, UQ	07-3346 3233 m.farre@awmc.uq.edu.au
This project assesses emerging and regulated disinfection by-product formation potential in different SEQ drinking water sources.		
Climate and Water	<i>Project Leader:</i> Dr Wenju Cai, CSIRO	03-9239 4419 Wenju.Cai@csiro.au
This project assesses and quantifies the impact of climate variability and change on water supply over the SEQ region.		
PRW in the Lockyer Valley	<i>Project Leader:</i> Dr Leif Wolf, CSIRO	07- 3214 2749 Leif.Wolf@csiro.au
This project researches the implications of using PRW as an adjunct to groundwater for irrigation in the Lockyer Valley.		
Total Water Cycle Planning Framework	<i>Project Leader:</i> Dr Shiroma Maheepala, CSIRO	03-9252 6072 Shiroma.Maheepala@csiro.au
The project aims to develop improved analytical methods and tools for quantitative assessment and decision-making for the development of total water cycle management plans SEQ.		
Water Quality Monitoring Technology and Information Collection System	<i>Project Leader:</i> Professor Huijun Zhao, GU	07-5552 8261 h.zhao@griffith.edu.au
This project has developed 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.		
Evaporation Loss	<i>Project Leader:</i> Adj Prof Stewart Burn, CSIRO	03-9252 6032 Stewart.Burn@csiro.au
This project is analysing technology to reduce evaporative losses from large SEQ dams and has undertaken detailed field analysis of the effectiveness of monolayers on a large farm dam.		
Water Smart Cities	<i>Project Leader:</i> Dr Sharon Biermann, CSIRO	07-3247 3007 sharon.biermann@csiro.au
This project will scope a framework for moving to Water Smart cities.		
Human Reliability Analysis Techniques	<i>Project Leader:</i> Professor Brian Head, UQ	07-3346 7450 Brian.Head@uq.edu.au
This project will scope human elements, eg judgement, decision making, perception, in risk management and quality control systems.		

Alliance Research Program 2011-2012

Leading water research and knowledge in South East Queensland

Urban Water Security Research Alliance



Science and Stakeholder Engagement Forum #4

Building Linkages, Collaboration and Science Quality

**Tuesday, 19 June 2012 [8:30am – 5:00pm]
Wednesday, 20 June 2012 [8:00am – 5:00pm]**

Venue: Catalina / Dehavilland Rooms - Royal on the Park, Cnr Alice and Albert Streets, Brisbane

Program

DAY 1 – Tuesday, 19 June				
Time	Session	Presenter		
8:30am	Registration and Welcome Coffee			
9:00am	Welcome and Opening Remarks	Chris Davis - Chair, UWSRA		
	Official Opening	Hon Mark McArdle, MP Minister for Energy and Water Supply		
9:15am	Keynote Speaker Water Reform Directions: Opportunities, Challenges and the Role of the National Water Commission	James Cameron CEO, National Water Commission		
9:45am	Guest Speaker Australia's Urban Water R&D – Needs and Capability Forum	Mark O'Donohue CEO, Australian Water Recycling Centre of Excellence		
10:15am	Morning tea			
	Theme: Reducing Water Grid Demand	Presenter	Theme: Total Water Cycle Planning and Management to Enhance Sustainability and Efficiencies	
11:00am	SESSION 1 – Stormwater (Chair – Tad Bagdon)		SESSION 2 – Reservoirs (Chair – Charles Lemckert)	
	Guest Speaker: Assessment of Alternative Water Options in Adelaide: The MARSUO and Optimal Water Resource Mix projects	Rosemary Leonard, Goyder Institute	Water Quality of Logan's Dam as a Function of Ecosystem Variables, Climate Fluctuations and Application of a Monolayer	
	Managing Urbanisation and Hydrological Change in South East Queensland	Stephanie Ashbolt	Photodegradation of Dissolved Organic Matter: The Impact on Monolayers	
	Can Bugs Live Anywhere? Habitat, Hydrology and the Health of Urban Creek Ecosystems in South East Queensland	Fran Sheldon	The Impact of Artificial Monolayers on Water Quality: Data from Tank and Field Trials	
	Potential Health Risks from Pathogens in Stormwater	Jatinder Sidhu	Integration of Pathogen / Chemical Removal in Reservoirs with Hydrodynamic Modelling	
	Panel Discussion / Questions		Panel Discussion / Questions	
1:00pm	Lunch			
2:00pm	SESSION 3 – Rainwater Tanks (Chair – Ted Gardner)		SESSION 4 – Purified Recycled Water (Chair – Wai-Tong Wong)	
	Guest Speaker: Rainwater Tank Study of New Homes	Helena Amaro, Sydney Water	Guest Speaker: National Perspectives on Potable Recycling – Barriers and Costs	
	Rainwater System Design for Urban Residential Households – Header Tanks and Pressure Vessels	Grace Tjandraatmadja	Enabling the Use of the Lockyer Valley Groundwater System as a Buffer in the South East Queensland Regional Water Grid – An Assessment Framework	
	Monitoring of Residential Rainwater Tanks in South East Queensland to Investigate Mains Water Savings and Volumetric Reliability	Shiv Umaphathi	Communication and Community Responses to Recycled Water	
	Panel Discussion / Questions		Panel Discussion / Questions	
3:30pm	Afternoon tea			
4:00pm	SESSION 5 - Rainwater Tanks (Chair – Ted Gardner)		SESSION 6 – Household Water Use (Chair – Danielle Butcher)	
	Towards the Quantification of Rainwater Tank Yield in South East Queensland by Considering the Spatial Variability of Tank Supplies	Esther Coultas	Guest Speaker: Water Efficiency – A Dry Topic - Exploring the Drivers for Water Utilities to Maximise Water Sales or Encourage Water Efficiency	
	Strategies for Managing the Condition of Rainwater Tanks in South East Qld	Magnus Moglia	Identifying the Drivers of Water Consumption: a Summary of Results from the South East Queensland Residential End Use Study	
	Community Acceptance of Policy Options for Managing the Maintenance of Rainwater Tanks: a Qualitative Study	Andrea Walton	Water End Use Feedback Produces Long-Term Reductions in Residential Water Demand	
	Panel Discussion / Questions		Panel Discussion / Questions	
5:30pm	Session Drinks and Networking Opportunity			
6:30pm	CLOSE – Day 1			

DAY 2 – Wednesday, 20 June				
8:00am	Registration and Welcome Coffee			
Time	Theme: Water Quality of Water Sources	Presenter		Presenter
8.30am	SESSION 7 – Disinfection By-Products (Chair – Paul Burrell)			
	Guest Speaker: Characterisation of DBP formation for Water Quality Management	Rolando Fabris, AWQC (WQRA)	SESSION 8 - Total Water Cycle Planning (Chair – Tad Bagdon)	
	Case Study - Occurrence of Non-Regulated Disinfection By-Products from the Capalaba Region's Distribution System	Maria José Farré	Guest Speaker: Collaboration in Developing Total Water Cycle Management Planning – Critical for Success	Tony Weber, BMT WBM
	Disinfection By-Product Minimisation by Organic and Inorganic Precursor Removal	Nicole Knight	Cost of Pollution and Options Evaluation – Case Study of Moreton Bay Regional Council Total Water Cycle Management Plan	Murray Hall
	Bioanalytical Assessment of the Formation Potential of Disinfection By-Products during Drinking Water Treatment	Beate Escher	Life-Cycle Perspectives for Total Water Cycle Planning	Joe Lane
	Panel Discussion / Questions		Assessing Urban Water Strategies for Total Water Cycle Management: an South East Queensland case	Magnus Moglia
10:30am	Morning tea			
11.00am	SESSION 9 - Wastewater Treatment (Chair – Cedric Robillot)			
	Guest Speaker: Factors that Mitigate the Formation of Emerging Disinfection By-Products in Recycled Water	Kathryn Linke, Curtin Uni (WQRA)	SESSION 10 - Water and Energy (Chair – Kirsten Shelly)	
	Ozonation and Biological Activated Carbon Filtration of Wastewater Treatment Plant Effluents	Julien Reungoat	Guest Speaker: Re-Imagining Sustainability	Michael Kane, ULDA
	Electrochemical Treatment of Reverse Osmosis Concentrate: Strategies to Minimise the Formation of Halogenated By-Products	Arseto Bagastyo	Implications of Resource-Efficient Technology on Peak Water Demand and Water-Related Energy Demand	Rodney Stewart
	Optimising Micropollutants Extraction for Analysis of Water Samples: Comparison of Different Solid Phase Materials and Liquid-Liquid Extraction	Fred Leusch	Water-Related Energy in a Brisbane Household – Modelling Analysis and Reduction Potential	Steve Kenway
	Panel Discussion / Questions		Decentralised Wastewater Systems: Robustness and Ecological Footprints	Meng Chong
12.45pm	Lunch			
1.30pm	SESSION 11 – Pathogen and Chemical Contaminants (Chair – Greg Jackson)			
	Reduction of Pharmaceutical Loads in Municipal Wastewater: would Onsite Treatment of Hospital Wastewater be Effective?	Kristell Le Corre	SESSION 12 – Resilient Water Future (Chair – Craig Walton)	
	Antibiotic Resistant Bacteria in Hospital Wastewater and STP; do they reach Surface Waters?	Mohammad Katouli	Guest Speaker: The Significance of Large Scale Seawater Desalination in Achieving Urban Water Security	Neil Palmer, NCEDA
	Faecal Indicators and Zoonotic Pathogens in Household Drinking Water Taps Fed from Rainwater Tanks	Warish Ahmed	Climate and Water over South East Queensland: Past and Future	Wenju Cai
	Panel Discussion / Questions		A Research Agenda for Water Smart Tropical and Sub-Tropical Cities and Towns	Sharon Biermann
3:00pm	Afternoon tea			
3:30pm	SESSION 13 – POSTERS - (Chair – Rodney Stewart)			
4.15pm	Awards for Best: Paper, Presentation and Poster – Don Begbie			
4.30pm	CLOSE			

OFFICIAL OPENING SPEAKER



Mark McArdle, MP

Minister for Energy and Water Supply, Queensland

Mark was appointed Minister for Energy and Water Supply on 3 April 2012. He is responsible for energy and water utilities, clean energy - including energy efficiency, and energy industry development and bulk water supply, distribution and retail arrangements. Born in Brisbane on 21 December 1956 he is married with a grown up family. Prior to election to Parliament as the Member for Caloundra on 7 February 2004, he was a practicing solicitor. Professional Qualifications - Solicitor of the Supreme Court, Queensland 1984; Graduate Diploma in Legal Practice (QIT, Brisbane) 1985; Solicitor of the High Court of Australia 1989; Approved Mediator, Queensland Law Society Inc. 1995; Certificate of Superannuation Management 1995; Accredited Family Law Specialist, Queensland Law Society Inc. 1999; Member, Federal Attorney-General Family Law Council Committee 2000; Member, Family Law Committee, Queensland Law Society 2000; Member, Queensland Law Society Family Law Specialist Accreditation Committee 2000. His interests include - gardening, reading, art, music, theatre, physical fitness, surfing, golf and squash.

KEYNOTE SPEAKER



James Cameron

Chief Executive Officer

National Water Commission, Canberra

James is the Chief Executive Officer of the National Water Commission. The Commission is Australia's national water reform agency. It provides independent assessments of the progress of Australia's federal and state governments in delivering on their reform commitments, as well as policy advice, advocacy and practical, financial support to help drive the water reform agenda. Mr Cameron joined the Commission in October 2009. Prior to his appointment, he had a broad ranging career in the Australian federal government across the industry, communications and cultural portfolios. In particular, he has held senior management roles responsible for the reform and regulation of Australia's telecommunications, broadcasting and media sectors, the development of the information and communications technology sector and managing Australia's arts and sport policy as well as its key cultural institutions. James holds a Bachelor of Arts (Honours) degree and a Graduate Diploma in Legal Studies.

GUEST SPEAKERS



Mark O'Donohue

Chief Executive Officer

Australian Water Recycling Centre of Excellence, Queensland

Mark is a researcher with a strong background in developing science policy partnerships for urban and environmental water. Prior to joining the Centre, he was the Director of Environmental Water Policy for the Department of Environment, Water, Heritage and the Arts in Canberra where he managed policy development and implementation for environmental water recovered as a result of the Commonwealth Government's \$3.1 billion investment through Water for the Future.



Rosemary Leonard

Senior Research Scientist

CSIRO, Social and Behavioural Sciences Group, Western Australia

Rosemary is a Senior Research Scientist, in the Social and Behavioural Sciences Group of CSIRO Ecosystem Sciences. She is also Associate Professor, School of Social Sciences, University of Western Sydney and Managing Editor of the refereed journal Third Sector Review. She is a recognised expert on social capital and leader of the social analysis component Optimal Water Resource Mix for Metropolitan Adelaide.



Helena Amaro

*Project Manager Water and Energy, Science and Technology
Sydney Water, New South Wales*

Helena is a Project Manager in Sydney Water's Science and Technology Division. She is responsible for research and development to better understand the role of smart cities in urban water management. She works in collaboration with other stakeholders to deliver research initiatives. Helena has been in Sydney Water for eight years. Her previous role was in the Strategic Directions team, project managing Sydney Water's Water Efficiency Strategy.



Ian Law

Principal, IBL Solutions, New South Wales

Ian is a Chemical Engineer with a Masters Degree in Public Health Engineering from the University of Cape Town in South Africa and is an Adjunct Professor at the University of Queensland. He was, until March 2003, CH2M HILL's Technology Director for South East Asia, Australia, and New Zealand and has since started his own business and trades as *IBL Solutions*. He has more than 30 years of experience in advanced wastewater and reuse projects in Southern Africa, South East Asia and Australia. He has published widely on the application of advanced reuse systems and the need to apply the concept of Total Water Management to all future water resource and wastewater planning and is actively promoting this concept in Australia. He currently serves on the Research Advisory Committees for the Australian Water Recycling Centre of Excellence (AWRCE) and the Urban Water Security Research Alliance (UWSRA).



Ray Beaton

*Manager, Water Resources Strategy
Yarra Valley Water, Victoria*

Ray is currently the Manager Water Resources Strategy at Yarra Valley Water. Yarra Valley Water provides water and sewerage services to 1.7 million people in the northern and eastern suburbs of Melbourne. Over a period of 33 years, Ray has held various positions in operations, planning, strategy and policy development for Water Utilities and Government. Ray has spent the last eight years managing many aspects associated with a prolonged drought, including strategic options for improving water security and managing demand. Ray's team has been instrumental in guiding Yarra Valley Water's investment in water efficiency which has led to benchmark low levels of water use and non-revenue water.



Rolando Fabris

*Senior Research Scientist, Water Treatment and Distribution
Australian Water Quality Centre, SA Water Corporation, South Australia*

Rolando joined SA Water in 1997 and has since developed experience in several areas pertaining to water treatment processes and characterisation of natural organic matter (NOM). Within the AWQC, he has had involvement in projects relating to online monitoring, distribution systems and coagulation processes, including alternative coagulants; multi-barrier treatments; identification of recalcitrant organic matter and membrane fouling components as well as disinfection by-product precursors. As part of this work, Rolando has achieved refinements in chromatographic and resin fractionation techniques to better characterise organic matter with applicability to drinking water, re-use and wastewater treatment optimisation. He has Honours in Chemical Technology and Bachelor of Applied Science, University of South Australia.



Tony Weber

Associate at BMT WBM Pty Ltd

Tony is an Associate at BMT WBM with over 23 years experience in the water industry. As the National Practice Leader for Water quality, Tony manages the water quality discipline across BMT WBM's Australian operations. Since joining BMT WBM, Tony has worked on a large range of water sensitive urban design, integrated water management, water quality and stormwater management projects. He joined BMT WBM after working as the Senior Program Officer with the Water Quality team in Brisbane City Council's Water Resources Program. In this role, Tony was responsible for Council's involvement in the development of MUSIC, the Cooperative Research Centre for Catchment Hydrology's urban stormwater decision support tool. In his current role, Tony provides expert advice, beta testing and peer review of catchment modelling activities across Australia and overseas and is currently assisting the eWater Cooperative Research Centre with the application of water quality modelling tools in the UK, Europe and South East Asia. Tony has a degree in chemistry from QUT and is a Visiting Fellow at the Australian National University's Fenner School.



Kathryn Linge

*Senior Research Fellow, Curtin Water Quality Research Centre
Department of Chemistry, Curtin University, Western Australia*

Kathryn is a Senior Research Fellow at the Curtin Water Quality Research Centre (CWQRC), Curtin University. She is an analytical chemist who has been working in the water industry since 2007, when she began leading CWQRC's contribution to a large collaborative project titled 'Characterising treated wastewater for drinking purposes following reverse osmosis treatment'. This was the first project in WA to study large-scale potable water reuse, and the first project in Australia to develop health guidelines for chemicals in recycled water. Since 2009 her research has focused on minimisation of disinfection by-products in both drinking and recycled water, and the efficacy of water recycling treatment processes such as reverse osmosis and advanced oxidation for chemical removal.



Michael Kane

*Assistant Director - Environment and Innovation
Urban Land Development Authority, Queensland*

Michael is Assistant Director Environment and Innovation at the Urban Land Development Authority (ULDA), a Queensland state government planning and development authority. This role involves Michael working with the ULDA's planning and development teams to develop and implement sustainable environmental and economic strategies and initiatives. Prior to this Michael was the Principal Advisor to CEO of the ULDA with responsibilities for strategic development during the period when the ULDA went from having responsibility for three urban development areas to fourteen in less than 12 months. Michael previously worked as an adviser for the Minister for Planning and Infrastructure in Western Australia with his area of responsibilities including WA Government's land development authorities and strategic planning.



Neil Palmer

*Chief Executive Officer
National Centre of Excellence in Desalination – Australia, Western Australia*

Neil has degrees in civil and public health engineering. His career spans 35 years in the Australian water industry, 20 years in Government and 15 years in the private sector. His professional experience includes the South Australian Engineering and Water Supply Department, the Fiji Public Works Department, Principal Wastewater Advisor with the South Australian Environment Protection Authority, Chief Engineer with United Utilities Australia and General Manager Technical Services with Osmoflo, the largest Australian desalination company. Neil is currently the CEO of the Australian National Centre of Excellence in Desalination, a partnership of 14 Australian universities and research organisations administering \$20m of Australian Government research funding. Neil is a Vice President of the Asia Pacific Desalination Association, a Director of the International Desalination Association and a member of the Institution of Engineers Australia. He is also a Life Member of the Australian Water Association and National Co-convenor of the AWA's Membranes and Desalination Specialist Network.

Guest Speaker Papers

Australia's Urban Water R&D – Needs and Capability Forum

O'Donohue, M.

Australian Water Recycling Centre of Excellence, Queensland

Overview

Australia's urban water research and development (R&D) investment is currently perceived as fragmented and confusing, and it is difficult to assess the current return on investment. Australia currently has numerous industries, state and commonwealth funded organisations investing in urban water R&D, and however, there is currently no statement of national needs and no coordination of the national investment approach.

The National Water Commission and other R&D brokers in the Australian Water R&D Coalition, such as the Australian Water Recycling Centre of Excellence, have agreed to implement a national needs and capability forum for urban water. The forum will address a national roadmap for existing urban water research, a national understanding of needs and priorities, and a national consensus on the most efficient mechanism to deliver on these needs.

The forum will assist Australia in becoming nationally efficient and internationally competitive with its investments in urban water R&D and will include senior government, industry and research representatives. This presentation will provide an overview of the national context that lead to the development of the forum concept and outline the expected approach and outcomes from the forum.

The Australian Water R&D Coalition:

Australian Water Recycling Centre of Excellence
[<http://www.australianwaterrecycling.com.au/coe/>]

Goyder Institute for Water Research
[<http://www.goyderinstitute.org/>]

National Centre of Excellence in Desalination
[<http://desalination.edu.au/>]

National Water Commission
[<http://www.nwc.gov.au/>]

Smart Water Fund
[<http://www.smartwater.com.au/>]

Water Quality Research Australia (WQRA)
[<http://www.wqra.com.au/>]

Water Services Association of Australia (WSAA)
[<https://www.wsaa.asn.au/>]

National Centre for Groundwater Research and Training
[<http://www.groundwater.com.au/>]

Urban Water Security Research Alliance
[<http://www.urbanwateralliance.org.au/>]

Assessment of Alternative Water Options in Adelaide: The MARSUO and Optimal Water Resource Mix Projects

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Summary

Adelaide will need to diversify and integrate its water sources to ensure its future water security. The present research found that stormwater and groundwater were more acceptable supply alternatives than recycled waste water and desalinated water. Further, community education markedly changed opinions of water sources and transparent processes and trust in the treatment and monitoring may be more important than the choice of water sources.

Keywords

Alternative water sources, stormwater, aquifer-recharge, community education, transparency.

Introduction

Securing Adelaide's water supply is a major challenge with inadequate natural rainfall and the Murray River under increasing pressure. The city will need to diversify and integrate its water sources for future water security. However, public attitudes can be a major obstacle to introducing alternative water sources, depending on the source and intended use of the water (Nancarrow *et al.*, 2010). Extensive research has been conducted on recycled water and identified numerous factors that can influence public attitudes, such as health issues, functioning and maintenance, equity and fairness, transparency and support for the decision making (Alexander 2010). However, to address Adelaide's water shortage, all potential water sources need to be investigated. Stormwater is already entering the drinking water supplies in some regional centres, either unintentionally as in Mount Gambier, SA, where stormwater recharges the local aquifer and Blue Lake, the main local source of the cities' drinking water, or intentionally as in Orange, NSW, where stormwater is diverted via wetlands, to a batch treatment plant, pumped into a reservoir and treated to drinking water quality (Dillon *et al.*, 2011).

Social analysis for the Managed Aquifer Recharge and Stormwater Use Options project (MARSUO) of the National Water Commission and Goyder Institute was previously conducted by Dr Kim Alexander. The social research explored the uses of stormwater through attitudinal research methods including focus groups a brief pre and post survey (N=36) and an Adelaide-wide survey (N=1,043) to examine participants' attitudes to stormwater treatment options, and aquifer storage. The social research investigated three treatment options.

Option 1 was wetland → aquifer → treatment plant → non-drinking.

Option 2 was wetland → aquifer → treatment plant → reservoir → treatment plant → drinking.

Option 3 was wetland → aquifer → treatment plant → drinking.

The focus groups presented treatment options and provided an opportunity to question scientists. Nineteen of the focus group attendees who gave email addresses responded to selected post survey questions. In the Adelaide-wide survey, treatment options were depicted pictorially, where questions related to the acceptability of stormwater relative to other water sources and treated in aquifers and wetlands prior to use. This paper focuses on two issues, differences between informed and uninformed responses and a comparison of stormwater with water sources.

Results

Comparison of Informed and Uninformed Responses

Perhaps the most important theme to emerge from the results was the difference between informed and uninformed responses. At the beginning of the focus groups 29% said they knew nothing about groundwater replenishment. The information gained from the presentation in the focus groups was considered very informative or extremely informative by 80% of the focus group participants and the attendees agreed that it had influenced their opinions about stormwater reuse. These differences emerged in three ways; the greater level of acceptance of ground water, the contrast between survey and focus group responses, and the lack of any distinction between Options 2 and 3.

1. Greater Acceptance of Groundwater

When focus group participants were initially asked if they supported using stormwater for groundwater replenishment, 34% strongly supported the use, though 43% had not formed an opinion. The post workshop data revealed that most participants (93%) were more likely to agree that stormwater could be used to refill groundwater after attending the workshop ($n=19$, Mean difference = 0.53, Std Deviation = 0.84, $p < 0.05$).

2. Focus Groups versus Adelaide-wide Survey

- By the end of the focus groups the idea of adding stormwater to the drinking water supply (Options 2 and 3) was supported or strongly supported by 97% of participants, with only 3% yet to form an opinion. In the Adelaide-wide survey only 70% supported or strongly supported Options 2 and 3 and approximately 20% thought that maybe they would protest against these options.
- In the focus groups Option 1 (third pipe for non-potable water) was only supported or strongly supported by 25% with 65% of participants undecided. They expressed concerns about the expense of the third pipe system and the lack of equity as not all residents would have equal access. In contrast, in the Adelaide-wide survey, this was the preferred option with 80% supporting or strongly supporting this option.

It is significant that 97% of the focus group participants had at least moderate confidence in the water utility, SA Water, to effectively manage the treatment and monitoring of the water and 94% had at least moderate confidence in their transparency. However, it was also seen as desirable for independent scientists to be involved in the monitoring of augmented water supplies.

3. Failure to Distinguish between Options 2 and 3

In both the focus groups and Adelaide-wide survey, there appeared to be no differentiation in participants' perspective between the potable Options 2 and 3. This contrasts with the view of the scientists who strongly support the benefits of multiple barriers. Thus, it appears that there were limits to the level of understanding achieved in the focus group presentation and discussion.

Comparison of Water Sources

Focus group discussion did not canvass opinions on a wide range of water options but participants did express some of the usual aversion to recycled wastewater. In the Adelaide-wide survey respondents were asked how important it was for the Adelaide community to be able to rely upon the proposed water supply options under drought and non-drought conditions. Figure 1 presents the mean importance rating for different water supply options split under drought and non-drought conditions. In general, respondents indicated that all water supply options were important, but more so during drought conditions. The two most preferred options, regardless of whether it was drought or non-drought condition, were water from rainwater and from Mt Lofty reservoir. Water from the desalination plant was seen as less important when compared with other supply options under both drought and non-drought situations.

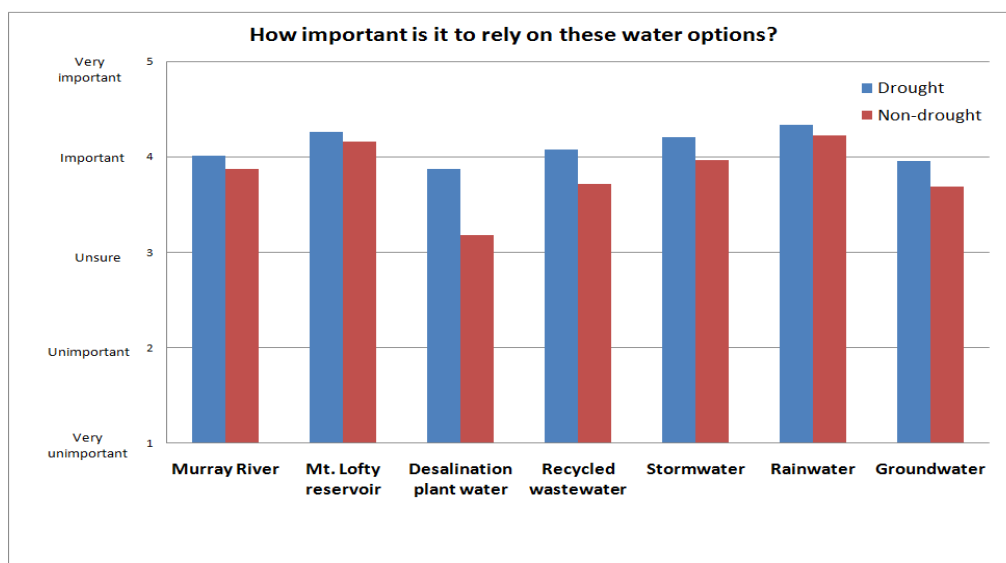


Figure 1. Relative importance of water source during drought and non-drought conditions.

Significance and Impact

Solving Adelaide's water problems is a long term project which requires the creative integration of a number of water sources. This is the focus of a new Goyder project lead by Dr Maheepala from CSIRO with the author leading the social analysis. Results from Alexander's social research suggest that stormwater and managed aquifer recharge will be considered by the community to be a possible water supply option in the future, particularly under drought conditions. Community engagement to inform residents of Adelaide is paramount and hence creative interactions need to be developed as in Singapore, where there are public displays, working models of the recycled water processes and guided interpreted tours. Regardless of the future water options chosen to augment water supplies, the community require reassurance that treatment and monitoring systems are working effectively and will not introduce health problems after ingestion. Trust in processes will increase with greater transparency and the presence of independent water quality monitors. How the water supply is managed may prove to be more important than the water sources that are introduced to mitigate future water supply problems.

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Rainwater Tank Study of New Homes

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Introduction

The environmental sustainability of rainwater tanks as a water solution for large cities depends on their ability to save water while using minimal energy. In the largest detailed rainwater tank monitoring study in Australia to date, Sydney Water remotely monitored rainwater tank water and energy use for more than 12 months for 52 real life installations at one-minute intervals. The study's principal objectives were to confirm that rainwater tanks in real life installations save water as expected and, if they were not, to identify opportunities to further increase water savings and reduce their pumping energy use.

Methodology

Monitored households were newly built homes, on average two years old, that had installed rainwater tanks as part of compliance with NSW's Building Sustainability Index (BASIX) water regulation requirements. These households were located all over the Sydney basin. The sample was recruited using a phone survey and a \$50 voucher was provided to the participating households to encourage program take up. Data were collected for 52 households between July 2009 and June 2010. Meters were installed to measure and log at one-minute intervals for 12 months.

Results

The monitored households saved an average of 21% of their total household water demand due to the rainwater tanks, which equates to around 38 kL per household per year of water savings. The maximum achieved savings was 96 kL per household per year. Total average household demand from the rainwater tanks, which includes rainwater use and top-up, was 59 kL per household per year. Energy use from the rainwater tank was relatively low at a household level, with a median energy use of 62 kWh per household per year (ie, approximately \$15 a year). The median energy intensity, which is the energy used per unit of rainwater used, was 1.48 kWh/kL.

The study identified and quantified a number of practical options to improve the efficiency of rainwater tanks by ensuring water savings are maximised and energy is used efficiently.

Conclusion

An environmentally sustainable tank system is one that is easy for householders to use and maintain if properly configured. It needs to be configured to maximise the potential for rainwater use by connecting as many end uses as possible and connecting as much roof area as possible. The tank needs to be sized according to these end uses. Using pumps that are energy efficient at low flow rates, possibly complemented with pressure tanks, can reduce energy use. Simple information could also be made available to customers to reduce the risk of easy losses (eg, alarms about leaks that increase energy use).

This type of system would increase average water savings and reduce rainwater tank pump's energy intensity to better than surface water supplies in some cases. Even with these improvements, rainwater tank installations would remain a relatively expensive water source with their levelised cost typically between \$4 and \$8 a kL.

National Perspectives on Potable Recycling – Barriers and Costs

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Summary

There is growing realisation within Australia and overseas that potable reuse is an option that must be seriously considered when considering options for future sustainable water supplies. Bold decisions must be taken today in order to ensure that future generations are not adversely impacted by inadequate decisions.

Keywords

Political will, potable reuse, community perceptions, stigma, sustainability.

Introduction

Water recycling is playing a significant role in the diversification of Australia's water supplies and we have seen tremendous growth in its application over the last years. There is still much to be done, however, if we are to reap the full benefit of the advanced forms of this option for water management into the future, particularly with the expected growth in our population.

As we strive to develop sustainable supplies for our cities into the future, there is increasing pressure to consider all options and in particular, the potable reuse (PR) option.

The Current Position

Australia has Guidelines (in place since May 2008) for the Augmentation of Drinking Water Supplies with water reclaimed from municipal effluents – the Potable Reuse option – with these guidelines being produced by an eminently qualified working party of scientists and health regulators, subjected to international and national refereeing and subsequently accepted by the then Environment Protection and Heritage Council, the National Health and Medical Research Council and the then Natural Resource Management Ministerial Council (EPHC 2009).

The Water Services Association of Australia (WSAA), representing Australia's major water utilities, in its Occasional Paper 25 of July 2010 (WSAA 2010) stresses the need for a diversified portfolio of water supply options to meet the future water needs of an increased population. It notes *'It is expected that the development of a diverse portfolio of water supply options including recycled water for non-drinking and drinking purposes, desalination, rural to urban water trading, rainwater tanks, groundwater, stormwater and dams will be required to mitigate the risks associated with population growth and climate change. There should not be any blocks to the different sources of supply and each case should be examined on its merits.'*

Further, WSAA notes *'It is imperative that there are no policy blocks in place that would preclude a source of water being considered for inclusion in a diverse portfolio of water supply options.'*

The National Water Commission also strongly supports consideration of water recycling on its merits as an option to be reviewed when determining future water supply sources, and notes that *water recycling – including for drinking purposes - can provide a significantly greater proportion of Australia's future urban water supplies. The Commission recognises there are intrinsic risks associated with recycled water. However, in our judgement, advances in science and improved regulatory arrangements mean that such risks can now be managed to levels of safety that are equivalent with other supply sources* (NWC 2010).

How is it then, that despite the above all stressing the importance of considering *all* water supply options, there are still two States (South Australia and Victoria) that have policies in place precluding the potable reuse option from consideration?

It is suggested that these policies are driven by a lack of 'political will' which in turn results from advice based on sensationalised media reports and/or perceptions of community concerns. This lack of 'political will' is causing a growing frustration in the industry as it strives to ensure that future water supplies are developed on a sustainable basis, very much as per the recommendations put forward by WSAA and the NWC. Water professionals in Australia have shown in papers published in *Water* and presented elsewhere at international and local conferences, that potable reuse, and indeed direct potable reuse, is a safe and sustainable water supply option that must be considered in the development of future water supply portfolios.

Potable reuse has recently been enshrined in legislation in California in that a Bill was passed by the Californian State Senate in October 2010 instructing that State's Department of Public Health to complete indirect potable reuse regulations and evaluate direct potable reuse. California thus views potable reuse as a viable option.

We must remove this divide between water supply reality and 'political will' if we are to ensure cost effective and sustainable water supplies into the future. How can this be achieved?

The Way Forward

Given that 'political will' is driven by perceptions of community attitudes – as evidenced by the Western Corridor decision in Queensland – there would appear to be a clear need to focus on the community at large as, if they accept the advantages of including potable reuse into the mix of options, the politicians will surely follow. It was Mahatma Ghandi who said "*If the people lead, the leaders will follow*".

While many investigations into community attitudes towards water recycling for potable purposes have been undertaken only a few have placed water recycling into context or note that water is used and reused around the world by downstream communities. There has been little to no research on whether a different presentation of water use and reuse can overcome the issues related to stigma and disgust created by a typical linear project scenario that focuses on the source - wastewater.

Groundbreaking research in this area has recently been reported on by the National Water Commission in Australia (NWC 2011) and the WateReuse Research Foundation in the US (WRF 2012). It clearly shows that the stigma can be overcome by using appropriate terminology, by putting water recycling into context and by ensuring the community is familiar with the urban water cycle of use and reuse.

The Australian Water Recycling Centre of Excellence (AWRCE) has taken up this challenge and has recently commissioned a project that will address one of its four Goals, Goal 3 – that of overcoming the barriers to *reclaimed water being viewed as an acceptable 'alternative water' for augmenting drinking water supplies*. This project has the objective of developing a National Demonstration, Education and Engagement Program that supports successful public engagement and addresses stakeholder concerns through the provision of contemporary scientific information on the urban water cycle and potable reuse. It will involve leading edge methods of communication to overcome known social barriers to acceptance and adoption.

This project also covers research into Governance and Pricing Practices with the aim being to identify the impediments to investment in potable recycling, compared to alternative water supplies.

Conclusions

Carpe Diem.

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Water Efficiency – A Dry Topic

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Summary

Various drivers for water utilities to manage demand are presented and discussed. An investment strategy for water efficiency that considers long term benefits and costs as well as the current supply situation is presented.

Keywords

Water, efficiency, supply, customer, economics, governance.

Introduction

The contents of this paper may assist water industry professionals in presenting a case for investment in water efficiency.

Water Conservation and Water Efficiency

For the purposes of this discussion let me attempt to clarify the difference between water conservation and water efficiency.

Water conservation is saving water with the aim of protecting supply security. During the drought, many of us faced a risk of running out of water, and we implemented numerous measures to **conserve water**. Many of these measures, particularly water restrictions, had a direct impact on customer behaviour and welfare, eg, limited garden watering, no car washing at home. Figure 1 shows actual storage levels and estimated storage levels without water conservation savings through Melbourne's drought.

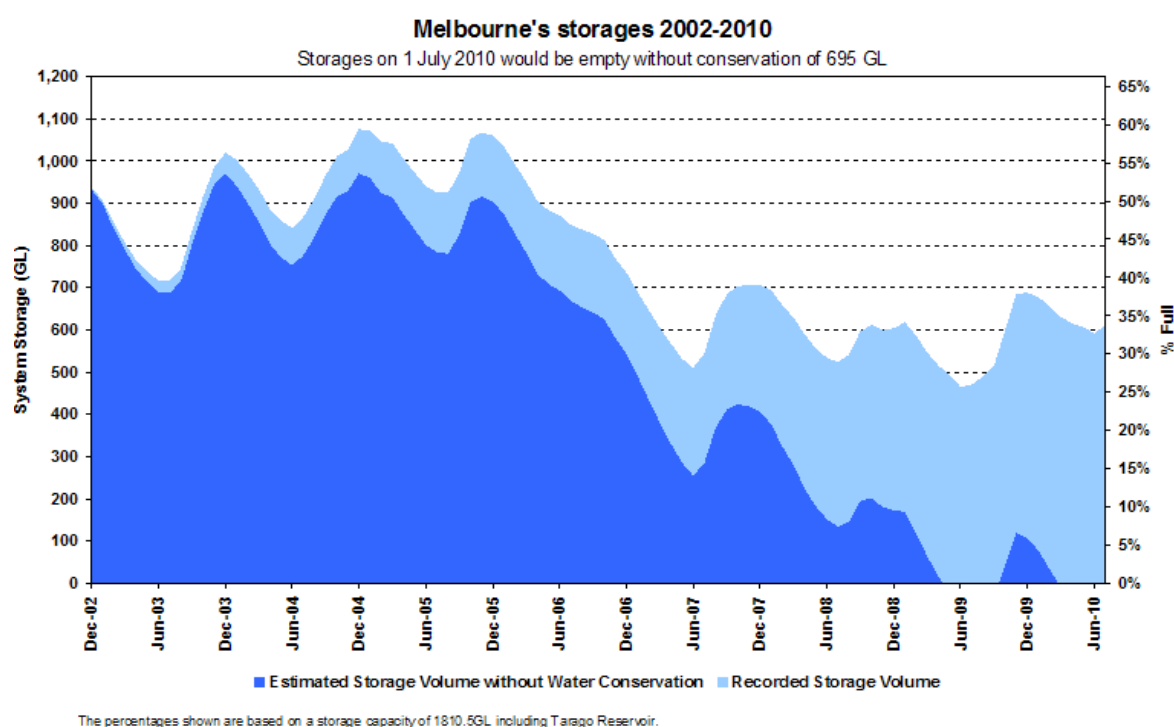


Figure 1. Melbourne Water Storages 2002-2010.

In the post drought era with rising storage levels we are less concerned about the risk of running out of water and more concerned about providing the water customers need to live and do business efficiently. In this context water efficiency is any measure that reduces the amount of water used per unit of a given activity, **without compromising** the achievement of the value expected from that activity.

Why Invest in Water Efficiency?

But why should a water utility invest in water efficiency? In fact, there are some drivers that on their own would encourage a water utility to increase water use by customers. For example, in Victoria water utilities receive their revenue on the basis of a return on capital, where a fair return as a percentage of investment is determined by an independent Economic Regulator, the Essential Services Commission (ESC).

Over half the revenue requirement is recovered through variable charges. This revenue requirement is divided by the forecast water sales to determine a price per kL of water.

Two interesting drivers emerge:

1. The more capital we have invested (eg, pipes, pumps treatment plants, etc) the more revenue we are entitled to receive.
2. Once the price per kL is set, the more water we sell the more revenue we make, or alternatively, the less we sell the less revenue we make.

Do we have a governance /incentive issue? Maybe.

To address these issues we have many arrangements including but not limited to the following:

1. An Act of Parliament and a licence that guides what we provide, who we service with, etc.
2. Government appointed Board.
3. Statement of Obligations from the Minister. Amongst other things this 15 page document requires us to prepare:
 - A Water Plan (being prepared for five years):
 - For the purpose of the ESC to make a decision with respect to prices.
 - Includes outcomes with respect to service and supply, future demands.
 - How we propose to deliver.
 - Revenue requirements and proposed price.
 - We must undertake effective consultation with customers on matters of concern to the customer.
 - A water supply demand strategy (prepared every five years):
 - Best mix of measures to maintain balance between supply and demand.
 - Includes targets to guide investment in water efficiency and water re use.
 - Consider:
 - 50 year outlook.
 - Total water cycle.
 - Social, environmental and economic costs and benefits.
 - Scenario based planning, adaptive management.
 - Risk and uncertainty.
4. The ESC is:
 - Victorian Independent economic regulator for electricity, gas, ports, water and sewer services.
 - Objective - to pursue the “long term interests of consumers”.

In my opinion, these arrangements do a pretty good job of managing issue number 1 - the level of investment we make in infrastructure.

Issue number 2 - demand forecasts and pricing - is subject to not only forecasting uncertainty but the vagaries of weather. Yarra Valley Water has proposed to our regulator that rather than imposing a price cap the regulator should allow a revenue cap.

All of these requirements and processes are designed to ensure the customer gets value from the payments they make to us, the water utility.

So if our job is to deliver value, what role does water efficiency play?

Well, what I can tell you is that benevolence arguments won't cut the mustard! Managing Directors and Economic Regulators want us to demonstrate how water efficiency translates to the bottom line for the community and customers.

In Melbourne and at Yarra Valley Water we've undertaken extensive research and investigation in the development of our next water supply demand strategy. Analysis undertaken during the development of the draft strategy (yet to be released) has identified reasons that support a long-term role for water efficiency programs in Melbourne. These include:

1. The community expects a continued focus on water efficiency:
"Many felt that the need to save water has now been embedded in the Melburnian psyche and would like to see this behaviour change become the norm. Households widely reported that they considered that they would be using about the same or a bit less water into the future."
 (Source GA Research for Melbourne's Water Supply Demand Strategy)
2. Customers benefit from water efficiency;
3. The environment benefits from water efficiency;
4. The water supply system benefits from water efficiency.

Two examples of how water efficiency can lower long term water utility expenditure are provided below.

Example 1 - Benefits of Deferring Major Supply Augmentations

Table 1 shows the results of a 50 year net present value (NPV) calculation of Yarra Valley Water's revenue requirement (per customer) for three water conservation expenditure. It can be seen that over the long term customers pay lower bills if a high level of investment in water efficiency results in low water demand.

Table 1.

Water Efficiency Investment and Demand Scenario	50yr NPV per customer \$
High Demand (no water efficiency expenditure)	\$435+-50
Baseline Demand (moderate water efficiency expenditure)	\$244+-30
Low Demand (high water efficiency expenditure)	\$197+-2

Notes:
 The lowest NPV is the best pricing outcome for YVW customers
 Nuances of current storage levels ignored.

Example 2 - Benefits of Reducing Distribution System Sizes

Peak day demand has reduced significantly over the last 30 years and significantly since our design standards were reviewed in 2006. Analysis of Yarra Valley Water's major growth area indicates that if we design and build for a 30% reduction in peak day demand, we will achieve a further 9% reduction in infrastructure costs associated with potable and recycled water supply systems.

A key message here is that customers don't see the benefit of efficiency savings unless we alter our design assumptions/standards.

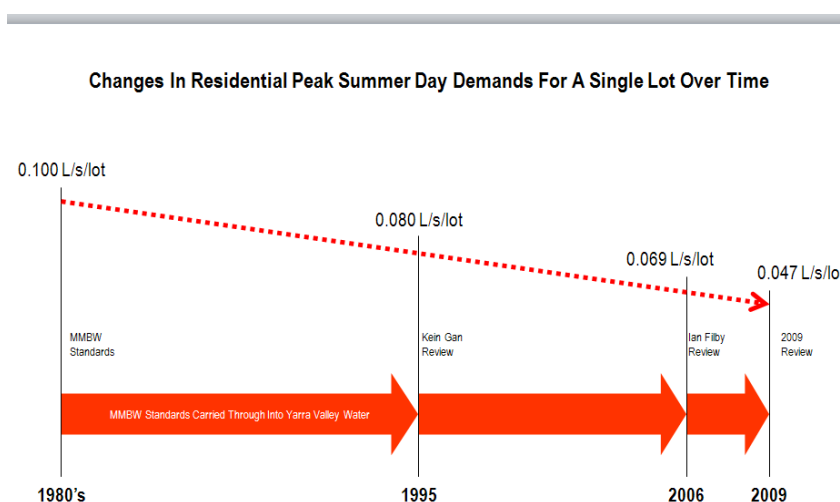


Figure 2. Changes in Design Standard Over Time.

A New Strategic Direction for Water Efficiency

In Melbourne the context for water efficiency has changed substantially with good inflows into storages and the impending completion of the Wonthaggi desalination plant in 2013. As a result of the changing context, Yarra Vally Water has adopted a new strategic direction to guide water efficiency expenditure in future years. This is summarised below and also shown in Figure 3.

1. We will invest in water efficiency, up to the amount that it is economic to do so, ie, where the cost is less than the alternatives being considered.
2. We will maintain a baseline program of customer engagement to provide a platform to launch more aggressive programs as needed, in accordance with Melbourne's annual Water Security Outlook (see below).
3. We will seek to manage water use within an efficient range. This is initially set at 160 L/person/day for residential water use, consistent with the achievement over the drought with a small allowance for bounce back. The efficient range will be regularly reviewed and is expected to decrease with improvements in appliance technology or changes in regulation.
4. Our demand forecast should be consistent with the efficient use range. If water use is above the approved forecast we will have extra revenue which can be used to educate and provide solutions to assist customers to achieve an efficient level of use. We may also be able to lower prices. If water use is below the approved forecast then the level of investment can be cut back to a baseline program of education, advice and assistance and we may also seek catch-up revenue in the next Water Plan.
5. Our water efficiency expenditure will be guided by our Community Cost Model.

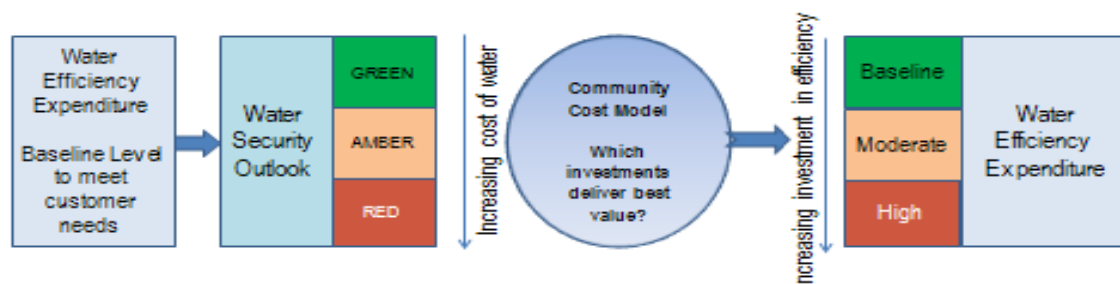


Figure 3. New Strategic Direction for Water Efficiency.

Baseline Investment

The baseline program should be scalable such that it is able to be expanded quickly to assist in times of water shortage and to manage potential for bounce back in demand. Contingency programs should also be developed to enable a wider service offering during times of water shortage or to manage the potential for bounce back in demand.

The baseline programs focus on:

- continuing (or ongoing) engagement with the community to encourage continued efficient water use behaviours;
- addressing the larger residential water uses including gardens, showers, washing machines and toilets;
- partnering with high water-using businesses to use water efficiently;
- targeting key non-residential water uses such as cooling towers, evaporative coolers, fire protection systems, restaurant kitchens, open space irrigation systems and aquatic centres; and
- actively engaging in the development and implementation of regulated water efficiency measures such as the Water Efficiency Labelling and Standards Scheme and building regulations.

The community benefit cost model is used to assist decision making such as the level of funding contributed by the water utility and directly benefitting customers.

Conclusions

The case for investment in water efficiency needs to and can be made on social, environmental and economic basis. Incentives that could encourage water utilities to encourage water use can be mitigated by the introduction of a revenue cap on water utilities. Investment strategies for water efficiency not only need to consider the long term benefits and costs, they also need to also consider the current supply situation, and developed accordingly.

Characterisation of Disinfection By-Product Formation for Water Quality Management

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Summary

This paper describes the first stage of a project to establish whether manipulation of water treatment plant operations would be sufficient to control the formation of disinfection by-products (DBPs) and their associated health effects, completed using one water source. Specifically, this related to optimisation of conventional treatment processes and disinfection strategy. A number of interesting findings with this single high-bromide water source have led to theories that require testing with other varied water sources to establish their general applicability.

Keywords

Disinfection by-products, trihalomethanes (THM), haloacetic acids (HAA), natural organic matter (NOM), water treatment.

Introduction

Australia is subject to extremes of climate, the extent and variability of which is predicted to increase due to the impact of climate change. The recent extensive drought in most of Australia resulted in increased salinity of natural water sources and this, together with the resultant increased use of alternative water sources such as desalinated water, has generally led to an increase in bromide concentration in drinking water supplies. Subsequent heavy rainfall has also produced significantly increased concentration of natural organics in source waters in certain areas of the country including Victoria and South Australia. The increase in natural organics and bromide reduces treatment effectiveness and increases disinfectant consumption and formation of DBPs. These water quality changes will therefore complicate effective treatment and disinfection of water supplies as utilities also struggle to simultaneously comply with the Australian Drinking Water Guidelines (ADWG) and/or state and contractual regulations.

Whilst the overall benefits of disinfection are well established, of particular concern is the formation and control of DBPs that can result in adverse health effects (NCI, 1976). Epidemiological studies indicate that the risk of developing bladder cancer is approximately two times higher in populations that use chlorinated water compared with populations drinking other types of water (Villanueva *et al.*, 2003). Other cancer types, as well as developmental effects, have also been associated with the use of chlorinated water (Hrudey, 2009). When multiplied over the very large population that use chlorinated water, this translates to a significant cancer burden that necessitates large and increasing health expenditure. It is therefore important to optimise water chlorination to maintain effective disinfection while reducing the formation of harmful DBPs. Animal studies indicate that the currently regulated DBPs are not potent enough to cause the observed human effects, so research is now evaluating alternative DBPs with higher potency (Hrudey, 2009). The brominated forms of various DBPs have been shown to be more potent genotoxins than their chlorinated counterparts. The relevance of this is that many of these compounds do not, as yet, have limit values in the ADWG and therefore are not monitored by water utilities, regardless of their potential impacts on the overall risks associated with long term chlorinated water consumption.

The key components controlling formation of DBPs are natural organic matter (NOM), bromide concentration, chlorine dose, pH and temperature. As bromide is heavier than chloride, increased bromide also results in increased concentration (on a weight basis) of the key DBPs - total trihalomethanes (THMs) and haloacetic acids (HAAs). The most common treatment process utilised by most utilities to remove NOM is conventional treatment employing coagulation, filtration and chlorine disinfection. Within previous projects undertaken by the project team, a range of effective techniques for characterising NOM were identified and used to better understand the coagulation process. Data obtained within these projects indicates that particular components of NOM may be key factors in the formation of chlorinated DBPs.

This project extends the previous work to identify the key factors controlling formation of brominated THMs and other DBPs to assess the toxicological impact on the water (Li *et al.*, 2011). Other DBPs measured included the formation of all nine haloacetic acids, the formation of total halogenated DBPs by absorbable organic halogens (AOX) and using bioanalytical tests to assess the cyto- and geno-toxicity of the water after disinfection. The extent that their formation may be reduced was evaluated by manipulating operational variables under a range of conditions. This included altering the disinfection process through dual-stage chlorination, simulating increasing bromide, as well as manipulating water quality using conventional treatment processes. When both project stages are complete, this may enable some identification of components of NOM that result in formation of DBPs and to

determine whether there is a particular concentration of bromide above which NOM removal becomes irrelevant to DBP formation. There is very little data available in Australia on the extent of HAA formation, apart from the three chlorinated HAAs that are currently in the ADWG. Once applied to a number of different water sources, this project should provide the opportunity to determine some generalised effects that changing conditions will have on DBP formation and the potential health implications that may result. The following describes the initial findings from Stage 1 of the project where the techniques were applied to a single high bromide surface water.

Results

In the first stage of the investigation (WQRA project 1041) using a Western Australian water source, a number of interesting findings were produced that require testing with other water sources to establish whether they are site specific or can be more generally applied to all water sources and treatment plant conditions. Results of Stage 1 showed that coagulation is an effective means of significantly reducing DBP formation and toxicity following chlorination. Significantly increasing alum dose above that required to achieve maximum DOC removal did not provide any additional benefits in terms of DBP or toxicity mitigation. Optimising coagulant dose was also identified as more critical in summer when small changes in coagulant dose significantly impacted the extent of DBP formation at higher average temperatures. Toxicity testing confirmed that at natural bromide concentrations, the treated and disinfected waters had low potential for acute health effects and could not be directly correlated to any of the measured DBPs.

HAA5 (US-EPA) and HAA3 (ADWG) only represented a minor portion of the total HAA9. Critically, the lowest correlation between HAA3 and HAA9 occurred within the optimum alum dose range where a treatment plant utilising enhanced coagulation strategies will operate.

Increasing the bromide concentration (ratio Br:DOC) resulted in increasing formation of all the brominated DBPs, with higher temperatures enhancing this effect. Higher bromide concentrations also seemed to favour THM formation at the expense of the unknown DBPs (as measured by AOX). Total HAA concentrations did not increase as much with increasing bromide concentration as THMs although the species distribution also changed to favour more brominated compounds. Toxicity also increased with increased bromide concentration and the higher proportion of brominated DBPs. The most significant correlations were found with the bromine-substituted haloacetic acids. THMs showed weak toxicity correlations, despite the fact that they constituted a large proportion of the total halogenated DBPs. Many water sources have lower bromide concentrations and lower bromide:DOC ratios so one of the interesting questions that Stage 2 needs to address is whether there is a bromide concentration below which no further reduction in DBP formation can be achieved.

Finally, treated water was dosed with chlorine in two stages to simulate a booster chlorination strategy and compared with the earlier single dose tests. The addition of less chlorine in the primary dose noticeably reduced both THM, and especially HAA formation. Final concentrations were up to 60µg/L less using the booster chlorination strategy in this water. Genotoxicity was also reduced with the booster chlorination strategy suggesting a possible link with decrease in formation of DBPs. This is a key conclusion and booster chlorination needs to be investigated further in waters of lower bromide concentration to determine if this is a possible strategy for reducing not only DBP formation but overall toxicity.

Conclusions

The interesting outcomes from this project to date may have significant potential in assisting water utilities to manage their treatment to minimise DBP formation and also to address future changes in DBP regulations with no additional capital investment. However, the application of the test strategy to only one water source limits the application of the outcomes more widely. Stage 2 (WQRA project 1048) is currently in preparation to test two additional water sources and additional project partners are being sought to help support this important work.

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Collaboration in Developing Total Water Cycle Management Planning – Critical for Success

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Summary

In recent years there has been a renewed focus on integration of all elements of the water cycle across Australia, especially with the extremes of flood and drought placing pressure on all facets of water management. This has resulted in the implementation of the Total Water Cycle Management Planning process in Queensland for local governments and significant development areas. These plans form the basis for planning water cycle infrastructure for the next 20 years and therefore need to be developed with the best knowledge, science and engineering to adequately integrate all elements of the water cycle. Fundamentally though, the most critical element for successful planning is collaboration. This collaboration is not just between the plan developers and their clients, but researchers, water providers, Council staff, Councillors, state agencies and other consultants. Without this, it is difficult to provide a detailed understanding of key parts of the water cycle and communicate the planning results effectively to key stakeholders.

Keywords

Collaboration, planning, water, cycle, management, local government.

Introduction

The planning process has been implemented in Queensland through a regulatory approach requiring Total Water Cycle Management Plans (TWCMP) being prepared within certain time limits as noted earlier. This has resulted in several different approaches by local authorities, with some engaging in detailed strategic planning followed by in-depth technical studies to resolve gaps in knowledge for some planning elements, whereas others have taken a broad strategic framework approach, mostly completed “in-house”, with some minor additional studies being completed as secondary exercises. Guidelines have been prepared for authorities to assist with plan preparation, however, these guidelines focus on the plan preparation process, rather than a detailed assessment of how to deal with the more complex issues of water cycle management that are being encountered.

In its truest form, Total Water Cycle Management (TWCMP) recognises the interrelationships between the human uses of water and its role in the environment, using the key principles of:

- Natural Cycles - minimising the alteration to natural flow and water quality regimes;
- Sustainable Limits - ensuring that the volume of water extracted from a source is sustainable for the community and the environment;
- Water Conservation - minimising water use and losses by reducing demand and by maximising efficient use and reuse;
- Diversity in New Supplies - considering all potential sources of water when new supplies are needed including reusing water and stormwater;
- Water Quality - managing the water cycle at all phases to preserve water quality for the community and the environment; and
- Water Quality ‘Fit for Purpose’ - aiming for water supply quality to be no better than is required for the proposed use, ie, not supplying potable water for uses that do not require potable quality (DIP, 2009).

In our approach to undertaking the development of both a Total Water Cycle Management Strategy for Moreton Bay Regional Council (MBRC) and the resultant detailed TWCMP, our primary planning vehicle has been an understanding and elucidation of the key drivers for water cycle management in a particular region complemented by a full quantification (where possible) of as many elements of the water cycle for which data is available. This was then formulated into a series of scenarios to address gaps identified and to minimise true environmental, economic and social costs in the delivery of water cycle infrastructure and assets through initial multi-criteria analysis and detailed numeric modelling. While this quantitative approach was extremely useful in developing a detailed understanding of the issues, impacts and management approaches, it was only possible through working collaboratively with Council and Unitywater staff, researchers from UWSRA and with other agencies such as the Queensland Water Commission.

Results

The most important part of the development of the plan was the involvement of key MBRC and Unitywater staff. Without their collaborative involvement, it would have been extremely difficult to identify repositories of corporate knowledge, existing literature and documents and most importantly, data sources. Several key individuals also were able to use their existing networks to obtain further information and coordinate consultation with other staff, including senior management. Of note was the need to secure “buy-in” early in the project with both senior management in Council and Unitywater, and also the Councillors. This helped create continuing support for the initiatives proposed, but we were also then able to use the science collaboration with UWSRA researchers to inform this discussion.

Of most value in this process was the use of the extended life cycle costing analysis (Hall 2011) for showing the tradeoffs between costs and efficacy. The current requirements on water retailers and distributors within Queensland are that they must demonstrate that the solutions adopted for their customers are those that are shown to be “least cost”. In developing the TWCMP, this has typically come down to a basic cost-effectiveness evaluation, that is, which solution delivers the required objectives or standards of service for the least overall financial cost. This is a simplistic view of what true least cost planning approaches should deliver. If the approach does not adequately address all externalities, then some costs may not be properly accounted for (see Lane *et al.*, 2010). In addition, if the benefits, both tangible and intangible, of such approaches are not accounted for, true cost-effectiveness may not be able to be delivered. Also, it may be that the cost burden for proposed management actions may be too high for the community to bear in order to achieve the required TWCM objectives.

In addressing this, we have found that when approaches of Extended Cost-Effectiveness Analysis (Hall 2011) are applied in a decision making context, it can demonstrate to decision makers how far towards achieving objectives they may get for a certain level of investment. This has been well received by political decision makers especially where benefits may not be adequately quantified. An example of the results of such an approach is to develop specific cost curves shown below.

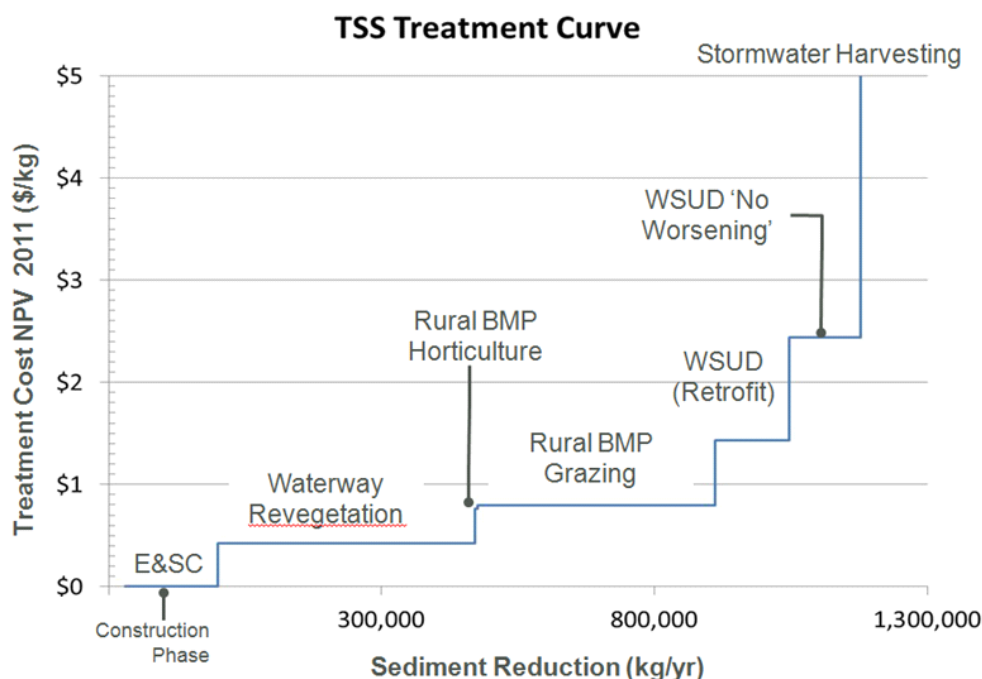


Figure 1. Extended Cost-Effectiveness Analysis Cost Curve (BMT WBM 2012).

These cost curves can also show a specific target in terms of a pollutant load reduction or potable supply demand reduction, however, we have also found that sometimes those targets are beyond that which can be adequately achieved with current technology or within current resources availability. These were only possible with the quantitative modelling and collaborative approaches. What is concerning is that these approaches are not standardised in guidelines for preparation of TWCMPs, leading to very different planning approaches across SEQ.

Conclusions

The TWCMP process was only possible through strong collaborative approaches across government and research agencies. This allowed the development of a comprehensive, but well understood plan that is now ready for implementation. Whether this approach is adopted in other TWCMP processes in the region remains to be seen, however, given the size of the infrastructure investments needed for true total water cycle management to deal with future land use changes and managing existing problems, such approaches are providing significant value to MBRC and would likely benefit other agencies if applied.

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Factors that Mitigate the Formation of Emerging Disinfection By-Products in Recycled Water

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Summary

Previous research indicates that some disinfection by-products (DBPs) can form within water recycling plants utilising membrane treatment processes, specifically as a result of the chloramination pretreatment step protecting reverse osmosis (RO) membranes from excessive biofouling. Here we report the results from laboratory studies and field sampling that demonstrate that key factors influencing *N*-nitrosamine formation are precursor removal in the wastewater treatment plant (WWTP), and appropriate control of chloramination in RO treatment. A pilot plant study of the impact of pH and contact time on DBP formation further highlighted the importance of chloramine speciation on a range of DBPs. In particular, the formation of iodinated-THMs is reported, with speciation influenced by iodide concentration.

Keywords

Indirect potable reuse, chloramination, disinfection by-product formation, *N*-nitrosamines, iodinated-THMs.

Introduction

Water scarcity is driving the increased use of recycled water. Where there is opportunity for indirect potable reuse, removal of contaminants from treated wastewater is now a growing research issue. Previous Western Australian research has shown that some disinfection by-products (DBPs), including *N*-nitrosamines, can form within water recycling plants utilising membrane treatment processes, specifically as a result of the chloramination pretreatment step protecting reverse osmosis (RO) membranes from excessive biofouling (Van Buynder *et al.*, 2009). The formation of DBPs within MF/RO treatment led to the detection of DBPs in post-RO water that were not otherwise present in the source secondary wastewater. Here we report further research into factors causing DBP formation, particularly focussing on *N*-nitrosamines. Experiments carried out included:

- *N*-nitrosamine precursor and concentration analysis in a full-scale wastewater treatment plant (WWTP) and advanced water recycling plant (AWRP) in the Perth, Western Australia.
- Laboratory experiments of *N*-nitrosamine formation in secondary wastewater under varying pH and chloramination regimes.
- Analyses of chloramine speciation and DBP formation in a pilot (100 kL/D) MF/RO treatment plant, including *N*-nitrosamines, trihalomethanes (THMs), haloacetic acids (HAAs), halonitromethanes (HNMs) and haloacetamides (HAAs).

Here we summarise the key factors found to influence *N*-nitrosamine formation, and also report on the formation of DBPs in the pilot plant study, including the first data for iodinated-THMS (I-THMS).

Results

Minimising *N*-nitrosamine Formation

Figure 1 shows that the two key barriers for *N*-nitrosamine precursor removal throughout the WWTP and AWRP were activated sludge treatment in Beenypur WWTP (~95% removal), and RO in Beenypur AWRP (~99% removal). Further testing at Beenypur WWTP demonstrated that *N*-nitrosamine precursor removal was linked to nitrification, and that variability in secondary wastewater NDMA precursor concentrations was controlled by primary wastewater precursor concentrations and ammonia removal, the main indicator of nitrification. While the variability in primary wastewater precursor concentrations requires further investigation, maximum *N*-nitrosamine formation in secondary wastewater occurred at pH 8, in agreement formation via oxidation of unsymmetrical dimethylhydrazine, suggesting that secondary amines may be a significant component of total precursors. The formation potential of amine-based cationic polyacrylamide polymers, used as coagulants in Beenypur WWTP, was minimal compared to secondary wastewater precursor concentrations.

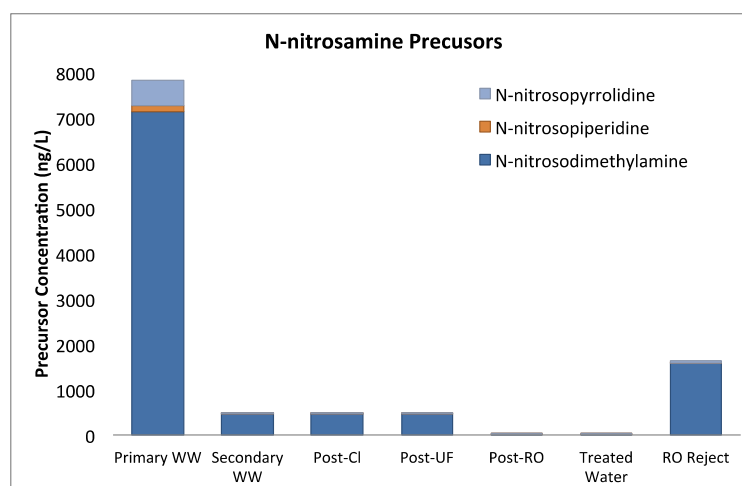


Figure 1. N-nitrosamine precursors, as measured by formation potential (Mitch *et al.*, 2003), in Beenyp WWTP and AWRP. NDMA has the greatest formation potential (and therefore highest precursor concentration) in both primary and secondary wastewater (~90% of all precursors), followed by *N*-nitrosopyrrolidine and *N*-nitrosopiperidine. All other *N*-nitrosamines showed negligible formation.

Laboratory experiments of pre-formed and inline-formed monochloramine (MCA) did not show significant differences in NDMA formation at a dose of 3 mg/L Cl_2 , but inline-formed MCA did form more NDMA and *N*-nitrosopyrrolidine than pre-formed MCA when a dose of 10 mg/L Cl_2 MCA was applied, in agreement with other studies (Farre *et al.*, 2011). Higher concentrations of dichloramine and organic chloramines were observed at 10 mg/L Cl_2 MCA, compared to 3 mg/L Cl_2 . The results suggest that the MCA dose should be controlled carefully in full-scale plants that use inline-formed MCA.

DBP Formation in a Pilot Plant

All DBPs tested (*N*-nitrosamines, THMs, HAAs, HNMs and HAAs) formed after chloramination in the pilot plant. Figure 2 shows concentration of total regulated (THM4) and iodinated THMs in the pilot plant (pH 6.5, contact time 30 min) using a novel SPME method for simultaneous detection of all ten THMs (Allard *et al.*, 2012). Even with iodide (I^-) < 10 $\mu\text{g/L}$, dichloriodomethane (CHCl_2I), bromochloriodomethane (CHBrClI) and dibromiodomethane (CHBr_2I) were formed upon chloramination, at ng/L concentrations. While I-THM concentrations (ng/L) were low compared to regulated THMs ($\mu\text{g/L}$), speciation was in agreement with the most common I-THMs found during drinking water surveys (Krasner *et al.*, 2006). While health guidelines do not exist for I-THMs, concentrations in post-RO water were three or more orders of magnitude lower than guidelines for regulated THMs.

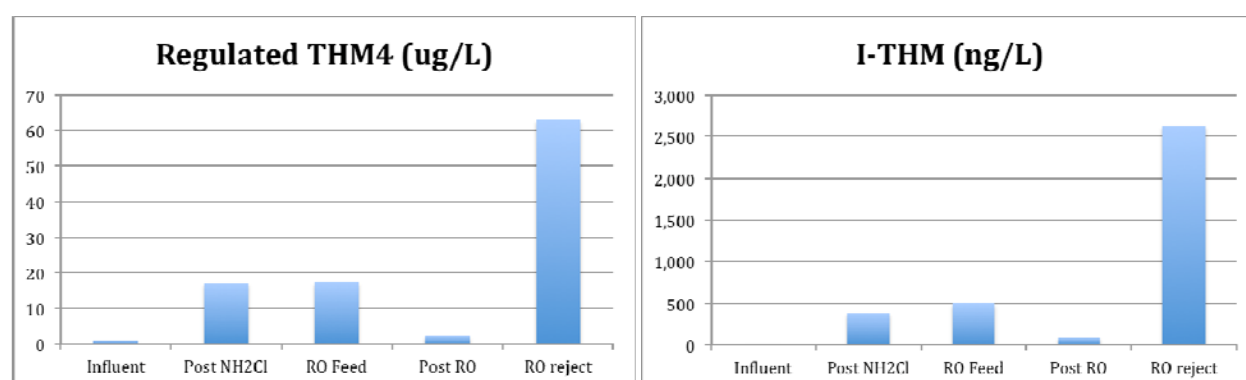


Figure 2. Formation of total regulated THM4 ($\mu\text{g/L}$) and iodinated THM concentrations (ng/L) in Beenyp Pilot Plant, with iodide < 10 $\mu\text{g/L}$.

Increasing I^- concentration led to increased I-THM concentrations (Figure 3). Both CHCl_2I and CHBrClI were detected at $\mu\text{g/L}$ concentrations when $\text{I}^- = 50 \mu\text{g/L}$ or $100 \mu\text{g/L}$. It is noteworthy that CHCl_2I and CHBrClI formation was higher for $\text{I}^- = 50 \mu\text{g/L}$ compared to $\text{I}^- = 100 \mu\text{g/L}$ because the Br^-/I^- ratio strongly influences I-THM speciation (Jones *et al.*, 2012). While the incorporation of I^- (22 nM) into THM is similar for both $\text{I}^- = 50 \mu\text{g/L}$ and $\text{I}^- = 100 \mu\text{g/L}$, a greater proportion of highly iodinated THMs formed with increasing I^- concentration, and these are considered more toxic.

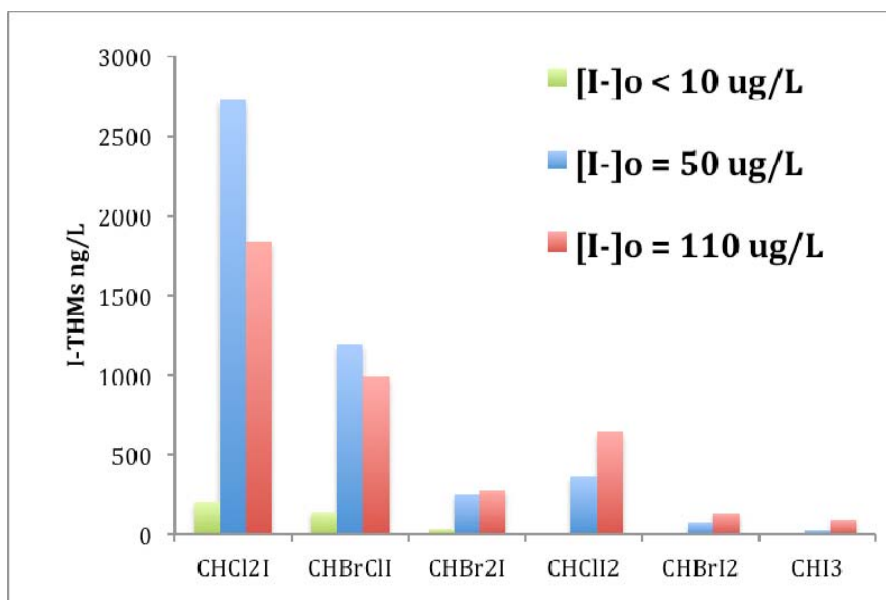


Figure 3. Effect of varying iodide concentration on I-THM speciation measured Post-NH₂Cl in Beenyup Pilot Plant.

Conclusions

- Nitrification is the key process for NDMA precursor removal in wastewater treatment.
- There was minimal difference in NDMA formation using either pre-formed or in-line formed MCA at 3 mg/L Cl₂, but inline-formed MCA did form more NDMA at higher MCA concentrations (10 mg/L Cl₂).
- All DBPs tested (*N*-nitrosamines, THMs, HAAs, HNMs and HAAs) formed after chloramination in the pilot plant, including iodinated THMs, although RO permeate concentrations were below health guidelines.
- More iodinated THMs were formed with increasing I⁻ concentration, favoring highly iodinated THMs at lower Br/I⁻ ratios.

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Re-Imagining Sustainability

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Summary

The Urban Land Development Authority (ULDA) as a planning and development agency adopts strategies and undertakes demonstration projects to road test the feasibility and sale-ability of urban sustainability initiatives. The ULDA is undertaking the planning for four major new communities of 325,000 people within South East Queensland (SEQ). The ULDA is also planning and developing 13 other urban developments across Queensland. ULDA seeks for sustainability to deliver housing affordability through reducing cost of infrastructure and providing for lower household operating costs. The sustainability vision is therefore underpinned by the principle of **Sustainability = Affordability**. The ULDA seeks to deliver affordable sustainability by:

- integrating sustainable principle into design of dwellings and infrastructure rather than seeking to add on additional components; and
- working with utilities, local government, and the development and building industries.

Keywords

Sustainability, affordability, infrastructure, operating costs.

Introduction

The Urban Land Development Authority (ULDA) as a planning and development agency adopts strategies and undertakes demonstration projects to road test the feasibility and sale-ability of urban sustainability initiatives. ULDA seeks for sustainability to deliver housing affordability through reducing cost of infrastructure and providing for lower household operating costs. The sustainability vision is therefore underpinned by the principle of **Sustainability = Affordability**. This road testing ensures that the policy requirements placed on the private sector are affordable, practical, reasonable and achievable. This ensures that sustainability is delivered in a way that ensures sustainability benefits of affordable living and high amenity are widely accrued and not just limited to the premium end of the development market.

ULDA Sustainability Agenda for the Four New Communities

The sustainability strategies for the four new communities are provided for in the Implementation Strategies section of the approved Development Schemes. The ULDA is seeking for the four new communities to be a 'model' new community embracing or even exceeding 'best practice' sustainability. Within the four new communities being delivered by the private sector in Caloundra South, Flagstone, Ripley Valley and Yarrabilba Urban Development Areas, there will be 135,000 homes constructed, housing 325,000 people over the next 30 years. The scale of new development will require over time significant infrastructure and capital investment. The challenges facing urban development generally within the growth areas of South East Queensland include:

- rising costs of infrastructure, household bills and lack of affordable land.
- the need to increase levels of sustainability within difficult cost constraints.
- difficulties of integrating systems and new technologies.
- lack of local or specifically applied data or research.

Appropriately targeted sustainability initiatives have the potential to deliver efficiency in resource use (of land, water, energy and waste) so as to reduce infrastructure and household living costs. The four new communities provide an opportunity to undertake development in a more sustainable and affordable way. The four communities will take approximately 30 to 40 years to fully develop and technologies, prevailing economic conditions, socio-demographic trends and attitudes will changes and evolve over time. The implementation strategies seek to respond to the challenge of delivering a 'model' community over a lengthy time period by establishing shorter term targets and goals, underpinned by a commitment to a cycle of research, data monitoring, review and, if warranted, amendment of standards, guidelines or targets. This approach establishes a cycle of continuous adoption of 'best practice' over time through a rigorous process of monitoring and review.

The Implementation Strategies for each of the four new communities provide for Actions, Demonstration/Pilot Projects and Stretch Targets and Goals.

Among the sustainability actions required are the ULDA working with developers, councils, government agencies, and utilities to develop strategies on:

- community education to promote the protection and enhancement of the natural environment.
- demand optimisation for water and energy efficiency and demand management strategies, including builder education.
- addressing the urban heat island effect to ensure urban amenity and lower energy use in dwellings and buildings.

Among the demonstration projects that the ULDA is seeking to facilitate include:

- alternative technology and service model projects for local renewable energy, water self sufficiency, and waste avoidance and recovery.
- affordable sustainable housing projects that reduces energy use and inputs to achieve zero emissions.

The short term 2016 Stretch Targets include:

- Potable water usage reduction to an average of 140 litres per person per day.
- Average household energy usage reduction to 15 kilowatt hours (kWh) per day.
- 20% peak energy demand reduction from 5 kilovolt ampere (kVa) to 4kVa average diversified maximum demand.

An example of the ULDA's role can be seen with Ripley Valley. For Ripley waste water is a key issue and ULDA is looking to facilitate a general consensus that an appropriate combination of scalable solutions and staging over the short, medium and long term to delay significant up front infrastructure costs, while still meeting the overarching objective for the Valley. This would see a 'greening' of the valley, with the potential for a mix of 'online' and 'offline' treatment methods and open space irrigation.

Fitzgibbon Water Sensitive Urban Design and Innovation

Fitzgibbon Chase is a ULDA development 12 km north of the Brisbane CBD. It is an affordable top selling 114ha residential development that demonstrates that water sensitive urban design (WSUD) is commercially viable and can add commercial value to a development when the WSUD is integrated into the core of the urban design. The Fitzgibbon Chase affordable residential development incorporates a number of leading WSUD initiatives and innovations, including:

- WSUD within all parks including Fitzgibbon Chase Park which forms a multifunctional space incorporating public open space with storm water flow paths.
- WSUD in the street scape including bio retention swales.
- WSUD within private open space with provision of a water sensitive landscaping rebate to all dwellings.
- Retention of 40% of the site as natural bushland including natural waterways.
- Fitzgibbon Stormwater Harvesting scheme which takes overflow from stormwater flows and treats it for non potable dwelling use.
- PotaRoo project which is intended to take water from household roofs for central treatment for potable household demand.

Energy Grid Strategies and Water Interface

Significant electricity grid issues arise with significant urban development both in SEQ and in regional Queensland. A key strategy is to develop energy efficient developments with energy efficient housing with a particular focus on peak energy demand. In the four new communities the ULDA is working with Energex and private developer to develop peak energy demand strategies that are aimed at reducing costs for Energex, developers and householders as well as delivering environmental benefits. In the regions, ULDA and Ergon are working towards more advanced peak energy strategies for our regional projects. The lessons learnt by ULDA, Energex and Ergon will be applicable to wider development and building industry.

A key issue impacting peak energy consumption is the urban heat effect and the need for air conditioning. Research is required to accurately identify the major causes of the urban heat island effect in suburban and urban development. Affordable mitigation strategies in-built into urban development also needs to be identified. The use of water within the urban environment is a key way to affordably address both energy and water infrastructure issues.

Research Proposal

To support the sustainability initiatives within the four new communities ULDA and CSIRO are proposing a visionary industry research alliance – the Sustainable Urban Development Research Alliance. The aims of the Alliance will be to achieve the sustainability vision set for the four new UDAs. The proposed Alliance will undertake research addressing issues and challenges facing urban development underpinned by the Alliance's First Principle of Sustainability = Affordability. The Alliance is an opportunity to create new commercial best practice for the urban development with a focus on affordability and sustainability.

Conclusion

The ULDA as a planning and development agency adopts strategies and undertakes demonstration projects to road test the feasibility and sale-ability of urban sustainability initiatives. ULDA seeks for sustainability to deliver housing affordability through reducing cost of infrastructure and providing for lower household operating costs. The sustainability vision is therefore underpinned by the principle of **Sustainability = Affordability**. The ULDA is seeking for the four new SEQ communities to be 'model' new communities embracing or even exceeding 'best practice' sustainability. ULDA is also working across regional Queensland. In the energy space ULDA is working with Energex, Ergon and private developers, to develop peak energy demand strategies that are aimed at reducing costs for Energex, developers and householders as well as delivering environmental benefits. The use of water within the urban environment is a key way to affordably address both energy and water infrastructure issues. To support the sustainability initiatives within the four new communities ULDA and CSIRO are proposing a visionary industry research alliance.

The Significance of Large Scale Seawater Desalination in Achieving Urban Water Security

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Summary

Following a protracted drought, seawater desalination has become a strong component of bulk water supplies for the five Australian mainland state capitals. On aggregate, seawater desalination can now provide up to 35% of current average annual water demand. After the drought break in 2010, extensive flooding occurred in the eastern states and there has been public criticism of the decision to build the desalination plants. However, construction of desalination capacity should be considered in the long term (50 years) during which time population growth will increase demand and droughts and possibly climate change will put existing conventional supplies under stress.

Keywords

Seawater, desalination, urban water, water security, climate resilient water, myths.

Introduction

It has become popular for some in the media with short memories to knock desalination. “Knee jerk reaction, political decision, why not build a cheap dam, expensive, energy guzzling, built to feed the profits of private water companies, destroys the marine environment, should only be turned on as a last resort” are just some of the epithets that have circulated about desalination since the end of the Millennium Drought.

The writer has spent the last 37 years working as a professional engineer in the water industry in Australia: 20 years in Government, 15 years in the private sector and the last two leading the National Centre of Excellence in Desalination Australia in Perth. He has been in the industry long enough to remember when water security was the key objective of State Government water utilities and massive expenditure on water infrastructure was expected and celebrated by the voting public.

This changed during the 1970s when, under the influence of economic rationalism, State Governments learnt about demand management, reduced spending on new water supplies and increased rates, turning water utilities into “cash cows”. It was this era in which “water efficiency” (restrictions) became fashionable. This all worked tolerably well with increasing population and concomitant demand, provided rain kept falling regularly. The Millennium Drought (1997 – 2009) changed all this, and despite draconian compulsory water restrictions, many Australian cities came close to running out of water. As a result, six large seawater desalination plants have been built to serve Perth, Gold Coast, Sydney, Adelaide and Melbourne.

Common Desalination Myths

A number of myths have developed about desalination. Let’s look at some of them.

Myth No 1 – The “Knee Jerk” Reaction

No responsible Government of a sophisticated economy can sit and do nothing when their water supply is about to run out. The proposition that the decision to build the Sydney Desalination Plant was a “knee jerk political decision” is nonsense. Governments are guided by their water utilities on infrastructure investments and decisions to spend nearly \$2 billion on new water supply infrastructure (the Sydney Desalination Plant) are never taken lightly. Indeed, the expenditure could be considered as “catch up” given the woefully inadequate investment in new water supplies since the 1970s.

Myth No 2 - Why not Build a Cheap Dam?

Dams take a long time to build and fill. Warragamba Dam, NSW, was approved in 1946 and was completed 14 years later in 1960. Thomson Dam in Victoria was approved in 1975 and completed eight years later in 1984. As a comparison, approval to build the Sydney Desalination Plant was given in November 2006 and it was opened just over three years later in January 2010. In 2010, plans to build dams at Traveston Crossing in Queensland and Tillegra in the Hunter Valley, NSW, were cancelled because of their potential impact on the environment. Dams are no longer seen as election winners. It would be extremely difficult under current circumstances to build large dams near populated areas, given the extent of public opposition.

Myth No 3 - Desalination is Energy Guzzling

Energy needed to produce water at the Sydney Desalination Plant is around 3.5 kWh/kL. The average household uses about 200 kL per year. Thus if all the water for a household came from desalination, the energy required is about the same as that required to run the normal household refrigerator. The smallest ducted household reverse cycle air conditioner runs at about 8 kW. Thus, if all the water for the household came from desalination, the energy required is about the same as running the ducted reverse cycle air conditioning system for 15 minutes. One feature of all the big Australia desalination plants is that they all purchase renewable energy from wind farms to offset all energy used in the desalination process. Their operational carbon footprint is very low.

Myth No 4 – Desalination is Expensive

To cover the cost of desalination, naturally water rates must increase. In the case of Adelaide's desalination plant, which will be capable of supplying up to half Adelaide's current water demand, the rates have steadily increased since 2008. For an average household using 190 kL/y, the average weekly increase between 2008 and 2012 is \$7.80. Now this also includes increases for such things as inflation, so the whole \$7.80 is not attributable to the cost of the desalination plant. But let's put this in perspective - \$7.80 is about what you pay for a glass of beer from a bar. So for the modest charge which is equivalent to less than the cost of just one glass of beer per week per household, Adelaide consumers have water security and do not ever have to face water restrictions again. That's a good deal.

Myth No 5 – Desalination Plants are Built to Feed the Profits of the Private Water Companies

Most of the big desalination plants in Australia have been built with some form of partnership between the water utilities and world leading desalination companies. In most cases, the construction was undertaken by Australian companies, but the process design was undertaken by experienced international desalination companies. The process has been guaranteed by the international desalination companies. There is much that could go wrong in such big and fast moving projects, so the modest profits need to be understood in terms of the large losses that can rapidly accrue if any part of the process fails. In all cases, the projects were tendered competitively with oversight and auditing as is normal for execution of a Government project.

Myth No 6 – Desalination Plants Destroy the Marine Environment

Australia's first major seawater desalination plant is at Kwinana, just south of Perth, on Cockburn Sound. Commissioned in November 2006, it has operated at 100% capacity (140 ML/d) continually since then – more than five years. Cockburn Sound has been a confined body of water since the construction of a causeway to HMAS Stirling on Garden Island in 1970. If ever there was going to be a problem with environmental impact of reverse osmosis concentrate, this would be the place. However, the experience of Perth's first desalination plant provides confidence that no environmental harm results from well designed and maintained return concentrate diffuser systems.

Myth No 7 – Should only be Turned On as a Last Resort

Water restrictions (using less) means reservoirs fall less quickly. As the levels drop further, restrictions become more severe, until ultimately the most extreme restrictions apply and even then without rain, the reservoirs will empty. Now there is an alternative to restrictions to slow the rate of reservoir depletion – running the desalination plants. The amount by which a water utility might run a desalination plant depends on the security of supply. If restrictions are to be avoided, then the desalination plant needs to run as a base load plant most of the time, with the reservoirs providing the peak load supply. The only time it really needs to be turned off is when the reservoirs are spilling.

Conclusion

The five Australian mainland capital cities have invested in desalination capacity up to 35% of their average demand. These climate resilient new water sources provide water security at an affordable price. Provided the desalination plants operate as base load plants, it means that water restrictions are likely to be a thing of the past. Energy consumption for desalination is much less than for other common household appliances (for example reverse cycle air conditioning), and the cost, if all household water came from desalination, is less than the cost of one glass of beer per week from a pub. Lastly, five years experience at 100% flow at Perth's Seawater Desalination Plant, which discharges concentrate through a well designed diffuser into the confined Cockburn Sound, has not caused any detectable environmental harm. In fact, the diffuser pipework is covered in marine life and resembles an artificial reef.

Papers

Can Stormwater Harvesting Restore Pre-Development Flows in Urban Catchments in South East Queensland?

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Summary

Increases in impervious area due to urbanisation have been shown to have negative impacts on the physical and ecological condition of streams, primarily through increased volume and frequency of runoff. The harvesting and detention of runoff has a potential to decrease this impact. This paper describes the effects of urbanisation on catchment flow and of stormwater harvesting on reducing those adverse impacts on a stream in South East Queensland (SEQ).

A largely undeveloped catchment located southeast of Brisbane city was calibrated and validated using the Stormwater Management Model (SWMM). This model was used to investigate the effect of a range of future increases in urbanisation (represented by impervious area) on stream hydrology as well as the potential of stormwater harvesting to return the catchments to pre-development flow conditions. Stormwater harvesting was modelled according to flow frequency measures specified in current SEQ development guidelines. These guidelines stipulate the capture of the first 10mm of runoff from impervious areas of 0-40% and the first 15mm from impervious areas of 40% or greater for urban developments.

We found that increases in impervious area resulted in increases in the mean, frequency and duration of high flows, and an increase in the mean rate of rise and fall for storm events in the catchment. However, the pre-development (non-urbanised) flow distribution was very flashy in comparison with all urbanised scenarios; ie, it had the quickest response to rainfall indicated by a high rate of rise to and fall from peak flow volume, followed by a return to zero flow conditions. Capturing the runoff according to the development guidelines resulted in a reduction in flow towards the flow distribution of a lower impervious area, however this was insufficient to meet pre-development conditions. This suggests a stronger influence of impervious areas in this catchment on the volume of runoff than flow frequency measures are able to ameliorate.

Keywords

Stormwater harvesting; urbanisation; urban stream syndrome.

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. Increased runoff brings with it a wide range of organic and inorganic pollutants, but importantly changes the hydrology of streams. Urban impervious area storm runoff can increase the frequency, magnitude and duration of high flow events and at the same time decrease infiltration and base flow. 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 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 [per day] from impervious surfaces where the total impervious surface is 0 to 40%.
- the first 15mm of runoff [per day] from impervious surfaces where the total impervious surface is greater than 40%.

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

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

$$\text{Capture volume (m}^3\text{)} = \text{Impervious area (m}^2\text{)} \times \text{target design runoff capture depth (mm/day)} / 1,000.$$

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.

The aim of this work is to examine the effectiveness of frequent flow management guidelines given in South East Queensland Regional Plan (2009) to mitigate impacts of urbanisation, using detailed calibrated models of catchments in the region. This paper presents a simulation modelling analysis of the impact of the FFMOs on stream hydrology across a range of potential urbanisation extents (percentage total impervious area values) for the first case-study catchment, Upper Yaun Creek.

Methodology

The methodology involves development of a hydrological model of the selected catchment to simulate hydrological behaviour of the catchment under a range of urbanisation scenarios, and assess the effectiveness of frequent flow management guidelines to replicate the predevelopment flow regime of the catchment.

The selected catchment, Upper Yaun Creek, is located near the township of Coomera south of Brisbane in South East Queensland (SEQ), Australia. The Upper Yaun catchment describes the portion of Yaun Creek catchment that is largely undeveloped forest, with only 3% impervious area, none of which is directly connected to the creek. The catchment has an area of 361 hectares, an average slope of 6.8% and a time of concentration of 1.90 hours (time for rainfall from the furthest part of the catchment to reach the outlet). Rainfall and stream flow data has been collected for this catchment from April 2009 to current day, using a 0.2mm tipping bucket rain gauge and a pressure transducer for continuous (6 minute) water levels, which were combined with rating curves developed from cross-sectional, slope and stream gauging data using in-house Hydstra software.

A hydrological model of Upper Yaun Creek was developed using the Stormwater Management Model (SWMM) (Huber, 2003). The model was calibrated using 12 of the 18 months of the recorded rainfall and stream flow data and validated using the remaining six months data. An hourly timestep was chosen for the model, as the time of concentration was greater than one hour and comparisons of the flow measurements on different timesteps concluded that the hourly timestep captured fairly well the peaks in the flow series, without being onerous in runtime of the calibration or simulation process. A shuffled complex evolution algorithm (Duan *et al.*, 1993) available for the MATLAB program was coupled with the SWMM executable to optimise the model parameters in calibration, using Nash Sutcliffe criterion of efficiency (NSE) (Nash and Sutcliffe, 1970) as a measure for goodness-of-fit with observed data. The NSE for calibration and validation periods (hourly timestep) was 0.78 and 0.33 respectively. The relatively poor validation score was likely due to low synchronisation of stream flow with rainfall data, and difficulty in replication of very low flows, as the NSE was better at 0.75 for the same period on a daily timestep. Performance was checked on a daily timestep as this is the timestep used here for FFMO performance analysis. The NSE during the calibration period on a daily basis was 0.81.

The model simulation was run for the period August 2000 to December 2011 using data from the Bureau of Meteorology's Oxenford Weir gauge, which is a continuous rainfall gauge located 3 km from the project's rainfall gauge. This rainfall data was chosen as it matched well to gauge data but had a longer record to explore a greater variety of climatic conditions. A range of urbanisation scenarios were modelled, represented by a change in the percentage impervious area of the subcatchments. These scenarios ranged from the current state (3% impervious), up

to 70% impervious, representing a high degree of urbanisation. In the absence of planning guidelines, urbanisation was assumed to commence in the middle portion of the catchment, up to a subcatchment imperviousness of 55%, then extending to the lower, followed by upper portions of the catchment to the total impervious area required of the scenario. Scenarios of 60% and 70% imperviousness were equally spread across subcatchments. A scenario of 0% impervious area was also simulated to represent pre-development conditions, and forms the basis of comparison for the impact of urbanisation as well as the effectiveness of runoff capture as per FFMOs.

Runoff capture scenarios followed the flow frequency management objectives. This involved harvesting the first 10mm of runoff from impervious surfaces each day (for catchments with 40% imperviousness or less); and 15mm per day for catchments with a higher proportion of impervious surface. For example, a 40% impervious runoff capture scenario required an impervious area of 144.4 ha (40% of the 360.1 ha catchment), with a maximum daily capture volume of 14,436 m³ (10mm of rainfall capture depth over an area of 144.4 ha). A time series of hourly flow capture volume (m³/s) was developed by using the hourly rainfall data to establish expected accumulated runoff from the urban area in millimetres, multiplied by impervious area. In other words, for each hour from the start of the day that rainfall occurred, flow was captured until the maximum daily capture volume was reached. This extraction was applied in the model as negative inflow at the node outlet of each urbanised subcatchment.

Results

Simulation results were reported as time series of hourly flow in cubic metres per second (m³/s) at the catchment outlet. However, since the FFMOs are stipulated on a daily basis, the flow series have been aggregated and assessed on a daily basis. Figure 1 shows the flow exceedance curves based on the aggregated daily time series. These curves illustrate the distribution of flow for select scenarios, by showing the probability of exceedance of a given flow in the record. The graph uses a logarithmic axis for the flow measurements to highlight the lower flows. High flows (low probability of exceedance) are to the left of the graph, with low flows (not appearing where zero), towards the right.

Figure 1 illustrates the impact of urbanisation on flow distribution for a range of urbanisation scenarios. It can be seen that even compared to a small degree of urbanisation (3-5%), a non-urbanised/pre-development catchment frequently runs dry, with no flow approximately 95% of the time. This is likely due to the infiltration of all pervious runoff to groundwater in the calibrated model, retaining the flow, as well as significant evaporation from surface pervious store. The difference between the urbanisation scenarios is not as great (as that between urbanised and non-urbanised), but a steady increase in flow volumes across the flow distribution can be observed, particularly for the medium flows, with the shape of the flow curves remaining similar. Despite the increase in higher flows with greater imperviousness, the lower flows remain similar, suggesting the creek becomes flashier, with greater flow peaks. Nevertheless, a small increase with urbanisation in the number of timesteps with flow occurring can be observed in the increase in the percentage time that above zero flows are experienced.

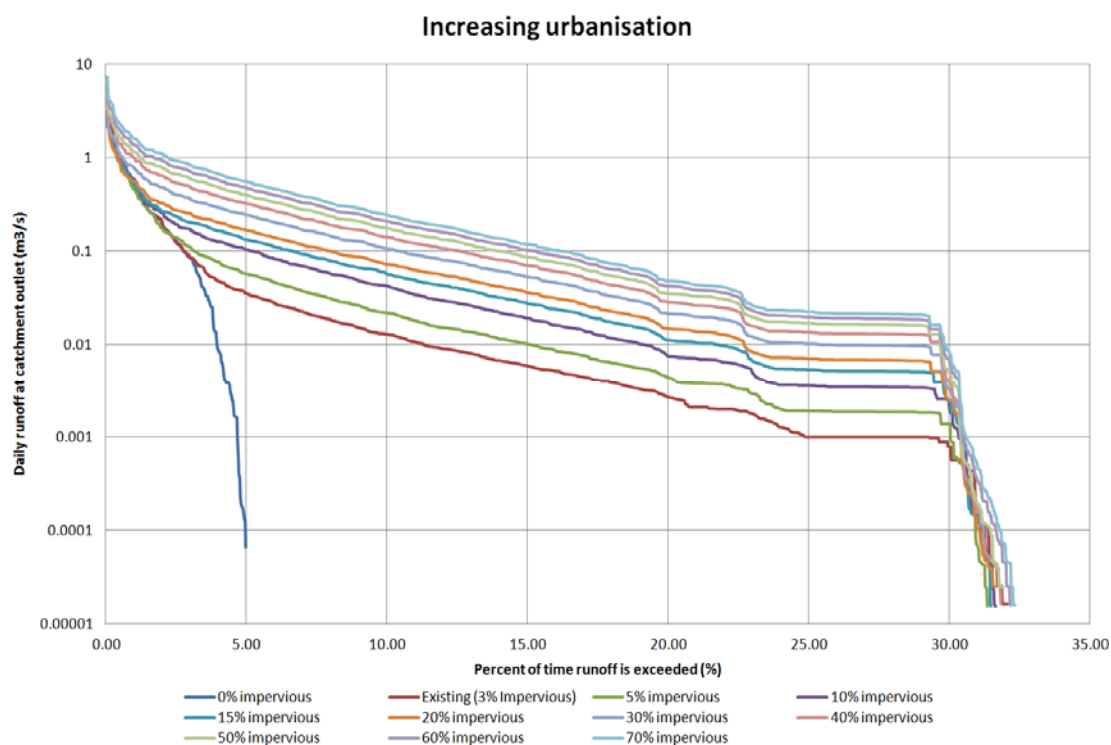


Figure 1. Flow exceedance curves for urbanisation scenarios (% impervious) within the catchment. These graphs illustrate the distribution of flow values in the time series by plotting the percent time (x-axis) within the runoff record that a given runoff (y-axis, log of m³/s/day) is exceeded.

Figure 2 compares a number of scenarios with and without runoff capture, and shows that the flow distributions of 5%, 20%, 40% and 70% impervious scenarios with runoff capture roughly match those of 3%, 15%, 30%, and 60% impervious scenarios without runoff capture, respectively. Whilst the capture does not reproduce the pre-development flow distribution, the flow curve more closely matches that of a lower impervious percentage, ie, minor increases in urbanisation of around 5% are ameliorated.

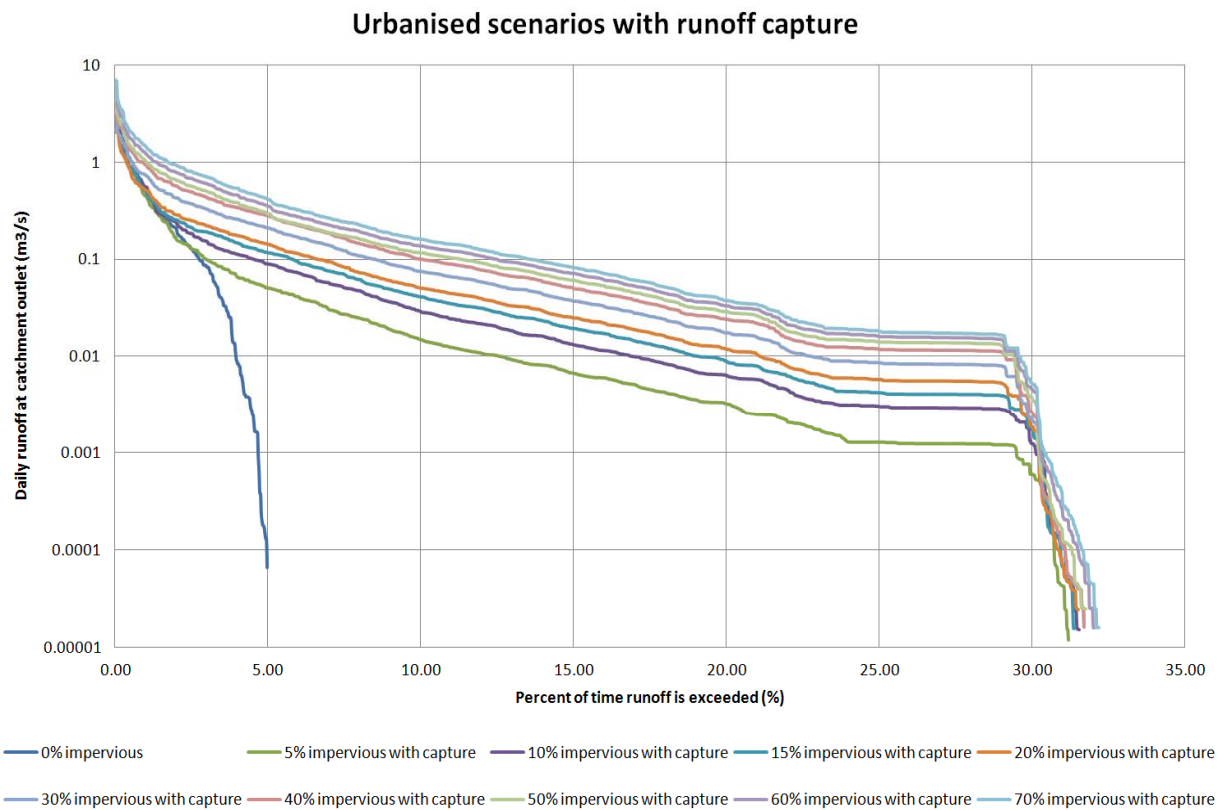


Figure 2. Flow exceedance curves for urbanisation (% impervious) with and without capture of runoff (capture) according to frequent flow management objectives. These graphs illustrate the distribution of flow values in the time series by plotting the percent time (x-axis) within the runoff record that a given runoff (y-axis, log of m³/s/day) is exceeded. Selected scenarios of urbanisation with runoff capture are compared to those of urbanisation without runoff capture to gauge the impact on the flow distribution.

Figure 3 summarises the change in mean and 90th percentile flow for each scenario. As was indicated by the flow exceedance curves, there is an increase in mean and 90th percentile flow with urbanisation. Runoff capture reduces these values. Figure 4 shows a similar increase with urbanisation (and subsequent reduction with runoff capture) in the number and duration of high spells. Figure 5 shows an increase in the rate of rise and fall of flow response to rainfall events with increases in impervious area, which decreases with runoff capture. The predevelopment (0% impervious area) scenario is significantly ‘flashier’ (higher rate of rise and fall, lower number of days with flow) than the urbanised scenarios. The high rate of rise for 0% impervious is likely due to the greater likelihood of zero flow for this scenario before a runoff event.

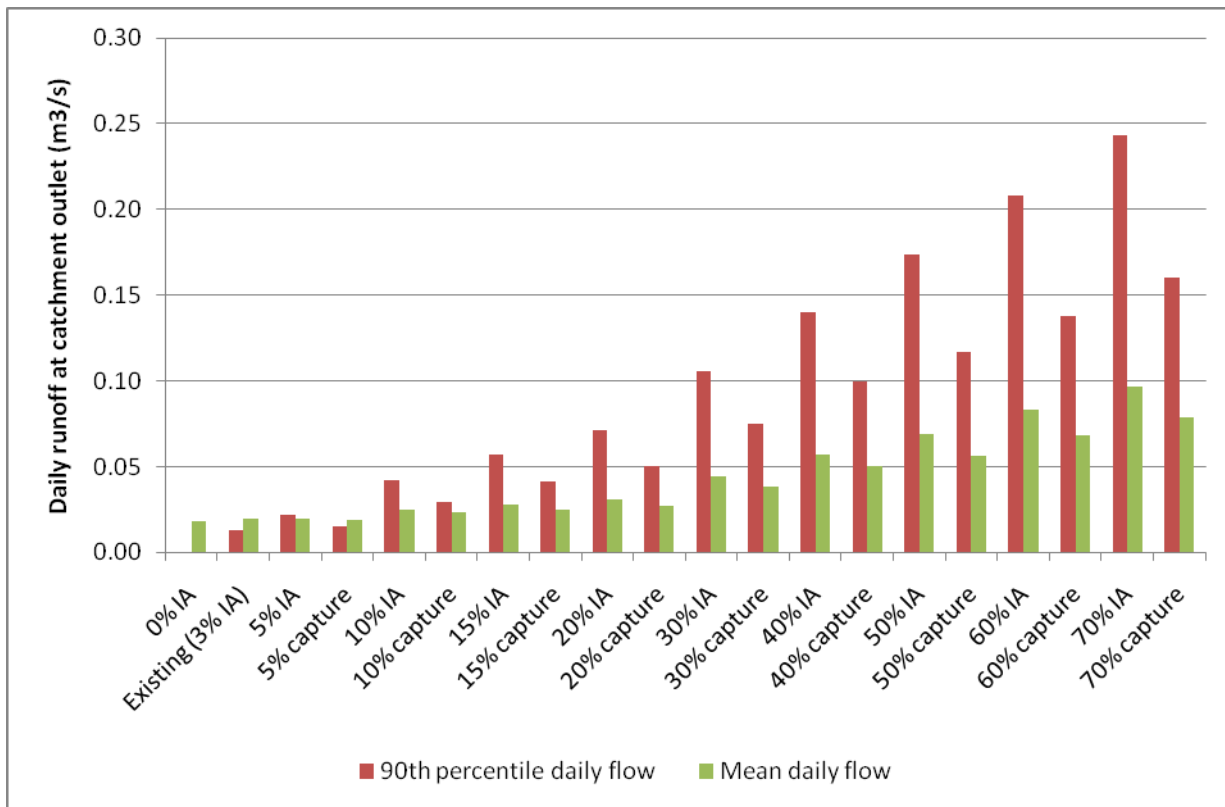


Figure 3. Mean and 90th percentile of daily flow (m3/s).

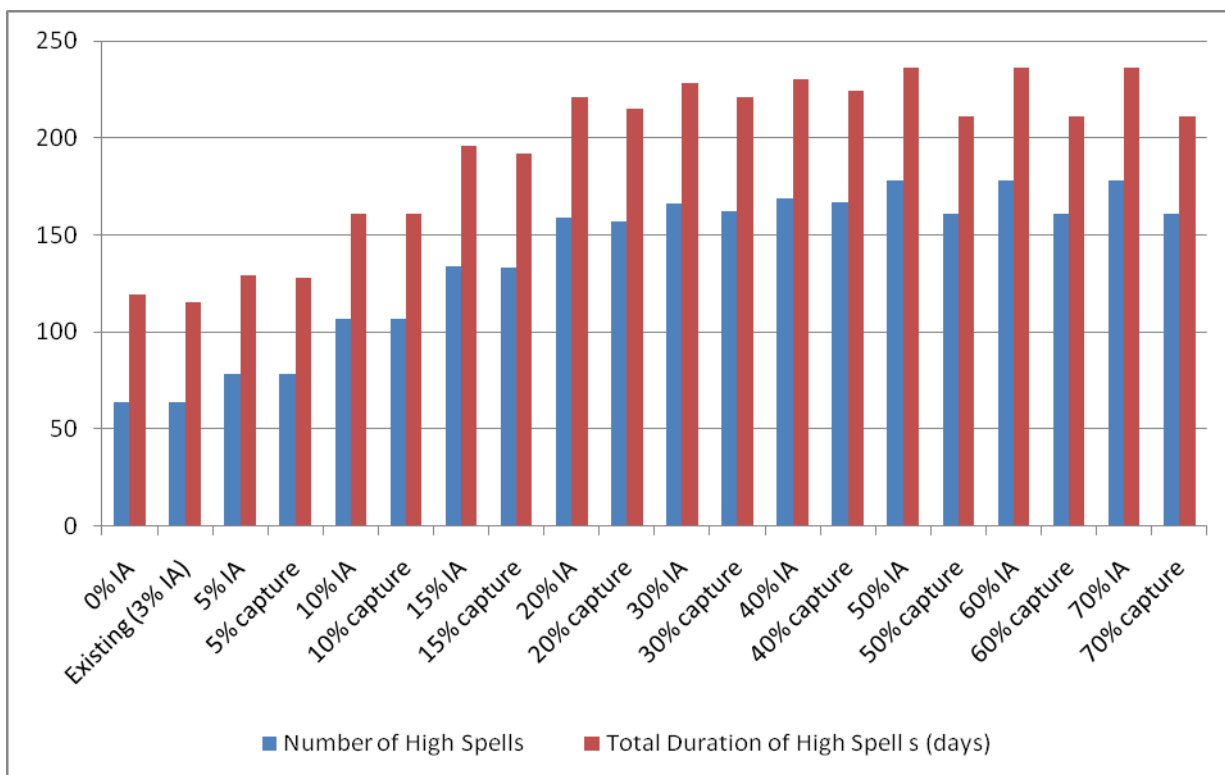


Figure 4. Number and duration of high spells in days for scenarios with various percentage impervious area (IA) with and without runoff capture (capture). A high spell is defined as a flow volume greater than five times mean flow.

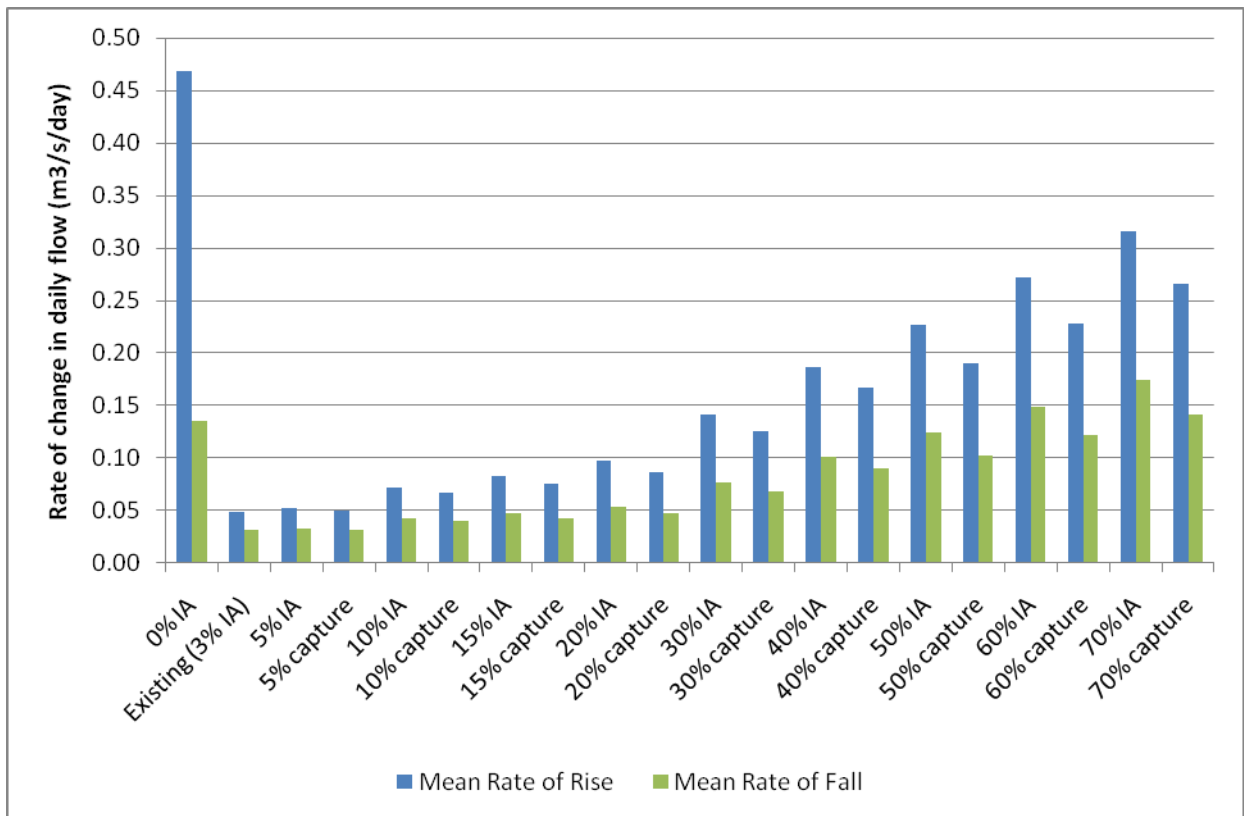


Figure 5. Mean rate of change in rise and fall of flow events.

Conclusions

Overall, the results of simulation indicate that there is an increase in flow magnitude, frequency and duration with urbanisation. Application of FFMOs indicate an effective reduction in flow magnitude, frequency and duration with runoff capture for a given urbanisation scenario. The rate of rise and fall of flow events was also decreased. For this catchment, the runoff capture stipulated in the frequent flow management objectives appears to shift flow regimes (and impacts) of urbanised catchments towards those with less urbanised (lower impervious area) conditions.

Whilst these results are not successful in replicating the unique pre-development conditions of Upper Yaun Creek catchment, this appears difficult due to the dramatic effect on the hydrological regime of the catchment of removing impervious areas in the model. This ‘de-urbanisation’, even from 3% to 0%, results in a significant increase in cease-to-flow conditions. In other words, the pre-development scenario is significantly ‘flashier’ than any of the impervious scenarios. This effect on hydrology appears to be much larger than the flow frequency rules can provide for in reducing flow. For this catchment, they do not appear to be of sufficient magnitude or suitable nature to replicate the pre-development flow conditions.

Further work in this project will be conducted both to investigate improvements to low flow performance of the calibrated model and to assess the effectiveness of FFMOs on additional catchments with different hydrological characteristics.

Acknowledgements

The authors acknowledge the significant contribution of Richard Gardiner of the Queensland Department of Environment and Resource Management in establishing and providing rainfall and stream flow gauging for the catchment as well as advising on interpretation of the data.

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Can Bugs Live Anywhere? Habitat, Hydrology and the Health of Urban Creek Ecosystems in South East Queensland

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Summary

Changes in hydrology through increased imperviousness are seen as one of the major drivers of poor health in urban streams. Flow change can impact instream components directly or through associated changes in water quality and habitat availability. This study explored three sites across an urban gradient in detail to detect the mechanism for poor instream health, as measured by macroinvertebrates. The macroinvertebrate assemblages of pool habitats in urban streams were vastly different to those of predominately forested catchments, whereas the riffle assemblages were similar. This suggests that habitat availability, or quality, may be a mechanism for poor instream health in urban streams and restoration efforts focused on restoring instream habitat through either in-channel restoration or stormwater management may have positive outcomes.

Keywords

Urban streams, macroinvertebrates, habitat, riffle, pool.

Introduction

Flow has a major effect on stream ecosystems. In urban environments, flow characteristics differ substantially from natural conditions due to the direct piping of water into stream networks and the increased amount of impervious (non-porous) surfaces in catchment areas (Walsh *et al.*, 2005). This can lead to sudden and large pulses of water in urban streams, events which may cause large scale erosion of instream habitat and dislodge organisms like invertebrates. Subsequently, dissolved oxygen levels may decline such that life can no longer be supported. In this way, the changes in stream hydrology caused by urbanisation can impact invertebrates directly or through changes in water quality and habitat availability (Figure 1). Degradation of urban streams is a worldwide problem (Paul and Meyer, 2001) with restoration attempts covering physical habitat restoration (instream and well as riparian) as well as flow modification (Bunn and Arthington, 2002). More recently, the focus has shifted to the role of stormwater removal in urban streams to restore more natural flow magnitudes and durations and reduce instream flow velocities during storm events (see Walsh *et al.*, 2005).

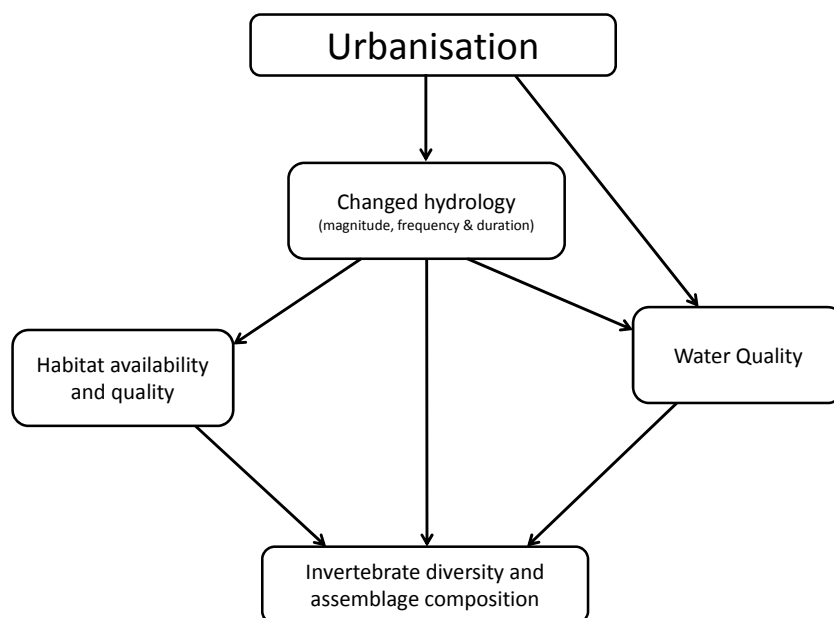


Figure 1. Conceptual model of the impacts on instream aquatic invertebrates caused by urbanisation.

In 2010, a survey of macroinvertebrates in 26 streams across Brisbane suggested a high degree of between-site variability with minimal differences between site ‘groups’ classified as “forested with little or no direct stormwater connection”, “Water Sensitive Urban Design (WSUD)” or “urban with direct stormwater connection” (C. Leigh, J.E. Dunlop and F. Sheldon, unpubl. data; Leigh *et al.*, in press). Taxa in the insect orders Ephemeroptera, Plecoptera and

Trichoptera (EPT) had lower relative abundances both in streams with direct urban connection and in streams with greater upstream catchment imperviousness. However, correlation between biotic and environmental variables in the directly-connected sites was lacking in comparison with other sites. This spatial study showed a strong impact of urbanisation on the sensitive groups of macroinvertebrates, namely EPT taxa, but did not provide a specific mechanism. Our conceptual understanding of the function of urban streams in sub-tropical Queensland suggested that the combined impact of a flashy hydrograph, where extreme velocities dislodged invertebrates, combined with inter-flow periods where water quality would become extremely poor provided a mechanism for poor urban stream health (Figure 2). Given this background we expected urban streams to be in the ‘poorest’ health from spring – autumn where the combination of higher air temperatures and more frequent flows would provide a mechanism for impact, and be in comparatively better health over the winter months. This paper describes the results of intensive sampling across three streams with different upstream characteristics in June 2011 (winter) to explore (a) differences in water quality and (b) differences in macroinvertebrate assemblages. We predicted that if hydrology directly impacted macroinvertebrates through dislodgement then differences would be minimised during the winter when flows are relatively stable. Further, as habitat may also be a key driver in community assemblage composition, we explored if habitat availability, in terms of the proportion of pools and riffles, differed between sites of differing levels of urbanisation.

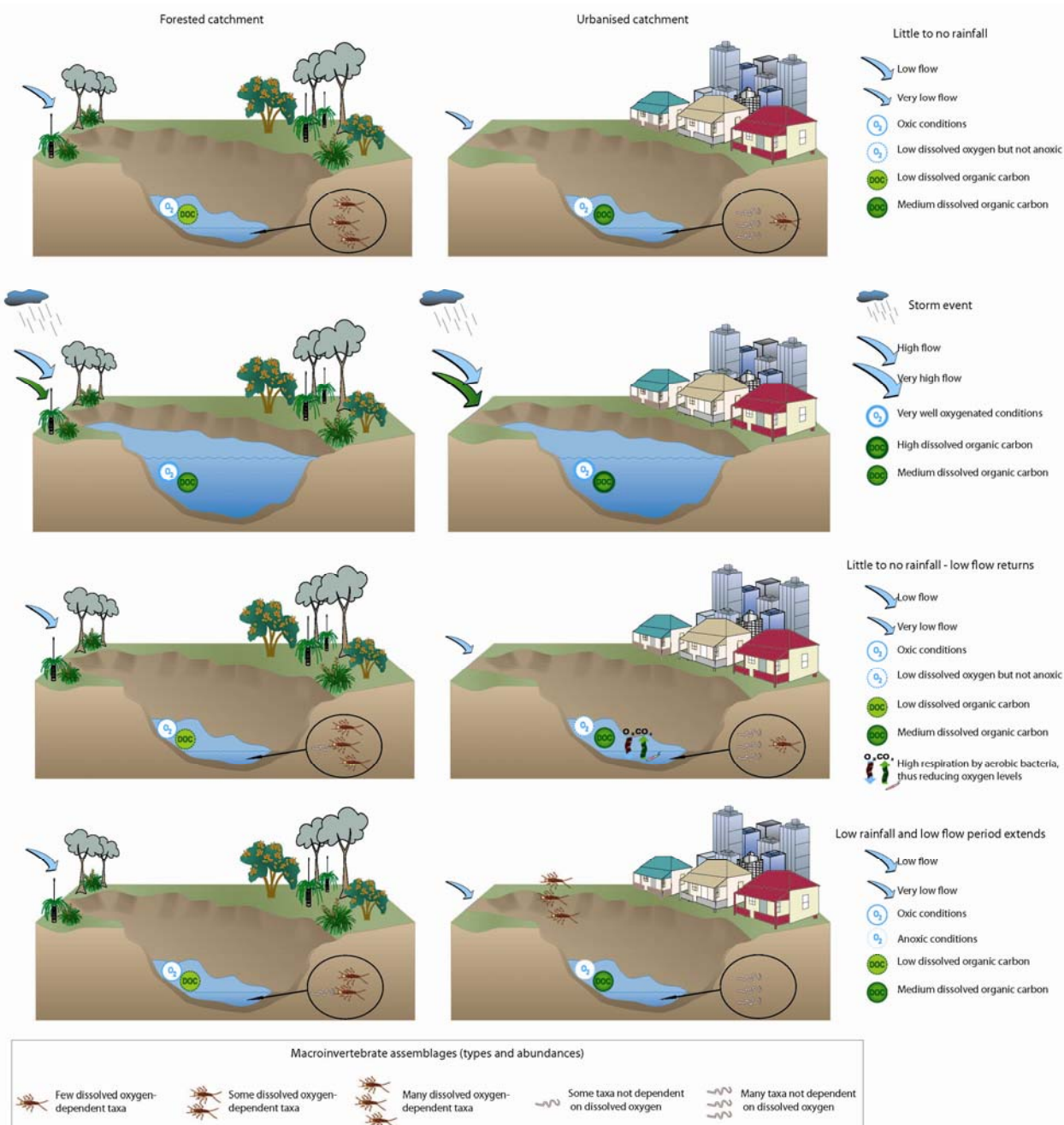


Figure 2. Conceptual model outlining the impacts on streams macroinvertebrates from urbanisation as mediated through flow and water quality changes.

Methods

Study Area

Three focus catchments within the Brisbane region of South East Queensland (SEQ) were chosen from the large range of imperviousness in the sites studied by Leigh *et al.* (in press). These were: (i) Tingalpa Creek, a forested catchment with 1% upstream catchment impervious area; (ii) Stable Swamp Creek, a highly urbanised stream with 36% upstream catchment impervious area; and (iii) an unnamed tributary stream to Blunder Creek downstream from Forest Lake within a Water Sensitive Urban Design (WSUD) region that includes sections with 14% upstream catchment impervious.

Sampling Methods

At each sampling site, a 100 m reach was stratified into the distinct habitats of ‘pool’, open standing water, ‘riffles’, areas of faster moving shallow water, and ‘macrophyte/snag’, areas of distinct aquatic vegetation and/or submerged wood often located in pool environments but not exclusively. Macroinvertebrates were collected in replicate samples from each habitat by sweeping a 500 µm mesh pond net over the habitat for 20 seconds. Samples were preserved in 70% ethanol and later washed through nested sieves (2000, 1000, 250 µm) and the organisms hand-sorted, enumerated and identified as far as practicable. At each sampling site, spot measurements were recorded of temperature, dissolved oxygen, pH and conductivity.

Statistical Methods

Differences in the summary community metrics richness, abundance, a measure of assemblage diversity (Margelef diversity (d)) and a measure of assemblage evenness (Pielou evenness (J)), evenness describes the distribution of the individuals between different taxa (see Clarke and Gorley, 2006), between sites for each habitat were explored using One-way Analysis of Variance (ANOVA) in the IBM SPSS Statistics v20 package with the Least Significant Difference (LSD) option for post-hoc comparisons. Patterns in macroinvertebrate assemblage composition between sites and habitats were explored using multivariate statistics from the PRIMER 6 software package (Clarke and Gorley, 2006). Data were $\log_{10}(x + 1)$ transformed before analysis; the Bray-Curtis coefficient was used as the measure of dissimilarity between samples and all taxa were retained in the analyses. SIMPER (Similarity of Percentages; Clarke and Gorley, 2006) from the PRIMER 6 software package was used to explore the species groups associated with the different habitats across the sites. ANOSIM (Analysis of Similarities; Clarke and Gorley, 2006) from the PRIMER software package was used to explore significant differences between sites and habitats based on assemblage composition. Assemblage differences between sites and habitats were visually displayed using the MDS (Non-Metric MultiDimensional Scaling; Clarke and Gorley, 2006) option in PRIMER 6; MDS was run using 100 random number starts. The two-dimensional solution presented here had a stress of 0.09, where stress is a measure of how well the ordination represents differences between samples; values lower than 0.2 are considered a good representation. Sample groups from habitats and sites were mapped onto the ordination plots.

Habitat Mapping

The proportion of pool and riffle habitat available along a 1 km reach of 20 urban streams was estimated by walking along the stream bed and measuring the length of each habitat type. The results were expressed as proportions of each habitat type across the three stream types – ‘forested’, ‘WSUD’ and ‘urban’.

Results

Water Quality

In June 2011, water quality parameters did not differ markedly across the three sites with the exception of samples taken downstream of a bridge on a tributary to Blunder Creek that was located between the different habitat types sampled at this site (Table 1). Water from the upstream site at Blunder Creek is forced through a gravel bar, the result of bitumen covering a sand slug upstream of the bridge. There is considerable reduction of iron (Fe) at this site and the downstream reaches contain large quantities of insoluble iron floc, causing a much lower pH and higher conductivity and turbidity.

Table 1. Spot water quality data collected from the three sites in June 2011.

Site	Dissolved Oxygen (mg/L)	pH	Conductivity (µS)	Turbidity (NTU)
Tingalpa Creek (forest)	9.1	7.725	281.5	23
Stable Swamp Creek (urban)	9.4	7.2	340	3.575
Blunder Creek upstream site (WSUD)	8.83	7.84	214.7	4.6
Blunder Creek downstream site (WSUD)	7.16	6.7	995	157

Community Assemblage Differences between Sites

Across the three sites in June 2011, 133 different taxa were collected. For pool samples, there was no significant difference between the sites for species richness ($F_{2,9} = 1.716$, $p > 0.05$) and Margelef diversity (d) ($F_{2,9} = 0.398$, $p > 0.05$). However, there was a significant difference in abundance ($F_{2,9} = 4.7$, $p < 0.05$) and Pielou Evenness ($F_{2,9} = 6.31$, $p < 0.05$), with Blunder Creek having significantly higher abundance (Figure 3) and lower evenness than Tingalpa Creek (Figure 3; Table 2). There were no significant differences in any summary community measure between sites for riffle habitats. However, for macrophyte/snag habitats there was a significant difference in Margelef diversity ($F_{2,6} = 9.58$, $p < 0.05$), with the Blunder Creek site having a significant lower overall diversity when compared with Tingalpa or Stable Swamp (Table 2).

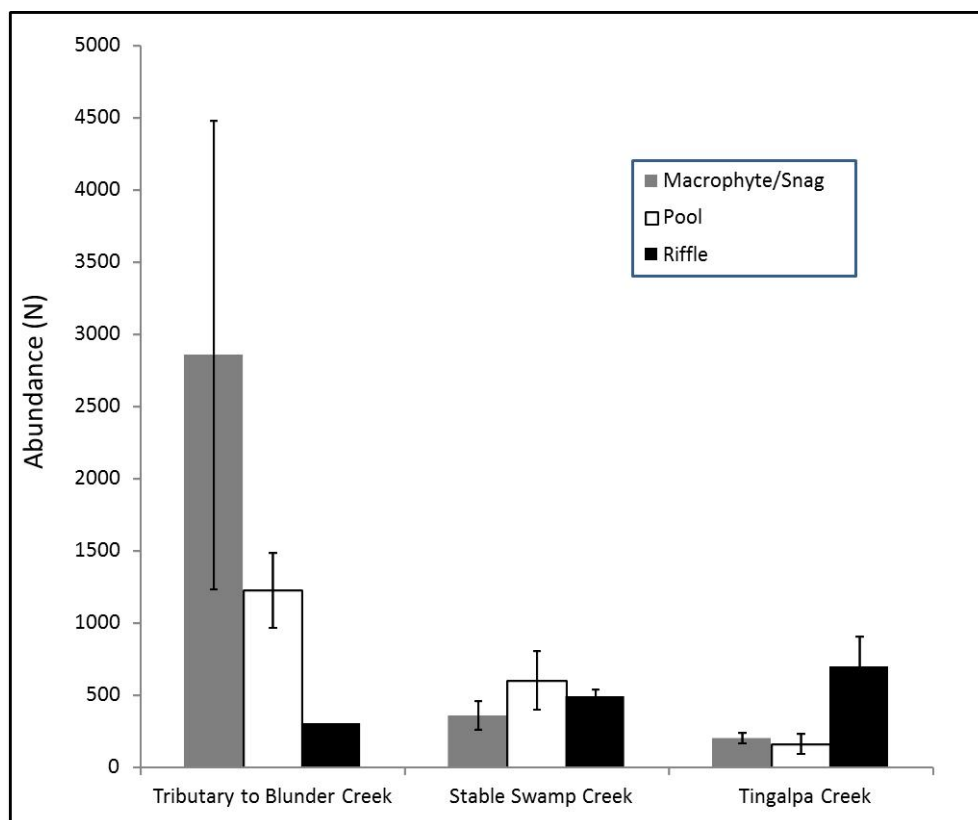


Figure 3. Mean ($\pm$ SE) total abundance for different habitats across the three sites; Tributary to Blunder Creek (WSUD), Stable Swamp Creek (Urban) and Tingalpa Creek (Forested).

Table 2. Mean ($\pm$ SE) for calculated summary community metrics for each habitat sampled at each site; Blunder Creek (WSUD), Stable Swamp Creek (Urban) and Tingalpa Creek (Forested).

	Richness (S)	Abundance (N)	Margelef Diversity (d)	Pielou Evenness (J')
Blunder Macrophyte	22.33 (3.8)	2858 (1619)	2.80 (0.25)	0.62 (0.03)
Stable Swamp Macrophyte	22.33 (1.7)	362 (99)	3.68 (0.11)	0.71 (0.01)
Tingalpa Snag	27.00 (3.6)	204 (35)	4.89 (0.51)	0.70 (0.02)
Blunder Pool	28.33 (2.7)	1227 (261)	3.89 (0.32)	0.62 (0.03)
Stable Swamp Pool	27.00 (3.1)	603 (204)	4.15 (0.49)	0.56 (0.06)
Tingalpa Pool	18.33 (6.7)	163 (71)	3.35 (1.10)	0.77 (0.04)
Blunder Riffle	20.00 (0)	308 (0)	3.32 (0)	0.68 (0)
Stable Swamp Riffle	17.67 (1.8)	493 (48)	2.68 (0.26)	0.68 (0.05)
Tingalpa Riffle	20.00 (0.6)	699 (206)	2.95 (0.13)	0.64 (0.01)

When focusing on the overall assemblage there were significant differences between sites regardless of habitats (ANOSIM Global R = 0.625, p = 0.01) and between habitats regardless of site (ANOSIM Global R = 0.628, p=0.01). All samples from the WSUD site, Blunder Creek, grouped close together regardless of habitat (Figure 4). Tingalpa Creek had the greatest between sample differences, represented by larger distances between points (Figure 4). While the highly urbanised site of Stable Swamp Creek had all pool samples grouping close to the Blunder Creek pool samples and riffle samples grouping close to riffle samples from Tingalpa Creek (Figure 4).

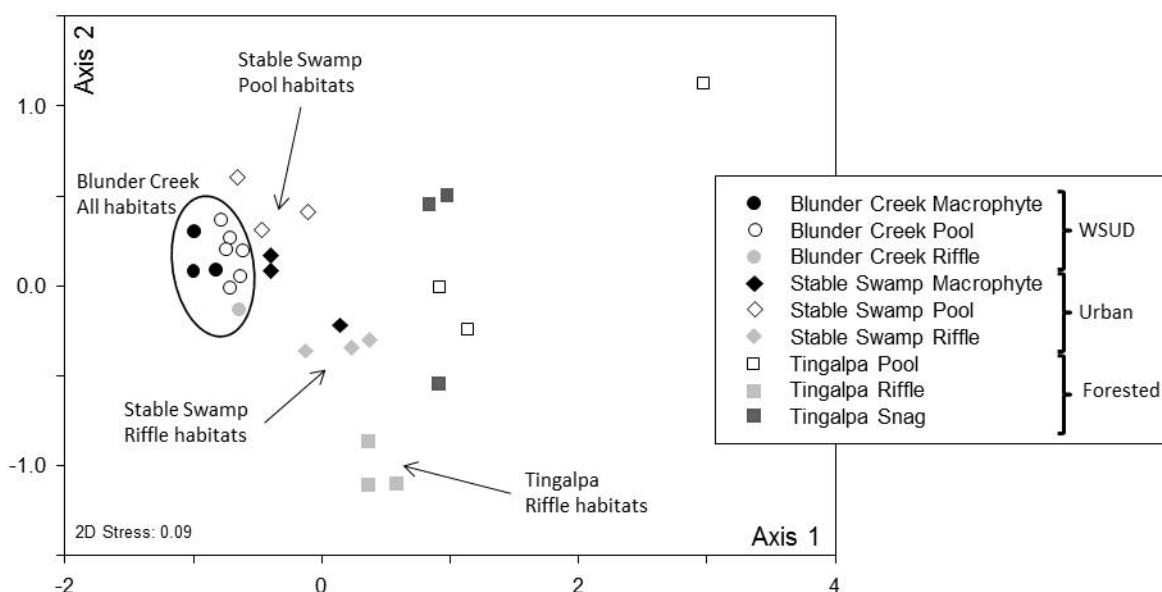


Figure 4. Two dimensional MDS ordination of samples based on species abundance. A priori 'site' and 'habitat' groupings have been mapped over the plot.

As the largest differences appeared to be between the pool and riffle habitats from Stable Swamp Creek and Tingalpa Creek, SIMPER was used to explore which taxa were driving within habitat similarity for each site. Interestingly, riffles from both Stable Swamp Creek and Tingalpa Creek were dominated by worms (Oligochaeta), the chironomids (Orthocladinae) and the caddisfly (*Cheumatopsyche* sp AV6), with riffles in Tingalpa Creek also containing the mayfly *Nousia*, not found in Stable Swamp Creek (Table 3).

Table 3. Percent contribution (SIMPER) for different taxa from habitats in Stable Swamp Creek (Urban) and Tingalpa Creek (Forested). Within group similarity is a measure of how similar samples within each group were to each other, if the value is 100% samples within the group are exactly the same, whereas 0% means completely different.

		Stable Swamp Creek (Urban)			Tingalpa Creek (Forested)		
Habitat (within group similarity)		Macrophyte (77%)	Pool (36%)	Riffle (61%)	Snag (43%)	Pool (20%)	Riffle (33%)
Microcrustaceans	Crustacea	26.97	29.39				
<i>Physa acuta</i>	Gastropoda	23.31	9.82				
Chironominae	Diptera	14.33	26.42	8.75	41.2		
Hydrobiidae	Gastropoda	11.8	10.32				
Oligochaeta	Oligochaeta	7.02		28.31			10.15
Orthocladinae	Diptera	4.78		27.12	9.18	26.46	20.12
<i>Cheumatopsyche</i> sp AV6	Ephemeroptera			10.14			33.52
Baetidae	Ephemeroptera			8.17	10.95		
Simuliidae				7.88		6.16	
<i>Caridina</i> sp.					5.07	35.63	
<i>Triplectides ciuskus ciuskus</i>	Trichoptera					8.82	
<i>Nousia</i> sp AV 1 or 2	Ephemeroptera						19.32
<i>Austrophleboides</i> sp.	Ephemeroptera				13.29		

Habitat Quantity

Habitat mapping suggested that pools were the dominant habitat type across streams in the Brisbane region of SEQ. There were no significant differences in the relative percentage of available pool and riffle habitat (Figure 5). However, although not quantified, there were observed differences in the ‘quality’ of riffle habitat across the three stream groups, with urban and WSUD streams having riffle habitats of sand and fine gravels, while forested streams have cobbles and boulder dominated riffles.

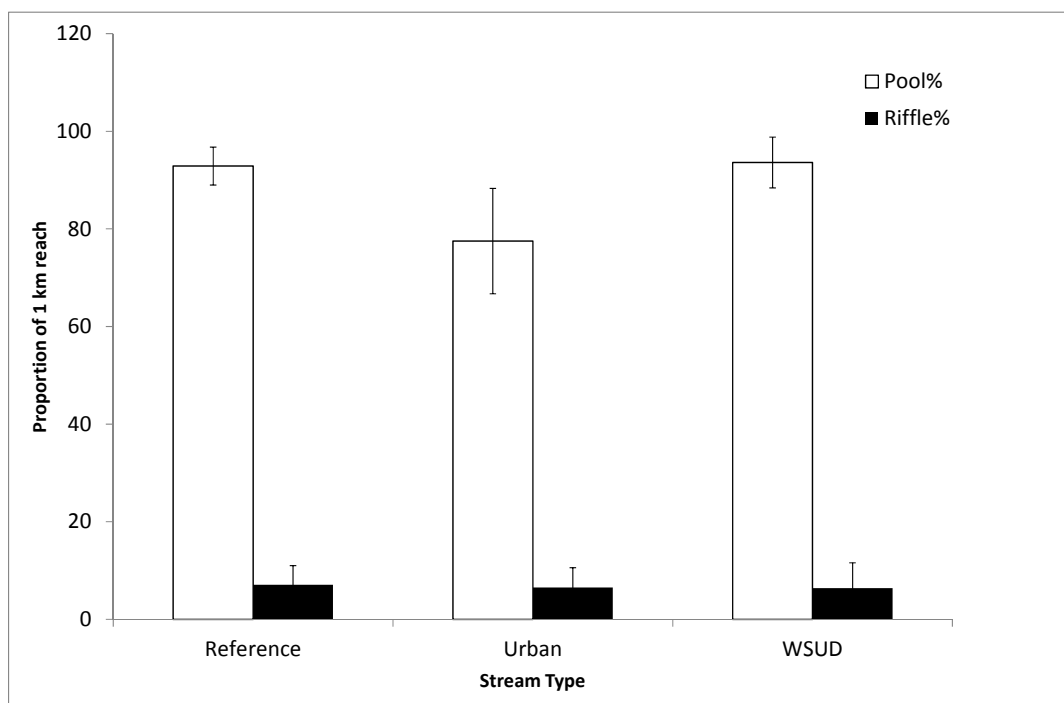


Figure 5. Relative proportion of pool and riffle habitat from a 1 km reach across 20 streams within the Brisbane region of SEQ.

Conclusions

This initial study of macroinvertebrate assemblage composition at the habitat level of urban streams in Brisbane suggests the main differences in assemblage composition between urban and forested streams occurs in pool habitats. Riffle habitats in urban streams contained some sensitive taxa and were more similar to riffle habitats of forested streams than the pool habitats of urban streams. Pool environments in the urban streams sites from this study were severely degraded, with a very unstable substrate of fine silts. They were also devoid of much structural habitat such as wood and leaf litter debris. In comparison, pool habitats in the forested sites on Tingalpa Creek were more structurally complex containing wood, leaf debris dams and emergent macrophytes.

While the habitat mapping did not suggest a lack of riffle habitat in urban streams, we suggest there may be differences in riffle ‘quality’ between urban and forested streams; urban stream riffles were observed to have smaller particle sizes (sands and gravels) compared to forested streams where riffles comprised boulders and cobbles. This suggests a possible mechanism for the poor health of urban streams, while flow itself may be partially responsible by directly dislodging invertebrates, its impact on sediment delivery to the stream through increased erosion may change habitat quality and therefore availability.

Future aspects of this study include measuring the quality of the riffle habitats along the urban stream gradient and exploring changes in riffle and pool assemblage composition over the pre- and post-wet summer period.

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Potential Health Risks from Pathogens in Alternative Waters

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Summary

The presence of enteric pathogens in the receiving waters following a storm event can be a serious public health concern. Stormwater samples ($n = 10$) collected from two sites in Melbourne, two sites in Brisbane and one site from Sydney were analysed for the presence of human adenovirus, *Campylobacter jejuni*, and sewage-associated markers, *Metahonobrevibacter smithii* *nifH* gene marker and *Bacteroides* HF183. In addition, the samples were also tested for the presence of selected pharmaceuticals and food additives (paracetamol, aspirin, acesulfame, and caffeine) which are linked with sewage contamination. Based on the *Enterococcus* spp. numbers detected in the stormwater runoff, the quality of the captured stormwater would not meet the recommended limits for category D (<501 Enterococci per 100mL⁻¹) under Australian guidelines for managing risks in recreational water. The regular occurrence of human adenovirus at two stormwater sampling sites during storm events in Brisbane in relatively high numbers (148 to 9×10^3 pdu L⁻¹) further highlights a potential health risk from the use of untreated stormwater. Among eight stormwater sites tested, five (63%) were positive for five markers, two (25%) were positive for four markers and the remaining site was positive for two markers, suggesting ubiquitous sewage contamination in the urban environment. Based on these results, it is recommended that some degree of treatment of captured stormwater should be carried out if it were to be used for non-potable purposes.

Keywords

Adenovirus, pathogens, stormwater, faecal pollution, microbial source tracking, faecal indicator bacteria.

Introduction

Captured urban stormwater could be an alternate source of water to augment non-potable and potable water supplies within cities and other urban areas. However, the potential presence of faecal contamination in stormwater runoff (Parker *et al.*, 2010; Sauer *et al.*, 2011; Sidhu *et al.*, 2012) could negatively impact public health if polluted water is used for recreational or non-potable purposes such as gardening and landscaping irrigation. Thus the identification of potential sources of faecal pollution in stormwater runoff is imperative to minimise public health risks associated with the exposure to captured stormwater.

Human enteric pathogens can find their way into stormwater and subsequently surface water through leaking sewer systems, sewer pumping station overflows, seepage from septic systems, agricultural runoff and the discharge of treated wastewater into aquatic environments (Gaffield *et al.*, 2003; Noble *et al.*, 2006; Rajal *et al.*, 2007). High numbers of enteric viruses, bacteria and protozoa have been reported in stormwater runoff indicating the presence of sewage pollution (Noble *et al.*, 2006; Sercu *et al.*, 2009; Cizek *et al.*, 2008). Hence, monitoring for faecal pollution in stormwater is required to make an assessment of health risks from exposure to harvested stormwater and the extent of treatment required prior to its use as alternative water.

Advances in molecular techniques have made it possible to rapidly detect enteric pathogens of concern in stormwater (Rajal *et al.*, 2007; Dyke *et al.*, 2009; Mull and Hill 2009). Therefore, direct monitoring of pathogens such as enteric viruses in addition to indicator bacteria may be a better approach for risk identification. Microbial source tracking (MST) methods have been used to identify potential sources of surface water contamination (Parker *et al.*, 2010; Sauer *et al.*, 2011). In addition, pharmaceuticals and food ingredients such as caffeine and acesulfame (Nakada *et al.*, 2008; Buerge *et al.*, 2009) have also been used as chemical markers for the presence of sewage in surface water. The use of MST tools/chemical markers in conjunction with monitoring for pathogens and indicators has the potential to provide information on the extent of faecal pollution and potential sources of contamination.

The primary aim of this study was to determine the presence of pathogens and sewage-associated markers in stormwater samples to determine potential health risks linked to the use of captured stormwater as alternative water.

Methods

Stormwater Sampling

Stormwater samples ($n=9$) were collected from the Fitzgibbon (FG) stormwater drain ($n=3$), Hornsby site (HN) ($n=2$), Banyan Creek (BA) ($n=2$), and Smith street (SM) ($n=2$). All samples were collected as composite samples following storm events. Water samples (20-40L) were collected in sterile plastic containers and transported to the laboratory for processing after the storm event. A brief site description and potential sources of contamination is presented in Table 1.

Table 1. Stormwater sites and brief site description.

Sites	Land Use	Potential Source of Faecal Pollution
Fitzgibbon Drain, Brisbane	Residential	Sewage pipe network, small numbers of horses and cattle
Hornsby, Sydney	Urban roads/carpark	Sewage pipe network
Banyan Creek, Melbourne	Residential	Sewage pipe network,
Smith Street, Melbourne	Commercial	Sewage pipe network, industry
Makerston Street, Brisbane	City, commercial	Sewage pipe network

Quantification of Faecal Indicator Bacteria (FIB)

Quantification of FIB (*E. coli* and *Enterococcus* spp.) was performed by the membrane filtration technique. Briefly, 1 and 10mL samples were filtered through 0.45µm nitrocellulose (Millipore) filters (47mm) and placed on respective selective agar plates in triplicate. *E. coli* was enumerated on Chromocult™ coliform agar (Merck) and *Enterococcus* spp. on Chromocult™ enterococci agar (Merck). Plates were incubated at 37°C overnight and then typical colonies were counted to determine the average number of colony forming units (cfu) 100mL⁻¹.

Detection of Adenovirus, MST and Chemical Markers

Stormwater samples (20 to 40L) collected from each site were concentrated by hollow-fiber ultrafiltration system (HFUS), using Hemoflow HF80S dialysis filters (Fresenius Medical Care, Lexington, MA, USA) as previously described (Hill *et al.*, 2005). Five mL of concentrate was centrifuged at 4750 rpm for 10 minutes and DNA was extracted from the resulting pellet with the PowerSoil DNA Kit (MOBIO Laboratories, CA, USA) following the manufacturer's instructions. Adenovirus was also detected by capture through adsorption to charged membranes. In brief, human adenovirus numbers in the stormwater samples were quantified from two sites in Brisbane. Stormwater samples (1L) were collected during the rising and receding limb of the hydrograph from each site during the storm event (Figures 1 and 2). Collected water samples were captured on negatively charged membranes by procedure outlined by Katayam *et al.*, (2002).

Real-time polymerase chain reaction (PCR) assays were performed using previously published primers, probes and cycling parameters (Heim *et al.*, 2003; Seurinck *et al.*, 2005; Ufnar *et al.*, 2006; Sidhu *et al.*, 2012). For each PCR experiment, positive controls (eg, corresponding plasmid DNA or genomic DNA) and negative control (eg, sterile water) were included. The PCR and quantitative PCR were performed using the Bio-Rad iQ5 thermal cycler (Bio-Rad Laboratories). Adenovirus numbers detected in the stormwater were presented as PCR detectable units (pdu L⁻¹).

Pharmaceutical (paracetamol, aspirin), caffeine and artificial sweetener (acesulfame) are known sewage ingress ion markers. One litre of collected stormwater sample was processed through a GF/C filter (Whatman) with 1.2 µm pore size. The filtrate was then passed through solid phase extraction cartridge and finally eluted with an appropriate solvent. The extract was finally detected with GC/MS and LC/MS. The whole analysis was done in the Forensic and Scientific Services laboratory, Queensland Government, Coopers Plains 4108 (National Association of Testing Authorities, Australia accredited laboratory). Among the eight samples tested to-date, seven (86%), six (75%), five (63%), and four (50%) were positive for acesulfame, caffeine, paracetamol and aspirin respectively (Table 2). Although the detected chemicals were below the guideline values (except caffeine), their presence in stormwater samples indicated possible sewage ingress ion in stormwater.

Results

The numbers of FIB in water samples collected after the storm event ranged from 40 to 6560 100mL⁻¹ for *E. coli* and from 1930 to 22600 100mL⁻¹ for *Enterococcus* (Table 2). The numbers of *Enterococcus* spp. were generally tenfold or more higher than *E. coli* across all sites in the stormwater runoff.

Among the eight samples tested, four (50%), eight (100%), four (50%), and eight (100%) were positive for themicrobial source tracking markers *nifH*, HF183, *C. jejuni* and human adenovirus respectively (Table 2). In this study, all eight sites (100%) were positive for HF183 and adenovirus, five (63%) were positive for five markers, two (25%) were positive for four markers and one site tested positive for two markers.

Table 2. Sewage chemical markers detected in stormwater samples collected in various catchments in Australia.

Sites	FIB*		Pathogens		MST markers		Food markers		Pharmaceuticals	
	<i>E. coli</i>	Enterococcus	Adenovirus	<i>C. jejuni</i>	HF183	nifH	Acesulfame	Caffeine	Paracetamol	Aspirin
Fitzgibbon										
8/12/2011	3600	16700	+	-	+	+	+	+	-	+
17/04/2012	3560	11800	+	+	+	+	+	+	+	
Hornsby										
15/02/2012	40	1930	+	-	+	-	+	+	+	+
20/02/2012	100	20900	+	+	+	-	+	+	+	+
Banyan Creek										
9/11/2011	3400	10200	+	-	+	-	+	+	+	-
6/02/2012	7200	22600	+	+	+	-	+	+	+	+
Smith Street										
30/09/2011	6560	152000	+	+	+	+	+	-	+	-
28/02/2012	-	7900	+	-	+	+	-	-	-	-

*= FIB numbers 100mL⁻¹

Five samples were collected over the hydrograph during storm event at Makerston Street and all tested positive for adenovirus (Figure 1). Adenovirus numbers varied from 148 to 400 pdu L⁻¹ with the highest number (400pdu L⁻¹) observed during the peak of the storm when water depth was > 0.40 meter.

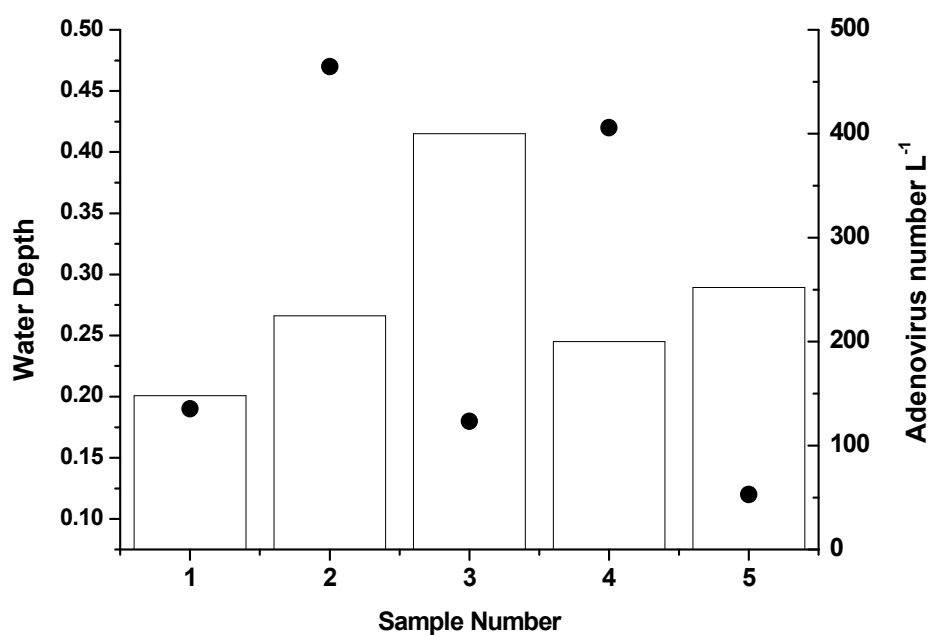


Figure 1. Adenovirus numbers over the hydrograph during the storm event at Makerston site. Dots represent water depth and bars represent adenovirus numbers.

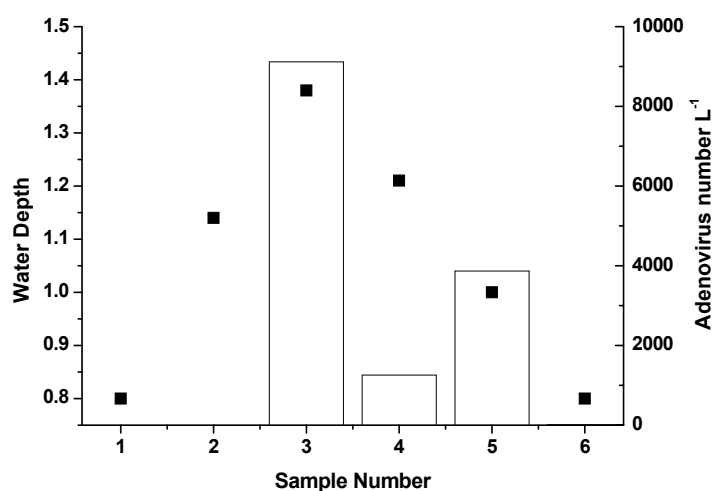


Figure 2. Adenovirus numbers over the hydrograph during the storm event at Fitzgibbon site. Dots represent water depth and bars represent adenovirus numbers.

At the Fitzgibbon site, higher numbers of adenovirus were detected ranging from 0 to 9000pdu L⁻¹, the maximum number of virus observed coincided with the peak of storm event which is represented by the maximum depth of the water flow at the site.

Conclusions

Urban stormwater runoff can mobilise and transport human pathogens from various non-point sources to receiving water bodies (Sidhu *et al.*, 2012). In this study, we tested stormwater samples collected after storm events for human pathogens, MST markers and chemical markers generally associated with sewage ingress.

Enterococcus spp. numbers detected in water samples collected from all sites after the storm event were generally higher than the recommended limits for category D (<501 *Enterococci* per 100mL⁻¹) under Australian guidelines for managing risks in recreational water by several orders of magnitude after the storm events (NHMRC, 2008). High FIB numbers observed after the rain events were also similar to what had been previously reported in the literature (Brownell *et al.*, 2007; Parker *et al.*, 2010).

Campylobacter jejuni was detected in four samples (50%) from all four sites tested in this study (Table 2), which suggest that there is a potential risk of presence of other bacterial pathogens such as *Salmonella enteric* in the stormwater. This is a cause of concern as *C. jejuni* and *Salmonella enteric* are major cause of bacteria dysentery in humans. At the stage of writing this paper data on the quantification of *C. jejuni* numbers in the stormwater number is pending.

Human adenovirus was detected in all stormwater runoff samples from all sites. In addition, relatively high numbers of adenovirus (148 to 9 x 10³ pdu L⁻¹) were detected at Fitzgibbon drain and Makerston Street sites in Brisbane during intensive sampling over the hydrograph. This finding is in agreement with previously reported adenovirus in surface water and stormwater numbers in the range of 1 x 10² to 10⁵ L⁻¹ (Sauer *et al.*, 2011; Hamza *et al.*, 2009; Muscillo *et al.*, 2008). However, presence of this human specific virus across all sites tested, especially in the stormwater runoff from commercial areas, suggests much wider sewage contamination than originally anticipated. The detection of adenovirus is also an indication that other human pathogens such as norovirus, rotavirus and *Cryptosporidium* could be in the water, thus further increasing the potential health risks. Further, investigation of the prevalence of other enteric virus (norovirus, rotavirus and *Cryptosporidium*) is currently under way to develop a better understanding of the extent of potential health risks.

The possible involvement of human sewage as a potential source of adenovirus and other pathogens was also investigated by using microbial and chemical source tracking methods. Out of eight samples tested, five (63%) were positive for five of the six chemical sewage contamination markers. This suggests that human faecal contamination is potentially the major source of microbial contamination of the stormwater. The wide presence of human adenovirus in the urban stormwater runoff indicates that there is potentially significant human faecal contamination, as opposed to contamination from animals, and thus has much higher public health implications. Presence of human-specific HF183 *Bacteroides* and *nifH* marker along with the acesulfame, caffeine, paracetamol and aspirin in most of the samples tested suggests ubiquitous sewage contamination in the urban environment. Consequently, some degree of treatment of captured stormwater prior to its reuse for potable and non-potable purposes would be required for public health risk mitigation.

The results of this study also suggested that the presence of high numbers of FIB in surface water after the storm events cannot be attributed to non-human sources alone as human enteric pathogens were also found at these sites. This observation has important ramifications for stormwater runoff management, as testing for pathogens such as human adenovirus at known “FIB hot spots” may be a more effective approach for proper health risk assessment and also for identification and abatement of sources of contamination.

Acknowledgements

This research was undertaken and funded as part of the Urban Water Security Research Alliance, a scientific collaboration between the Queensland government, CSIRO, The University of Queensland and Griffith University. The authors would like to acknowledge the financial support obtained by the Cities as Water Supply Catchments program funded by the National Water Commission, the Victoria Smart Water Fund and a broad range of governmental and industry partners as listed on the program’s website (<http://www.watersensitivecities.org.au/programs/cities-as-water-supply-catchments/>)

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Water Quality of Logan's Dam as a Function of Ecosystem Variables, Climate Fluctuations and Application of a Monolayer

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Summary

Time series for ecosystem variables and water quality of Logan's Dam water storage were obtained for 2009-2011. Sampling was designed to include one year prior to application of an evaporation-reducing monolayer and one year after. Continuous application of the monolayer did not eventuate after the first year, however short-term trials were undertaken and the impact of these was assessed in the context of other factors' influence upon water quality and ecosystem processes. While distinct population changes were seen after monolayer application, this period also coincided with large water inflows to the dam and, as a result, the reason for the observed changes is not clear. Variation in water level, turbidity and density of cyanobacteria, probably driven by ammonia, could determine water quality in Logan's Dam. Monolayers may affect these variables through their effect on evaporation and gas exchange. Based on the present findings, we suggest that if monolayers would slow down the decline of water level during drought, this could decrease turbidity and increase zooplankton grazing, both of which would improve water quality.

Keywords

Evaporation losses, monolayer, ecosystem response, water quality.

Introduction

Monolayers offer a potential means by which to reduce evaporation from water storages. Monolayers are chemical films one molecule thick (~2 millionths of a mm), which produce a diffusion barrier on the water surface which increases the apparent surface boundary layer thickness, thereby, increasing resistance to evaporation. Application of monolayers can significantly reduce evaporation from water storages and save water during periods of drought. This study was conducted on Logan's Dam, a small water body near Gatton, in South East Queensland (SEQ), chosen as an appropriate model site to investigate evaporation processes and the effects of a monolayer applied to the whole lake. While monolayers have been shown to modify physical properties of water-air interface (Barnes, 2008), the knowledge about their effects on water quality is insufficient. Water quality in lakes is strongly influenced by ecosystem processes. Ecosystems are naturally highly variable due to seasonal changes and inter-annual variation in climate. So the tasks of the present investigation were: (1) to evaluate ecosystem variability; (2) to identify drivers of water quality; and (3) to predict potential effects of monolayer on these drivers and ecosystem processes associated with water quality. During the first 12 months of observations on Logan's Dam, the amount of rainfall was relatively low (595 mm as compared to the long-term average of 770 mm). After that, during the following 12 months, total rainfall in the area reached 1310 mm. So, the ecosystem underwent a shift from near-drought conditions to flooding. Hence, this study also provided a rare opportunity to analyse in a "natural experiment" the effects of a strong hydrological change.

Results

Fifteen time series based on fortnightly observations, which characterised physio-chemistry, nutrient concentrations and dynamics of biological components of the ecosystem were obtained for Logan's Dam from 2009 to 2011. Detrital turbidity and phytoplankton abundance, due to their high levels and extensive variation, turned out to be the most important water quality factors in Logan's Dam.

Lake Classification and Phytoplankton Limiting Factors

The observed values of phytoplankton biovolume (mean = 8, max = 108 mm³L⁻¹) and nutrient concentrations placed Logan's Dam ecosystem into the category of eutrophic-hypoeutrophic lakes (Wetzel, 2001). Total phosphorus, filterable reactive phosphorus, total nitrogen, and nitrogen oxides varied at concentrations which suggested that phytoplankton growth was not limited by these nutrient species (Figure 1). In contrast, ammonia stayed below threshold limiting concentration during long periods (Figure 1) and was correlated to total phytoplankton biovolume (Figure 2A). No such correlations were found for any other nutrient species. This suggested that phytoplankton could be limited by ammonia. Ammonia is believed to be the most preferred nutrient for cyanobacterial growth (Von Ruckert and Gianni, 2004; Donald *et al.*, 2011). It was suggested that its concentration in water can be affected by monolayers which reduce NH₄ volatilisation (Cai *et al.*, 1987). During the two years of observation, light conditions could also be limiting for algae because of very high turbidity (165-415 NTU) resulting in low Secchi transparency

(0.1-0.2 m). During warm seasons, phytoplankton was dominated by potentially toxic cyanobacterium *Microcystis aeruginosa*. This species explained strong fluctuations in total phytoplankton biovolume which spanned more than two orders of magnitude (Figure 3D). In large water storages of SEQ, the maximal range of variation may not exceed 54 times (Matveev, unpublished data), which is much less than in Logan's Dam.

Correlates of Turbidity

During the relatively stable dry period of 2009-2010, turbidity varied between 170 and 370 NTU and was strongly negatively correlated to water level (Figure 2B). During the wet period of 2010-2011, turbidity fluctuated at a very high level (330 – 380 NTU) and no correlation with water level was found (Figure 1D). Turbidity probably had a negative effect on zooplankton biomass in the dam (Figure 2C), reducing its grazing potential. It has been shown to inhibit zooplankton in other lakes too (Hart, 1990; Kirk, 1992; Butler, 1995). High zooplankton grazing is usually considered an important variable associated with good water quality (Carpenter and Kitchell, 1993). Water level was negatively correlated to electric conductivity (Figure 2E), which could be a result of the dilution effect of rains (Figure 2E).

Synchronisation of Peaks – Monolayer Effect?

Several monolayer treatments of Logan's Dam took place between January and late April 2011 (Figures 1 and 3, blue vertical lines). No significant responses were observed in the dynamics of nutrient species (Figure 1). On the other hand, the time series of pH (Figure 3C), phytoplankton biovolume (Figure 3D), zooplankton biomass (Figure 3E) and mean crustacean length (Figure 3F) showed distinct peaks, which roughly coincided with the period of treatment. Different timing within the period of treatment could be caused by different time lags determined by the nature of the variables, provided the monolayer had a significant effect. However, it is not obvious that the synchronisation of peaks was caused by the monolayer. Firstly, in the absence of the physical effect, ie, reduction of the evaporation by the monolayer (McJannet, personal communication), the effect could have been limited by chemical influence. Secondly, confounding factors such as large pumped inputs to the dam during this time could also be responsible for the peak synchronisation. Unfortunately, prolonged application of the monolayer did not take place due to supply issues therefore effects of monolayer on water quality are not certain. While the possibility of the chemical influence of the monolayer should not be dismissed, additional experimental testing will be needed.

Dry vs Wet Periods

Although results from monolayer application were uncertain, sample collection occurred through a distinct change from dry conditions (2009-10) to wet conditions (2010-11), and this provided a unique opportunity to consider the dynamics of ecosystem processes and water quality through these periods. The highest biomass of plankton in reservoirs of SEQ is usually reached between October and April. A Student's t-test adjusted for serial dependence (Brockwell and Davis, 2002) was used to compare means for October-April in 2009-10 and in 2010-11. The assumed null hypothesis was that means of the variables under consideration were not significantly different in the dry and wet periods. Of all time series analysed, five variables showed significant differences at the level of $P < 0.05$. Mean zooplankton biomass and ammonia concentration have declined with the coming of the wet period, while turbidity and the concentrations of nitrogen oxides and filterable reactive phosphorus have increased in Logan's Dam. Mean biovolumes of total phytoplankton, cyanobacteria and grazable algae have not changed. It is interesting that while turbidity and chemical species responded to flooding conditions, phytoplankton showed considerable resilience to the strong climatic variation and its mean biomasses have not changed. Inhibition of zooplankton biomass could be due to higher turbidity (Figure 2C and Butler, 1995). Very high turbidity observed in 2010-2011 was probably a co-limiting factor for phytoplankton growth together with light limitation, explaining its stability.

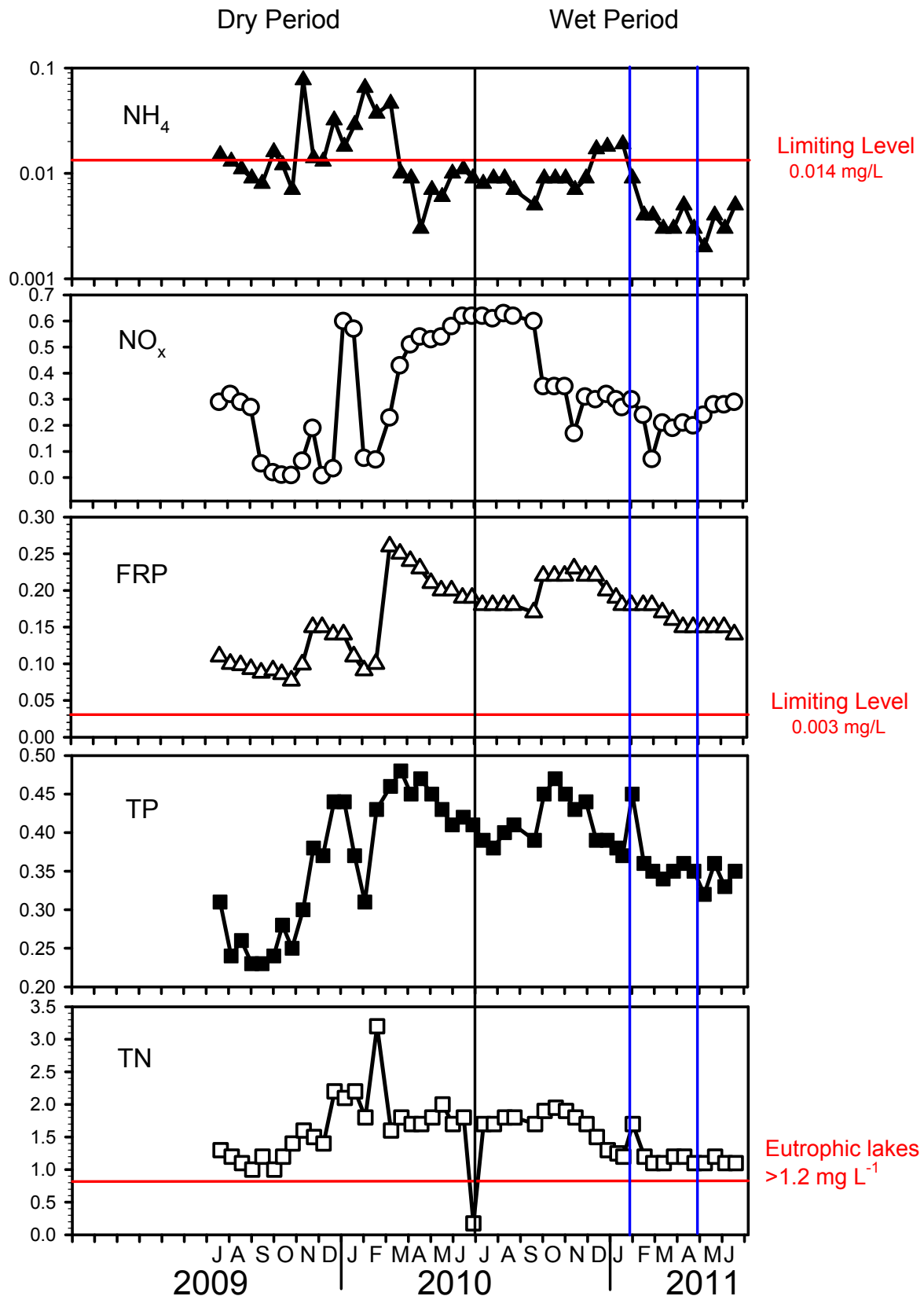


Figure 1. Dynamics of nutrient concentrations in Logan's Dam. NH₄= ammonia, NO_x= oxides of nitrogen, FRP= filterable reactive phosphorus, TP= total phosphorus, TN= total nitrogen. Black vertical line separates dry and wet periods, blue lines show the period of monolayer application. For NH₄ and FRP, red horizontal lines show threshold levels limiting phytoplankton growth.

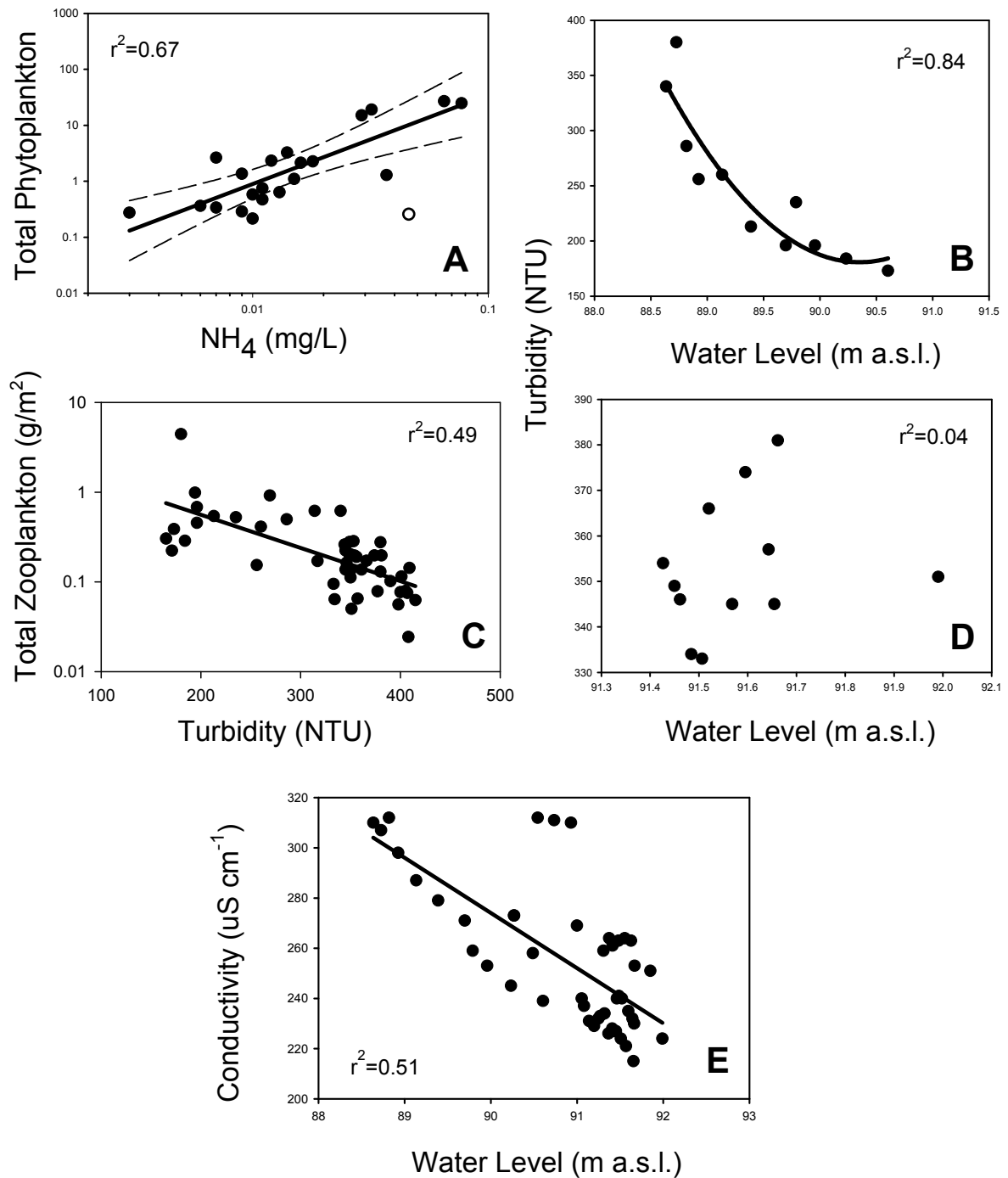


Figure 2. Correlations between Logan's Dam water quality variables. **A.** Phytoplankton biovolume vs. ammonia concentration. The hollow circle outside the dashed lines (99% confidence limits) applies to monolayer application. **B, D.** Turbidity vs. water level during the dry and wet periods accordingly. **C.** Zooplankton biomass vs. turbidity. **E.** Electric conductivity vs. water level.

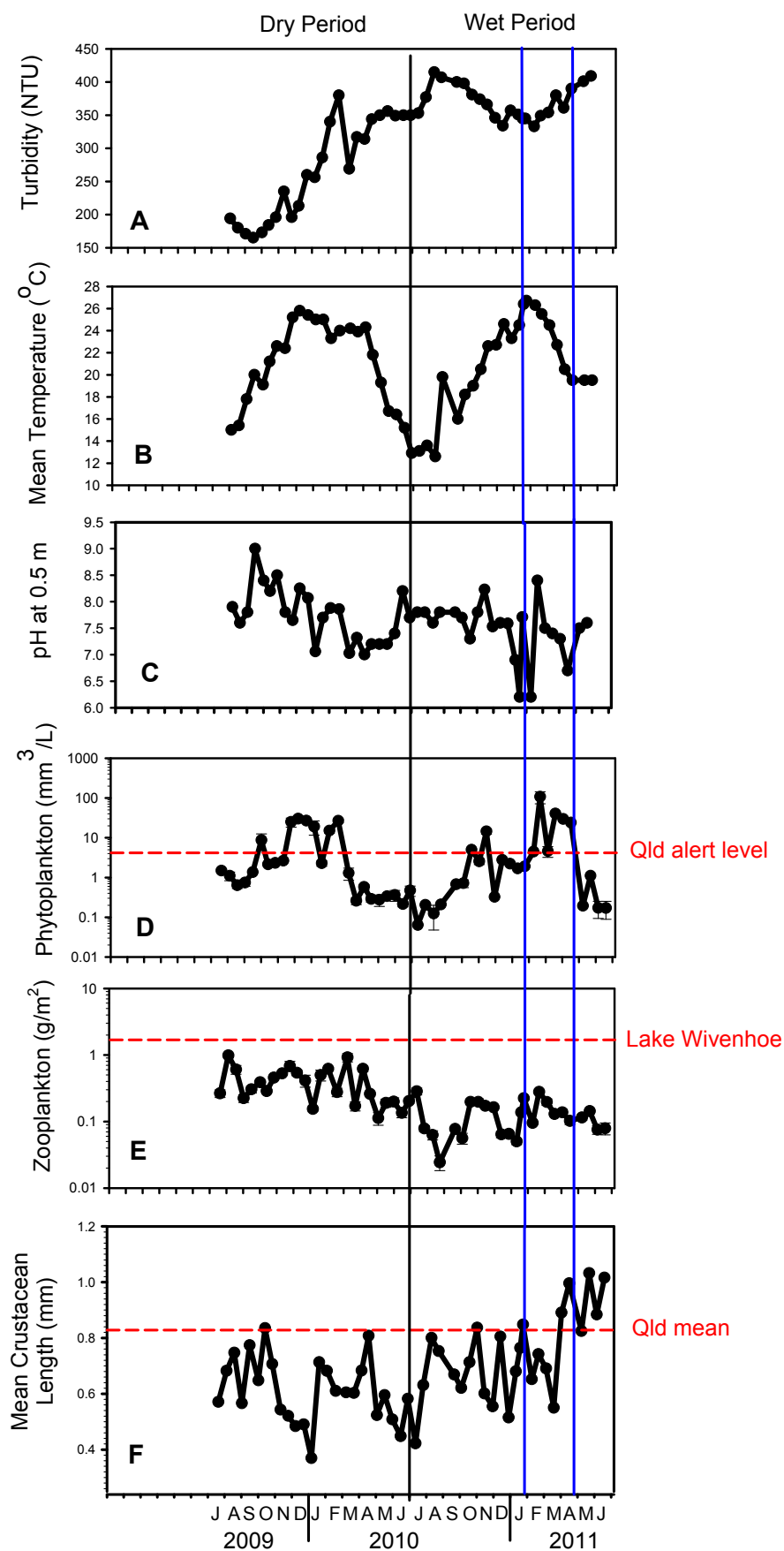


Figure 3. Seasonal changes in physio-chemical and biological parameters. Black vertical line separates dry and wet periods, blue lines show the period of monolayer application.

Conclusions

Although this study was unable to make any robust conclusions as to the ecological and water quality impacts of monolayers which could be used for reducing evaporation it did show that strong seasonal and inter-annual variation in ecological variables associated with water quality can be expected. This variation must be taken into account in future monolayer treatments of water storages. While distinct population changes were seen after monolayer application, this period also coincided with large water inflows to the dam and the reason for the observed changes is not clear.

Variation in water quality was largely driven by organic turbidity and abundance of the cyanobacterium *Microcystis*, but not by zooplankton grazing. Time series suggested that ammonia could control *Microcystis* dynamics. Within the dry year, turbidity was negatively correlated with water level, which suggested that if a monolayer would slow down evaporation; this may reduce turbidity and improve water quality. Biomasses of total phytoplankton, cyanobacteria and grazable algae did not differ between dry and wet years, demonstrating ecosystem resilience to climate variation. Water conductivity in Logan's Dam was negatively correlated to water level, probably illustrating the concentrating effect of evaporation and diluting effect of rains.

Water level, turbidity and density of cyanobacteria were among the most important drivers of water quality in Logan's Dam. Consequently, the factors which control these variables should receive serious consideration in monolayer experiments. As monolayers are believed to affect water level through reduction of evaporation, turbid systems like Logan's Dam may experience decreased turbidity and increased zooplankton grazing, both of which would improve water quality. Natural disturbance effects, such as heavy rains, have the potential of reducing total concentration of ions in the lake and modifying ratios of nutrient species, but in spite of this, phytoplankton may show surprising stability in biomass dynamics.

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Photodegradation of Dissolved Organic Matter: The Impact on Monolayers

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Summary

The loss of monolayer compounds from the surface of a water body is one of the major causes of variable field performance in reducing evaporative loss. The degradation of susceptible molecules by indirect photochemical reactions involving dissolved organic matter (DOM) in the aquatic microlayer may occur within minutes, reducing the half-life of the molecule by up to 94%. The susceptibility of three commercially available monolayer compounds Hexadecanol, Octadecanol and 2-octadecoxyethanol to indirect photodegradation was assessed using water sampled from eight water bodies (including clear, brown, turbid brown and black water bodies) in South East Queensland (SEQ). A decision matrix was developed using water quality attributes such as colour, source water, catchment type and season, to select the most appropriate monolayer based on photoreactivity of the dissolved organic matter in the water body.

Keywords

Monolayers, photodegradation, dissolved organic matter.

Introduction

A monolayer is a mono-molecular film applied to the surface of a water body to reduce evaporation, by inhibiting the escape of water molecules in the vapour phase (Barnes, 2008). Monolayers have shown the potential to reduce evaporation by up to 60%. However, other studies have shown much lower reductions in evaporation, with Craig *et al.* (2005) reporting a zero percent reduction in evaporation during field testing of the commercial product WaterSavr (a formulation of Hexadecanol and Octadecanol) on a municipal water storage at Capella, Queensland. Other field studies lasting several weeks further highlighted the variability in field performance with water savings ranging from 10-40%.

Factors responsible for variable field performance include mechanical disruption of the monolayer film due to wind and wave action. Wind damages the monolayer by blowing the film off the water surface (beaching), or into a concentrated area leaving most of the storage exposed (Barnes, 2008). The introduction of impurities or contaminants to the monolayer, such as dust or pollen landing on the surface, may also disrupt the monolayer. However, under low wind speeds, the spontaneous respreading properties of the monolayer soon re-establishes the film if sufficient molecules of the monolayer remain. Monolayer molecules such as Hexadecanol are susceptible to volatilisation, with half the compound lost to the atmosphere in as little as 48 hours after field application. Volatilisation, dissolution into the subsurface water and beaching can be minimised if longer carbon chain, more wind-resilient compounds producing a higher surface pressure are applied (Barnes, 2008). However, the high surface pressure induced from naturally occurring microlayers may prevent the monolayer from spreading over the entire water body, leaving areas of water exposed (Pittaway and van den Ancker, 2010). The microlayer is the biologically active zone at the air-water interface where hydrophobic, dissolved organic matter concentrates.

Monolayer molecules can also be broken down by microbial degradation or photodegradation. Microbial degradation is generally due to bacteria present in the water body (usually adapted microlayer species) using the monolayer compound as substrate. Monolayer compounds may also be susceptible to degradation from solar radiation, due to the high exposure at the water surface. Both direct and indirect photochemical reactions may occur when monolayer molecules are exposed to light, in particular ultra violet radiation (Barnes, 2008). Photochemical reactions take minutes to hours to degrade susceptible molecules, whereas microbial degradation requires hours to days. The ultra violet range ($\approx 295 - 400\text{nm}$) of solar radiation is the most reactive wavelength associated with these photochemical reactions. In South East Queensland (SEQ), solar radiation can reach intensities of $1,000 - 1,200 \text{ W/m}^2$ and greater on the hottest clear days of midsummer. On sunny days, a monolayer film can receive greater than 30 MJ/m^2 of total solar radiation (Bureau of Meteorology, 2011). Direct photolysis involves a molecule undergoing a chemical change due to the absorption of photons by chromophores (Zafiriou *et al.*, 1984).

Monolayer compounds have been tested for susceptibility to direct photodegradation (degradation by solar radiation, in particular UV radiation, eg Turnbull, 2006). Commercially available monolayer compounds Hexadecanol (C_{16}OH) and Octadecanol (C_{18}OH) were applied to distilled water and exposed to solar radiation (from a solar radiation generator) for seven days. Losses due to volatilisation were determined by placing distilled water monolayer samples in the dark. For Hexadecanol, losses due to volatilisation only were 22.8%, and losses due to direct photodegradation were 23%. For Octadecanol, losses were 18.6% and 18.8% respectively. These results confirm the monolayer compounds Hexadecanol and Octadecanol are not susceptible to direct photodegradation.

Monolayers are also subjected to indirect photochemical reactions when applied to the surface of a water body. In indirect photolysis, a reaction is initiated by light absorbed by a chromophore present in adjacent molecules, not the monolayer molecule itself. If the chromophore is regenerated it plays a photocatalytic role. If the chromophore changes irreversibly, it has undergone direct photolysis, which simultaneously causes indirect photolysis of other substances present (Zafiriou *et al.*, 1984). In these indirect photochemical reactions, Dissolved Organic Matter (DOM) present in the natural microlayer of all water bodies reacts with photons from solar radiation (in particular, UV radiation) to form reactive compounds at the surface of the water body which can then breakdown the molecules of susceptible monolayer compounds.

Indirect photodegradation of monolayer compounds will vary from one water body to the next depending on the nature of the photoreactive compounds present in the water body at the time the monolayer was applied. The photoreactivity of a water body is dependent on several variables including water colour, catchment type, source water, and season. These variables alter not only the quantity of photoreactive compounds in the water, but more importantly, the quality or photoreactive nature of the compounds. In our study, storages with similar environmental attributes had similar photoreactive properties.

In this paper, the susceptibility of three monolayers to direct and indirect photodegradation when applied to four selected natural water bodies characterised for their photoreactive properties was investigated. The results from these studies were used to construct a decision support matrix for the selection of the most appropriate commercially available monolayer to apply to a water body based on photoreactivity. Results from a clear water body (Figure 1(a) - Lake Dyer), a brown water body (Figure 1(b) - Narda Lagoon), a turbid brown water body (Figure 1(c) - Logan's Dam) and a black water body (Figure 1(d) - Pittaway Pond) will be reported.



Figure 1. Water bodies in the Lockyer Valley, SEQ, used to compare the susceptibility of three monolayer compounds to indirect photodegradation, (a) is Lake Dyer, (b) is Narda Lagoon, (c) is Logan's Dam and (d) is Pittaway Pond.

Methods

The three monolayer compounds compared in our study were Hexadecanol ($C_{16}OH$), Octadecanol ($C_{18}OH$) and 2-octadecoxyethanol ($C_{18}E_1$). Microlayer samples were collected from water bodies (Figure 1) filtered through $0.45\ \mu m$ syringe filters and added to petri dishes. Each of the three monolayer compounds was applied to the surface of water samples in petri dishes. The samples with monolayer applied were irradiated outside under natural solar radiation for eight hours. The amount of monolayer compounds present before and after irradiation was quantified by Gas Chromatography – Mass Spectroscopy (GCMS).

Photoreactivity of the water bodies was assessed as the breakdown rate and half-life of a specific concentration of the pesticide Pentachlorophenol (PCP) applied to water samples exposed to solar radiation. PCP is susceptible to direct and indirect photodegradation (Wong and Crosby, 1981), breaking down more rapidly in the presence of dissolved organic matter (Chi and Huang, 2004). For the photoreactivity assay, microlayer and subsurface water samples from the four water bodies were filtered ($0.45\ \mu m$) PCP added at a rate of $250\ mg/L$ PCP. The samples were placed in petri dishes and exposed to solar radiation for 30 minutes, with petri dishes removed from the light every 5 minutes. The PCP was then extracted into hexane after acidification with concentrated HCl and the remaining PCP was quantified using GCMS. Results were expressed as the time in minutes required for half of the original concentration of PCP to degrade (half-life).

The four water bodies characterised for photoreactivity and dissolved organic matter were used to compare the susceptibility of the three monolayer molecules to indirect photodegradation. Water from the microlayer of the four water bodies was placed in a petri dish. Each of the monolayer compounds were prepared in absolute ethanol and applied to the water surface of separate petri dishes at a rate of three times the theoretical monolayer (Brink *et al.*, 2010). The petri dishes were exposed to solar radiation for 8 hours, with samples removed from the light at every 30 minutes. The monolayers were then extracted into hexane and the remaining monolayer concentration was determined by GCMS. The susceptibility of the monolayer molecules to indirect photodegradation was calculated as the half-life of the original concentration expressed in hours.

Results

The losses recorded on distilled water for each monolayer were comparable to published volatilisation rates (Brooks and Alexander, 1960), confirming that Hexadecanol and Octadecanol do not undergo direct photodegradation (Turnbull, 2006), and that 2-octadecoxyethanol is also not susceptible to direct photodegradation.

Our results (Figure 2) confirm that PCP has a half-life of approximately 26 minutes when irradiated in distilled water (Wong and Crosby, 1981). The degradation of PCP in this case is direct photodegradation, as there is no DOM present for indirect photodegradation. Control samples placed in the dark for 30 minutes indicate PCP is not susceptible to volatilisation.

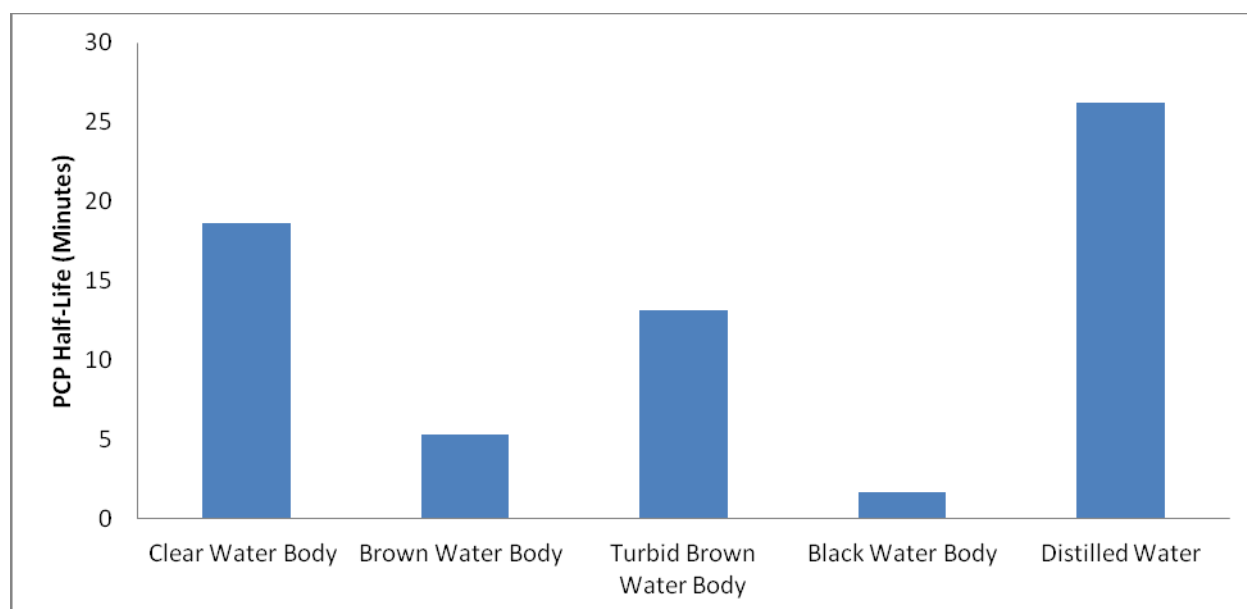


Figure 2. Half-life of 250 mg PCP when applied to microlayer samples taken from four water bodies exposed to approximately 1.8 MJ/m² of solar radiation for thirty minutes. Clear water body is Lake Dyer, brown water body is Narda Lagoon, turbid brown water body is Logan's Dam and black water body is Pittaway Pond.

The water bodies with more reactive DOM present in the surface microlayer decreased the half-life of the PCP to a greater extent than those storages with less reactive DOM. The reactivity of the DOM was measured as Permanganate Index (Hot Permanganate assay; Rump, 1988) per milligram Dissolved Organic Carbon (DOC) (measured on a Shimadzu TOC-V Total Organic Carbon Analyser). Lower standardised Permanganate Index results indicate higher reactivity. DOM reactivity is inversely proportional to PCP half-life. The clear water body had the lowest concentration of DOC and the least reactive DOM, and the longest half-life of PCP (after 19 minutes exposure, a reduction of 29%) (Figure 2). The turbid brown water body had the highest concentration of DOC; however the DOM was very unreactive resulting in the second longest PCP half-life (13 minutes exposure, a reduction of 50%). The brown water body had a high DOC concentration and DOM reactivity leading to it having the second shortest PCP half-life (five minutes exposure, a reduction of 80%). The black water body a high DOC concentration and a very high DOM reactivity, resulting in it having the shortest PCP half-life (two minutes exposure, a reduction of 94%).

The four water bodies characterised for photoreactivity and dissolved organic matter were used to compare the susceptibility of the three monolayer molecules to indirect photodegradation. The susceptibility of the monolayer molecules to indirect photodegradation was calculated as the half-life of the original concentration expressed in hours (Figure 3).

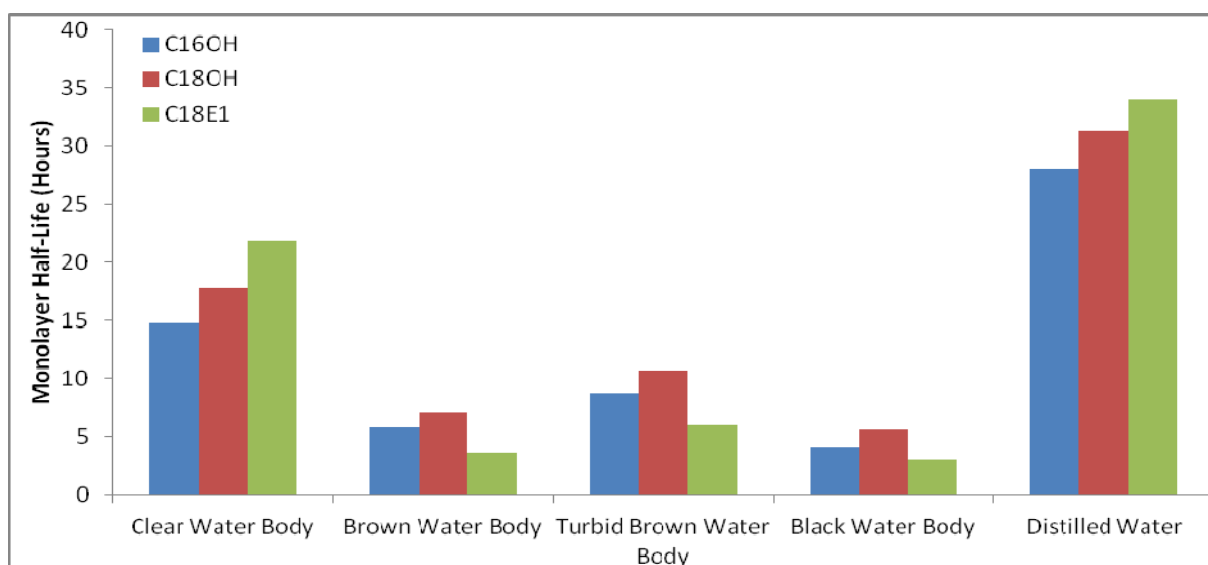


Figure 3. Half-lives of monolayer compounds C₁₆OH (Blue), C₁₈OH (Red) and C₁₈E₁ (Green) applied to microlayer water sampled from a clear water body (Lake Dyer), a brown water body (Narda Lagoon), a turbid brown water body (Logan's Dam) and a black water body (Pittaway Pond). After application to the water surface the monolayers were exposed to 14.16 MJ/m² of solar radiation, for a duration of 8 hours.

In the distilled water control, volatilisation was the predominant mechanism accounting for monolayer loss as there was no DOM present to facilitate indirect photodegradation, no pre-adapted microbial inocula, and the duration of the assay was less than eight hours. Results confirm the longer carbon chain molecules C₁₈OH and C₁₈E₁ were less susceptible to volatilisation than C₁₆OH, with the octadecyl ether head group more resilient than the fatty alcohol head group (Figure 3, and Barnes, 2008).

All three monolayer compounds were susceptible to indirect photodegradation in the presence of DOM, with half-lives reduced substantially below the results recorded for distilled water. The clear water body reduced the half-life the least, with the longer chain C₁₈E₁ and C₁₈OH molecules more resilient than the C₁₆OH molecule (reductions of 48%, 37% and 22% respectively). As with volatilisation, the octadecyl ether head group was more resilient to indirect photodegradation than the fatty alcohol head group. However, in the turbid, brown water, both fatty alcohol molecules (C₁₈OH and C₁₆OH) were more resistant to indirect photodegradation than the octadecyl ether molecule (C₁₈E₁). The greater susceptibility of the C₁₈E₁ monolayer to indirect photodegradation was also evident in the most photoreactive water sample, from the black water body. The longer chain, fatty alcohol molecule C₁₈OH was the most resilient (80% reduction), followed by C₁₆OH (85% reduction), with C₁₈E₁ highly susceptible (89% reduction).

Across all three coloured water bodies, the best performing monolayer was C₁₈OH. The compound most sensitive to indirect photodegradation was the C₁₈E₁ monolayer, which suffered a reduction in half-life ranging from 79 to 89% when placed on coloured or turbid water. The relative reduction in the half-life of the three monolayer compounds (Figure 3) is proportional to the reduction in the half-life of PCP (Figure 2), which is a function of the reactivity of the DOM present in the water. The more reactive the DOM, the greater the rate of direct photodegradation of those molecules susceptible direct photodegradation. Also, the more reactive the DOM, the greater the generation of more chemically reactive molecules produced during direct photodegradation of DOM (eg singlet and triplet oxygen species) driving indirect photodegradation. Seasonal sampling data indicates the concentration and reactivity of DOM in eight water bodies monitored in SEQ differs both spatially, and over time. The major water catchment variables correlated with photoreactivity include water colour, source water type, catchment type, rainfall, and season.

A decision tree (Figure 4) was developed using these variables to select the most appropriate monolayer for a specific water body. Dry and wet refer to the occurrence of rainfall intensity likely to dilute DOM in large, clean water bodies, or conversely, to add to the concentration of DOM in brown and black water bodies receiving high amounts of leaf and bark litter in overland flow. The output of the decision tree highlights the impact that susceptibility to photodegradation may have on the field performance of a candidate monolayer molecule. Under clean water conditions, the monolayer C₁₈E₁ is highly resilient with a half-life equivalent to approximately 32 hours. However, in the presence of higher concentrations of reactive DOM, the half-life dropped to as little as 2 hours. Monolayer products must be biodegradable to minimise environmental harm, but they must also be resilient for a matter of days to be cost-effective as a strategy for reducing evaporative loss. Susceptibility to photodegradation must be a key selection criterion in monolayer product development programs to avoid the variable field performance of previous trials that discouraged the commercial adoption of the technology.

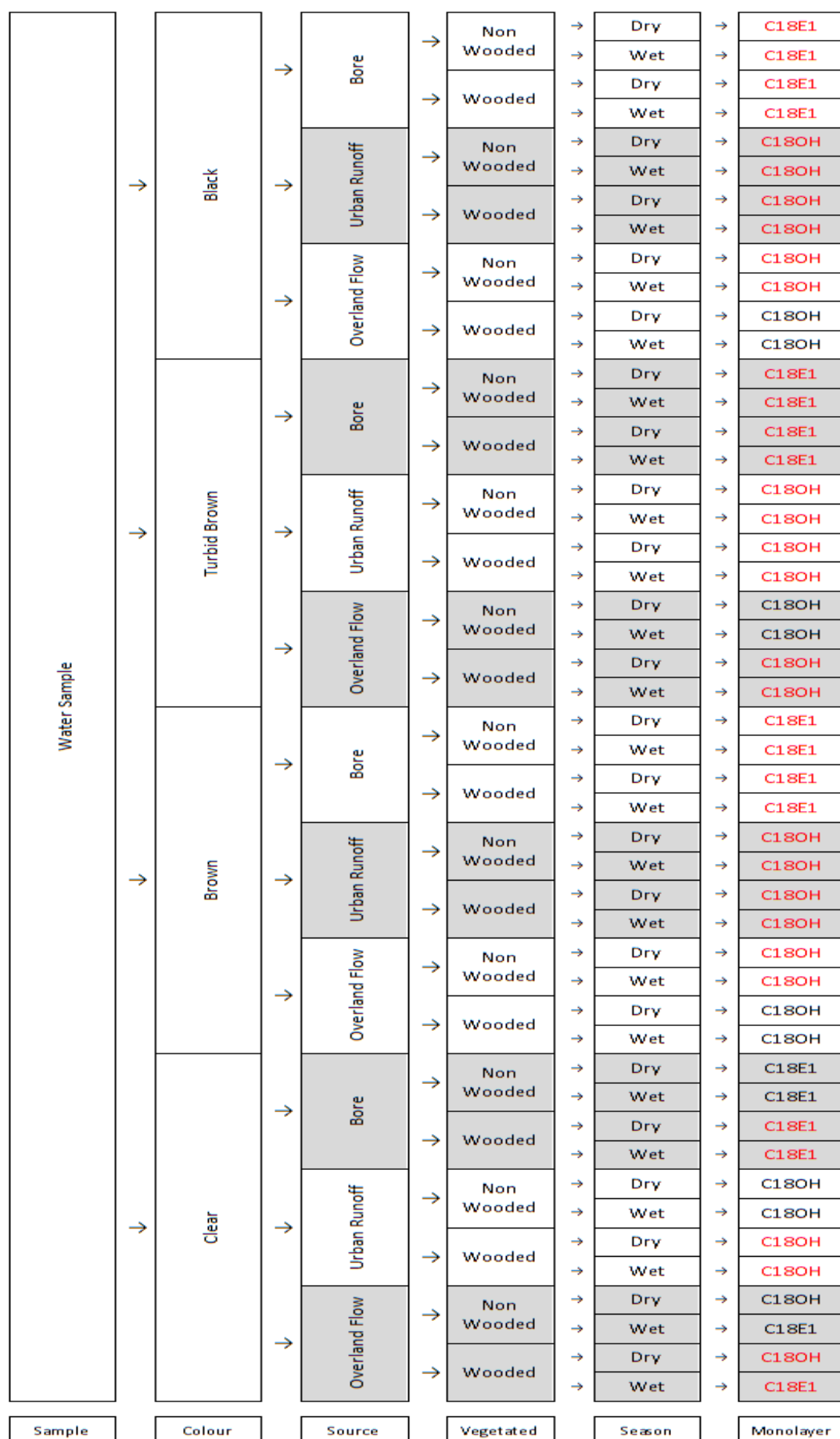


Figure 4. Decision tree for the selection of the most appropriate commercially available monolayer compounds based on photoreactivity of DOM in the water body (C18E1 – 2-octadecoxyethanol and C18OH – octadecanol). Monolayer recommendations - black are determined from actual experimental results, those in red are based on attributes of those water bodies. NOTE: C16OH – hexadecanol is not recommended for any water body due to its susceptibility to volatilisation.

Conclusions

The loss on monolayer compounds from the surface of a water body is one of the major causes of variable field performance in reducing evaporative loss. Susceptibility to wind and volatilisation are key factors that affect monolayer performance, and the results of our research indicate susceptibility to indirect photodegradation must also be considered. Of the three commercially available monolayer compounds studied, C₁₈E₁ showed the most resilience to indirect photodegradation when applied to a clean water body. However, when applied to brown, turbid brown or black water, it was the least resilient. Of the two fatty alcohol molecules, C₁₈OH was the most resilient in the presence of DOM, with a half-life varying from approximately 11 hours to 6 hours as the concentration and reactivity of the DOM increased. Monolayers must form a condensed film to be effective, and must have a half-life of at least 6 days or so to be cost-effective. The Decision Support matrix developed in our study can be incorporated into a Decision Support Framework (Brink *et al.*, 2010) to ensure the most appropriate monolayer product is selected for a given storage. Accounting for all factors that affect monolayer performance is necessary if the variability encountered in past field trials is to be reduced, and susceptibility to photodegradation is a key factor that must also be included.

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The Impact of Artificial Monolayers on Water Quality: Data from Tank and Field Trials

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Summary

In conjunction with the Logan's Storage water quality monitoring program, a tank trial was conducted using suspended physical covers to control wind speed and light intensity. The monolayer octadecanol was applied twice weekly to one tank, continuously for 4 months. An unspecified monolayer was applied at Logan's on selected days with wind less than 12 m sec⁻¹ and no rain. Contrary to expectations, the monolayer did not increase water temperature. A condensed monolayer insulates the air-water interface, buffering water temperature from heat gain and heat loss. Phytoplankton populations in the tanks were more sensitive to changes in light intensity, water stratification, and toxins off-gassing from a new high density polyethylene liner, than the continuous application of the octadecanol monolayer. These results highlight that even under the extreme case of continuous application over four months, there was no conclusive evidence of the octadecanol monolayer adversely affecting water quality.

Keywords

Monolayer, air/water interface, microlayer, suspended covers, enrichment, water quality.

Introduction

A key limitation of artificial monolayer research is film disruption caused by wave turbulence and wind greater than about 12 km hr⁻¹ (Barnes, 2008). Monolayer molecules may be lost via beaching on the shoreline, or may be emulsified and mixed in subsurface water. Artificial monolayers retard evaporation at the molecular level, by calming capillary waves and reducing the energy available for evaporative loss at the air/water interface. The energy available for evaporation is inversely proportional to the resistance imposed by the liquid thermal and gaseous boundary layers (Katsaros, 1980), the zone most affected by the presence of a monolayer. Ecologists refer to this biologically significant zone as the microlayer (Norkrans, 1980).

Both energy (heat) and mass are transferred when water molecules evaporate (Wells *et al.*, 2009). The removal of heat at the surface lowers the temperature, increasing the surface tension (free energy) of water within the microlayer (liquid thermal boundary layer) above the free energy of the bulk water. The natural convective circulation induced as this cooler, gravitationally unstable water descends (Katsaros and Garrett, 1982), increases the energy available for evaporation. Under light winds, the presence of a condensed monolayer calms capillary waves and natural convective circulation, reducing evaporation. Theoretically, this change in the energy flux within the microlayer should increase the temperature of the water, and reduce gaseous exchange (McJannet *et al.*, 2008). Ecologists are concerned that what may seem a small change within the microlayer, may adversely affect key biological processes that occur within this zone.

The Evaporation Loss project at Logan's Storage in the Lockyer Valley monitored the impact of applying an artificial monolayer on water quality. In conjunction with this study, two 10 m diameter tanks at the University of Southern Queensland (USQ) were fitted with suspended physical covers and lined with high density polyethylene (HDPE) to control wind speed and seepage. The monolayer applied twice-weekly to one of the tanks was a fatty alcohol (octadecanol, C18OH), formulated to improve spreading (Herzig *et al.*, 2011). The monolayer applied to Logan's Storage was an undisclosed experimental product. At both locations, water quality parameters were monitored every two weeks. At Logan's, the limnology was also monitored (V Matveev, this volume). A 70 mL sample of the water column was used to inoculate the three tanks in the USQ study, which were filled with municipal potable water. Changes in bacterial populations known to degrade monolayers (phenol-degrading bacteria, Pittaway and van den Ancker, 2010b), biochemical oxygen demand (BOD) and phytoplankton were monitored in microlayer and subsurface water sampled at the start and end of each cover trial. At Logan's, the population of phenol-degrading bacteria and the concentration of total organic carbon were also monitored immediately prior to and during monolayer application.

In the USQ tank study, two covers differing in colour and wind resistance were applied to two of the tanks in series (black covers, white covers, and black and white covers), with the third tank remaining uncovered. C18OH monolayer was applied twice-weekly to one tank only, at a rate equivalent to three times the amount required to form a condensed monolayer. The surface to volume ratio of the USQ tanks was much smaller than Logan's storage (water depth 0.7 m in 10m diameter tanks, Logan's nominal depth 4 m area 16 ha), indicating any adverse impacts of monolayer application should be more readily detected in the tanks. Evidence of the formation of a condensed monolayer was assessed by comparing the change in the surface deviation temperature (water surface temperature

minus the subsurface temperature, Gladyshev, 2002) averaged on an hourly basis for the covered tanks with and without monolayer application (monolayer and clean respectively). For Logan's Storage, two consecutive days before and during monolayer application when wind speed was predominantly less than 12 km hr^{-1} and relative humidity less than 60%, were selected for the comparison of surface deviation temperatures.

Results

The black fabric reduced the velocity of the air under the cover by 95%, and reduced light intensity by 86% (Gallego-Elvira *et al.*, 2010). The white fabric reduced air velocity by 44% and light intensity by 60%. However, the energy absorbed by the black cover during the day heated the air under the cover, increasing the air to water heat flux ($T_{\text{air}} - T_{\text{subs}}$ in Figure 1). The calm conditions below the black cover maintained a condensed monolayer, evident as a change in the surface deviation temperature (surface – subsurface temperature, $T_{\text{float}} - T_{\text{subs}}$ in Figure 1). Theoretically, applying a condensed monolayer should increase water temperature (McJannet *et al.*, 2008). Our results indicate the monolayer insulated the microlayer from the high air to water energy flux generated under the black cover. This insulating effect of artificial monolayers reducing heat gain under a positive heat flux, and reducing heat loss under a negative heat flux, has been reported in the literature (the dual effect, Gladyshev, 2002). Selecting two consecutive days from the Logan's Storage trial when turbulence was sufficiently low for a condensed monolayer to form also confirmed the formation of a condensed monolayer during the January application period (Figure 2). The air to water heat flux at Logan's was lower than the energy flux under the black cover, but again, monolayer application appeared to reduce the surface deviation temperature (infrared sensor used for water surface temperature, $T_{\text{WIR}} - T_{\text{subs}}$) by about 2°C . Data for Logan's Storage is more limited, as wind was often above 12 m sec^{-1} , which disrupted and beached the monolayer product.

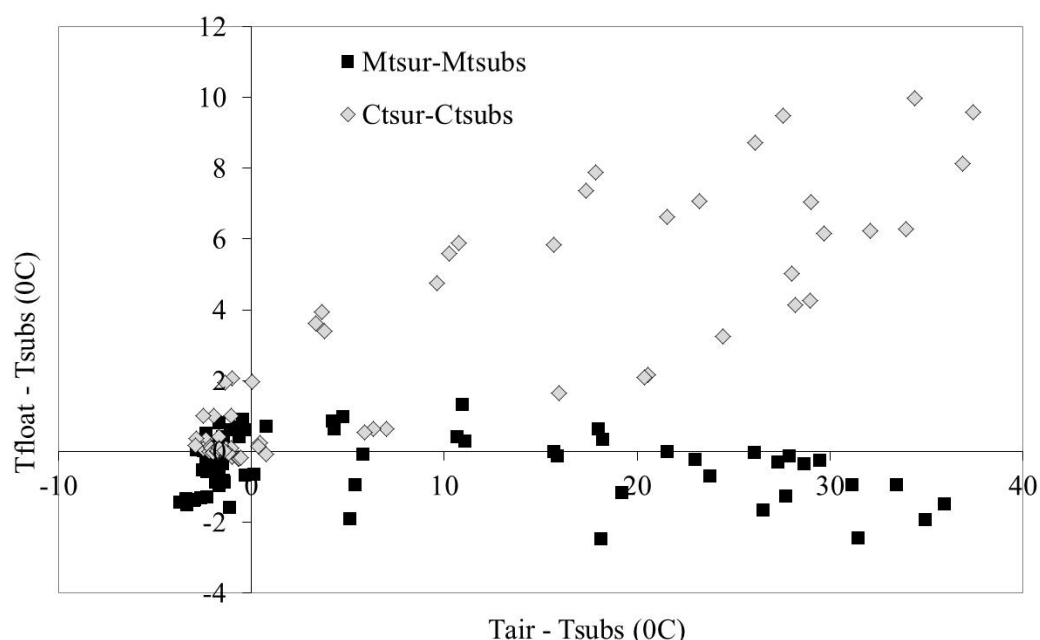


Figure 1. Confirmation of a condensed monolayer in above-ground tanks covered with a double black polyethylene cover. The hourly average surface deviation temperature recorded over 2 days on the clean water tank (Ctsurf – Ctsubs) is proportional to the air to water heat flux ($T_{\text{air}} - T_{\text{subs}}$). The condensed monolayer has insulated the water from heat gain, with the surface deviation temperature (Mtsurf – Mtsubs) well below that of the clean water surface.

The reduction in light intensity and the thermal stratification induced by the calm conditions under the black cover in the tanks, favoured cyanobacterial (blue-green algal) populations (Figure 3). Prior to cover installation (<21/1/2010), chlorophyte (green) algal species predominated, dropping three orders of magnitude in the covered tanks (T1 and T2) by the end of the black fabric trial (23/2/2010). The greater light intensity and isothermal water temperatures induced by turbulence under the white cover restored chlorophyte populations close to levels in the uncovered tank, dropping once again when the black fabric was reinstalled.

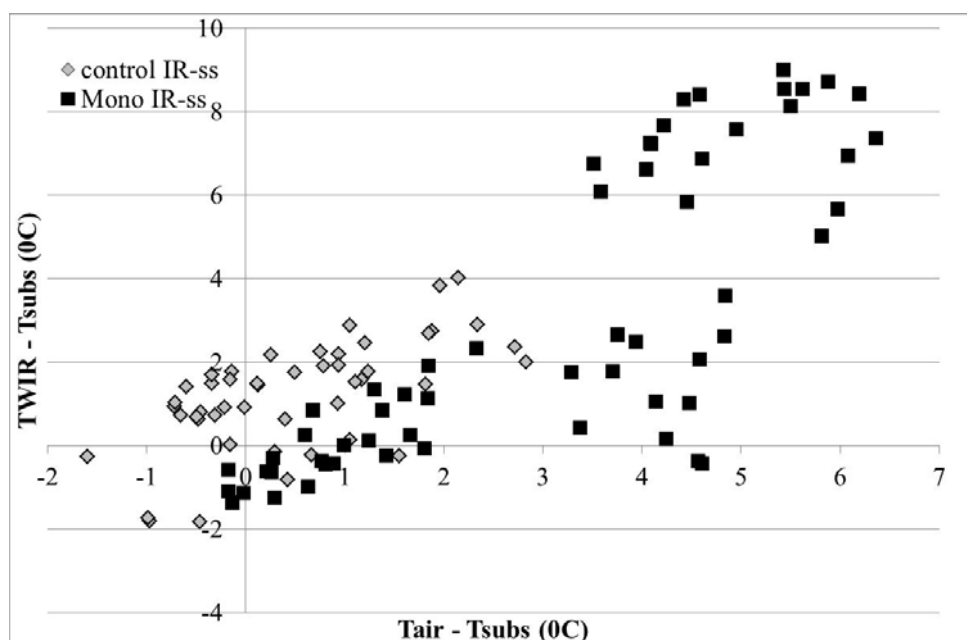


Figure 2. Confirmation of a condensed monolayer at Logan's Storage. The surface deviation temperatures before (control IR-ss) and after monolayer application are proportional to the air to water heat flux ($TWIR - T_{subs}$), but the monolayer reduced heat gain by about 20°C. The prevailing wind over the selected 2-day periods in January 2011 was predominantly less than 12 km hr⁻¹ and relative humidity was predominantly less than 50%.

Cyanobacterial populations were less affected by reduced light intensity and water stratification in the covered tanks, but some other factor inhibited populations in the uncovered tank (Figure 3) at the start of the trial. The inhibitory factor had dissipated by the end of the black cover trial, and may be associated with the placement of a new HDPE liner in Tank 3 to replace a leaking liner. Off-gassing hydrocarbons from the heat-welded liner may have been differentially toxic to phytoplankton (Paabo and Levin, 1987), as indicated by the very low species diversity index in Tank 3 at the start of the trial (Table 1). Cyanobacteria are known to be sensitive to polyphenols at concentrations that have no adverse impact on chlorophytes (Gross and Sutfield, 1994).

Table 1. Comparison of species richness of the predominant phytoplankton sampled from a depth of 0.4 m above the base of each Tank (10 m diameter, depth 0.7 m), at the start and end of each cover trial. Species richness was measured as the number of predominant genera identified per mL of water sample.

Sampling Date	Tank 1 (Covered)	Tank 2 (Cover and C ₁₈ OH)	Tank 3 (Uncovered)
19/1/2010 - Black cover	4	8	1
23/2/2010 - White cover	5	4	6
25/3/2010 - Black cover	4	3	5
15/4/2010	3	4	4

Table 2. The biochemical oxygen demand and the most probable number of phenol-degrading bacteria in microlayer water samples taken at the start and end of each tank cover trial. Bacterial populations were quantified as the most probable number (MPN) of colony forming units (CFU) present in 100 mL of the microlayer water sample.

Sampling Date	Biochemical Oxygen Demand (mg mL ⁻¹)			MPN Phenol-Degrading Bacteria (CFU 100 mL ⁻¹)		
	Tank 1 (Covered)	Tank 2 (Cover & C ₁₈ OH)	Tank 3 (Uncovered)	Tank 1 (Covered)	Tank 2 (Cover & C ₁₈ OH)	Tank 3 (Uncovered)
19/1/2010 - Black cover	4	4	3	2	1	9
23/2/2010 - White cover	2	2	6	2	2	2
25/3/2010 - Black cover	4	2	2	12	9	12
15/4/2010	2	2	10	12	9	12

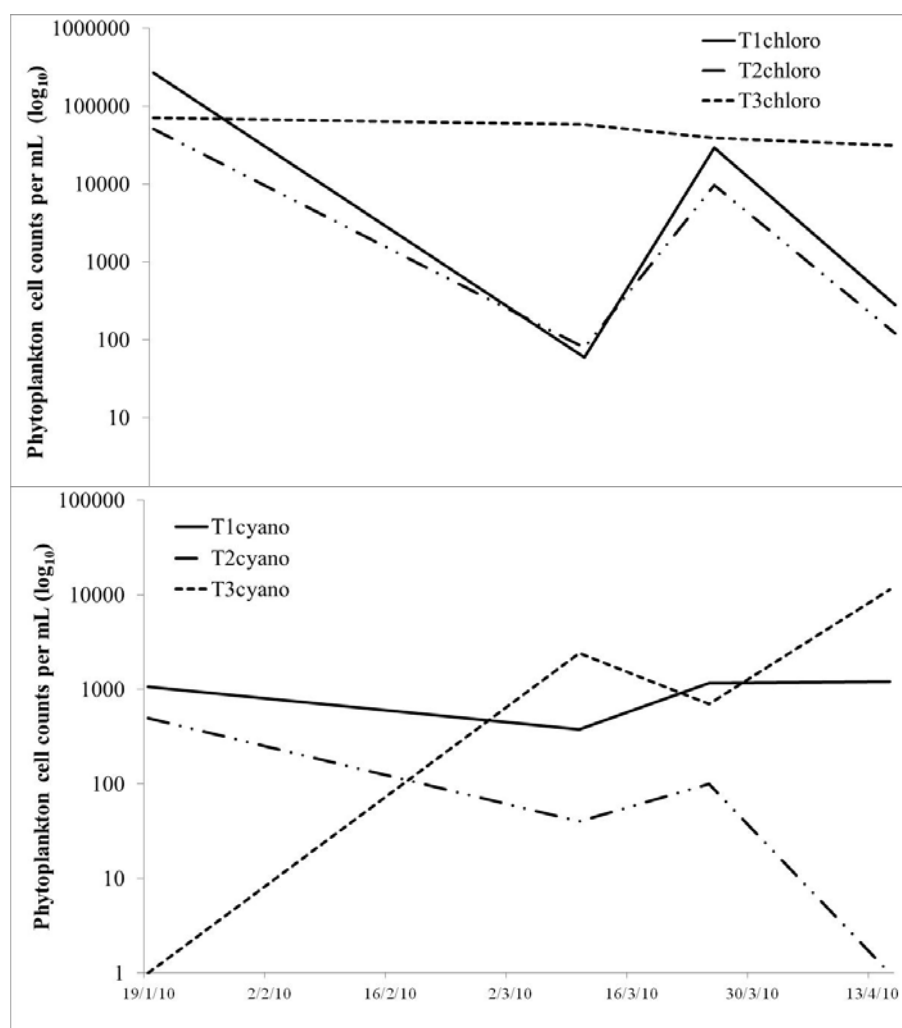


Figure 3. Cell counts per mL of water sample of predominant chlorophyte (chloro) and cyanobacterial (cyano) phytoplankton species sampled before the installation and replacement of covers on the 10m diameter tanks. Water was sampled from a depth of 0.4 m above the base of each tank, 0.3 m below the benchmark depth of 0.7 m. The black covers were installed on tanks 1 and 2 (T1, T2) on 21/1/2010, replaced with the white covers on 24/2/2010, with the black covers reinstalled on 26/3/2010. Tank 3 (T3) remained uncovered over the duration of the trial.

Monolayer application did not adversely affect species richness (Table 1), but the steady reduction in cyanobacterial cell counts in Tank 2 relative to Tank 1 over the duration of the trial may indicate some cumulative inhibition over time. Cyanobacteria were more sensitive to by-products from heat welding HDPE, and may also be more sensitive to the C18OH monolayer formulation. There was no evidence of an increase in the biochemical oxygen demand (Table 2), or in populations of bacteria most likely to degrade the monolayer (phenol-degrading bacteria, Pittaway and van den Ancker, 2010b). These results imply that twice-weekly applications of the C18OH monolayer formulation did not substantially increase the concentration of biologically degradable organic carbon, and did not increase bacterial populations known to degrade condensed monolayers (phenol degrading bacteria, Table 2). The sensitivity of the methodology used in the tank trials was sufficient to detect the adverse impact of the new HDPE liner in Tank 3 (reduced species richness in Table 1, and higher phenol-degrading bacterial concentrations at the start of the trial in Table 2), inferring that if the regular application of monolayer was having an adverse impact, we should have detected it.

There was also no evidence of an increase in or enrichment of the total organic carbon in microlayer and subsurface water samples at Logan's Storage (Figure 4 and Table 4), adjacent to the north-western or south-eastern shores where the concentration of monolayer should be greatest (blown by the prevailing wind). Monolayer application did not appear to enhance populations of phenol-degrading bacteria during and after dosing, with the highest value recorded from the south-eastern shoreline before monolayer application commenced (Table 4). Rain and wave turbulence are known to disrupt and disperse condensed monolayers, and the compounds used to-date have half-lives of days, producing few if any by-products (Pittaway and van den Ancker, 2010b). Application rates of the order of tens of litres of monolayer formulation per hectare of storage are comparatively dilute, likely to be well below natural concentrations of the more microbially resilient aquatic humic substances present in the microlayer of water storages in South East Queensland (Pittaway and van den Ancker, 2010a).

A key feature of monolayer application is the potential to apply product from automated dosing stations, only when conditions are conducive to the formation of a condensed monolayer (wind less than 12 km hr^{-1} , light or no rain), at times when the value of water saved offsets the cost of monolayer application (Brink *et al.*, 2010). The continuous, twice-weekly application of C18OH for 4 months during the tank trials represents an extreme case. Under the calm conditions below the black covers, a condensed monolayer was maintained for weeks, with very little change evident in water quality. Change in light intensity and water turbulence had a much greater impact, as did the installation of the new HDPE liner in the uncovered, clean water tank.

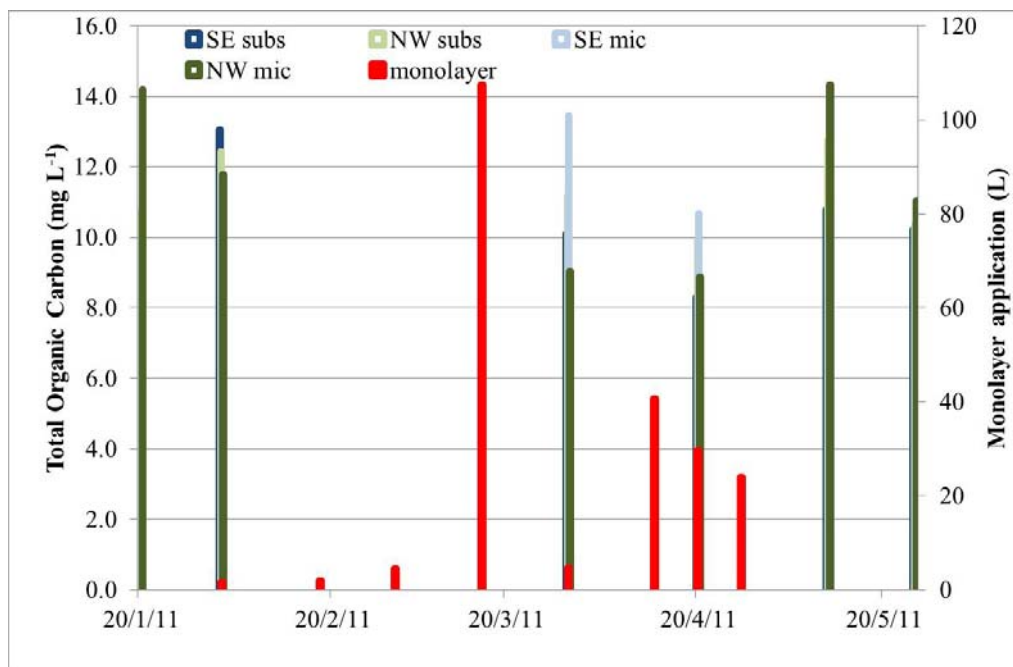


Figure 4. The concentration of total organic carbon in microlayer (mic) and subsurface (subs) water samples taken from the south-eastern (SE) and north-western (NW) shores of Logan's Storage immediately before, during and after monolayer application. The product applied was an undisclosed experimental formulation, applied at a rate equivalent to 9 times the monomolecular rate. During the application period, the prevailing wind was from the east-northeast.

Table 4. Enrichment values for total organic carbon (TOC) and phenol-degrading bacteria (measured as Most Probable Number, MPN) sampled before during and after monolayer application at Logan's Storage. Data for the south-eastern (SE) and north-western (NW) shores are given separately. Data in bold indicates sampling within 2 days of monolayer application. Monolayer was applied between 25/1-9/2, 23-30/3 and 6-18/4/2011.

Sampling Date	MPN SE Shore	TOC SE Shore	MPN NW Shore	TOC NW Shore
11/12/10	4	1	0.5	1.3
20/1/11	10	1	0.03	1.3
2/2/11	1	0.9	0.5	0.9
30/3/11	1	1.3	0.3	0.8
20/4/11	1	1.3	3.6	0.9
11/5/11	1	0.9	2.6	1.1

Conclusion

The installation of the wind-impervious black covers on the tanks provided ideal conditions for studying the impact of a condensed monolayer on water quality. The climatic conditions that prevailed during monolayer application at Logan's Storage were not as conducive, but the 2°C buffering of surface deviation temperatures during monolayer application under calm conditions indicates a condensed monolayer had formed (Figure 2). This is the first time the insulating effect of a condensed monolayer on surface water temperature has been confirmed in tank and field studies. Scientists have theoretically calculated that the reduction in latent heat loss that occurs when evaporation is suppressed by a monolayer must increase the temperature of the water (McJannet *et al.*, 2008). Results from our tank and field trials indicate that at the molecular level, the formation of a condensed monolayer insulates the microlayer from heat gain and heat loss by increasing the resistance to molecular and heat diffusion. Monitoring from both the tank (Pittaway, unpublished data) and field trials (Pittaway *et al.*, 2011) indicates monolayer application does not reduce dissolved oxygen or pH in the water, and there is no evidence that monolayer application increases the biochemical oxygen demand (Table 2).

There is some evidence that twice-weekly C18OH application over four months in the tank trial may adversely affect some cyanobacterial populations (Figure 3), but the installation of a new HDPE liner in the uncovered tank had a greater impact. Monolayer technology remains the only cost-effective strategy for reducing evaporative loss on large reservoirs, and results from this study suggest adverse impacts on water quality will be minimal. Improvements in automated application (Brink *et al.*, 2010) ensure monolayer product will only be applied under climatic conditions conducive to the formation of a condensed monolayer at times of peak water demand when supply is limited. Whilst candidate monolayer compounds continue to be selected for biodegradability and low toxicity (Barnes, 2008), the molecular scale at which artificial monolayers exert their effect will ensure minimal adverse impact on water quality.

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Integration of Pathogen and Chemical Removal in Reservoirs with Hydrodynamic Modelling

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Summary

A research project has been undertaken to measure the decay of enteric microorganisms and the degradation processes of micropollutants to quantify the treatment capacity of South East Queensland (SEQ) reservoirs. At the same time, work has been done on developing a three-dimensional (3-D) hydrodynamic model of Lake Wivenhoe. This paper reports on a preliminary assessment on how the hydrodynamic model can assist in predicting the health risks posed by pathogens and micropollutants in Wivenhoe Dam. The outcomes of this assessment demonstrated that there is potential for combining the hydrodynamic model with pathogen and chemical fate and behaviour data. The resulting successful combination could provide a powerful management tool for Seqwater and regulators.

Keywords

Pathogens, trace organics, decay, reservoirs, hydrodynamic models.

Introduction

Microbial pathogens and micropollutants, such as pharmaceuticals, personal care products and hormones, are commonly detected in urban and rural waters (Dan *et al.*, 1997; Lund, 1996). However, their fate and behaviour in natural water bodies is little understood, especially with respect to natural attenuation capacities of reservoirs. Therefore, a UWSRA research project was designed by CSIRO in partnership with Seqwater to measure the decay of enteric microorganisms and the degradation processes of micropollutants to quantify the treatment capacity of SEQ reservoirs. Outcomes from this work are expected to directly inform the risk assessment of pathogens and chemicals that may occur in the rivers and reservoirs (from either catchment-based sources or purified recycled water (PRW) discharges) and guide contaminant monitoring and management programs.

Seqwater has also been developing a three-dimensional (3-D) hydrodynamic model of Lake Wivenhoe in partnership with the University of Queensland. The model of Lake Wivenhoe was based on a uniform rectilinear grid (100m horizontal and 1m vertical resolution) derived from recently acquired bathymetric data. The model uses forcing data (eg, meteorological data, inflow/outflow data) as well as various in-lake measurement data (eg, temperature profiles, water surface elevation records). There is interest in determining how well this model can integrate with non-hydrological data such as the assessment of the fate and behaviour of pathogens and chemicals in the reservoir.

This paper provides information on a preliminary assessment on how the hydrodynamic model could assist in predicting the health risks posed by pathogens and micropollutants in Wivenhoe Dam.

Methods and Results

Pathogen Decay

Pathogen decay was studied in-situ using diffusion chambers in Wivenhoe Dam as previously reported by Toze *et al.* (2009, 2011). A range of conditions, seasons and enteric microorganisms were tested and the amalgamated results are given in Table 1. During the series of experiments the levels of Wivenhoe Dam changed from less than 20% capacity to 100% and then to the changed conditions post 2011 floods.

The results show that in all the experiments under all conditions, the bacteria decayed faster than adenovirus and *Cryptosporidium*. The exception was *Campylobacter jejuni* at 15m depth where the T90 (time for a 1 log loss) was similar to the T90 for *Cryptosporidium*. Before the 2011 floods, adenovirus had the slowest T90 times of greater than 30 days.

Table 1. Log1 decay times (T90 days) of pathogens in Wivenhoe Dam.

Experiment	Variables Tested	<i>E. coli</i>	<i>S. enteric</i>	<i>C. jejuni</i>	MS2	Adenovirus	<i>Cryptosporidium</i>
General Decay		2	7	8	5	11	13
Indigenous Microbes	Filtered	3	NR	-	4	40	-
	Unfiltered	2	NR	-	3	32	-
Pre-Flood	Surface	3	5	4	-	45	11
	15m	3	5	14	-	NR	-
Post-Flood	Surface	3	3	3	-	27	46
	15m	3	7	3	-	12	46

After the 2011 floods the huge change in water conditions in Wivenhoe Dam had an influence on the T90 times for some of the tested microorganisms while having no effect on the decay of others. The changed conditions had minimal influence on the decay of the bacteria apart from *Campylobacter* at 15m depth (Table 1). In contrast, the changed conditions slowed the decay of *Cryptosporidium* oocysts and increased the decay of adenovirus. The reasons for these changes in decay are yet to be determined.

Trace Chemical Photodegradation

A photodegradation test was undertaken under laboratory conditions using Wivenhoe Dam water and compounds identified by Seqwater as potentially problematic in SEQ reservoirs. The stability of the selected contaminants from exposure to solar radiation was assessed using a Suntest Solar Simulator (Atlas Material Testing Technology) containing a 1500 W xenon lamp, filtered to exclude the wavelength range <300 nm. The exposure intensity was set at a nominal irradiance values between 500 and 750 W/m² to approximate midday solar irradiance at 27° latitude, corresponding with Brisbane (NASA Atmospheric Science Data Centre). Decay of the compounds was measured at different times by residue analysis by liquid chromatography and tandem mass spectrometry (LC/MS/MS). Rates of breakdowns were established from the time series of data and half-lives were calculated.

Biodegradation was found to be of minimal importance, even where an additional quantity of organic carbon (DOC) or a microbial inoculum was added. In contrast, photolysis was found to be an important degradation pathway for a number of compounds, particularly propranolol (PRL), sulfamethoxazole (SFM), diclofenac (DCF), triclosan (TCS) and triclopyr (TCP). Indirect photolysis, most likely due to the reaction with the OH[•] species, also led to the degradation of the majority of the test micropollutants, with half-lives ($t_{0.5}$) ranging from a few days to a few hours for most compounds (Table 2).

The high degree of light attenuation in all the natural waters, however, is likely to limit the importance that solar irradiance has on micropollutants that exist within the tested water systems. The degree of photolysis will be highly dependent on a number of potentially transient water quality parameters, such as turbidity (reducing effectiveness of photolysis) or precursors of reactive species (such as [NO_x], which is converted to OH).

Assessment of factors affecting photolysis within these systems should therefore be assessed to further validate the importance of photolysis as an attenuation pathway. Also, based on the physicochemical properties of a number of these compounds (such as high K_{ow} values), other potential attenuation processes, such as sorption, and the factors which can influence their importance should also be considered in assessing the attenuation processes occurring within these water systems.

Table 2. Summary of photolysis rates (days) and half-lives (days) of breakdown of chemicals under simulated solar radiation.

Compound	Photolysis Rate		Average Half-life	SD
	Experiment 1	Experiment 2		
Atenolol (ATL)	4.1	3.7	3.9	0.28
Benzotriazole (BZT)	4.8	na	4.8	na
Methotrexate (MTX)	3.1	1.8	2.45	0.92
Metoprolol (MET)	3.1	na	3.1	na
Trimethoprim (TRM)	3.1	4.5	3.8	0.99
Venlafaxine (VEN)	1.9	1.5	1.7	0.28
Propranolol (PRL)	2.6	0.18	1.39	1.71
Cyclophosphamide (CPP)	16	>30	>30	na
Sulfamethoxazole (SFM)	0.82	0.59	0.71	0.16
Atrazine (ATR)	5.8	1.7	3.75	2.90
Carbamazepine (CBZ)	3.3	4.7	4.0	0.99
Diethyltoluamide (DEET)	6.1	na	6.1	na
Diuron (DIU)	3.4	1.5	2.45	1.34
Sertraline (SER)	na	0.9	0.9	0.00
2,4-dichlorophenoxyacetic acid (2,4-D)	3.3	3.3	3.3	0.00
Triclopyr (TCP)	3.1	0.13	1.62	2.10
Diclofenac (DCF)	0.2	0.02	0.11	0.13

Hydrodynamic Model Simulations

Selected simulation results of the maximum amount of PRW in Wivenhoe Dam at Logan's Inlet and Pelican Island under starting storage levels of 20, 40 and 80% of dam capacity are given in Table 3. Examples of selected simulations demonstrating the change in the amount of PRW in Logan's Inlet and Pelican Island over a one year period are given in Figures 1-3. A summary of the PRW inflow scenarios used in the model for the selected simulation runs is provided in Table 4. Results also suggest that catchment inflows, Somerset Dam releases and mixing induced by the Splityard Creek hydropower station have the potential to increase mixing and significantly dilute the PRW (results not shown). As expected, the simulation results showed that PRW concentrations within Logan's Inlet are likely to be high (40-100% of PRW inflow concentration) at extremely low lake storage levels ($\approx 25\%$ storage) and/or under conditions of high PRW inflows (ie, 180 Ml d^{-1}). Simulation results suggest that more mixing and dilution is generally likely to occur within Logan's Inlet at higher storage levels resulting in reduced maximum PRW concentrations within the inlet. This is thought to be caused by the somewhat constricted entrance to Logan's Inlet during low storage levels. As lake storage levels increase, the entrance to Logan's Inlet becomes less constricted and allows more exchange with the main part of the Lake, thus increasing dilution.

The identification of the importance of both the PRW inflow rate and initial lake storage level on the PRW concentrations at the main dam wall off-take point is useful for reservoir managers who might need to meet specific guidelines for PRW mixing and detention times within the system. Further work is required to assess the model (ie, validation against measured PRW inflows) in this regard. Simulation results also highlight the importance of catchment inflows, releases from Somerset Dam and mixing induced by the Splityard Creek hydro-power on PRW dilution. In particular, the Somerset Dam releases and the Splityard Creek hydro-power station flows might offer managers additional options for manipulating PRW concentrations as well as the degree of mixing (both horizontal and vertical) within the southern portion of the reservoir.

Number of Pathogen Log Removal or Chemical Half-Lives Based on Model Simulation

The concept on the ability to combine the model simulation from the hydrodynamic model with the T90 pathogen decay rates and chemical photodegradation half-lives was tested to determine the efficacy of combining the different data sources to determine the amount of loss of pathogens and chemicals as water moves through the reservoir. This initial test was undertaken by approximating the time to 50% of the water at Logan's Inlet or Pelican Island to consist of PRW when the reservoir was at 20%, 40% and 80% of storage capacity (Table 5). The maximum or averaged T90 and photodegradation half-lives were then divided from this time to estimate the total number of pathogen Log reductions or chemical half-lives would be expected to occur in this time frame.

The results of this initial test are provided in Table 6. For pathogens, the calculations show that there is a major difference between the number of Log reductions that can be expected for each of the microbial types. The bacteria were calculated to have a total number of Log reductions ranging from 48 for the 20% storage capacity scenario to greater than 99 for the modelled 80% storage capacity scenario. In contrast the number of Log reductions for *Cryptosporidium* ranged from 16 to 30 for the 20% and 40% storage capacity scenarios but dropped to around eight for the 80% storage capacity scenario. It needs to be remembered that the 80% scenario was treated as a post-flood condition which further demonstrates that the flood has influenced the removal of some pathogens, in this case for *Cryptosporidium* significantly reducing the number of Log reductions achieved. To further progress this research to the stage where it has potential to be used as a management tool, the ability to combine decay data with hydrodynamic models needs to be further refined to enable the tracking of pathogen decay over both distance as well as time. The combination of the model/decay output with quantitative health risk assessment (eg, Quantitative Microbial Risk Assessment) will also enable a determination of how the human health risk changes in various locations and the prediction of the input of localised pollution sources on health risks throughout the reservoir.

A similar result was obtained for the degradation of organic chemicals in the reservoir. The number of half-lives achieved varied from 65 to 120 which would mean that the chemicals would all fall below detection limits within a very short distance in the reservoir. It should be noted that the chemical degradation data used here was photodegradation. If the water in which the chemical(s) are dissolved move to lower depths in the reservoir (ie, not at the surface) then the influence of sunlight becomes reduced and other removal mechanisms take more prevalence. If, as the research has indicated so far, other common chemical degradation pathways have little influence on the persistence of these chemicals, then the resulting number half-lives will be much less and thus much larger movement of the chemicals through the reservoir will occur.

Table 3. Summary of example PRW inflow scenarios.

Model Run	PRW Inflow Rate [ML d ⁻¹]	Starting Storage level [% Full Water Supply]	PRW Temperature / Salinity
626	180	20	Warm/Low
627	22.5	40 [58.00 mAHD]	Warm/Low
628	45	40	Warm/Low
629	90	40	Warm/Low
630	180	40	Warm/Low
631	22.5	80 [64.64 mAHD]	Warm/Low
632	45	80	Warm/Low
634	180	80	Warm/Low

Table 4. Maximum simulated PRW concentration at selected monitoring locations.

Scenario Run	Dam Wall	Pelican Island	Logan's Inlet
626	33.63%	47.89%	109.26%
627	1.89%	27.68%	55.36%
628	15.06%	106.52%	121.73%
629	14.94%	112.79%	113.11%
630	15.16%	100.72%	102.42%
631	1.04%	3.72%	6.21%
632	2.16%	8.61%	12.62%
634	7.91%	25.29%	28.93%

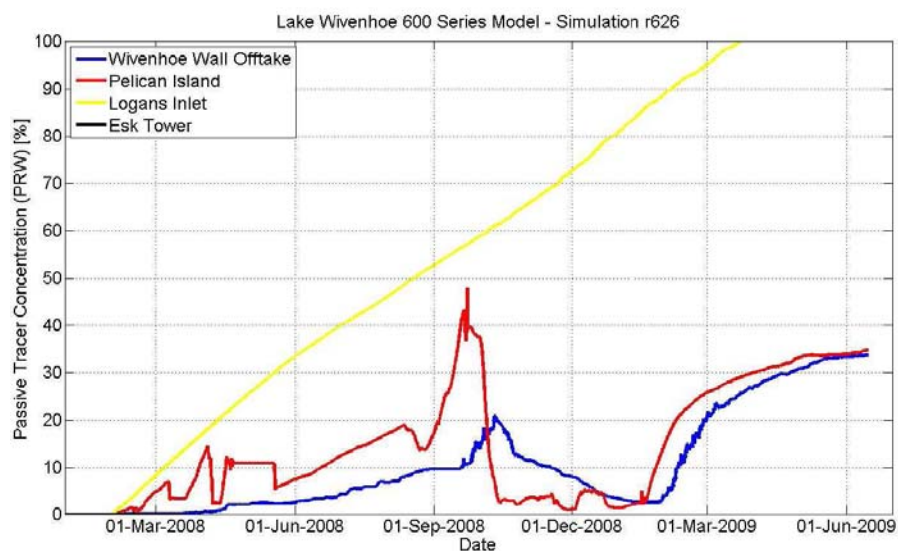


Figure 1. Example simulation of PRW flow in reservoir with storage level at 20%.

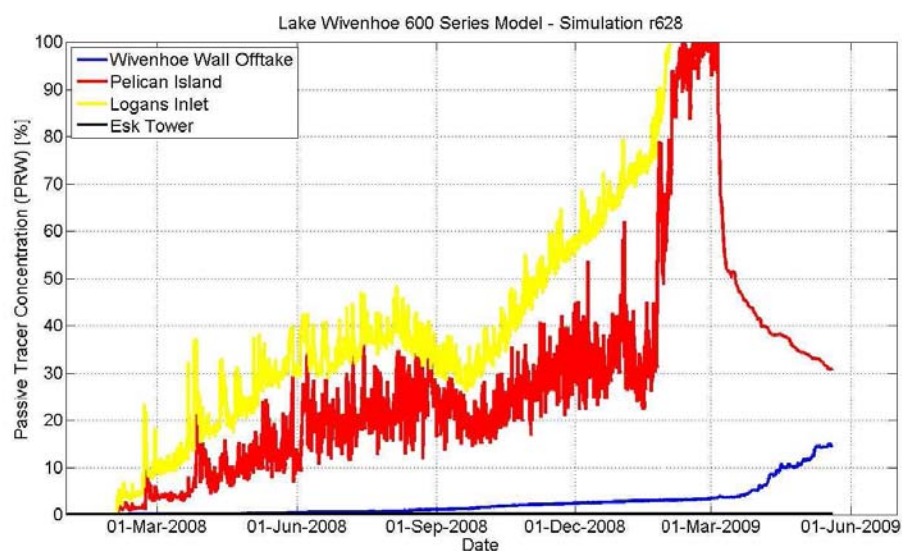


Figure 2. Example simulation of PRW flow in reservoir with storage level at 40%.

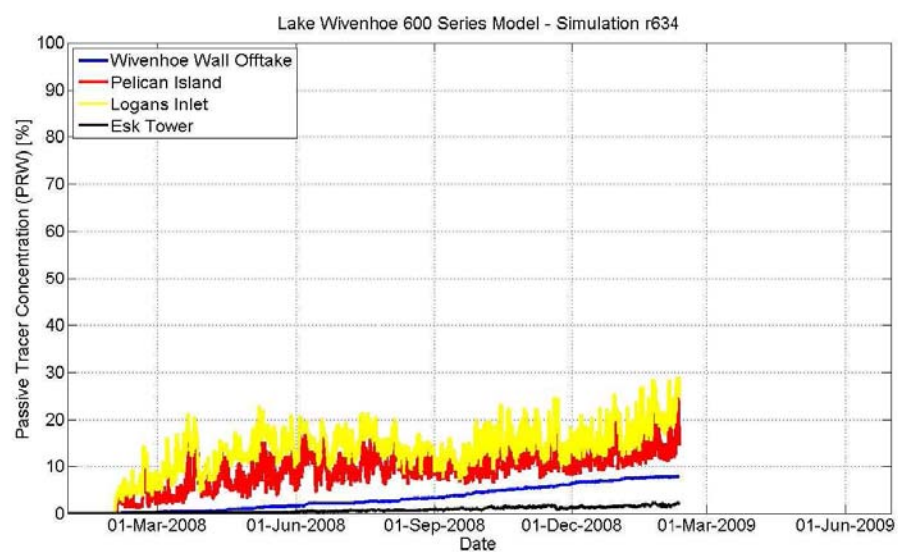


Figure 3. Example simulation of PRW flow in reservoir with storage level at 80%.

Table 5. Estimated times for PRW to reach 50% of water at locators in Wivenhoe Dam.

Storage Capacity Scenario	Time for Achieving 50% PRW (days)	
	Logan's Inlet	Pelican Island
20%	184	214
40%	245	337
80%	>365	>365

Table 6. Number of pathogen Log reductions and chemical half-lives under different model simulations based on the approximate time for 50% of PRW^a.

Bacteria /Chemicals	Pre-Flood ^b	Post-Flood ^c	20%		40%		80% ^d	
			Logan's	Pelican Is	Logan's	Pelican Is	Logan's	Pelican Is
Number of Log Removals	T90 (days)		Number of Log Removals					
Bacteria	3.8 ^e	3.7 ^f	48.4	56.3	64.5	88.7	>99	>99
Adenovirus	45	27	4.1	4.8	5.4	7.5	>14	>14
<i>Cryptosporidium</i>	11	46	16.7	19.5	22.3	30.6	>8	>8
Number of half lives	Half Life (days)		Number of Half Lives					
Chemicals	2.8 ^g	nd	65.7	76.4	87.5	120.4	na ^h	na

^a Time for 50% PRW at Logan's Inlet and Pelican Island estimated from Figures 1-3 and given in Table 4.

^b Pre-flood T90 times were tested on the 20% and 40% model simulations.

^c Post-flood T90 times were tested on the 80% simulation.

^d It was assumed that the 80% simulation was water from post-flood conditions.

^e Average T90 for all three bacteria from all pre-flood experiments from Table 1.

^f Average T90 for all three bacteria from all post-flood experiments from Table 1.

^g Average of photodegradation half-lives for all chemicals in Table 2.

^h na = not applicable as the results of degradation of the organic chemicals in post-flood water is not available at this time.

Conclusions and Impact

While this is a preliminary assessment of the ability to combine hydrodynamic modelling with results from the study of the fate of pathogen and micropollutants, the initial results indicate that there is good potential for a successful outcome. While there remains a large amount of work on further developing the hydrodynamic model and studying the factors controlling the decay of pathogens and degradation of chemicals, the outcomes of this preliminary study indicate that the successful combination of all these elements, in conjunction with health risk modelling could provide Seqwater as well as health and water regulators with a powerful new water management tool.

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Rainwater System Design for Urban Residential Households –the Role of Header Tanks and Pressure Vessels in Improving Energy Efficiency

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Summary

Pumps are most energy effective when performing in their best efficiency range, at which energy requirements are minimised. However, in urban dwellings typical end uses for rainwater can have flow and volume requirements which cause the pump to operate at a high specific energy for rainwater delivery. In an attempt to reduce the overall energy for rainwater delivery a number of rainwater system configurations have been proposed. We examined some of the most common components (pump sizing, header tanks, pressure vessels and infrastructure pipe size) to determine their potential to increase the energy efficiency of rainwater pumping systems for a single storey dwelling. Whilst all measures can improve energy efficiency, dwelling characteristics and appliance design can limit the effectiveness of some system components, such as gravity fed header tanks. Greater access to information on how system components operate and service requirements in urban dwellings is the most effective tool to improve design to achieve increased energy efficiency.

Keywords

Rainwater systems, specific energy consumption, urban rainwater supply.

Introduction

The installation of rainwater systems in urban areas across Australia surged during the drought period. Initially promoted via financial incentives up to June 2011, in most capital cities (ACT Government, 2011, Queensland Government, 2011, Government of South Australia, 2011, Northern Territory Government, 2007, State Government of Victoria, 2011, NSW Government 2011, Water Corporation, 2011, Hobart City Council, 2011), rainwater tanks were subsequently, included in new housing requirements in NSW, Queensland and SA. Concerns about the additional energy burden that rainwater pumping would have into the future (Retamal *et al.*, 2009; Hall *et al.*, 2008; Lane and Gardner, 2009; Lane *et al.*, 2010), and the realisation that knowledge on the operation of rainwater systems in urban dwellings was limited (Retamal *et al.*, 2009), and yet it differed from the traditional operation in rural areas, led to research aimed at understanding and reducing the energy requirements of urban rainwater systems.

The specific energy requirements for rainwater supply to individual dwellings in Australia were examined in a number of in-situ studies. This included: a complex of 4-6 high value houses in Queensland (Gardner *et al.*, 2006; Beal *et al.*, 2008); 8 houses in New South Wales (Retamal *et al.*, 2009) and 31 houses in Victoria (SEWL, 2009). The studies demonstrated that the specific energy for rainwater pumping could vary from 0.6 to 11.61 kilowatt-hours per kilolitre(kWh/kL) among the various dwellings.

A significant part of variance is attributed to the individual system designs and operation patterns at dwelling level (Retamal *et al.*, 2009; Cunio and Sproul, 2009; SEWL, 2010; Tjandraatmadja *et al.*, 2011). Retamal *et al.*, (2009) associated the theoretical energy losses of a system to various system components, including mismatch of end uses, pump technology and design, pipe friction and system design.

Mismatch between End Uses and Pump Selection

The role of individual system components on pump energy demand has been examined, theoretically (Retamal *et al.*, 2009) and experimentally (Cunio and Sproul, 2009; SEWL 2010). Minimum service requirements (flow, volume and pressure) apply to a range of household appliances sold in Australia (Tjandraatmadja *et al.*, 2011, SEWL 2010), and these can restrict the energy consumption and service from a pump. Mismatch between end uses and pump sizing was a major influence on the energy use of the system as verified by various researchers. Showers, washing machines, toilet cisterns and dishwashers fill up at flow rates of less than 15 litres per minute (L/min) (Tjandraatmadja *et al.*, 2011; SEWL, 2010). Likewise, except for taps and showers, volume requirements per appliance run are often restricted, eg, 3 to 6 L for a toilet cistern or 50 to 60 L for a full top loading washing machine tub (Tjandraatmadja *et al.*, 2011).

Cunio and Sproul (2009) and Tjandraatmadja *et al.*, (2011) verified that the common motor capacity of pumps adopted in urban dwellings could often be oversized for individual end uses in households, such as the filling of toilet cisterns. Whilst the two studies verified that the smaller size pumps they tested were less energy intensive for same applications, Hauber-Davidson and Shortt (2011) and SEWL (2010), when testing a larger sample of eight different pumps, verified that pump design also had a great influence and that different brand pumps with similar motor capacity could at times display very different energy requirements.

On the other hand, Cunio and Sproul (2009) also verified that under-sizing a pump could limit its service, eg, one of the small pumps they tested (0.18 kW engine) was unable to fill a toilet cistern, as it did not provide sufficient pressure.

Pipe Size

Retamal *et al.*, (2009) estimated that infrastructure pipe friction losses would in theory contribute on average to only 2% of energy losses in pumping systems. Tjandraatmadja *et al.*, (2011) verified that measured friction losses in their system (15mm diameter pipe) ranged from 1% to up to 6% for various pump sizes depending on flow requirements for various household appliances. On the other hand, Cunio and Sproul (2009) verified that combining a low pressure pump and larger diameter pipe could reduce pumping energy by over 50% (from 1.7 to 0.79 kWh/kL), when shifting from a high pressure system (0.55 kW pump) with 19mm diameter pipe to low pressure pumps (0.18 and 0.22 kW) with 40mm diameter pipe for supply of toilet cisterns. Whilst they did not verify the influence of pipe size and pump size separately, they estimated that by increasing pipe diameter from 12.5mm to 25mm diameter, a system with 20m length could reduce power use by 95% (Cunio and Sproul, 2008).

Header Tanks

Supply of rainwater to a header tank followed by gravity flow of the rainwater to appliances is expected by most researchers (Retamal *et al.*, 2009; Cunio and Sproul, 2009; SEWL, 2010) to be the least energy intensive system for rainwater supply, however the set-up has not been verified experimentally. Cunio and Sproul (2009) estimated that the specific energy required for supply of rainwater to a toilet cistern could have been reduced to less than 0.04 kWh/kL with a header tank at a height of 5m, but they had not verified the set-up experimentally. However, verification is needed of the header tank height required to fulfill the minimum water supply requirements for appliances.

Other Components

Hauber-Davidson and Shortt (2011) and SEWL (2010) examined the performance of mains switch pressure controllers, a variable speed drive (VSD) pump, and a 5 L pressure vessel in the supply of rainwater to selected end uses in a two storey dwelling (toilet cistern, front loader washing machine, shower head and garden hose) (SEWL, unpublished). For the single VSD pump test, the stand-by energy outweighed the reduction in operating energy achieved for the pump examined (SEWL, 2010). Mains pressure switches were seen as beneficial by the researchers, as they prevented start-ups for low volumes of water, such as small leaks or a dripping tap. A similar function was achieved by the 5 L pressure vessel as it was able to provide low volumes (1-2 L) before starting the pump (Hauber-Davidson and Shortt, 2011). However, the overall reduction in energy for the system tested was small. Larger size pressure vessels were not trialed in those studies.

Hence, questions remain about the benefits that can be achieved from the incorporation of the various components into a rainwater pumping system. Are header tanks the ultimate low energy supply system and can they provide suitable service with energy savings? What energy reduction can larger capacity pressure vessels achieve? Are there real advantages in increasing piping size?

This paper aims to answer some of those questions by examining the impact that some individual components (header tanks and pressure vessels) and system design can have on the energy requirements for rainwater pumping.

Methods

The experimental set-up adopted was a controlled laboratory environment that replicates typical end uses and contains a range of appliances including a washing machine, toilet cistern, tap and a dishwasher (characteristics are outlined in Table 1). The system set-up has been previously described in Tjandraatmadja *et al.*, (2011). It replicates the internal plumbing of a house supplied with rainwater. Rainwater was supplied from an 850 L rainwater tank using 17m of 18mm diameter polyethylene pipe (Sharkbite™) and a range of fixed speed pumps.

Table 1. Appliance characteristics (Source: Tjandraatmadja *et al.*, 2011).

Appliance	Water Efficiency Labeling Scheme (WELS) Rating	Average Water Consumption (L/wash)	Flow Rate during Fill (L/min)
Washing machine top loader (5.5 kg capacity)	4 stars	113 (2 fills)	9.6-13
Dishwasher	2.5 stars	16 (>6 fills)	3.3-4.1
Toilet cistern	-	3 (half flush) 6 (full flush)	4-6.3
Tap	-	Not applicable	Up to 35 l/min

Table 2 shows the range of system configurations trialed to evaluate the energy and service conditions. System components adopted included three different external constant speed pumps A, B and C, with respective motor capacities of 0.20 kW, 0.55 kW and 0.75 kW; pressure vessels with nominal capacities of 8, 18, 40 and 60 L and a 300 L header tank equipped with a float valve. The header tank was placed at a height of 2.64m from the house floor, equivalent to the height of ceiling beams of a dwelling. Parameters monitored during rainwater supply included the pressure in the system, power consumed by the pumps, flow rate, volume of water and the specific energy for water supply (in kWh/kL). The monitoring instrumentation adopted had full-scale deflection accuracies of $\pm 0.15\%$, $\pm 0.5\%$ FSD and $\pm 0.16\%$ at 20°C for pressure, current and flow respectively. A monitoring frequency of 0.2 seconds was adopted to ensure the capture of energy peaks. Specific energy was calculated as the ratio of total energy to volume of rainwater supplied during an event.

Table 2. Rainwater system configurations.

System	Pump Motor (kW)	Pipe Diameter (mm)	Pressure Vessel (L)	Switch Type
Pump Only	0.2, 0.55, 0.75	18	-	Mains, pressure
Pump with Pressure Vessel	0.2, 0.55, 0.75	18	8, 18, 40, 60	Mains, pressure
Pump with Header Tank	0.2, 0.55, 0.75	18	-	Mains, pressure

Results

The water use patterns for the appliances investigated were previously described in Tjandraatmadja *et al.*, (2011) and were summarised in Table 1. Toilet cisterns are single fill appliances, whilst washing machines and dishwashers are filled in multiple events. As previously verified in Tjandraatmadja *et al.*, (2011) appliances which can be supplied at high flow rates allow pumps to operate more efficiently at lower specific energy of water supply. Thus in principle, energy usage can be restricted either by reducing the time of operation of a pump or operating it at higher flow rates.

Header Tanks

A 300 L header tank was selected. The tank dimensions could be fitted into the ceiling vault of a single storey dwelling and the effective volume, 257 L, can supply all appliances requirements in a household. For comparison, total water use in South East Queensland (SEQ) was recently reported to average 145.3 L/p/d and the water demand for typical rainwater applications (washing machine, toilet and irrigation) ranged from 137 to 233 L/hh/d in 2010 (Beal *et al.*, 2010).

Figure 3 and 4 compare the energy requirements for pumping rainwater to the appliances in this study. Filling a header tank requires the least specific energy. This is because filling occurs at high flow rates (>20L/min). The average specific energy ranged from 0.39 to 0.66 kWh/kL, depending on the pump adopted. When considering the volume of water delivered by a header tank, direct provision of rainwater to individual appliances (equivalent to one washing machine run, three dishwasher runs, 15 half cistern fills and 10 full cistern fills) has a significantly higher average energy burden ranging from 0.24 to 0.86 kWh, compared to 0.10 to 0.17 kWh for the header tank (Figure 4).

However, the service pressure provided by gravity was only 20 kPa, which was below the minimum requirements for operation of household appliances in Australia (minimum pressure 31-100 kPa). By changing the toilet cistern valve with a low pressure valve (min. 15 kPa) we were able to fill the toilet cistern in less than four minutes – under mains pressure the cistern fills in one minute. Based on estimates, to achieve 50 kPa minimum pressure, the header tank would have to be placed at a height greater than 5m. For comparison, the smallest pump A (0.2 kWh) operates at 180 kPa when filling the toilet cistern.

Hence, whilst a header tank is in principle able to reduce the overall energy requirements for water delivery by pumps, independent of pump size; and given that appliances are currently designed to operate under pressure, provision by gravity would require the placement of the header tank above the current ceiling height of a single storey dwelling. Discussion with manufacturers is required to understand if modifications are possible to allow appliances to operate at low pressure.

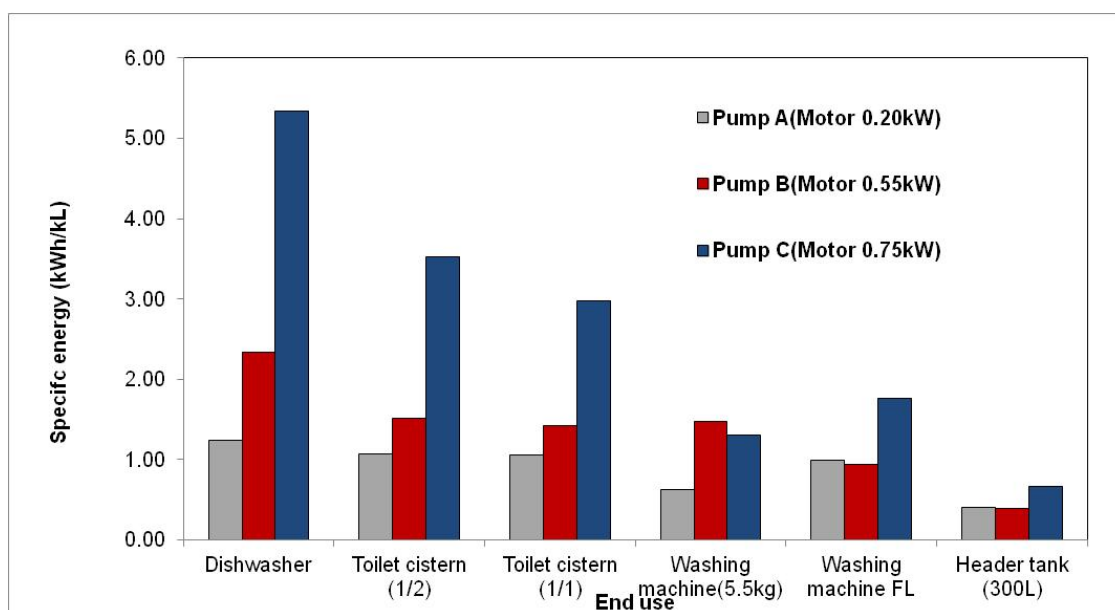


Figure 3. Specific energy requirements for supply of rainwater to a range of appliances.

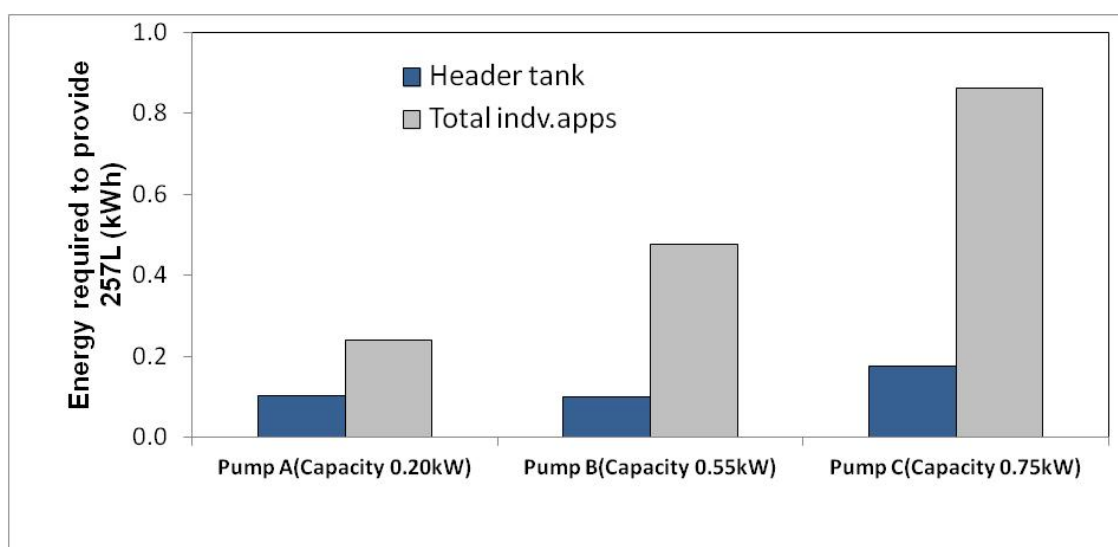


Figure 4. Energy required for delivery of rainwater to individual appliances and to a header tank (257 L).

Pressure Vessels

Pressure vessels are designed to provide a specific volume of water to a pumping system under pressure and thus prevent the pump from frequent start-ups. The operation of a pump as it fills a pressure vessel is shown in Figure 5. The pressure in the system gradually decreases before the pumps starts. Upon start-up, the pump operates at maximum flow, which is gradually reduced as the pressure vessel fills up. The full vessel will continuously deliver water to the system until the trigger pressure in the vessel is reached.

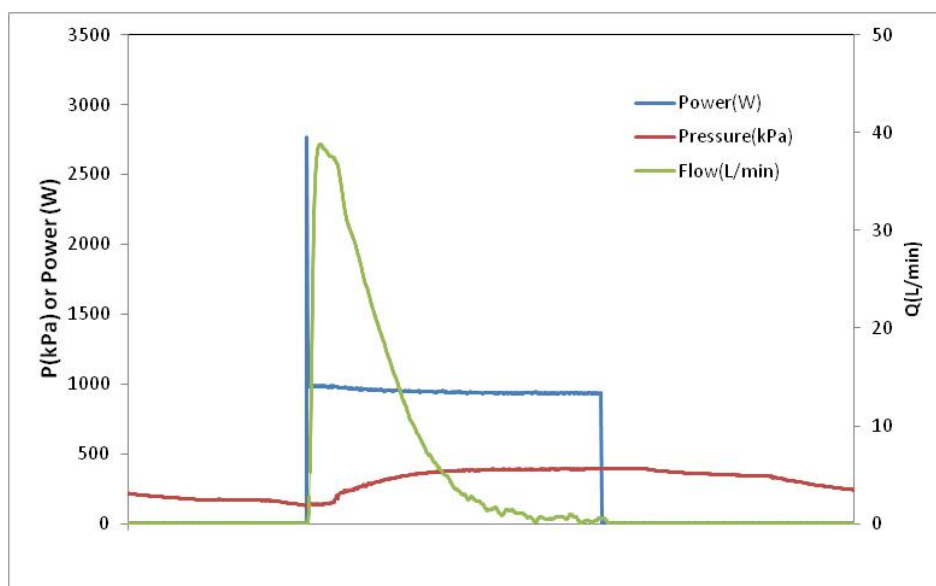


Figure 5. Operating conditions for the filling of a pressure vessel by pump C (0.75 kW).

Thus the effectiveness of a pressure vessel in decreasing the operating time and frequency of start-ups of the pump will depend on the volume that it can supply and the pressure settings of each pump. The actual volume supplied by a pressure vessel is significantly less than its nominal volume and it can typically range from 1/4 to 1/3 of the nominal volume. For instance, our 18 L pressure vessel coupled with pump C (0.75 kW) could deliver 6.3 ± 0.5 L before refilling was required. Such volume was sufficient to fill the toilet cistern twice or to provide small volumes of water to the dishwasher or the washing machine before the pump start-up. 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), which leads to more efficient pump operation particularly for low volume and low flow end uses.

Figure 6 shows the specific energy required for pump C to supply a range of end uses with and without an 18 L pressure vessel. The addition of the pressure vessel reduced the energy required, this was most markedly observed for applications which required delivery of low volumes such as hand washing (2 L/wash), the dishwasher (multiple supply events of < 4 L) or intermediate supply during the washing machine cycle. Delivery of larger volumes of water such as during the fill of the washing machine (> 50 L/fill stage) were less impacted.

Hence, in selecting a pressure vessel it is necessary to consider the volume requirements for individual appliances (end uses) in a dwelling. This explains why 5 to 8 L pressure vessels had very little impact on the energy required for operation of a pump.

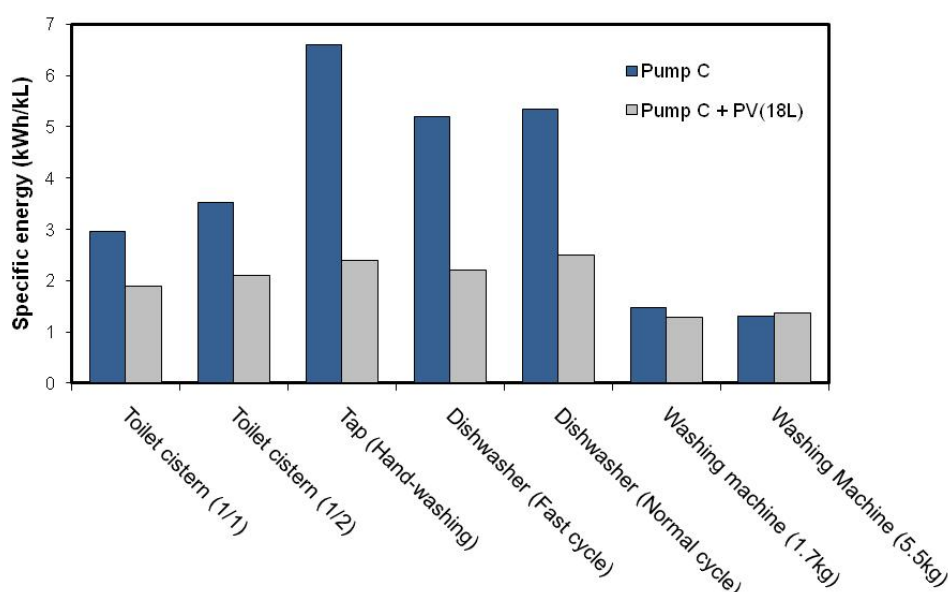


Figure 6. Pump C (0.75 kW) energy required with and without an 18L pressure vessel (PV).

Conclusions

Pumps are most effective when performing in their best efficiency range, at which energy requirements are minimised. In addition, design of the rainwater pumping system and proper selection of ancillary components can also contribute to reduce the energy required for rainwater pumping. To achieve optimal energy use for rainwater supply, it is necessary to understand the operation and limitations of the various system components. In this paper, the efficacy of header tanks and pressure vessels in reducing energy and delivering suitable service has been highlighted.

Header tanks have the potential to produce the most marked energy reduction of all the devices adopted in a rainwater system. However, given that minimum operating requirements for common appliances are based on pressurised systems, insufficient pressure would be generated by placement of a header tank in the roof cavity of a single storey dwelling for satisfactory operation of appliances such as washing machines and toilet cisterns. Yet, adoption of header tanks and gravity feed could be re-evaluated upon redesign of appliances with water inlet valves for low pressure/flow supply or for types of dwellings where header tank height could be increased.

Pressure vessels on the other hand, if properly sized, have the potential to reduce the energy consumption and maintain suitable pressure and flow for appliances. A pressure vessel can help smooth out low-flow, low-volume, high-energy intensity water requirements, while maintaining system pressure. It is important to ensure that the volume of water released will correspond or exceed the minimum end use volume requirements. Pump settings will also impact the released volume and will need to be explored in other publications.

Whilst all measures can improve energy efficiency, dwelling characteristics and appliance design can limit the effectiveness of some system components, such as gravity feed by header tanks. Overall, giving householders greater access to information on how system components operate and water requirements of various appliances in urban dwellings is the most effective tool to improve design and increase energy efficiency.

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Monitoring of Residential Rainwater Tanks in South East Queensland to Investigate Mains Water Savings and Volumetric Reliability

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Summary

The main aim of this paper is to present the outcomes of a monitoring study conducted on 20 detached households with rainwater tanks in South East Queensland (SEQ), Australia. The purpose was to quantify mains water savings and volumetric reliabilities of household rainwater tanks. A water balance analysis was conducted to evaluate and substantiate the actual reductions in mains water being achieved. Diurnal pattern analysis for the 20 homes was also conducted using real-time water demand data, for individual homes and as a cluster. The volumetric reliability, the ratio of the rainwater usage to the total household water demand, was found to average at 29% for the 20 homes. Around 18% of the non-potable water demand in the households is sourced from potable mains water, which is supplied as backup into the system. The long-term diurnal pattern analysis showed that an average 28% of the total peak hour water demand was met by rainwater collected in household rainwater tanks. This finding provides scope for further investigation into improving the effectiveness of rainwater tanks and rainwater collection systems design.

Keywords

Rainwater tanks, water end use, sustainable development, urban water, water demand management.

Introduction

Over the past few decades, there has been significant growth in population and constant changes in lifestyle due to rapid economic growth, which has had a great impact on urban water demand in Australia and across the world (Whittington *et al.*, 2002; Johnson and Handmer, 2002). Calls to satisfy the growing demand on potable water resources have been met with numerous challenges owing to an expanding urban community and problems arising from potential impacts of climate change (Mwenge Kahinda *et al.*, 2010; Zhang *et al.*, 2009). The increasing scarcity of fresh water in Australia attributed to various factors has impelled state authorities to broaden the range of sustainable water management practices to meet the water demand in urban areas. In South East Queensland (SEQ), state and local governments have introduced various strategies such as imposing water restrictions and rebate programs like the WaterWise Rebate Scheme (2006) to encourage the uptake of water efficient household appliances. Most recently, the Queensland Development Code (QDC) MP 4.2 was established with aims to significantly reduce dependency on traditional mains water supply from the water grid; one of the most significant steps taken to complement the state government's commitment to achieve potable water savings in new residential dwellings (DIP, 2010). This has led to the use of various alternative water sources such as rainwater tanks to supplement mainstream water supply sources on a fit-for-purpose basis and relieve strain on traditional centralised water supply systems.

To conform to QDC MP 4.2, every new household in SEQ is expected to save 70 kL of potable mains water. One of the recommended measures to achieve this target is through installation of a 5 kL rainwater tank connected to either a 100 m² roof catchment area or 50% of the total roof area (whichever is lesser) in Class I dwellings (detached houses). The rainwater tank (RWT) system is internally plumbed to household appliances such as washing machine cold-water tap, toilet cisterns and at least one external irrigation tap. The internally connected fixtures where the water demand is met by the rainwater tank also require back-up water supply connections; either through trickle top-up or automatic switching valve system to ensure continuous supply of water (DLGP, 2010). A number of studies have previously attempted to validate the potential mains water savings achievable through plumbed rainwater tanks through different methods such as: rainwater system tank modelling (Coombes and Kuczera, 2003); paired statistical analysis using water billing data between household cohorts with and without plumbed rainwater tanks (Beal *et al.*, 2011); and benchmarking the water usage in households having plumbed rainwater tanks to the corresponding regional water usage (Chong *et al.*, 2011). However, the substantiation of these previous research outcomes to estimate the effectiveness of the newly adopted water saving strategies is imperative to make improvements in sustainable water management initiatives.

The study aimed to validate mains water savings and volumetric reliabilities of rainwater tanks by monitoring the actual water usage at 20 new residential households in SEQ with plumbed end-uses from mandated rainwater tanks. Smart water-flow monitoring systems were installed at each household to obtain continuous data of water usage from the mains water meter, water allocation between internally and externally plumbed end-uses and rainwater tank top-up supply from the mains at a fine pulse resolution. The 20 monitored households were selected from across four new greenfield urban residential developments in SEQ to quantify the effects of localised rainfall pattern and household occupancy rate. The continuous monitoring data obtained was analysed mainly for per capita water use,

mains water savings from installed mandated rainwater tanks, diurnal water usage patterns as well as the supply reliability of the rainwater tank system. Studies have shown that diurnal water use patterns help provide significant insight into peak hour mains water reductions achieved through usage of rainwater, which in turn help determine design criteria for urban water supply network infrastructure (Lucas *et al.*, 2010). Hence, further analysis was carried out on the diurnal water demand pattern for the 20 monitored households as a cluster. This study is expected to help achieve a better understanding of the performance of mandated rainwater tanks in the *real world* as well as provide validation for mains water savings and demand profiles in comparison to other desktop studies. It is also expected that the findings from this study would provide important information for future urban water resource planning, given that a large number of mandated rainwater tanks are being adopted across Australia. However, as this study was conducted using a small data sample, further analysis involving a larger data set will be conducted to account for factors influencing variability in per capita consumption.

Methodology

Sample Size and Sample Selection

A group of 20 detached residential homes built after 1 January 2007, including those applying for extension permits, were selected for this study to include only those homes that were required to comply with QDC MP 4.2. This ensured that all homes had internally plumbed rainwater tanks installed on their properties. The chosen samples of 20 homes were spread across four local government areas (LGAs) in the SEQ region namely, Pine Rivers, Caboolture (both now part of the Moreton Bay Regional Council), Redland City Council and Gold Coast City Council.

Metering Setup at Twenty Mandated Households in SEQ

Each of the 20 homes is monitored at four relevant points in their rainwater systems to study the individual behavioural characteristics of rainwater consumption. Smart pulse meters were used to monitor the total water supply to the household through traditional mains and water flows in and out of the rainwater system. All pulse meters installed on the rainwater tanks and garden taps were calibrated at the rate of 0.5 L/pulse and all the potable mains water meters pulsed at 5 L/pulse, except for two households where the potable mains meters were set to pulse at 100 L/pulse due to different types of mains meter setups. All meters were capable of providing continuous data to wired data loggers located on-site. Figure 1 depicts the points at which flow meters are installed at each household.

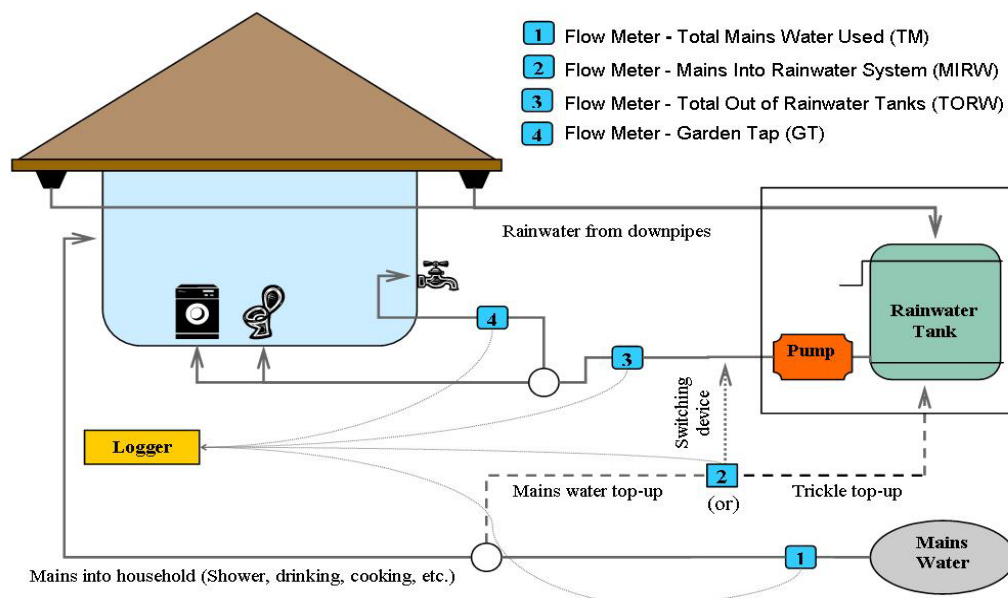


Figure 1. Illustrative diagram of metering system setup at each household.

Total Mains (TM)

The total mains water is the total potable water being supplied from the mains water line. This is an important parameter to determine variations in total water usage and daily consumption patterns and characteristics. In dwellings with mandated rainwater tanks, TM acts as the sole potable water supply to most internal home fixtures such as showers, cooking/drinking, and internal faucets, with the exclusion of filling of toilet cisterns and operation of wash machines where rainwater is used as the priority source.

Mains Water Top-up into Rainwater System (MIRW)

Mandated rainwater tank systems incorporate a mains water top-up system, which enable uninterrupted supply of water to the plumbed end-uses in the absence of rainwater. There are two types of top-up mechanisms usually installed at the 20 homes; nine homes operate on the “*trickle top-up*” mechanism and 11 homes operate on a “*rainwater switch*” mechanism. The *trickle top-up* mechanism operates on a “float” arrangement when every time there is a drop in water level below a stipulated point in the rainwater tank, a fixed volume of mains water is delivered into the tank. Whereas in the *rainwater switch* system, mains water bypasses the tank and pump systems and delivers directly to the designated end-uses until there is sufficient rainwater available in the tank again.

Total Water Supplied from the Rainwater Tank (TORW)

There is a difference in the metering of water supplied from a RWT system into the household depending on the type of top-up employed at each household. In a *trickle top-up* mechanism, the water exiting the RWT system was measured immediately after being pumped out of the tank. In this system, water leaving the RWT contains either rainwater or mains water (top-up) or a mixture of both. In a *rainwater switch* system, the water exiting the RWT is comprised solely of rainwater collected by the system. Thus, the total water supply from the RWT system is being metered accordingly.

Garden Tap (GT)

At least one garden tap in all homes is supplied water through the RWT system. The water supplied to the garden tap (Flow Meter 4) was also metered to differentiate external from internal end-use water demand.

Data Validation and Analysis Procedures

A comprehensive validation of the data was carried out prior to periodic monthly analysis. A complete logger data set consists of four water demand/supply data series for all 20 homes (excluding one home where the mains water flow data series was not available). The data was validated in terms of data presence, consistency, range and format, both manually and by using the Microsoft Excel® ‘macro’ function to ensure ambiguous data was excluded from the analysis.

Diurnal Pattern and Peak Water Demand Analysis

The diurnal water demand patterns were plotted by converting the available “per-minute” data for each home into diurnal hourly demand values and consequently combining all patterns to generate a cluster-of-20 homes diurnal demand pattern. The diurnal water demand pattern was used to identify peaks and trends in water usage and to generate further information for urban water planners.

Assessment of Mains Water Offset (Volumetric Reliability) in Diurnal Pattern Analysis

The volumetric reliability of the rainwater tanks is defined as the ratio of the rainwater available from the RWT system to the total household water demand during the monitoring period. The volumetric reliabilities for the households indicate the percentage of mains water offset being achieved in the systems through the usage of captured rainwater as a supply source for designated plumbed end uses (eg, filling the toilet cisterns, washing machine cold tap and external garden tap). The difference between the readings from flow meters three and two will be the rainwater supplied in this configuration. The rainwater offset for each RWT system was calculated using Equation 1 below:

$$Rv = \left(\frac{\sum_{t=1}^T (TORW - MIRW)}{\sum_{t=1}^T TM + \sum_{t=1}^T (TORW - MIRW)} \right) \dots\dots\dots (1)$$

Where:

Rv = Volumetric Reliability of the system (%)

T = Total monitoring/assessment time period (7 months)

t = Time step in minutes

TORW = Total water supplied from the rainwater tank system (kL)(flow meter 3)

MIRW = Mains water top-up into the rainwater tank system (kL) (flow meter 2)

TM = Total mains used in household (kL)

Results and Discussion

Per Capita Water Consumption

Figure 2 shows that the average household per capita water use for the 20 monitored households over a seven month monitoring period was 139 litres/person/day (L/p/d), which is significantly lower than the reported per capita water demand of 158 L/p/d (QWC, 2011) in the SEQ region between April-November 2011. Results showed that most homes located in 'above average' to high rainfall climatic regions displayed relatively higher per capita water use that led to higher mains water savings from mandated rainwater tanks. This was particularly apparent in homes located in Redland (eg, IPT5 and IPT15) and Gold Coast (IPT18) LGAs, where the recorded rainfall was found to be 695 mm and 657 mm, respectively. However, monitored homes (such as IPT12, IPT13 and IPT14) located in the 'below average' rainfall climatic region (Pine Rivers = Caboolture = 545 mm) (ie, Caboolture and Pine Rivers LGAs) showed significantly lower per capita water use and mains water savings from mandated rainwater tanks in comparison with the rest of the monitored homes. Further rainwater tank modelling studies will be conducted to investigate the impact of various physical factors (ie, connected roof area and effective tank size) and household characteristics (ie, types of plumbed end-uses) associated with the rainwater uptake. This would be an attempt to develop a more in-depth understanding of the performance of mandated rainwater tanks supplying rainwater as an alternative water source.

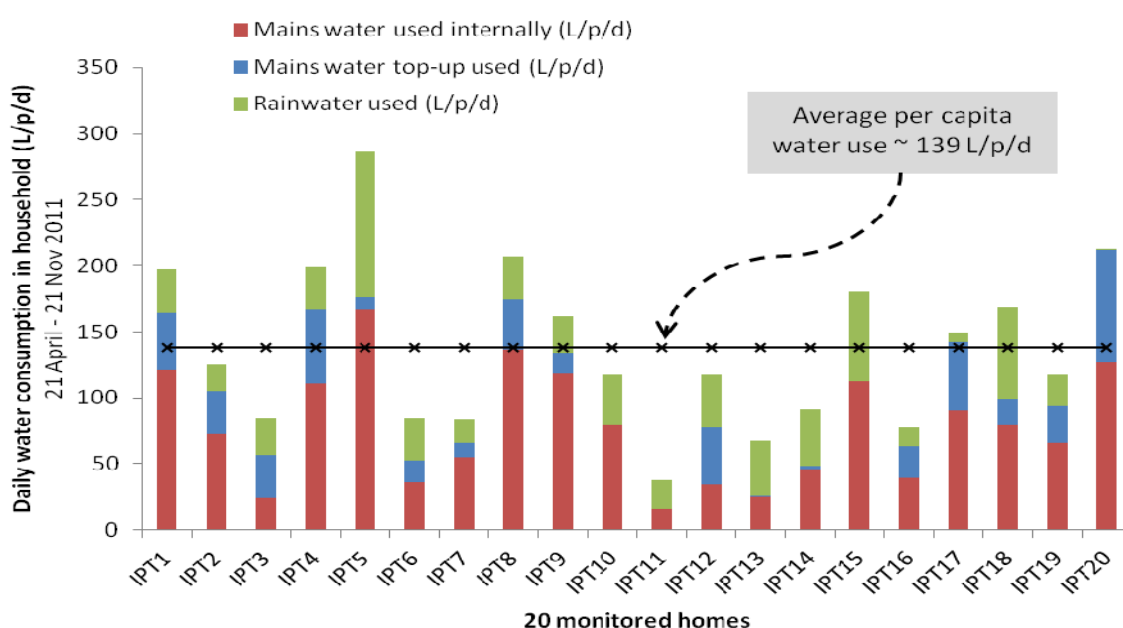


Figure 2. Per capita water consumption across the 20 monitored homes.

Diurnal Water Demand Pattern Analysis

Figure 3 demonstrates the diurnal pattern for water consumption in the cluster-of-20 homes (recorded over a 4-month period) normalised against the peak daily water demand. This facilitates an easy interpretation of the water balance fractions between the monitored streams of Direct Mains Water (DMW, the difference between the total mains used in the household and the mains water top-up into the rainwater tank), Mains into the Rainwater Tank (MIRW) and Total Rainwater (TRW) supplied from the RWT system.

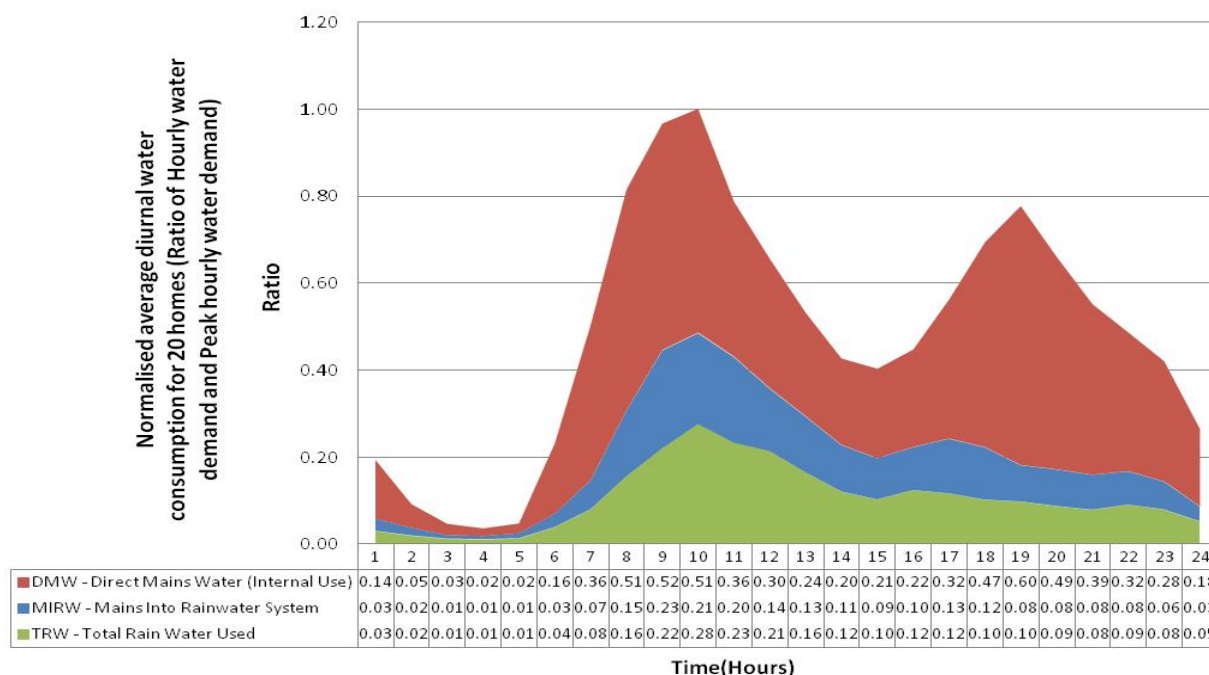


Figure 3. Normalised average diurnal water consumption patterns for a cluster-of-20 monitored homes.

From Figure 3, it can be observed that there are two distinct water demand peaks, representing the morning and evening peak water usage. The morning peak occurs between 8:00 and 11:00 hours, while the evening peak water consumption occurs between 18:00 and 20:00 hours with significantly higher usage of internal mains water supply and declining demand in rainwater supply. Around 49% of the average hourly household water demand during the morning water peak (at 10:00 hours) is met through the mandated rainwater tank supply system (either by captured rainwater or mains water top-up or both). Of the total demand during the morning peak hour, an average 28% is met solely through the rainwater available in the tanks. On the contrary, only 18% of the average hourly household water demand for the evening water peak (at 19:00 hours) was substituted from the mandated rainwater tank supply system, with the contribution solely from rainwater as a source being around an average of 10%.

This finding shows that the evening water peak is mainly attributed to end use fixtures that draw water directly from the mains water supply (ie, shower, kitchen taps/drinking water supply, dishwasher, etc). Previously, Thyer *et al.*, (2007) found that increase in typical end-uses such as toilets and showers provided the greatest increase in average water use resulting in higher peak hour mains water demand. In this study, the morning peak hour demands for the cluster-of-20 homes is directly influenced by more prominent and consistent water supply from the rainwater tanks to typical end uses such as toilet flushing, clothes washing and garden tap. However, the evening peak was seen to be a factor of other typical end-uses (eg, shower, kitchen taps/drinking water, dishwasher) that were dependant on mains water supply.

Volumetric Reliability of Rainwater Tank System

The average of the volumetric reliabilities across the 20 individual homes was determined to be 29% (Figure 4). In the previous phase of this wider study, Chong *et al.*, (2011) conducted a benchmark analysis to establish the average mains water offset achieved in IPT households (n=691) across Pine Rivers, Caboolture, Gold Coast and Redland LGAs, relative to the daily average water consumption in the SEQ region on the whole. For 2009, the mains water offset achieved through rainwater use in IPT homes was found to be 15%, 18%, 34% and 36% for the four areas respectively, with the overall average rainwater offset achieved in IPT households across all four areas being ~26%. Similarly for 2010, the rainwater offsetting the mains water in the same set of IPT homes was 22%, 25%, 33% and 31% respectively, with an average offset of ~28%. Thus, these figures match closely the average rainwater offset (volumetric reliability) of 29% currently achieved in this study at the 20 homes.

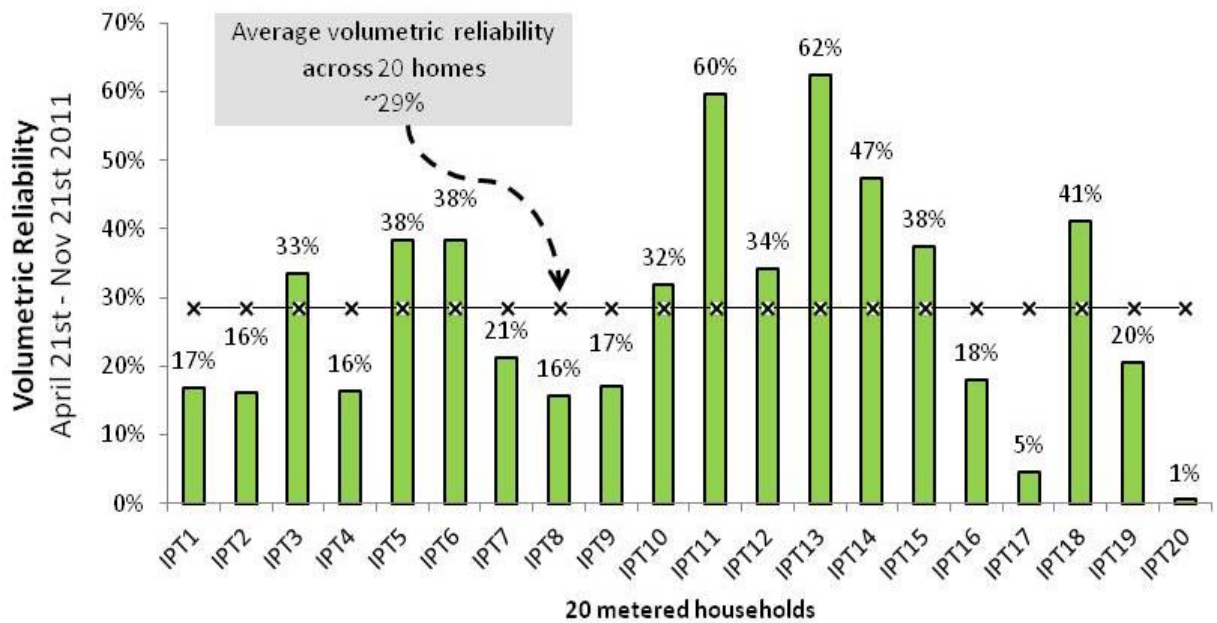


Figure 4. Estimated volumetric reliability of rainwater tank systems at 20 monitored homes.

From the analysis, it was also observed that the availability of rainwater had considerable influence on the supply reliability of RWT systems over a longer period of time. It was found that rainfall events did not influence the internal daily water demand from the RWT systems, with the exception of homes where the external garden tap was regularly used. This was supported by the long-term monitoring data which showed that there was little or no water drawn from the external garden tap during most rainfall events. The seasonal effects on the rainwater savings achieved at the 20 homes will be studied further as more complete data for the total monitoring period (a minimum of 12 months) becomes available, allowing a more comprehensive analysis of the behaviours of rainwater tank systems.

Conclusions

The study has provided improved insight into the real world performance of mandated rainwater tanks in SEQ. As the implementation of mandated rainwater tanks is still in its infancy, any monitored information on their contribution towards mains water savings is important for water professionals and policy makers. Results showed the average household per capita water use for the 20 monitored households was 139 L/p/d, which is significantly lower than the reported average per capita water demand across the SEQ region (158 L/p/d) for the same period. This could be attributed to lower rainfall during the analysis period. The average volumetric reliability of the rainwater tank systems to meet daily household water demands (29%) is comparable to previous analyses conducted to determine the mains water offset by the tanks.

Two distinct water demand peaks that are representative of the morning and evening peak water usage are evident from the average diurnal water demand pattern of the cluster of 20 monitored homes over four months period. From diurnal pattern analysis, on an average 28% of the morning peak hour demand is met solely through the rainwater available in the tank, whereas the rainwater contributed only an average offset of 10% in mains water consumption during the evening peak period.

Differences in household water usage based on local rainfall patterns were also found. Lower water consumption in areas with lower rainfall could possibly be attributed to householders' water use behaviours in these areas. Based on known parameters such as roof area, tank size and water consumption in each home, modelling will be conducted to determine the expected water saving outcomes in each individual dwelling for comparison with measured results. The findings from this study would provide information that would be significant for the future of urban water resources management and planning in SEQ, or even Australia wide.

Acknowledgements

This research is being conducted as part of the Urban Water Security Research Alliance (UWSRA), a research collaboration between the Queensland Government, CSIRO, The University of Queensland and Griffith University. Authors would also like to acknowledge Don Begbie, Director UWSRA for his continual support.

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Enabling the Use of the Lockyer Valley Groundwater System as a Buffer in the South East Queensland Regional Water Grid – An Assessment Framework

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Summary

This paper presents a planning and modeling framework which addresses the complex issues related to the use of groundwater as a managed common pool resource. Around 300-360 GL total groundwater storage volume is available, with top-up from an external source of water reuse. The world's third largest indirect potable reuse scheme (according to design capacity), located in South East Queensland (Australia), provides the example for this framework demonstration. In this case study, we analysed the implications of the supply of purified recycled water (PRW) to the Lockyer Valley to augment irrigation water resources using a combination of field research, water quality and quantity modeling and unstructured stakeholder interviews. We found that the use of PRW required the shift from a simplified end-of pipe planning exercise to the application of a more holistic framework, with well defined management strategies, that both recognise and balance the need to mitigate environmental risks and optimally deliver a reliable supply to the local community from the new high value water resource. The research undertaken in the Lockyer suggests that environmental and supply risks are manageable, but that rigorous evaluation needs to be performed. Detailed hydro-economic modeling of the scheme and a follow up on the first climate change impact calculations is recommended to assess long term cost benefits of the scheme.

Keywords

Wastewater reuse, groundwater, integrated water resources management, purified recycled water, peri-urban.

Introduction

The water supply security of South East Queensland (SEQ) has been increased recently by the provision of advanced wastewater treatment plants for potential 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 (Traves *et al.*, 2008). 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 advanced oxidation. Figure 1 depicts the seven safety barriers in a flow diagram in connection with upstream agriculture (Figure 1).

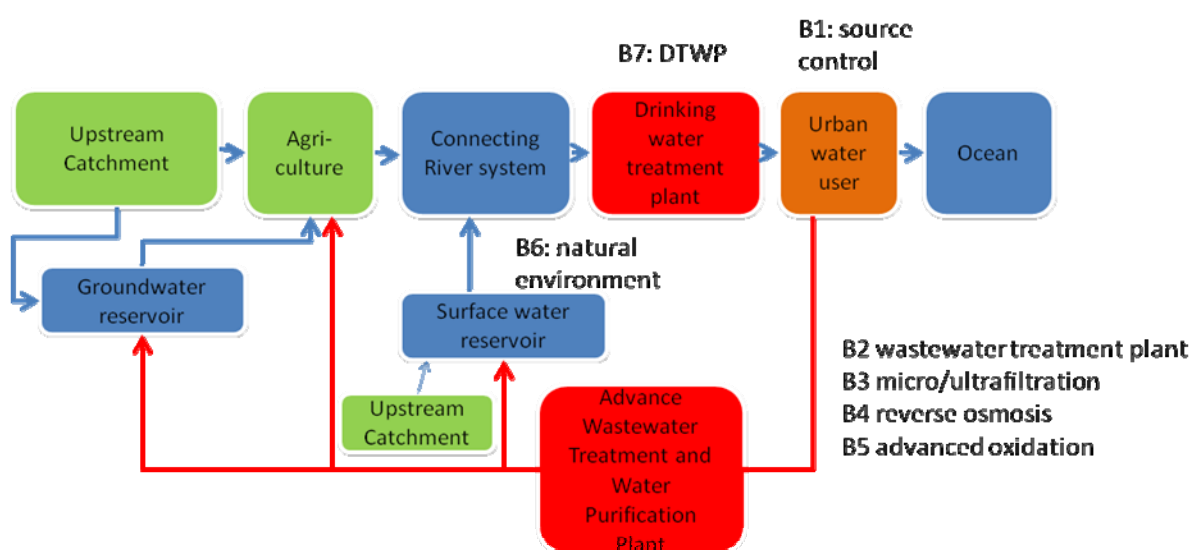


Figure 1. Urban-rural water exchange in the indirect potable reuse scheme for SEQ. Abbreviations B1-B7 relate to safety barriers in the indirect potable reuse scheme.

In this scheme, waste water from the Brisbane metropolis is treated to potable water standard (then branded as purified recycled water, PRW) with the intent to supplement the Wivenhoe dam reservoir (approx 1,200 GL capacity). Water from the Wivenhoe reservoir discharges into the Brisbane River and is tapped further downstream for the drinking water treatment plant at Mt Crosby. Using PRW to augment Wivenhoe Dam only when the combined key SEQ water grid storage capacity drops to 40% reflects the current one operating strategy for the SEQ Water Grid. However, simulations based on long term historic flow records (1880-2010) indicate that reservoir levels would have dropped below the 40% trigger level only twice with existing and committed infrastructure projects (Ref SEQ Water Strategy 2010). This infrequent requirement leaves significant volumes of PRW available to supplement non-potable uses such as agriculture. Against this background we analysed the possibilities and implications of using PRW as documented in the following sections.

Results and Findings

Problem Mapping and Stakeholder Consultation

Development of the management framework commenced with a problem mapping phase based on stakeholder consultation. The stakeholder consultation involved the SEQ Water Grid Manager, Queensland Water Commission (QWC), a number of different units and offices of the then 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). The stakeholder consultation opened up distinctively different positions on the delivery scenarios and the potential management of the PRW supply.

To convey these different positions, a conceptual problem map provides a means to communicate system complexity and interconnectedness, in a tangible and easily accessible way, which acknowledges how different technical, economical, political and cultural factors affect the fulfilment of the key objectives (Tiberghien, 2011). For the Lockyer Valley, we applied this technique with slight modification, to identify the factors which lead to the fact that the PRW supply scheme to the Lockyer is currently neither implemented nor in the detailed design phase (Figure 2). As will be explained in the section below, we suggest that the key factors hindering the implementation are: (i) unknown financial returns for the scheme operator; (ii) uncertain demand; and (iii) opposition from irrigators who are concerned about high water prices and changes in regulations upon PRW introduction.

Purified recycled water comes at a relatively high production cost (>\$AUD 500 per megalitre), due to the extensive treatment required for indirect potable reuse. This cost can be compared to current irrigation water charges of a maximum of \$AUD 30 per megalitre in the Lockyer Valley. For this reason, irrigators will go a long way to avoid using PRW at all, as long as groundwater is available. A business case for irrigators to use PRW only exists where supply reliability is compromised due to reduced groundwater levels severely restricting the available pumping supplies from bores/wells, and/or because of groundwater becoming more salty during low groundwater level periods. In other years, such as the very wet 2010/2011 season with catastrophic flooding in the Lockyer, the demand for PRW would be close to zero. Clearly, when defining management options to protect groundwater levels, the decision of where the target groundwater levels should be will differ between an environmental, regulator or water users perspective, and will dictate future PRW demand.

Three major supply scenarios are currently discussed:

1. Delivery of PRW to individual farm gates;
2. Delivery of PRW to major reservoirs, distribution via existing pipes to selected farmers and releases to the creek, with resulting recharge to groundwater aquifers from the creek; and
3. Direct injection of PRW into the groundwater via recharge bores.

While supply Scenario 2, which has the lowest investment costs in distribution infrastructure, initially appeared the preferred solution at the beginning of the research project in 2010, stakeholder opinion now tends towards supply Scenario 1. This change is triggered by the realisation of the complexity of common pool resource management, which is a barrier to farmer's acceptance of the PRW supply. Supply Scenario 1 provides farmers with increased control of their individual PRW use and consequently on the costs associated with using the additional resource. Still all three scenarios share the requirement to decide on the appropriate or optimum mix between low cost natural water resources and high cost imported water resources. This comes back to the question of defining the sustainable yield for this aquifer system (without augmentation) and the water volumes required to meet ecosystem targets. In recognition of the uncertainties associated with the estimation of the sustainable yield and the large climatic variability, we therefore propose to use defined target water levels as a trigger to decide on the necessity of PRW use (Figure 4).

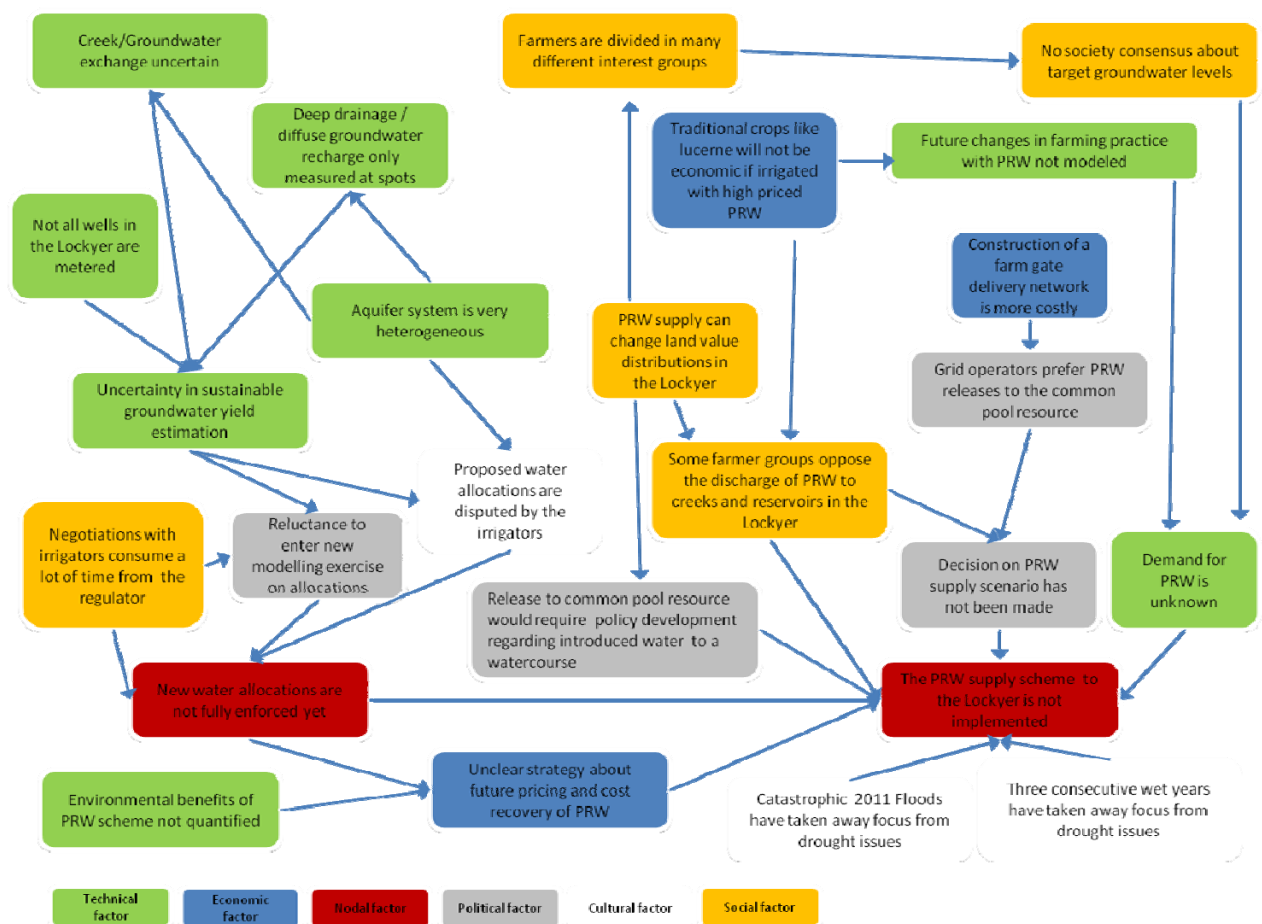


Figure 2. Conceptual map of technical, economic, political, cultural and social factors impacting on the supply of PRW to the Lockyer Valley.

Assessment Framework

Based on the experiences from the Lockyer Valley and a review of current literature, we propose a transferable assessment framework as a guide to future investigations either in the Lockyer Valley or elsewhere (as depicted in Figure 3). The framework applies not only to PRW, but also importing wastewater from alternate sources, and of variable qualities, requiring a similar scheme.

In **Tier 1** we suggest an ‘issues’ screening, starting with a simple water demand estimate and a comparison of water quality parameters. When importing to a natural environment, the key concern is not the compliance with drinking water guidelines alone, additional key parameters to assess are Sodium Adsorption Ratio as an indicator for potential soil structural impacts and salt content in relation to the natural water resources of the area. If the supplied water is already of drinking water standard, such as the case for PRW, the water quality impact assessment for downstream water supply systems can be limited in depth. Otherwise, minimum travel times on the path from irrigation to the next downstream drinking water plants should be investigated as part of a quantitative microbiological risk assessment. The low ion and nutrient content of PRW is not expected to have detrimental impact in this ion-rich setting. The issues screening in tier one can already identify available information on the likelihood and severity of the range of hazards, both of which are then detailed in the following tiers to arrive at a comprehensive environmental risk assessment.

The key objective of **Tier 2** is to provide a sufficient understanding of the natural system and the identification of environmental risks. This starts with a review of past landuse changes in response to different water availability and a mapping of areas which could be irrigated in future under additional availability of imported water. If metered use data is available, land use and irrigation water demand can be tested for correlation. Soil water balance models such as included in APSIM or Howleaky (Robinson, 2009) can be used to model irrigation demand and deep drainage under different landuse scenarios. In addition, Tier 2 should involve investigations of the potential salt accumulation and salt mobilisation. This can be achieved in simplified form by catchment salt balances, the collection of existing soil salinity distribution information or the drilling of soil salinity profiles. Numerical salt transport modeling for the unsaturated zone can be used, as necessary, to estimate the speed of salt release to the groundwater in areas which previously did not receive intensive irrigation. If indicated as a potential risk in Tier 1, Tier 2 may also include more detail on the response of the aquatic ecology to the new water source.

The aim of **Tier 3** is to quantify the demand for imported water from an ecosystem perspective, compare this demand with the reliability of supply perspective and identify potential consensus positions which recognise the tradeoffs and costs involved. This requires a model of the groundwater system's response to climate, withdrawal, irrigation strategies and potential direct injection of imported water. Outputs from this model can inform stakeholder consultation, with the aims of eliciting preferences and allowing deliberative multi criteria evaluation.

Finally, **Tier 4** is designed to provide additional robustness to the exercise by allowing the risks to the future operability of the concept to be considered. At the highest level of complexity, coupled hydro-economic models estimate the impact of different water availability and prices on the future land use and allocation strategies and then subsequently use the new land use information to generate new estimates of water availability, thus allowing for the full feedback loop in this coupled system. This hydro-economic model may also incorporate more complex operational rules and be an instrument to test the performance of different adaptive management strategies. Where available, downscaled climate change scenarios should be considered for increased robustness. The availability of the additional knowledge from the hydro-economic coupling will likely trigger a new round of stakeholder consultation and multi-criteria evaluation.

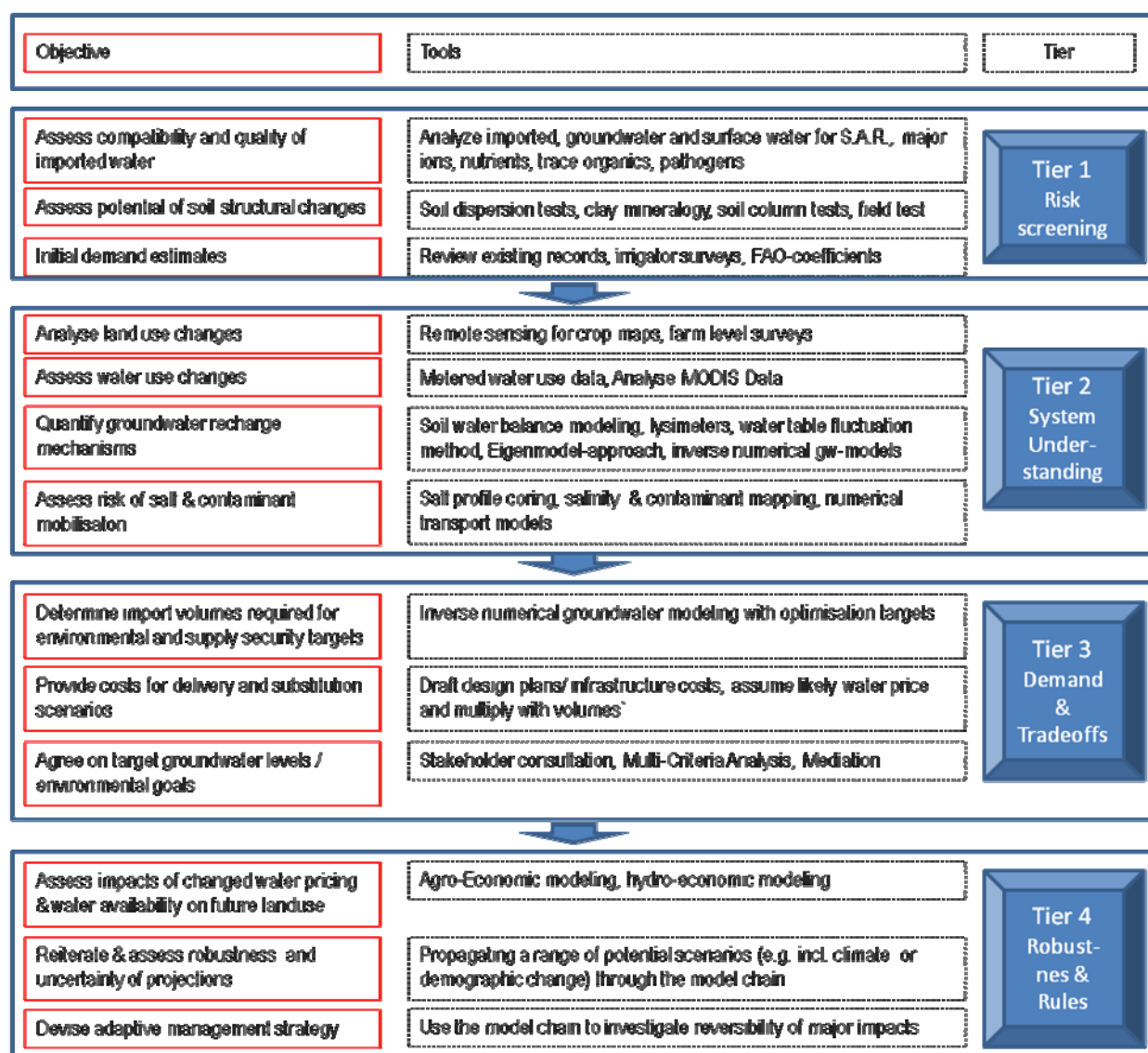


Figure 3. Multi-tier assessment framework for the import of recycled water to peri-urban agriculture.

Application of the Tiered Assessment Framework to the Lockyer Valley

Within the UWSRA research project “PRW in the Lockyer Valley”, a wide range of investigations were undertaken, which correspond to and which helped to develop the above mentioned framework. Table 1 summarises the applied methods and the key results in an abbreviated fashion. At the time of writing, substantial work is still ongoing, largely related to Tier 3. Hydro-economical modeling, as suggested in Tier 4, is not within the scope of the current research project.

Table 1. Elements of the assessment framework which were applied to the Lockyer and selected results.

	Methodology Applied	Results / Impacts
Tier 1	Hydrochemical analysis including Sodium Adsorption Ratio of PRW, surface water and groundwater; mechanical and spontaneous dispersion test.	SAR of PRW < 2, E.C. around 220 uS/cm, PRW water chemistry very close to current surface water chemistry, no spontaneous dispersion, low risk expected.
Tier 2	Analysed historical land use change using multi-spectral Landsat Thematic Mapper (Landsat 5 TM) imagery with atmospheric correction.	Bare land appears to change significantly according to water availability between 31 % (wet period Sep 10) and 56 % (dry period Jul 06) of the area (entire LV model zone).
Tier 2	Analysed water use changes.	Metered surface and groundwater use strongly declined in the period 1992-2010, correlating to lower availability of water, increased irrigation efficiency but probably also problems with aging meter infrastructure. Metered ground- and surface water use for the central Lockyer declined from ca 19,120 ML/a in 1992/1993 to ca. 4,413 ML (2009/2010 metered) or 6,568 ML/a (2009/2010 adjusted for suspected meter failures). Water demand estimated with soil water balance models ranged between 10-16 GL/a with an average of 13.5 GL/a for the central Lockyer and 36 GL/a for the entire Lockyer if a landuse like in September 2010 is assumed. Especially for the last decade, modelled water demand is exceeding the actual recorded usage in the Central Lockyer. This suggests that the significant variations in crop areas, caused by varying groundwater availability, and increased irrigation efficiency are not adequately being accounted for. Ageing meter infrastructure may also be a contributing factor.
Tier 2	Groundwater recharge assessed: Groundwater table fluctuation method applied.	Different groundwater table fluctuation methods suggest annual recharge volumes between 0 ML/a and 40 GL/a for the central Lockyer; long term average groundwater recharge as a combination of creek recharge and diffuse recharge ranges at 10-11 GL/a. In the period 1987-2011, groundwater storage in the Lockyer varied between 31% and 91% of historic recorded full groundwater storage in the central Lockyer.
Tier 2	Groundwater recharge focussing on creeks using the Eigenmodel approach combined with a numerical groundwater model.	Ongoing. Aquifer parameters were determined using an Eigenmodel approach for 11 wells in various distances to the Lockyer and Laidley creeks. The fast response of the wells suggests that recharge from creeks is the dominant recharge mechanism in the Lockyer Valley. The final recharge assessment will only be available with the calibrated numerical groundwater model.
Tier 2	Diffuse groundwater recharge assessed: Forward modelling of deep drainage using soil water balance modelling.	Deep drainage rates suggested to range between 89-112 mm/a based on soil water balance modelling. Long term deep drainage maps constructed were for different fixed landuses. Long term deep drainage figures assuming crop rotation currently range around 5.6 GL/a but may be reduced further with progress in soil water balance modelling.
Tier 2	Groundwater recharge assessed: Numerical modelling of the unsaturated soil zone with Hydrus, sensitivity study.	Time lag between deep drainage pulses and groundwater recharge pulses varies between 7 months and 5 years.
Tier 2	Salt risk: Catchment salt balances: Estimating the amount of salt added to the system.	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 compared to chloride input via rainfall of approximately 1084 t/a. The additional salt loading could be offset by increased baseflow to the Lockyer creek and increased lateral groundwater movement. PRW input to soil is small compared to current use of salty groundwater which put a first order estimate of 9800 t/a chloride to the soils of the Lockyer.
Tier 2	Salt risk: Analysis of salinity trends in groundwater.	Rising salinities exist before PRW supply in most of the upstream side valleys, no major trends were observed in the central parts of the alluvium. High salinities occur outside and at the fringe of the alluvium. Suggest not to use additional water to irrigate high salinity areas outside the main alluvial aquifer.
Tier 2	Salt risk: Deep soil profiles taken down to 20 m.	Major amounts of salt already washed out of the soil profile in areas where irrigators switched to surface water. Spatial heterogeneities prevent construction of salt flux maps.
Tier 2	Salt risk: Hydrus modelling of salt flux.	Modeling suggests that irrigation with 327 mg/L Cl (medium salty groundwater) will lead to 75% of the current salt being retained in the soil profile after 25 years whereas irrigation with 19mg/L Cl (PRW) would lead to less than 10% of the current salt load still present after 25 years in the profile at Forest Hill.
Tier 3	PRW Demand to fulfil ecosystem service and grid buffer goals: Coupled numerical surface groundwater modelling exercise using customised Modflow and IQQM.	Proof of concept modelling provided, initial results indicate a requirement of an average 37 GL/a water import to keep the entire Lockyer valley permanently at a 100% groundwater storage level.
Tier 4	Robustness: Limited irrigators survey on potential landuse changes under different water availability.	All six irrigators interviewed would expect to expand their irrigated area under availability of PRW, two out of six would consider to stop using groundwater (van Opstal 2010).
Tier 4	Robustness: Climate change impacts. Downscaled climate change scenarios were processed with IQQM and subsequent information on changed surface water flows was fed into a novel Eigenmodel tool at representative bores.	For the ECHAM scenario (as a median of the climate scenarios) the number of months which are below the 25% quantile of the historic groundwater levels in a period of 1,200 months increases by 373,330 and 266 months.

Demand for Recycled Water under Different Management Goals

As a ‘proof of concept’, the investigations suggested in tier 3 were undertaken using a modified Modflow model with a customised script (Moore, Woehling *et al.*, 2011). This was used to determine the required volumes of PRW to meet different target groundwater levels throughout the valley, in a variable climate (Wolf and Moore, 2011) (Figure 4). PRW demand was defined as the amount of water required to top up the aquifer, at any point in time. In this simplified ‘proof of concept’ form, the augmentation is provided directly to every model cell which indicates a groundwater level deficit. In a real world sense, the PRW would not be injected throughout the aquifer. Instead, most likely it would be delivered to farmers via piped supply and reduce their need for groundwater pumping. This reduction of pumping rates would lead to increased water availability in the aquifer. The figures modelled do however not account for the spatial dimension of this substitution yet, eg, if farmers dominantly in the lower Lockyer will be supplied with PRW, this will not lead additional water availability for the upper Lockyer valley. Alternatively, the other supply scenarios considered, viz. delivery of PRW to major reservoirs, and/or direct injection of PRW into the groundwater via a few bores would also give slightly altered PRW demand figures. Nevertheless, this ‘proof of concept’ analysis, does provide a starting point in the quantification of PRW demand that would be required to meet environmental targets, whilst ensuring supply reliability.

The current projection of PRW demand is based on a very limited economic analysis with no consideration of environmental benefits. The next model generation will need to specify which areas are likely to receive PRW substitution, considering the costs of building supply networks. Possibly, it will be cheaper to install more efficient irrigation systems instead.

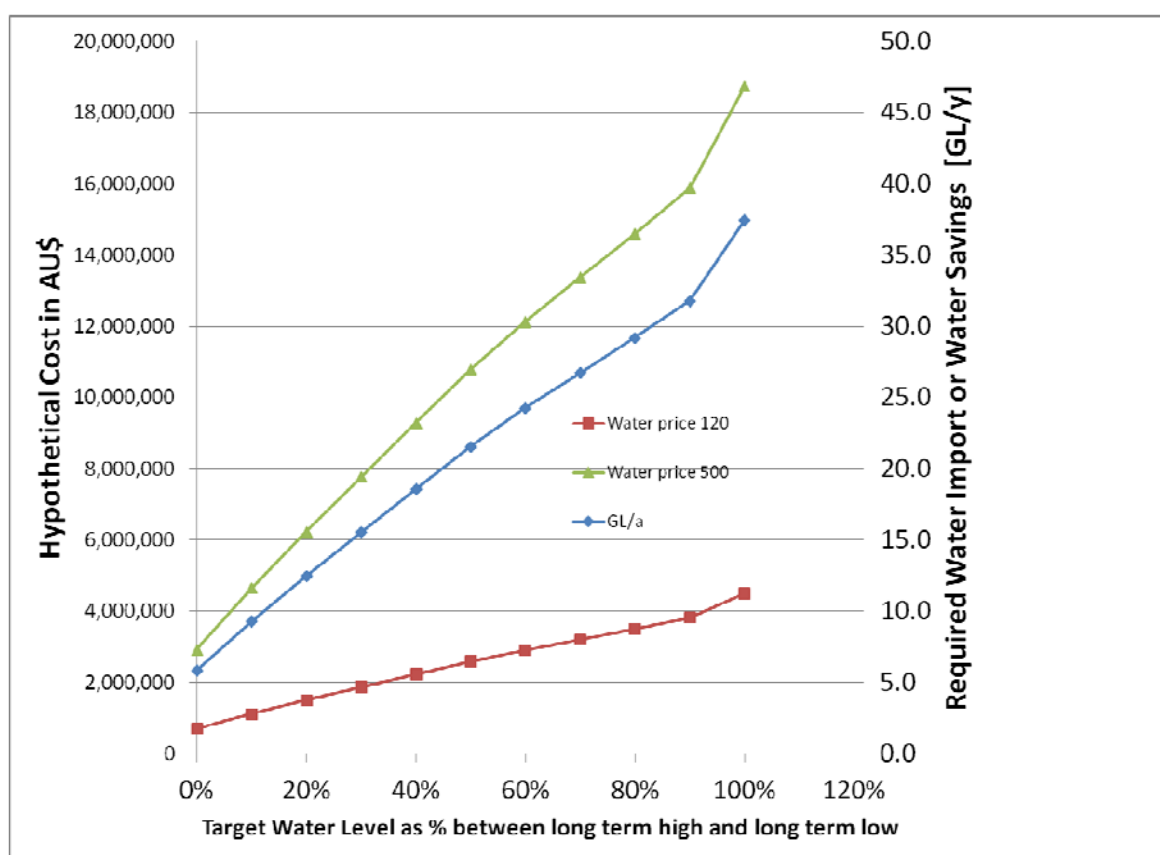


Figure 4. PRW imports or water savings required for different target water levels and hypothetical costs for two hypothetical unit water prices for imported water supply (Wolf and Moore, 2011).

A key benefit of maintaining high groundwater levels is the increased resilience against drought. Starting from high groundwater levels such as recorded for the year 1990 with an estimated stored water volume of 106 GL in just the central Lockyer, the alluvial aquifer would be able to supply the highest irrigation demand of 22 GL/a recorded for 1993 in the central Lockyer for more than three years until the the characteristic low water levels recorded between 2006-2007 (with an estimated groundwater storage of 36 GL) would be reached (assuming negligible lateral flow out of the alluvium in that period).

To allow for a rapid assessment of climate change impacts, Eigenmodel tools were fitted to the water level records of three bores in the Lockyer during a 21-year period. The Eigenmodel predicts groundwater levels using climate and surface water levels as input parameters. Since the impact of downscaled climate changes scenarios on surface water levels in the Lockyer was estimated by DERM using IQQM (Gooda, Voogt *et al.*, 2011), it was possible to use this data as an input to the Eigenmodels. Further validation on these initial figures is ongoing, however initial and simplified assessments indicate that the local climate scenarios derived from IQQM (Gooda, Voogt *et al.*, 2011) could result in low groundwater levels (eg, less than the 25% quantile of the historic runs) occurring twice as frequently in a median climate change scenario (ECHAM5).

Significance and Impact

The described research documents the evolving mindset from a simplified ‘end-of pipe’ planning exercise (which focused on the cost and infrastructure to supply the additional water) to the application of a more holistic framework which recognises both environmental risk and the need to provide a management strategy which delivers a substantial benefit from the new high value water resource to the local community. The research undertaken in the Lockyer suggests that while environmental and supply risks are manageable (Wolf, Moore *et al.*, 2011), the cost effectiveness of options of PRW supplementation needs to be examined via hydro-economic modeling. Simplified Eigenmodel assessments of climate change impact on three bores using downscaled climate scenarios (Gooda, Voogt *et al.*, 2011) suggest that low groundwater levels would occur twice as frequently in a median climate change scenario. This would obviously impact on water reliability in the future. If this is confirmed by more detailed studies, then the increased demand for PRW in the Lockyer Valley would need to be actively managed, using the groundwater system’s total storage of ca. 300-360 GL, as a climate buffer. Historic variations in groundwater storage during the last twenty years suggest a variation in storage volume of 70 GL just in the central Lockyer. The described assessment framework is transferable to other areas or other water sources than PRW (eg, also for a large scale import of Class A treated wastewater).

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Communication and Community Responses to Recycled Water

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Summary

When alternative water supply systems, such as purified recycled water (PRW), are proposed and implemented to supplement existing water supplies, it is crucial that communication about these systems is effective and meaningful. This project investigated the role of communication, specifically whether different communication techniques can influence community responses towards recycled water. Initial interviews with scientists involved in Urban Water Security Research Alliance (UWSRA) projects revealed uncertainty about what constitutes effective communication and how to best package information about PRW for the community. This work led to a comprehensive literature review detailing examples of communication about recycled water schemes as well as particular communication frameworks and theories applied in other domains. Evident from the literature review was a lack of experimental investigation of communication theories, such as the Elaboration Likelihood Model, in the recycled water domain. Therefore, two experimental surveys investigating communication about PRW were conducted, with a particular focus on the different information processing routes used by individuals. This paper details these experimental studies and the resultant implications for future communication about PRW schemes.

Keywords

Purified recycled water, community acceptance, Elaboration Likelihood Model, communication of risks and benefits.

Introduction

Investigation into the use of alternative water supply schemes to help augment current water supplies has become common practice both in Australia and around the world. Recycled water schemes, desalination plants and the utilisation of existing groundwater supplies are examples of such schemes proposed to help ease the pressure of contested water use that has emerged from population growth and changing rainfall patterns (Water Services Association of Australia [WSAA], 2005). Successful and unsuccessful attempts to implement alternative schemes across the world suggest that the acceptance of a scheme by local communities is paramount (Po, Kaercher and Nancarrow, 2003; Ingram *et al*, 2006; Hurlimann and Dolnicar, 2010). Local communities need to be at least informed of such schemes, making communication critically important to planning and implementation. This study, conducted as part of the Systematic Social Analysis project of the UWSRA, considers the role of communication in shaping community responses towards a purified recycled water (PRW) scheme in South East Queensland (SEQ). We begin by providing some examples of previous research investigating effective communication of recycled water, followed by a summary of communication conceptual frameworks that have been established in the social psychological field. Our research is then presented, aiming to fill a gap in the literature as well as provide practical communication advice for those planning and implementing recycled water schemes.

A review of the literature on how to best communicate scientific and technical information (see Green, Fielding, Leviston and Price, 2010) reported on a number of key elements necessary for effective communication that were based on climate change communication research conducted by Moser (2010). These key communication elements include:

- **Goals:** Communication goals need to be established. These may include education, engagement, building trust, or changing behaviour.
- **Audience:** Identify the audience(s) and understand their values, attitudes, beliefs, worldviews, social norms and context.
- **Framing:** Communication needs to be framed in ways that appeal to existing issues and concerns or highlight important aspects of the issue.
- **Messages:** Must be accessible, consistent, respect diversity and not overwhelm the receiver.
- **Messengers:** Trusted messengers are needed. Audiences can differ in their trust of messengers.
- **Channels:** Communication should be delivered through a variety of channels to appeal to diverse audiences.
- **Effect:** It is crucial to evaluate the effectiveness of communication. Evaluation should be aligned with goals and continually occurring.

Whilst these components can guide communication across a number of domains, there is still a gap in knowledge about how to best package information about recycled water by scientists and project leaders when trying to reach different audiences and stakeholders (Russell and Green, 2009).

Communication about PRW often begins with the provision of information about the scheme, which can be a powerful tool in raising awareness and changing behaviour if conveyed correctly (Marks and Zadoroznyj, 2005; Trumbo and O’Keefe, 2005; and Khan and Gerrard, 2006). However, many researchers have noted that the public are commonly treated as though they have a “knowledge deficit” (Libutti and Valente, 2006) and that providing more detailed technical information will be sufficient to prompt community acceptance, behaviour change and reduce controversy (Russell and Hampton, 2006; Nisbet and Mooney, 2007; Russell, Lux and Hampton, 2009). Russell and Lux (2009) suggest that it is important to investigate what the appropriate message is and what level of information is required, rather than assuming there is a complete knowledge deficit and expecting that providing facts and figures will address this deficit.

The role of information provision as a means of building trust has also been discussed in the recycled water literature (Hurlimann and McKay, 2004; Nancarrow *et al.* 2007). A key consideration is that the development of alternative water sources, such as recycled water, is challenging the basic trust that communities have in authorities to deliver traditional and safe household water (Marks and Zadoroznyj, 2005). Price, Fielding and Leviston (2012) investigated an unsuccessful attempt at implementing recycled water in Toowoomba and report that messengers are not viewed uniformly by the community, as trust in key campaign messengers (eg, local government and local business figures) is influenced by individual worldviews. Hurlimann and Dolnicar (2010) also report on the Toowoomba case study and emphasise the important role that the interest group Citizens Against Drinking Sewage (CADS) had in getting a negative message about recycled water out to the community first, hence creating a lack of trust and support.

Attributes of successful recycled water communication programs have been discussed in the literature. Marks and Zadoroznyj (2005) found that the recycled water schemes with extensive formal water communication strategies (eg, websites, newsletters, and invited public attendance/input at board meetings) resulted in residents with a higher awareness of regulations and rules. Khan and Gerrard (2006) suggest that a focus on the benefits in all messages will encourage greater community acceptance. However Miller and Buys (2008) found that certain arguments for and against recycled water can resonate differently with different genders, highlighting the difficulty in finding an appropriate frame that reaches a wide audience. Persuasive communication techniques have also been commonly noted in the recycled water literature (Russell and Hampton, 2006; Russell *et al.*, 2009), for example the use of marketing professionals by scheme proponents to identify barriers and design programs to overcome them (McKenzie-Mohr, 2000). Many successful programs have been attributed to appropriate community engagement processes which tend to encourage a better understanding of community attitudes, values and beliefs, and hence more creative communication methods and mutual understandings (see PUB, 2008 and Ingram *et al.*, 2006 for examples).

Conceptual Frameworks

Communication and persuasion research is common to social psychology, yet many of these theories and frameworks have not yet been applied to the recycled water domain. Experimental social psychology research has identified a number of factors that influence the effectiveness, or persuasiveness, of messages. Messages have three underlying components: 1) the structure, which includes argument sidedness, conclusions and sequential order; 2) the content, including evidence and emotional appeals; and 3) the language, such as the speed, intensity and power of the rhetoric used (Perloff, 2010). One-sided messages present only one perspective, whereas two-sided messages present information that opposes and supports a particular perspective. Two-sided messages have been found to be more persuasive than one-sided messages, provided that the message refutes the opposing perspective presented (Allen, 1998; O’Keefe, 1999). Two-sided messages can also enhance the credibility and trustworthiness of the messenger by explaining why opposing perspectives are incorrect (Perloff, 2010).

Understanding the way information is processed is important for communication. The Heuristic-Systematic Model (HSM) (Chaiken, Liberman and Eagly, 1989) and Elaboration-Likelihood Model (ELM) (Petty and Cacioppo, 1986) are two well known theories that describe the processes by which communication influences attitudes. These theories identify two different information processing routes. Effortful systematic modes of information processing are contrasted against less effortful modes based on heuristics or peripheral cues (Pierro *et al.*, 2005). The *central route* described in ELM requires people to carefully consider message arguments and implications, whereas the *peripheral route* involves focusing on cues that are not central to the message, such as the messenger’s attributes or external contextual factors, like colour and sound. When evidence is centrally processed, or elaborated upon, it can result in people modifying their attitudes, which is presented in ELM as determining persuasion.

When an issue is relevant or interesting, people are more motivated to process information via central processing routes which can result in longer-lasting changes in attitudes (Petty, Haugtvedt and Smith, 1995). Furthermore, when people demonstrate high involvement, they are more persuaded by longer message lengths (Kruglanski and Thompson, 1999) and messengers with high expertise (Petty, Harkins and Williams, 1980). Conversely, when people have low involvement in an issue, they are less motivated to process message arguments, instead focusing on peripheral cues (eg, the source of the message; Petty, Cacioppo and Goldman, 1981). Message complexity, or the difficulty associated with information processing, influences whether central or peripheral routes are taken (Pierro *et al.*, 2005).

The persuasiveness of communication is also contingent on the attributes and traits of the person receiving the message. There are a number of psychological traits and preferences that influence information processing. For instance, individuals who demonstrate a psychological need for closure, or who are averse to ambiguity and uncertainty, are more persuaded by information presented early in the sequence (Pierro *et al.*, 2005) and by low quality arguments (Petty *et al.*, 1981). Political conservatives have been found to emphasise in-group authority and purity when making decisions (Graham, Haidt and Nosek, 2009; Haidt, Graham and Craig, 2009) which may be salient to the recycled water context by driving opposition via the 'yukfactor'. Cultural solidarities, or worldviews, are thought to influence information processing as they reflect preferences for different types of risks and policy solutions. The four worldviews: Egalitarian, Hierarchical, Individualistic and Fatalistic, each represent a different assumption about ideal society and have been linked to risk perceptions (Dake, 1991; Lima and Castro, 2005; Steg and Sievers, 2000), making it a relevant concept in determining people's attitudes towards PRW. In addition, people are motivated to justify or rationalise social systems as fair and balanced which can prevent them from changing their attitudes towards the environment (Feygina *et al.* 2010). As such, system justification tendencies may influence people's attitudes to recycled water, by motivating them to form opinions that support the status quo and deny the need for alternative water treatments that benefit the environment.

Method and Results

The examples of social psychological communication theories demonstrate there is a need to experimentally test whether certain aspects of communication can shape individual responses towards PRW. It is hoped that the results will help to better inform recycled water communication strategies for planning, implementation and duration of a PRW scheme. This section details the two experimental studies conducted during 2011 and 2012.

Study 1

This study examined whether different recycled water messages impacted on residents' attitudes towards purified recycled water in an experimental setting. Broad aims were to investigate whether:

- (a) the provision of complex messages about recycled water treatment are more effective in influencing community attitudes to recycled water than simple messages;
- (b) one-sided messages detailing only the advantages of recycled water are more effective than two-sided messages that acknowledge, but refute, criticism of recycled water; and
- (c) the influence of individual differences, such as conservatism, political orientation and coping with ambiguity, on responses to recycled water information.

A series of hypotheses were tested in a two-by-two-by-two experimental design, which was devised to assess the influence of argument complexity (complex or simple information) and sidedness (one- or two-sided arguments) on participants' support for PRW over two time periods. An initial survey assessed participants' characteristics, traits and baseline attitudes towards PRW. Participants were contacted again three weeks later, and were provided with brief introductory information about the study and the PRW treatment process. Participants were then randomly assigned to one of four experimental conditions, in which they received messages about PRW that were either: 1) complex one-sided; 2) complex two-sided; 3) simple one-sided; or 4) simple two-sided arguments. Participants from the SEQ community aged 18 and over were recruited by a social research company, via access to an online panel of research volunteers, to participate in the study. Altogether 415 participants completed both the Time 1 and Time 2 surveys.

Overall, participants tended to be moderately accepting of PRW at Time 1, before any experimental communication strategies were tested. Participants mostly agreed that the water would be safe, could be added to dam water and that they would be prepared to drink it. The experimental conditions demonstrated that providing complex information increased support for potable recycled water and that trust in the Queensland (Qld) Government increased when both the arguments for and against recycled water were included in the information. However, participants that were initially ambivalent towards PRW increased their support when provided with a one sided (only portraying arguments for recycled water) message. There was also evidence that scientists were the preferred messengers over government and that individual characteristics of respondents influenced their acceptance of recycled water prior to the provision of information as well as their responses to different types of information. Specifically, egalitarians, those who have an emphasis on the group and believe in the importance of protecting nature, were more likely to support PRW than fatalists, who perceive nature and humanity as unpredictable and unfair (Verweij *et al.*, 2006). Participants with a low tolerance of ambiguity also recorded higher levels of support when they received complex rather than simple arguments, perhaps indicating that more information can clarify any misconceptions. A summary of survey results as they pertain to the aims and hypothesis of the study are provided in Table 1.

Table 1. Summary of survey results matched with project aims and hypothesis.

Aims	Summary of Results
a) The provision of complex messages about recycled water treatment are more effective in influencing community attitudes to recycled water than simple messages.	Participants receiving complex information recorded higher mean support for recycled water at Time 2 (T2). A small significant interaction was observed for complexity and time, with participants receiving complex information recording significantly higher mean support at T2. However, participants in the complex condition did not have significantly higher mean support than those in the simple group at T2.
b) One-sided messages detailing only the advantages of recycled water are more effective than two-sided messages that acknowledge, but refute, criticism of recycled water.	No significant main effect was observed for argument sidedness on support. No significant interactions were observed for time and sidedness.
c) The influence of individual differences, such as conservatism, political orientation and coping with ambiguity, on responses to recycled water information.	Moderate positive relationships were observed between support for recycled water and comfort with technologies, trust in information from the Qld Government, trust in information from scientists, and system justification.
Hypotheses	Summary of Results
H1: Participants' support for recycled water at T2 will be influenced by argument i) complexity; and ii) sidedness.	A small significant effect was observed between participants with the complex argument condition and support for recycled water at T2. Argument sidedness did not have any influence on support.
H2: Participants' trust in information from the Qld Government will be influenced by argument i) complexity and ii) sidedness.	Participants receiving two-sided information had a higher mean trust in information from the (Qld Government at T2 (slight significant effect). No effects between argument complexity and trust in the Qld Government.
H3: Participants that oppose purified recycled water at T1 will have higher support levels at T2 for the two-sided argument conditions.	Those that opposed recycled water at T1 did not significantly change their support at T2 in one and two- sided conditions. Participants who were uncertain whether they support recycled water or not responded positively to one-sided arguments (ie, significantly higher support for recycled water at T2).
H4: Participants that support recycled water at T1 will have higher support levels at T2 for the one-sided arguments conditions.	Participants that supported recycled water at T1 did not record significant differences in support at T2 when receiving either one or two-sided arguments.
H5: Participants that oppose recycled water at T1 will have higher levels of trust in information from the Qld Government in i) complex and ii) two-sided conditions.	Those that opposed recycled water at T1 recorded higher mean levels of trust in information from the Qld Government at T2 for complex information (when compared to simple) however, the difference was not significant. There were no significant interactions between support groups (those that oppose, are ambivalent or support it), trust in information from the Qld government and sidedness.
H6: Participants with low tolerance of ambiguity will respond more positively to simple one-sided arguments than: i) complex two-sided arguments; and ii) people with high tolerance of with ambiguity.	Participants with low tolerance of ambiguity had higher mean levels of support for complex rather than simple conditions (small significant effect). Participants with high tolerance of ambiguity had higher mean support for recycled water under complex conditions than the low tolerance group but this difference was not significant.
H7: Those with high purity moral relevance will have: i) lower levels of support for recycled water; ii) higher discomfort with ambiguity; iii) more right-wing political orientation; and iv) higher levels of system justification.	Participants with high purity moral relevance demonstrated significantly lower tolerance of ambiguity than the mid and low purity moral relevance group.

Study 2

The second experimental study was developed to further investigate communication processing pathways with a particular focus on the Elaboration Likelihood Model of communication (Petty and Cacioppo, 1986). When revisiting the literature it appeared that personal relevance plays a particularly important role in how people process information. Previous social investigations conducted as part of the UWSRA project had demonstrated interesting findings in terms of people's perceptions of risks, benefits and the 'yuck factor' associated with PRW (Browne *et al.*, 2008). Therefore, Study 2 aimed to incorporate the above findings into a series of research questions examining whether particular information processing routes impact on an individual's acceptance of and attitudes towards recycled water. For example:

- Is the ‘yuck factor’ a form of peripheral processing? Specifically, do those for whom PRW is of low relevance demonstrate more negative emotions to PRW?
- Does risk communication hinder rather than promote acceptance of recycled water? Is communication about the benefits of PRW more effective in fostering support?
- Do participants for which PRW is of high issue relevance respond better to messages about the low risks of PRW by demonstrating higher levels of support?
- Do participants for which PRW is of low issue relevance respond better to messages about benefits of PRW by demonstrating higher levels of support?
- Do conditions of risk, benefits and risk and benefits combined impact on participants’ level of support, emotion and perception of information?

The survey was piloted with 180 participants, specifically to test a number of issue relevance statements. The strongest measure of central processing was then included in the final survey, along with a series of manipulation checks and measures to determine attitudes towards PRW and risk. Both surveys were administered online and with 180 individuals participating in the pilot and 800 in the final survey.

At the time of writing the results from Study 2 were not yet available. These will be presented at the UWSRA 4th Science Forum in June 2012.

Conclusions

The introduction of alternative water supply systems, such as PRW, requires effective and thoughtful communication strategies. The two empirical studies presented here have investigated elements of communication such as message complexity, message sidedness, personal issue relevance, and portrayal of risks and benefits. In addition a number of individual difference measures that may shape responses to recycled water were included in the survey.

The results from Study 1 indicate that scientists are the preferred messengers over government and that individual characteristics of respondents can influence acceptance of recycled water. Therefore, effective communication about PRW could entail tailoring information to suit specific groups. Study 2 provides insight into the usefulness of the Elaboration Likelihood Model for communicating and processing information about PRW. Both studies demonstrate the intricacies of communication and suggest a range of ideas for effective communication about PRW. These findings are relevant to not only those working within the UWSRA but also for other places around the world where PRW is being considered or is already in place.

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Towards the Quantification of Rainwater Tank Yield in South East Queensland by Considering the Spatial Variability of Tanks

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Summary

This paper reports a study aimed at quantifying the yield from rainwater tanks in South East Queensland (SEQ) by considering the spatial variability of tanks. The methodology involves Monte Carlo simulation of tank storage behaviour by considering the spatial variability of input variables, ie, tank sizes, inflows to the tank, household water demand and losses associated with the tank inflow. Probability distributions and probability based methods have been used to represent the spatial variability of input variables. The results have indicated that the error introduced to the average annual yield of a system with multiple internally plumbed rainwater tanks can be in the order of 14% when spatial variability is ignored. For single family new residential households (ie, detached dwellings) in SEQ, the rainwater tank yield found through stochastic simulation is 37 kL/h/y, if the tank is used for the same household end uses. Further work is in progress to include a large sample of tank sizes and roof areas as well as water uses of both new and existing single family residential houses in Gold Coast, Ipswich and Sunshine Coast.

Keywords

Rainwater tank yield, stochastic simulation, potable water savings, rainwater harvesting.

Introduction

Domestic rainwater tanks are promoted as an alternative source of water supply in Australian major cities. Understanding the amount of water supply (ie, yield) from rainwater tanks is essential for urban water supply planning, in particular to assess the long-term supply security. A common approach used by practitioners to quantify the yield from a system consisting of a large number of domestic rainwater tanks is, linear extrapolation of the supply obtained from a single tank based on average tank and water demand characteristics. This approach assumes that the yield of individual domestic rainwater tanks in a given area is the same for all the tanks and, that the tank characteristics and household water uses have linear relationships with the tank yield. However, several studies have shown that the amount of water supplied from rainwater tanks varies with such factors as prevailing climate, tank volume, area of the roof connected to the tank and household water use (Fewkes and Butler, 2000; Fewkes and Warm, 2000; Coombes and Barry, 2007; Mitchell, 2007; Ghisi, 2010; Basinger *et al.*, 2010; Khastagir and Jayasuriya, 2010; Palla *et al.*, 2011 and Neumann *et al.*, 2011). These studies have also shown that the tank and water use characteristics have non-linear relationships with the tank yield. Also, studies on household end use measurements have clearly shown that the volume of water use by individual end uses varies from house to house (Roberts, 2005; Willis *et al.*, 2009; Beal and Stewart, 2011).

The spatial variability of supply from domestic rainwater tanks in an urban area has also been reported in Beal *et al.* (2012) and Chong *et al.* (2011a and 2011b). Unlike the above-mentioned studies based on computer simulations, these studies have analysed water consumption data obtained from households with and without rainwater tanks. Both studies have examined rainwater tank supply in South East Queensland (SEQ), Australia, where installation of internally plumbed rainwater tanks for toilet, clothes washing and garden uses, is a mandatory requirement in all new houses (Queensland Water Commission, 2010). Beal *et al.* (2012) have used 2008 water consumption data, and have found that rainwater tank yield in the SEQ varies from 20 kilolitres/household/year (kL/h/y) to 95 kL/h/y with a mean of 50 kL/h/y. Chong *et al.* (2011a and 2011b) have used 2009 and 2010 consumption data, and have found that rainwater tank supply varies from 24.5 kL/h/y to 88.5 kL/h/y with a mean of 58.8 kL/h/y. These studies clearly show that supply from rainwater tanks can vary spatially. At present, further studies are in progress in the SEQ to quantify the spatial variability of tank sizes and tank inflows. Some outcomes of such studies are used in the analysis reported in this paper.

All these studies indicate that it is not realistic to assume the factors that can affect the supply from domestic rainwater tanks would remain uniform in a given urban area, which could be a suburb, a town or a city. As mentioned earlier, these factors include tank size, connected roof area, losses from the roof due to different roof material, prevailing climate, household occupancy rates, household end uses to which tank water is used and the water usage behaviour of individual household occupants. Therefore, it is likely that an approach that uses linear upscaling of the yield of a single tank with average characteristics to determine the yield of a system with multiple rainwater tanks, can introduce errors. The main reason for such errors is the non-linear dependency of the rainwater

tank yield on the parameters that define household water use and the tank (Mitchell *et al.*, 2008; Neumann *et al.*, 2011; Maheepala *et al.*, 2011).

Yield of a system with multiple rainwater tanks has been examined in a number of studies (Mitchell *et al.*, 2008; Xu *et al.*, 2010; Neumann *et al.*, 2011; Maheepala *et al.*, 2011; Mashford *et al.*, 2011 and Coultas *et al.*, 2011). All these studies have considered the spatial variability of the above-mentioned factors. They have shown that the use of average values for rainwater tanks characteristics as well as for household water demand can result in an overestimation of the supply from a system with multiple tanks. The reported overestimations are in the order of 14% for Melbourne-based data (Mitchell *et al.*, 2008; Xu *et al.*, 2010; Maheepala *et al.*, 2011), 18% for Canberra-based data (Maheepala *et al.*, 2011) and 14% for Brisbane-based data (Coultas *et al.*, 2011).

This paper reports a study aimed at quantifying potable water savings of domestic rainwater tanks in SEQ, which is one of the fastest growing urban regions in Australia, covering about 22,420 km². The analysis reported in the paper can be considered as an extension to Coultas *et al.* (2011), which reported preliminary results of a study undertaken to quantify the yield of rainwater tanks in SEQ, using assumed probability distributions for tank sizes, effective roof areas and household end water uses, based on literature sources. The reason for using assumed data in Coultas *et al.* (2011) was the lack of observed (or measured) data at the time of undertaking the study. Observed data relevant to SEQ have become available since then, and this paper reports an analysis undertaken to quantify the yield of rainwater tanks in SEQ, considering the spatial variability exhibited in the observed data of tank characteristics and household demands.

Methodology and Results

To quantify the yield of rainwater tanks in SEQ, we used Monte Carlo (or stochastic) simulation, which was a method for iteratively evaluating a deterministic model using sets of random numbers as inputs (Kroese *et al.*, 2011). Our method involved simulation of storage behaviour of a rainwater tank using sets of tank and household end use characteristics sampled either directly from probability distributions, or from a large number of plausible values generated using probability based methods. Rainwater tank characteristics included tank size, connected roof area and losses from the roof. Two types of losses were considered: initial and continuing loss of water from the roof. Tank characteristics were sampled from probability distributions. Water demands of household end uses were sampled from a set of plausible time series, which were generated using a probability-based method for predicting the household water use.

The rainwater tank model described in Mitchell *et al.* (2008) was used for the stochastic simulation of rainwater tanks. The Mitchell *et al.* (2008) rainwater tank model was a water balance model, capable of simulating the processes of rainfall, roof runoff, and tank storage behaviour. It consisted of two modules: rainfall-runoff module, which computed the amount of roof runoff into the tank, and storage module, which computed the amount of water stored in the tank. The model allowed each tank parameter be specified either as a continuous probability distribution with a minimum and a maximum value, or as an average value.

Table 1. End use frequency statistics of 61 SFR houses in Brisbane (data source: Beal and Stewart, 2012).

Statistic	Frequency (events per day)							
	Toilet Half Flush	Toilet Full Flush	Tap	Shower	Bath	Dishwasher	Clothes Washer	Irrigation
Mean	4.87	4.21	58.70	2.13	0.13	0.55	0.71	0.12
Std Dev	3.97	2.68	33.42	1.99	0.28	0.68	0.56	0.19
Skewness	1.67	1.29	1.13	5.11	2.12	1.94	2.93	1.94

Table 2. End use event mean volume statistics of 61 SFR houses in Brisbane (data source: Beal and Stewart, 2012).

Statistic	Mean Volume of End Use Event (litres/event)				
	Toilet Half Flush	Toilet Full Flush	Tap	Dishwasher	Clothes Washer
Mean	3.89	7.44	1.19	6.55	99.45
Std Dev	1.10	1.58	0.51	8.82	69.06
Skewness	-0.49	1.23	1.18	1.76	1.10

Table 3. Shower flow rate and duration statistics of 61 SFR houses in Brisbane (data source: Beal and Stewart, 2012).

Statistic	Shower Event Flow Rate (litres/minute)	Shower Event Duration (minutes)
Mean	7.82	5.72
Std Dev	3.18	2.22
Skewness	2.32	1.23

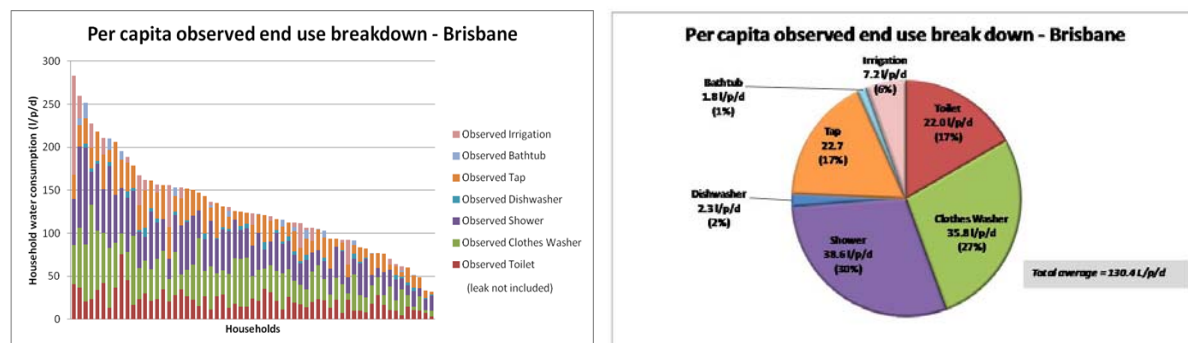


Figure 1. Observed per capita water consumption of 61 houses in Brisbane (data sourced from: Beal and Stewart, 2011).

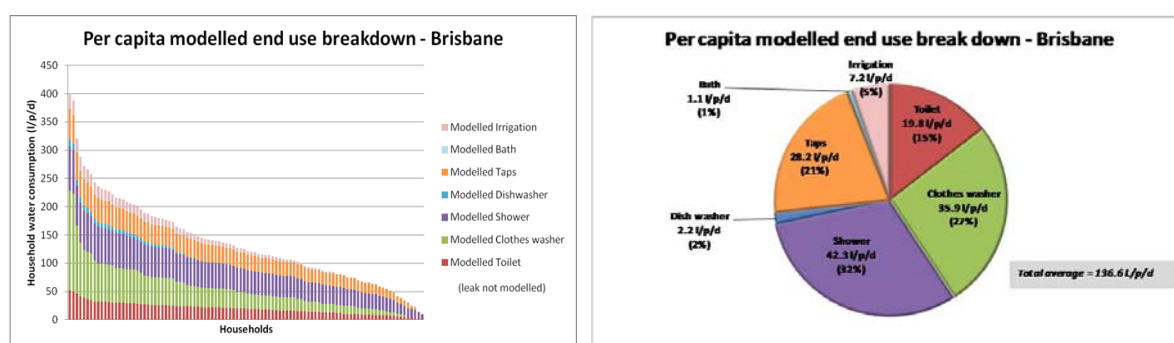


Figure 2. Modelled per capita water consumption of 100 houses in Brisbane.

The probabilistic demand model described in Duncan and Mitchell (2008) was modified and used to simulate the water demands of household end uses. The household end uses included in this model were: toilet use, tap use, showers, baths, dishwashers, clothes washers and garden irrigation. Modifications to the household demand simulation method included changing fixed volumes of water used for taps, clothes washer, dishwasher and baths to probability distributions, which allowed accounting for the spatial variability present in such end uses. The probabilistic demand model quantified the water demand of each end use through a two stage process. The first stage defined the probability of an end use starting in a given time step using diurnal data. The second stage quantified the volume of water use by that end use, during the given time step, using probability distributions of frequency of that end use (input variables are shown in Table 1) and volume per event (input variables are shown in Table 2 and Table 3). All end use demands were generated at one minute intervals, which were then aggregated to any higher order time step. The probabilistic demand model was calibrated using Brisbane's measured data sourced from Beal and Stewart (2011).

Table 4. Observed and modelled household end use demands for Brisbane.

	Household End Use Water Demand in Litres/Person/Day							Total
	Toilet	Clothes Washer	Shower	Dishwasher	Tap	Bathtub	Garden	
Observed Mean	21.9	35.8	38.6	2.3	22.7	1.9	7.2	130.4
Observed Standard Deviation	11.9	20.7	20.9	2.5	11.4	3.9	17.4	55.1
Modelled Mean	19.8	35.9	42.3	2.3	28.2	1.1	7.6	136.6
Modelled Standard Deviation	10.3	30.8	15.4	3.4	10.6	0.02	7.1	38.7

The measured data represented 61 single family residential (SFR) households, contained a mixture of efficient and non-efficient household appliances, and an estimated amount of water considered to be lost through leaks. Leaks were not considered in our analysis simply because the probabilistic demand model did not have an option to model leaks. The average household consumption during the measured period (ie, 14-28 June 2010) without considering leaks was 130.4 Litres/person/day (L/p/d) (Figure 1 and Table 4). The simulated or modelled value of household consumption was 136.6 L/p/d (Figure 2 and Table 4). Comparison of the observed end uses of 61 houses and the modelled end uses of 100 houses are shown in Figure 1, Figure 2 and Table 4. The calibrated demand model was used to generate 100 plausible demand time series over the simulation period (ie, January 1960 to December 2010), in order to feed into the Monte Carlo simulation to determine the tank yield.

The probability distributions for tank variables and household end uses were constructed from the observed data of 20 new SFR households. Details of the rainwater tank parameters used for the Monte Carlo simulation are given in Table 5. In line with Queensland's current water savings target (Queensland Development Code, 2008; Queensland Water Commission, 2010), toilet use, clothes washers and garden irrigation were supplied from the rainwater tank. The rainwater tank simulation assumed that the supply from the tank was switched to mains supply, when the tank was empty (ie, no trickle supply). Behaviour of the rainwater tank was simulated on a daily basis. Simulation was carried out over a period of 50 years, from 1960 to 2010. The simulation process involved computation of the daily supply from the tank over the simulation period, over a large number of iterations. For each iteration, a set of tank parameters was sampled from the probability distributions given in Table 5, and a time series of demand was sampled for each end use being supplied from the tank, from a sample of 100 plausible demand time series, generated from the above-mentioned calibrated demand model. An iteration could be viewed as daily simulation of tank behaviour of a detached dwelling with an internally plumbed rainwater tank over a 50 year period. Number of iterations was varied from 100 to 35,000, and for each case, the average annual yield was computed from the daily time series of tank supply. It was noticed that the average annual yield became almost a constant when the number of iterations was greater than 10,000 (Figure 3). Hence 10,000 iterations were considered as adequate to represent the spatial variability of tank supplies, for our study.

Table 5. Parameters values used for Monte Carlo simulation of rainwater tank behaviour of 50 years.

	Tank Size	Effective Roof Area	Initial Loss	Continuing Loss	Occupancy
Units	KL	m ²	mm	%	No.
Minimum	0.79	27.00	0	0	1
Mean	4.67	76.60	0.5	15	2.6
Maximum	5.61	135.00	1.75	30	6
Probability Distribution	Normal	Normal	Normal	Normal	
Standard Deviation	1.06	28.84	0.5	5	
Sample size	20	20	0 ¹	0 ¹	61

Note 1: data not available for SEQ. Used Melbourne-based data reported in Xu *et al.* (2010).

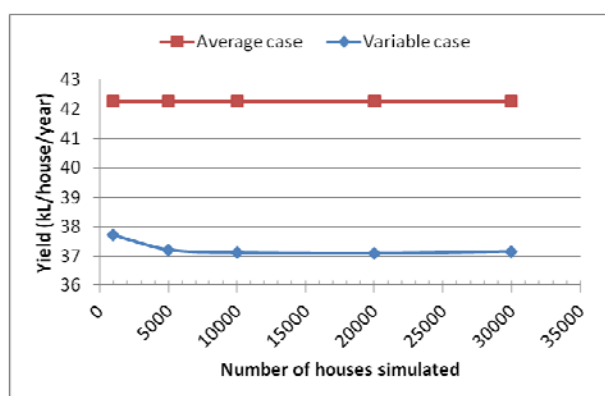


Figure 3. Expected rainwater tank yield for different runs of stochastic simulation.

The average rainwater tank yield for 10,000 tanks (or iterations) is shown in Figure 4. The average annual yield vary from 12.6 kL/h/y to 88.7 kL/h/y, with a mean value of 37.12 kL/h/y and a standard deviation of 9.97 kL/h/y. That is, the long-term, expected rainwater tank yield in the SEQ for SFR households is 37.12 kL/h/y, if the tank water is used for toilet flushing, garden watering and clothes washing (ie, the red horizontal line shown on Figure 4). To examine implications of the common practice for computing tank yield, a simulation was performed by using average values obtained from the 10,000 iterations, for tank parameters and demand time series. The tank yield for the average case was 42.28 kL/h/y (ie, the purple horizontal line shown on Figure 4), which was about a 14% overestimation compared to the yield obtained by considering the spatial variability of tank supply.

The tank yield computed through stochastic simulation was compared with the tank yield computed by the Queensland Water Commission (QWC). The QWC study involved the use of billing records of 1,841 single family residential (SFR) houses in Brisbane during the period from January 2011 to June 2011. This sample had 120 SFR houses with internally plumbed rainwater tanks (IPR) and 1,721 SFR houses without IPR. It was assumed that the houses with IPR supplied toilet, clothes washer and external irrigation from the rainwater tank. The sample with IPR was called 'sample B' and the other sample was called 'sample C'. Both samples were considered to be consisting of efficient appliances for toilets, clothes washers and showers. The average household consumption of sample B and C were 381 litres/household/day (L/h/d) and 481 L/h/d, respectively. Based on a comparison of the average water consumption in sample B and C, the average yield obtained from a rainwater tank in Brisbane was estimated as 38.46 L/p/d (or 36.55 kL/h/y based on the average occupancy rate of 2.6 people/household, which is the same for the sample of 61 houses used in our study).

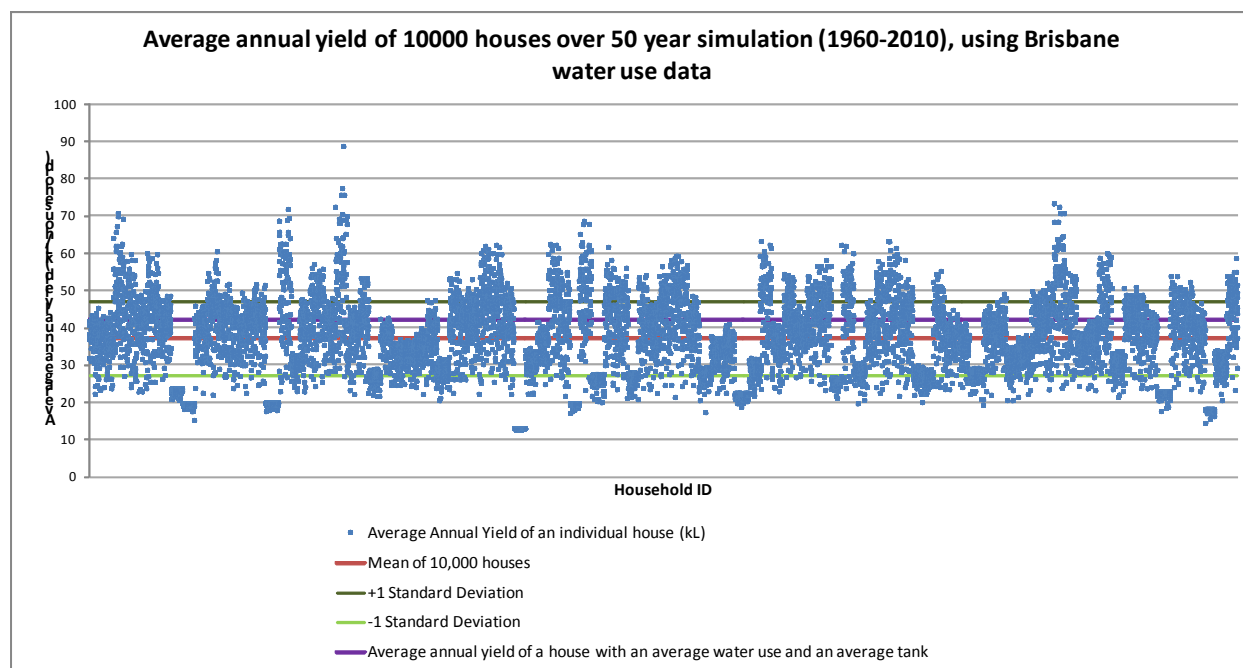


Figure 4. Monte Carlo simulation of 10,000 domestic rainwater tanks with each tank's behaviour simulated on a daily basis over 50 years.

In summary, our study indicated 39.09 L/p/d (or 37.12 kL/h/y), as the rainwater tank yield in SEQ, which is of similar order of magnitude to the QWC's study which indicated 38.46 L/p/d (or 36.55 kL/h/y) as the rainwater tank yield in SEQ. Our study considered the spatial variability of tank characteristics and household water use and used stochastic simulation to capture that variability, in order to quantify the tank yield, whereas the QWC study compared billing records of SFR houses with and without rainwater tanks in Brisbane. The fact that the tank yield obtained from our study is of similar order of magnitude to the tank yield estimated from the billing records (difference is 0.63 L/p/d), indicates that stochastic simulation has the ability to capture the spatial variability present in rainwater tank supplies across an urban area in a reliable manner.

Conclusions and Work in Progress

In this study, we have showed that potable water savings obtained with domestic rainwater tanks in SEQ can be quantified and upscaled to the regional scale satisfactorily and reliably, through the use of stochastic or Monte Carlo simulation technique. Upscaling of rainwater tank yield is essential to assess the security of water supply at the SEQ regional scale. The tank yield found through the stochastic simulation of supply and demand behaviour of rainwater tanks in the SEQ is 37 kL/h/y, which is of similar order of magnitude to the tank yield estimated from the billing records by the QWC. If the spatial variability present in tank water supply and household water consumption is ignored, the tank yield is 42 kL/h/y, which is about a 14% overestimation compared to the tank yield that takes into account the spatial variability of tank supply.

However, it should be noted that for the study reported in this paper, the spatial variability of rainwater tank characteristics were derived only from 20 new single family residential houses in SEQ and the spatial variability of household water use was derived from 61 new and existing single family residential houses in Brisbane, which might not be an adequate sample to represent the SEQ region. Hence the above-mentioned results should be used cautiously for the SEQ region. The study is in progress to include a large sample of tank sizes and roof areas as well as water uses of both new and existing single family residential houses in Gold Coast, Ipswich and Sunshine Coast.

Acknowledgement

This study has been funded by the Urban Water Security Research Alliance, which is a research partnership between the Queensland State Government, CSIRO's Water for a Healthy Country Research Flagship Program, Griffith University and the University of Queensland. We would like to sincerely thank Tad Bagdon, Mark Askins, Patricia Hurikino and Phillip Chan of the Queensland Water Commission, for their valuable advice and providing access to results of their study on estimating rainwater tank yield in SEQ.

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Strategies for Managing the Condition of Rainwater Tanks in South East Queensland

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Summary

This paper reports on a study on the topic of rainwater tank management. The study has been based on exploration of literature, interviews and surveys in combination with focus groups and workshops; as well as some synthesis of the elicited information, including by the means of computational models. The study has been prompted by the emerging realisation that rainwater tanks installed in response to the recent drought in South East Queensland (SEQ) have not always been kept in adequate condition. This poses risks in terms of health, as well as in terms of potentially losing the supply benefits that water planners hoped to achieve with rainwater tanks as being incorporated in the regional strategic water planning under integrated urban water management approaches. This paper provides some discussion on what is known about rainwater tank conditions in SEQ, and then proposes some strategies for the management of rainwater tank conditions. Although, the study has been conducted for a local region in Queensland, Australia, however the investigation approach can be implemented in any part of the globe where on-going operation of rainwater tanks is essential from overall supply considerations.

Keywords

Rainwater tanks, risk management, decentralised systems, governance and planning.

Introduction

Rainwater harvesting and collection in tanks as a supplement to water provisions is adopted widely in many cities around the world. Uptake of this practice is driven by the need to provide water when other sources are inadequate for ensuring safe and sustainable supply (Sharma *et al.*, 2008; Zhang *et al.*, 2010). For example, in Australian cities, household rainwater tanks have been promoted by various levels of government by means of legislation, rebate schemes and subsidies, as a way to reduce the severity of water restrictions, and to improve the supply-demand balance. As a consequence, the uptake of rainwater tanks for suitable dwellings¹ is above 40% in some major capital cities such as Brisbane and Adelaide. In South East Queensland (SEQ), new single dwellings are required to have rainwater tanks and/or other non-mains water sources installed (such as greywater treatment plants, communal rainwater tanks, dual reticulation or treated storm water), in order to meet specified mains water savings targets (Queensland Government, 2008). In places such as these, rainwater tanks provide an increasingly important contribution to urban water supplies and it can therefore be argued that it is important to protect the government's and community's investment in such infrastructure. Despite this, there is currently very little understanding of the current state of the rainwater tank asset stock.

In the case of SEQ, being used for purposes of illustration in this paper, rainwater tank uptake has increased markedly. From 2007 to 2009, the uptake of rainwater tanks in the State of Queensland increased by 15.9%, with rainwater tanks installed in 38% of households in Queensland by 2009 (Australian Bureau of Statistics, 2007; 2009). Brisbane experienced the largest uptake; with rainwater tanks in 43% of suitable dwellings (Australian Bureau of Statistics, 2010). Rebate programs contributed to over 230,000 domestic rainwater tanks installed between 2006 and 2008 (Queensland Government, 2005), and, since 2007, changes in the local development code, referred to as the Queensland Development Code MP4.2 (Queensland Government, 2008), resulted in an additional 30,000 tanks installed in new homes (Gardiner, 2010). Rainwater tank uptake is highest in new dwellings, with 57% of new houses less than one year old being connected to a rainwater tank (Australian Bureau of Statistics, 2010). The costs of installing and operating rainwater tanks are in many circumstances competitive compared to alternative water sources, such as additional dams or desalination plants, but this competitiveness depends on the exact costs of maintenance (Tam *et al.*, 2010). From a water management and planning perspective, rainwater tanks provide some public good benefits, in the form of water savings and can assure an independent supply during water restrictions, plus provide reductions in peak stormwater runoff. Private benefits include the independent supply of water of reasonable quality at times of water restrictions. This considerable investment in private infrastructure is considered by water planners in their projections, thereby being able to delay major investments in public infrastructure such as desalination facilities or new dams. It is therefore of critical public interest to protect this investment. In Australia however, some water planners are becoming increasingly worried about the condition of the stock of rainwater tanks in private backyards, but there is limited data to gauge whether there is a real need for concern.

¹ In this context, the Australian Bureau of Statistics define a "suitable dwelling" as a separate house, semi-detached, row/terrace house, townhouse, etc.

This paper reports on a study to identify and explore approaches to managing rainwater tanks. It builds on the work on understanding and describing the context of rainwater tank management in SEQ (Moglia *et al.*, 2012a), exploring the available data and professional judgments on rainwater tank conditions (Moglia *et al.*, 2012b) and focus groups and workshops to find out about community attitudes (Walton *et al.*, 2012). Subsequently, the authors have undertaken interviews with selected experts in urban water management to quantify a Bayesian Network model, which empirically describes the design of a management strategy for rainwater tanks.

Discussion

What is Known about the Condition of Tanks?

Ensuring the ongoing condition of rainwater tanks is critical, and the exact condition of relatively new rainwater tanks in SEQ is unclear, particularly for those installed before 2007. This has presented the program of research with a significant dilemma, especially during its early stages. As a result, two streams of research activities were undertaken to address this issue; surveys of professionals and plumbers; and surveys of tank owners.

Surveys of Professionals and Plumbers

Literature and interviews were used to identify the types of things that can go wrong with rainwater tanks, with the main issues being (Moglia *et al.*, 2012a): gutters can get blocked; the mosquito meshing can be broken or removed; pumps may malfunction or be broken; there may be structural concerns about the tank or its foundation; there can be problems with automatic switching valves or trickle top-up settings; and there may concerns about the water quality. In addition, data was collected from professionals and plumbers, using email based surveys, as to their estimates of expected frequency of such concerns occurring (Moglia *et al.*, 2012b). Plumbers, who are perhaps likely to see tanks which have problems, estimated that about 46% of pumps were broken; 35% of mosquito meshing were removed or broken; 40% of automatic switching valves or top-up settings had problems; 20% of tanks had water quality issues, and 5-10% had possible structural problems related to the tank itself or its foundations (Moglia *et al.*, 2012b). The professional judgements were less negative, and more varied, but on average estimated: 19% of tank systems with pump problems; 21% broken or removed meshing; and 2% of tanks with structural condition problems (Moglia *et al.*, 2012b). This indirect data is obviously not ideal and the authors recognise the need to obtain more accurate information. One small-scale, and unpublished, survey of tank conditions in SEQ of mandated tanks indicated minimal tank maintenance behaviour occurring.

Surveys of Maintenance Behaviours

At present, householders have the responsibility for management of their rainwater tanks, thus they need to know how to do the required operation and maintenance tasks, and to be motivated to do so. The extent of scientific data on the practices and motivations of householders in relation to rain water tanks is scarce, with only a few studies available on the issues (Gardiner, 2009, 2010; Gardiner *et al.*, 2008; Tilbrook, 2009). Upkeep and maintenance of rainwater tanks is closely linked to the level of engagement and knowledge of householders regarding their tanks and rainwater collection system (Gardiner, 2010). In rural and remote areas where households depend solely on rainwater for their water needs there is a long history of rainwater use for all purposes, including drinking, and a track record of appropriate maintenance. However, in urban areas where reticulated water supply is the main source of water, rainwater is connected mainly to garden taps, laundry cold water tap and toilet cisterns. In addition, rainwater tanks systems must be fitted with mains water back-up to ensure continuous water supply. As a result there is large variance in the understanding of rainwater risks and attitudes towards maintenance (Gardiner, 2009, 2010; Gardiner *et al.*, 2008).

During 2007-2008, Gardiner (2010) conducted a survey of 1,051 people in SEQ and verified that although 95% of tank owners reported confidence in managing their tanks, a significant number did not conduct proper maintenance. For instance, 50% of the sample of mandated tank owners reported never having conducted maintenance such as the cleaning of gutters or screens, inspection of the inside of the tank, and only did so if a problem was detected (Gardiner, 2010). It seems likely that a key factor in these findings is that the tanks were relatively newly installed, and that maintenance requirements would therefore be limited. White (2009) conducted a survey of 279 SEQ households with rainwater tanks regarding operation and maintenance (O&M) practices and concluded that maintenance of rainwater tanks was adequate, with tank owners reporting on average 6.2 hours per year on gutter maintenance, with 76% performing self-maintenance, 12% relying on professional service and 12% relying on visiting friends or family. However, it was also reported that long-term behaviour would be difficult to gauge as the majority of tanks were less than three years old and 86% less than one year old at the time of that survey (White, 2009).

Tilbrook *et al.* (2009) undertook a survey of 145 homes in Lake Macquarie in New South Wales, Australia, where rainwater tanks are part of the mandated BASIX program for achieving water savings. Most tanks were installed in 2005 and the survey was undertaken in 2009, so the tanks in question were approximately four years old. One of the thematic areas of the survey was on the issue of tank knowledge and maintenance. More than 50% of respondents rated their knowledge of the required maintenance tasks as poor. Some of those who considered themselves to have good or fair knowledge of rainwater tank maintenance also adjusted their response after answering the following maintenance questions, saying that they had less knowledge than first anticipated. Furthermore, it was found that less than half the respondents carried out regular maintenance on the gutters (32%), the first flush device (23%), and the inlet on the tank (39%). Nearly 30% of respondents said they had gutter guards on their gutters and thus had not provided any maintenance to the gutters. Only two respondents said that they had de-sludged their tank.

Tucker *et al.* (2011) undertook a large scale survey of 1,984 households in SEQ to provide greater insights into attitudes and behaviours of rainwater tank owners. Results indicated participants with mandated rainwater tanks were found to have lower levels of motivation than retrofitters suggesting that they may experience a sense of lack of control and independence when relating to their tank, and their subsequent drive to engage in maintenance behaviour may lack self-directed motivation, and it may be seen as a meaningless activity.

As a whole, whilst a number of surveys have been undertaken, the focus of each has varied, and the subsequent results context dependent and at times inconsistent with each other. Only Tucker *et al.* (2011) have really explored motivations and the psychology of householders in relation to rainwater tanks, and there is a significant need for better understanding those issues. The motivation to display O&M behaviours seems variable, and subject to complex socio-psychological factors, and hence strongly contextual and dynamic. There seems to be some important factors that do contribute to motivation, including education, providing opportunity for choice, and a perception of achieving private and public goals. What works in terms of motivating householders in one community, may not necessarily work in another community. Therefore, it has been suggested that an adaptive approach to management would be appropriate in such cases (Moglia *et al.*, 2011; Moglia *et al.*, 2010).

Management of Rainwater Tanks

Faced with the knowledge that some form of intervention is required to ensure that rainwater tanks are being kept in adequate condition, the obvious next step is to think about how to manage rainwater tanks in order to achieve this. Recognising that this is a policy issue that concerns a number of different stakeholders, such as the SEQ community, state government and businesses, it is appropriate to consult widely to identify the evaluation criteria for a management strategy as well as the bounds of what is feasible. Interviews have been undertaken with key state government departments, the Queensland Water Commission (QWC), plumbers, tank manufacturers and engineers. Furthermore, a workshop was undertaken with such key stakeholders to elicit a list of possible management strategies; and subsequently community focus groups were run in order to explore community attitudes to various options (Walton *et al.*, 2012). There are two key requirements of a rainwater tank stock that have been identified through interviews, and these are as follows:

1. Health: it is critical that rainwater tanks do not pose a health risk to the community. The risk to the community comes in three forms, a) from drinking contaminated water, b) by allowing mosquito breeding in tanks and related spread of mosquito born disease (ie, primarily dengue fever), and c) potential physical injury related to damage of tank foundations or attached structures.
2. Water supply: it is also important that rainwater tanks contribute in the long term as an additional water supply source as this is the main function of tanks. Water planners need to be sure of tanks providing this water in the future and this projected availability of water supply will help avoid or delay further water supply investments (such as desalination plants, or further grid connections).

Supply Risk Mitigation

The sole purpose of tanks is that they provide water and hence allow water planners to avoid or delay major infrastructure investments. The tank failure events that pose risks to the ability of providing water are simply: blocked gutters (reduced supply), broken or faulty pumps (no supply), or faulty or broken plumbing (reduced or no supply). The main mitigation strategy for all of these types of issues is to make sure that the tanks and their various parts are installed and designed correctly and do not break subsequently. In fact, interviews suggest that these three factors each play an almost equally important role in an overall mitigation strategy. In other words, a tank that has been designed well, and installed well, but is not maintained, will experience problems. A tank that has been designed poorly, but is installed and maintained well, will experience problems. And a tank that has been installed poorly, but has been designed well and is adequately maintained, will have problems. Hence, the risks are: 1) poor design, 2) poor installation, and 3) poor maintenance. The mitigation strategies for these types of risks are shown in Table 1.

Table 1. Water supply risk mitigation strategies.

Water Supply Risk	Mitigation Strategy
Poor Design	Guidelines and standards for designs of tanks need to be updated in light of the experience with urban rainwater tanks in the last few years. In particular, more attention needs to be given to designs that minimise maintenance requirements. Ensuring compliance with revised standards is also important.
Poor Installation	There is a need to provide better training for plumbers to increase their skills in rainwater tank installations. Also, the process for certification of tank installation currently allows for some poor installations to occur and not be dealt with. The regulation regarding certifications needs to be improved. Plumbers specialised in rainwater tank installation report that, according to their high standards, currently as much as 70% of tanks are poorly installed (mostly DIY).
Poor Maintenance	The only feasible and acceptable mitigation strategy that has been identified by experts and the community is for householders to have the responsibility to maintain their tanks. Currently, community, experts and plumbers all report largely inadequate maintenance activities by a majority of households. Social scientists indicate that a complex set of socio-psychological factors influence the likelihood of householders maintaining their tank, and this implies that managers and policy makers need to pay close attention to their interactions with the community, and may also invest in education programs, public campaigns, and provision of information to support this mitigation strategy. This however appears to be the most difficult of all the mitigation strategies to implement.

Health Risk Mitigation

The health risks can be mitigated by the following actions, as per Table 2.

Table 2. Health risk mitigation strategies.

Health Risk	Mitigation Strategy
Drinking of Contaminated Water	Communicating a clear message to the community not to drink rainwater. It is extremely hard if not impossible to make rainwater safe without treatment. If individuals choose to drink the water it is their own responsibility.
Spread of Mosquito Borne Disease	It is absolutely critical that mosquito meshing are not removed or damaged. There needs to be a high level of certainty that this meshing is intact for 99% of tanks.
Physical Injury due to Damaged Tanks / Foundations	Tank owners need to be aware of the dangers, and need to be encouraged to act if risk of physical injury exists.

The mitigation strategies in Table 2 would hence involve a communication strategy; to make sure that householders are aware that drinking rainwater is not advised, and that householders need to be active in mitigation of any risk of physical injury relating to their tanks. The more difficult issue is to create certainty that 99% of mosquito meshing are in good condition. This would require some level of maintenance, combined with a representative sample of tanks.

Governance Certainty

Another key aspect of a management strategy that has been clearly stated by Queensland Health and the QWC in particular is the need to have certainty that the health risks and the supply risk are adequately managed. Generating certainty would require regular random samples to achieve high confidence statistical estimates regarding the condition of the rainwater tank population in SEQ. To enable such random sample strategies to be undertaken, there is a need for having one centralised database of rainwater tanks in SEQ. Proposed random sampling frequencies would be annual for mosquito meshing; and perhaps every three years for supply capacities. The exact number of tanks that would need to be sampled, or the sampling strategy, would need to be defined in a statistical fashion on the basis of an initial smaller sample of tank conditions.

Conclusions

This paper has reported on some of the current understanding regarding rainwater tank management that has been generated as part of our study on this issue. This understanding has helped us identify some strategies for rainwater tank management, and these have been briefly described. What seems pretty clear is that there is no simple solution to this problem, but any strategy would need to address a set of issues simultaneously. A Bayesian Network model currently under development shows exactly that, as shown in Figure 1.

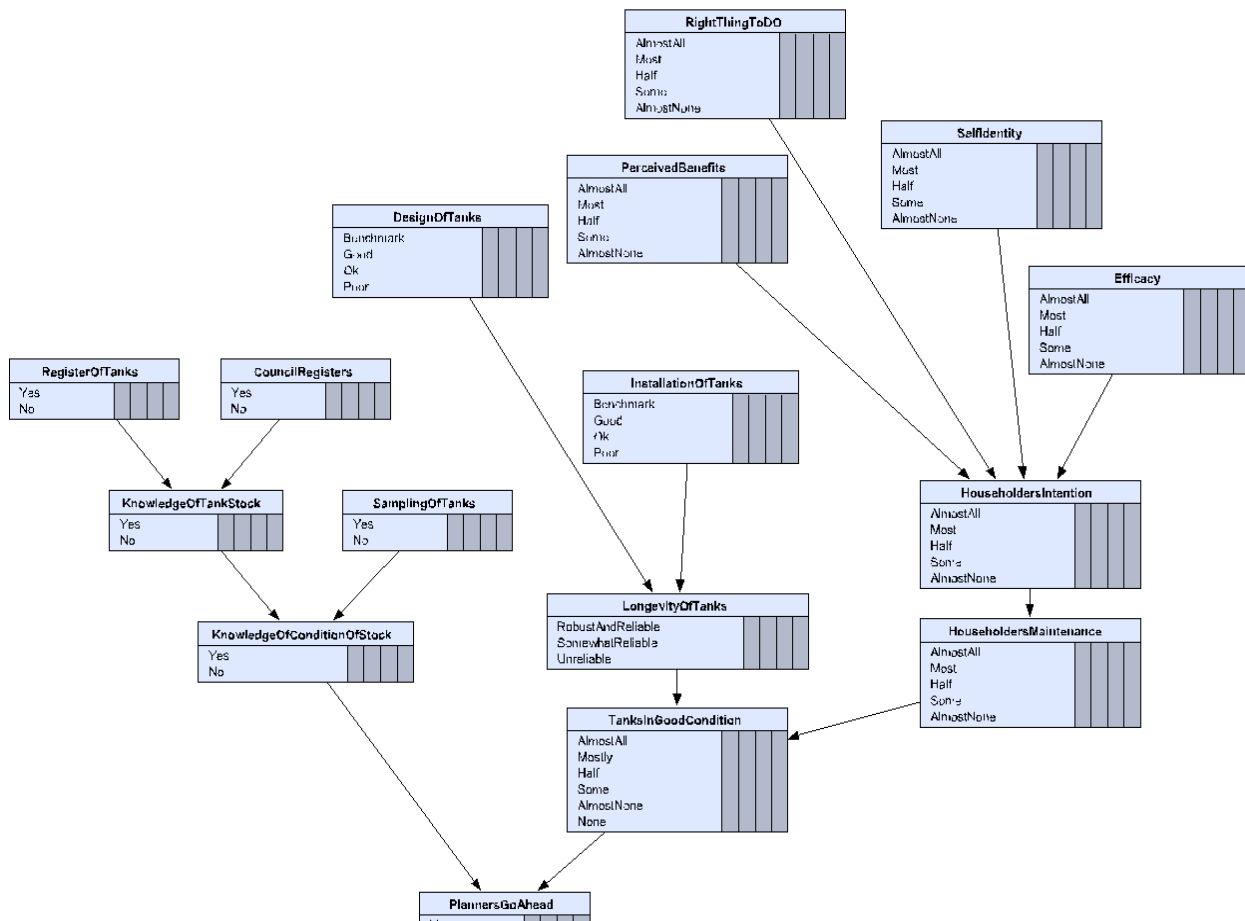


Figure 1. Bayesian Network model to assess rainwater tank management strategies.

This model has been set up and numbers in the model elicited based on expert judgments (there is still some scope for improvement based on additional information). Whilst the details of this model will be published in a forthcoming publication, we can already conclude that to be able to achieve a good likelihood of both low health and supply risk, as well as certainty of this low risk, it is necessary to implement all of the above strategies: improve installation practices, improve design practices, improve maintenance practices, and undertake random sampling of tanks. It is also clear that whilst improvements in design and installation practices should be possible to achieve relatively easily (albeit at a cost); improvements in maintenance practices seems to be difficult to achieve due to the complex nature of the influencing socio-psychological factors. This also very much relates to the fact that “an intention to maintain a tank” is not necessarily translated to actual maintenance behaviours.

The emerging recommendation from this study is that immediate attention should go towards improving design and installation practices, as well as compiling a database of rainwater tanks across SEQ. The second step would be to develop strategies for promoting tank maintenance behaviours, and the implementation of such strategies would most likely need to incorporate some sort of monitoring activities in order to find out what works and what does not work. Rainwater tank management is not rocket science, but seems to have been neglected so far. There is a need to more rigorously assess the extent of the problem and monitor trends over time as the basis for effective management of rainwater tanks to ensure positive health and supply outcomes.

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Community Acceptance of Policy Options for Managing the Maintenance of Rain Water Tanks: a Qualitative Study

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Summary

This paper reports on focus group research that explored community attitudes towards policy options for encouraging ongoing maintenance of rainwater tanks. Eleven policy intervention ideas were presented to community members, ranging from self-management to regulatory approaches. Self-management approaches and changes to regulations that support improved tank design and installation were viewed most favourably. More coercive approaches that regulated ongoing maintenance, through use of tank registers and inspections, were viewed least favourably. Analyses identified seven drivers of community reactions to the various policy options; some related to the benefit of owning a tank, and some directly to the issues related to undertaking tank maintenance.

Keywords

Policy acceptance, rainwater tanks, rainwater tank maintenance.

Introduction

Installation of rain water tank systems in South East Queensland (SEQ) has become widespread in response to concern for water security in the region. Over the last five years, it is estimated that more than 300,000 rain water tanks have been installed in homes around the Brisbane, Gold Coast and Sunshine Coast areas (Australian Bureau of Statistics, 2010). Some tank installation has been supported by state government rebate schemes, which resulted in many home owners retrofitting tanks to their existing properties. Since 2007, installation of a tank has become a mandatory requirement of building a new home in the region (Queensland Government, 2008).

Although the uptake of rainwater tanks has water security benefits, there are questions regarding the ongoing maintenance of such systems. To date, tank owners generally have been left to maintain their own tanks. The extent to which this maintenance is occurring is unclear, although there is some evidence to suggest that home owner maintenance of tank systems is minimal (Mankad, Chong, Umapathi, and Sharma, 2011; Moglia, Tjandraatmadja, and Sharma, 2011). This trend could result in problems in the long term: an unmaintained tank is assumed to contribute to eventual tank failure; possible tank abandonment; risk of mosquito-related problems if tanks are left in a state of disrepair; and reduced likelihood of not achieving the envisaged mains water savings. Thus, there are implications both for public health and strategic water planning if rain water tank systems function sub-optimally, as well as the potential degradation of public infrastructure.

In light of these considerations, this research investigated community acceptance of various policy and intervention options that could be used to encourage tank maintenance at the household level. The research represents part of a three-year integrated research program, which has addressed the social science aspects of decentralised water systems. This present study was qualitative and used focus groups as the method for collecting data. Prior to the focus group research, a workshop involving government and industry stakeholders identified a range of possible policy intervention options. In the current study, community attitudes towards these options were sought, in particular with regard to perceived acceptability and effectiveness. The specific research questions addressed were: 1) what are the views of the community towards the various policy options?; and 2) what are community members' preferred options for ensuring the ongoing performance of household rainwater tanks?

Methodology

Six focus groups, with a total of 42 participants, were conducted. Two groups each were conducted with three different research populations: people who had acquired a tank through a government assisted rebate scheme (retrofitted tank owners); people who had a tank because it was mandated as part of building a new house (mandated tank owners); and people who didn't own a tank. Focus groups are well suited to exploring a specific topic among different population groups (Hair, Lukas, Miller, Bush, and Ortinau, 2008). This is achieved through the interactive nature of focus groups where spontaneous discussions generate diversity of views and reveal differences, not only within groups, but also between groups (Flick, 2010).

A semi-structured discussion protocol was developed based around the intervention strategies identified in a prior stakeholder workshop. The protocol was designed to explore each strategy and determine the perceived impact the strategy would have on effecting tank maintenance behaviour. Open-ended questions were used to explore the benefits and barriers associated with each strategy idea. We also sought to understand participants' perceptions of acceptability, and developed questions about what people liked and disliked about each strategy. We used a

perceptual mapping technique to summarise the groups' views on each strategy. By consensus, groups positioned each idea onto a shared map that defined differing levels of policy effectiveness and policy acceptability.

Eleven policy ideas were put to each group, and each idea was discussed in terms of its potential effectiveness in promoting maintenance and its likely acceptability by the community. Table 1 displays the eleven ideas. The perceptual mapping activity followed, with each strategy idea positioned on the map and shifted in light of ongoing discussions of other strategies. The final part of the protocol was a voting process, in which participants indicated their preferences for the various strategy ideas.

Table 1. Policy ideas presented to the focus groups.

1.	Leave it to householders to manage for themselves; the status quo
2.	Provide support to householders: fact sheets; helpline; directory of services, web-based resources
3.	Increase awareness: promote benefits; highlight consequences poor maintenance; reminders and prompts
4.	Home service, like Climate Smart: tank owners pay to have someone come and inspect the tank
5A.	Create a register of tanks – rely on tank owners co-operation
5B.	Create a register of tanks – make it compulsory to have tanks registered
6A.	Inspect tanks – make it compulsory to have tanks inspected for correct installation when first installed
6B.	Inspect tanks – make it compulsory to have tanks inspected every few years
6C.	Inspect tanks – make it compulsory to have tanks inspected when the house is sold
7.	Tank design – make it compulsory to improve the design of the tank so that less things will need maintaining
8.	Maintenance information – make it compulsory to be given information about tank maintenance when installed

Results

Participants seemed unaware of the need for tank maintenance. The initial information that was provided to participants, which outlined a typical tank maintenance schedule, appeared new and surprising to most participants. This finding was consistent with previous research which has suggested that tanks owners are not aware of the need for maintenance, and that maintenance typically is not occurring.

In general, there was community support for self-management approaches, improvements to standards that govern design and installation, and home based services modelled on the Climate Smart program. Findings suggested that if tank owners were provided with options for getting their tank maintained, ranging from 'do it yourself' information through to paid maintenance services, that this would enable many tank owners to maintain their tank. This approach, combined with an effective communication program aimed at creating awareness and highlighting benefits, would increase motivation for keeping a tank maintained. In contrast, there was minimal support for inspections aimed at ensuring ongoing maintenance, or for creating tank registers. Tank inspections at the point of sale could be viewed more favourably if integrated into the current pest and building inspections.

The most negatively viewed strategies were those involving registration of tanks. Not only was a register viewed with suspicion (as a potential precursor to inspections, fees and fines), but also as an ineffective option for achieving tank maintenance. Participants suggested this type of approach could lead to people abandoning their tanks. A register, and associated fees and inspections, would add to the cost burden of tank maintenance and erode any savings benefit of having a tank. Also, it would mitigate the notion of autonomy, an important motivation for having a tank. However, there was evidence that a potential public health risk, for example mosquito-borne illness, could influence these views, and could result in greater acceptance of regulatory approaches for ongoing maintenance. Figure 1 displays these findings.

Some differences in attitudes towards the various policy interventions existed between the three research populations, although our limited samples don't allow us to quantify these differences. The main apparent differences were between non-tank owners and tank owners. Non-tank owners appeared to be the least accepting of the status quo, and were also more accepting of suggestions for regular inspections than were tank owners. This difference could reflect the fact that, as non-tank owners, they are not directly affected by these types of decisions.

In addition, mandated tank owners were the most negative group regarding inspections at time of sale of the property, describing this sort of requirement as 'another thing to do'. This reaction could be due to their more recent experiences with buying and selling property and the concomitant inspections associated with a sale, given that this group had presumably purchased a new home within the last four years.

Different attitudes between mandated and retrofitted tank owners might be expected, because of the typically greater direct cost of the tank for mandated tank owners. However, no such difference was evident. It could be that the cost of the tank to the mandated owner is subsumed within the overall cost of the new home. As a consequence, mandated tank owners did not seem more concerned about maintaining the value of their tank than retrofitted owners. Further, mandated tank owners did not report any added benefit of saving more water than a retrofitted tank owner, even though they typically have their tanks connected to more water using devices. This lack of regard for increased water saving, a potential saving of a third of overall water use (Chong *et al.*, 2011), could be because the mandated owner was unaware of how much water they were ‘saving’, given that they had no way to assess the contribution of the tank to their household water consumption.

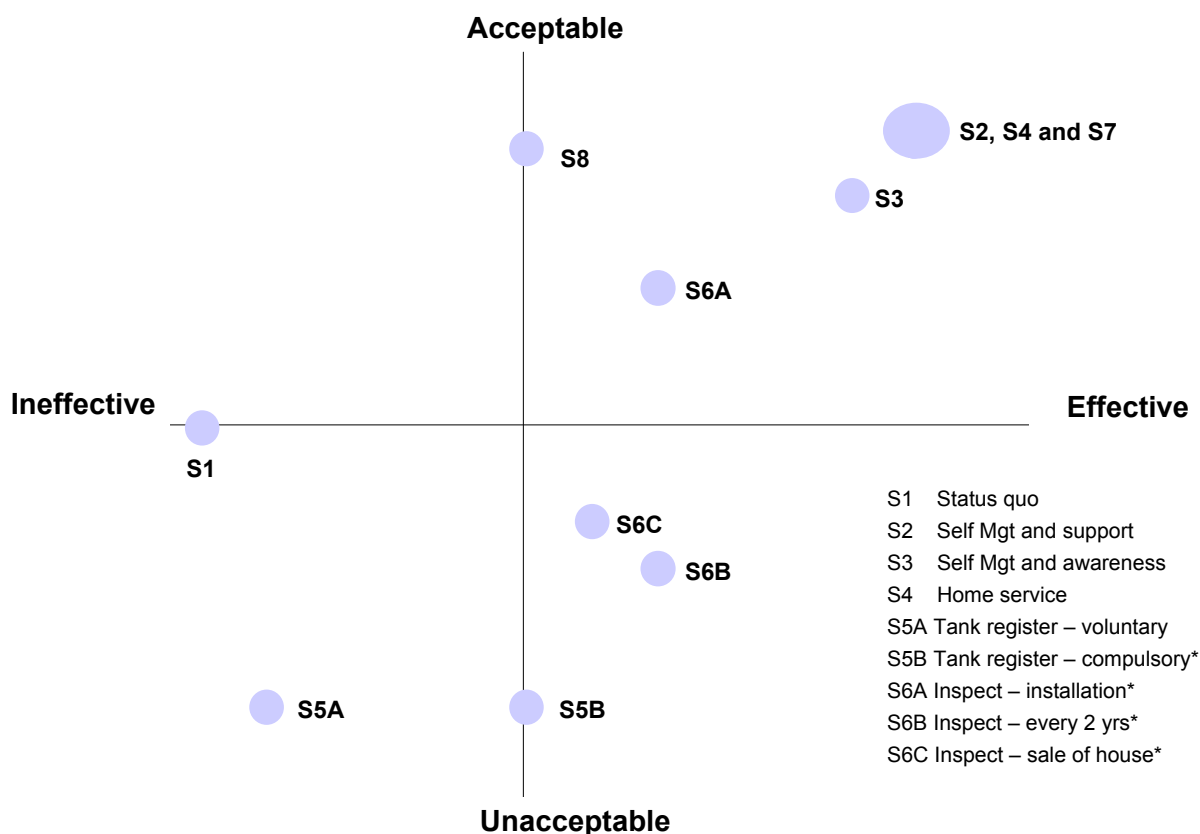


Figure 1. Perceptual map of strategy ideas: All focus groups.

Factors Relevant to Tank Maintenance Behaviour

Analysis revealed seven potential drivers of prevention behaviours and acceptance of policy interventions for fostering such behaviour: each of these is discussed in turn below.

Relative cost-benefit. For many people, the potential advantage of saving money on water costs was an important motivating factor for maintaining a tank, and any erosion of this benefit would act to demotivate the tank owner. Cost savings appeared to play a significant role in attitude formation and for many these were fundamental to the original purchase decision. However, tank owners reported that the costs actually saved from having a tank were probably marginal. Even though research suggests tanks contribute to water savings of up to one third for an average size dwelling (Chong *et al.*, 2011), this does not seem to translate to a proportionate savings in costs. In part, this is due to the format of the utility bill, in which only a small portion reflects the variable cost related to water usage, with a larger portion reflecting fixed costs via infrastructure charges. Thus, participants assessed any potential strategies that incurred significant costs as unattractive, because of the likely erosion of any cost advantage. Even interventions such as registers and inspections implied that there would be some sort of ensuing cost, for example, a licence fee to register or an inspection fee with any inspection process, and that this cost would be additional to any actual maintenance costs. This presumed cost would reduce the financial benefit as a motivation, and likely result in people reconsidering using their tank, especially if it began to malfunction.

Agency. The ability to act autonomously, and be an agent of one’s own decisions, appeared to be a major influence on attitudes toward tank maintenance behaviour and policy interventions. For many, the idea of having freedom to ‘do what I want with my water’ was a significant motivation for getting a tank in the first place. This notion seemed to extend to attitudes related to managing the tank. The idea that an intervention might try to ‘enforce’ tank maintenance was viewed very negatively. The importance of agency is highlighted by public concerns about ‘the nanny state’, where citizens react negatively to overly paternalistic policy approaches by government (Prendergrast

et al., 2008). Research has suggested that public perceptions of unfairness, infringement of freedom, and perceptions of coercion have been associated with lower levels of policy acceptance (Cherry, Kallbekken, and Kroll, 2011; Ericksson, Garvill, and Nordlund, 2006). In this present study, some participants indicated they would engage in counter-productive behaviour, such as abandoning their tank, if coercive regulatory approaches were introduced. Rather, participants wanted choices and options for managing tank maintenance. For some the preferred option was self-management, for others it was access to a home-based service, or information and web-based resources. Participants also supported interventions that did not directly affect them. For example, they were very supportive of any improvements to tank design and installation, even if these changes resulted in additional cost to the initial outlay of the tank. It was also seen as important that a tank owner should be aware of tank maintenance issues in advance. Full disclosure of costs involved with maintaining a tank would enable more informed decisions, and would create awareness of responsibilities involved in owning a tank.

Environmental awareness. An awareness that a tank could contribute to safeguarding against water shortages was a perceived benefit to all groups of participants. Participants were positive about the idea of having a tank as a way of securing additional water during periods of a drought. Although the region was currently experiencing an abundance of water, participants felt that droughts were cyclical events and likely to return. There was a suggestion that if this were the case, their motivation for having and maintaining a tank would be increased. This view is in line with previous research which has demonstrated that a perceived threat, such as a drought, is a motivator for adopting a tank or other decentralised water systems (Mankad, Tapsuwan, Greenhill, Tucker and Malkin, 2010).

Efficacy. The feeling of being capable of maintaining a tank was a major issue. Capability in terms of knowledge, skills and abilities were all important factors. Participants suggested efficacy could be improved through increased awareness, knowledge of the activities involved, provision of 'how to do it' facts, and information about relevant service providers. Thus, participants viewed any interventions that supported improvement in efficacy as both acceptable and effective. Conversely, interventions that didn't improve efficacy were judged negatively. For example, participants did not see how tank registers or mandatory inspections would result in better tank maintenance. Efficacy is widely recognised for its importance in achieving behaviour change. A person's belief that he or she has the necessary skills and abilities to undertake the target behaviour, and the control over any environmental constraints is a powerful variable in fostering behavioural change (Armitage and Conner, 2001; Azjen, 1991).

Participants also indicated that being provided with information alone would not be very effective. Rather, information needed to be used in tandem with other motivation strategies such as creating awareness and highlighting the benefits of keeping a tank well maintained. This view was supported by those participants who reported being provided with a maintenance information brochure at the time of installation, but never having bothered to look at it again, unaware of the value of keeping their tank maintained.

A final factor impacting efficacy was a busy lifestyle. Many participants felt that their lives were full, and their resources for attending to additional tasks were limited. The need to expend time, attention and care on activities for maintaining the tank would be viewed as difficult, especially if the burden increased to include inspections and registration. If however, some of these tasks could be included in other routine 'house maintenance' tasks, it would not only improve the likelihood of getting it done, but also be more acceptable. For example, if tank maintenance could be included in pest control, or swimming pool inspections, this would ultimately improve efficacy for tank maintenance behaviour.

Moral norms. For some participants there was a sense of feeling obliged to look after their tank; that it was 'the right thing to do'. This sense of moral obligation seemed to be based on two main reasons; not wanting to waste the financial resources invested in the tank (both at a personal level and at a public investment level); and not wanting to create a potential public health risk. If equipped with suitable capability, these individuals felt they would be motivated to keep their tank maintained. A sense of moral obligation, or personal norm, has been described in the literature as a powerful motivator of pro-social behaviour, particularly in the environmental field (Harland, Staats and Wilke, 1999; Spinks, Fielding, Russell, Mankad and Price, 2011).

Self identity. Some people saw themselves as 'someone who keeps things maintained and in good repair'. These individuals viewed maintaining a tank as no different from keeping the air conditioner regularly serviced, or the carpets annually cleaned, and indicated if they knew what and when things were to be done they would do so. Furthermore, some saw themselves as a 'do it yourself' type of person and suggested all that they needed to know was what was required and that they would keep their tank maintained. This suggests if efficacy levels were improved this type of person would be motivated to keep the tank maintained. Self-identity has been an important predictor of behaviour across many domains including the environmental field (Whitmarsh and O'Neill, 2010).

Low perceived need for intervention. Participants indicated the level and type of intervention for encouraging tank maintenance needed to reflect the level of need for the intervention. Because many participants were 'only' using their tank water for external garden use, they found it hard to understand that this type of use warranted any interventions that were viewed as too 'heavy handed'. On the other hand, if the tank water was used for drinking then

they could understand the need for more regulatory interventions. Along this same theme was the recognition by participants that if a situation arose related to a public health risk, such as increased prevalence of mosquito-borne illnesses, then they would be more amenable to regulations, including registers and inspections to manage this type of situation. Therefore, participants seemed unconvinced as to the need for an intervention response beyond self-management approaches, and viewed any alternate intervention with suspicion. In particular, tank registers were viewed very negatively, not only because tank registers were not perceived to assist the tank owner in maintaining a tank, but also because participants felt that government should already have this type of information. Participants felt that data related to rebate schemes and new home builds would already exist, and therefore, they did not understand the need for having a register.

Discussion, Implications and Future Research

The community's acceptance of various policy interventions seemed to relate to attitudes and views around seven factors, some which were fundamental to the decision making of getting a tank in the first place, and some more directly related to performing tank maintenance behaviour. A positive view towards savings on water costs, the appeal of being autonomous and able to use water freely, and an awareness of the environment were original drivers of tank purchase, particularly for retrofitted tank owners. These factors were considered benefits of owning a tank, and confirmed as such by mandated tank owners, although to a lesser degree. A sense of self-efficacy in performing the tank maintenance behaviours, a concern for doing the right thing, and self-perceptions of being someone who looks after things, were also important factors underpinning tank maintenance behaviour. These concepts are depicted in Figure 2.

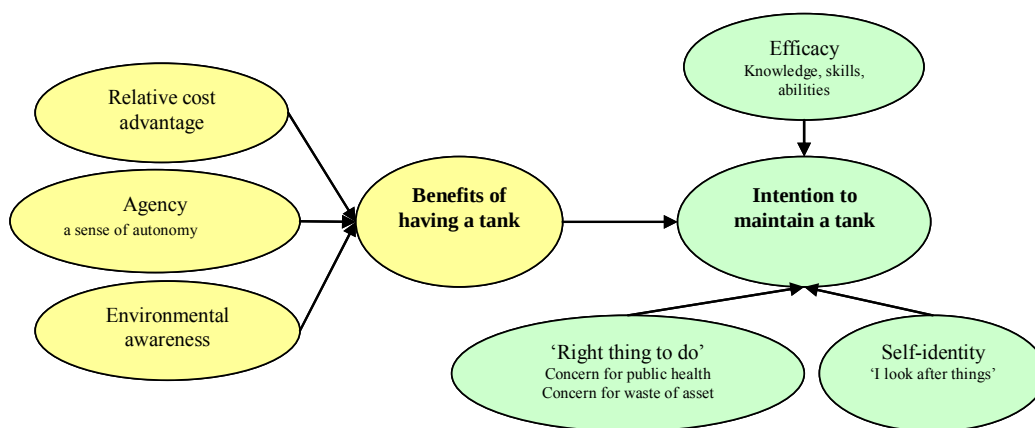


Figure 2. Model depicting psychosocial factors important to tank maintenance behaviour.

As a consequence of these underlying motivations, any policy interventions that directly affected these factors were reflected in corresponding positive or negative attitudes towards the interventions. For example, interventions that promoted efficacy were deemed to be acceptable and effective by the participants. Any interventions that potentially eroded any relative cost advantage were assessed negatively. In particular, policy ideas that interfered with a sense of agency were viewed most negatively. Most participants did not see tank maintenance as requiring a paternalistic approach from government. Rather, more regulatory types of approaches were seen as being ‘nannied’ by the state, and seen as unnecessary for an issue such as keeping a rain water tank well maintained. As a consequence, these types of approaches were viewed with suspicion and scorn, with suggestion that any such attempts to regulate tank maintenance could result in tank owners choosing to abandon their tanks. The caveat to this view was in relation to a public health risk. If a situation arose in which poor maintenance of rainwater tanks was assessed to be a cause of mosquito-borne illness, then attitudes would likely soften towards a more regulatory approach to tank maintenance.

Instigating change using self-management approaches incorporates some of the features of a social marketing program. Communicating the benefits of undertaking tank maintenance behaviour, highlighting the costs of not keeping a tank maintained, reducing perceived barriers to adopting the behaviour, and increasing a person's self-efficacy for performing the behaviours are all principles of a social marketing approach (McKenzie-Mohr and Smith, 1999). However, also fundamental to this approach is the concept of segmentation. Recognising that tank owners are not homogenous, but rather a disparate group with differing needs and levels of motivation for tank maintenance, has bearing on consequent interventions. Developing interventions to target each type of tank owner has the potential to improve outcomes (Andreasen, 2006). The present research suggests that differences exist among retrofitted tank owners, mandated tank owners and prospective tank owners, although the extent of these differences is unclear. In addition, segmentation based around usage appears reasonable. Research indicates there are primarily two patterns of use for tank water: garden and outdoor use; or outdoor use combined with indoor toilet and laundry functions. Findings suggest that usage patterns could also contribute to differences in views and attitudes towards tank maintenance, and underline the need to introduce non-regulatory approaches.

Another consideration is the choice of the target behaviour on which to focus the intervention. Choosing which behaviour to change is an important aspect of planning any intervention (Steg and Vlek, 2009), and this seems relevant to tank maintenance. Tank maintenance can be viewed as an umbrella term, incorporating two different types of activities; regular six-monthly checking of gutters and screens; and periodic de-sludging of the tank. These different activities require different levels of efficacy from the individual. It is possible that there are different attitudes and views associated with each of these behaviours that comprise tank maintenance, and that the final intervention needs to address these two types of behaviour differently.

A final point to highlight is the influence of context in affecting community acceptability and determining tank maintenance behaviour. This research was undertaken within the context of an abundant water supply situation. In this circumstance, the need for decentralised systems to provide water security is reduced, compared to when these measures were first encouraged and adopted. The research findings suggest that views and attitudes might be different if the region was again experiencing drought conditions. Similarly, the presence of a heightened public health risk associated with mosquito-borne illnesses would also likely influence motivation and policy acceptance. The literature describes the powerful effect of context to potentially alter attitudes and other psychological variables associated with pro-environmental behaviour (Stern, 2000; Steg and Vlek, 2009). The influence of a drought context and public health risk context on motivation for tank maintenance behaviour, and acceptance of policies that promote such behaviour, could also be investigated in further research.

Qualitative research cannot make claims about the wider population's attitudes. The focus groups aimed for depth of findings rather than breadth, and covered a limited number of different strategy ideas. The opportunity for the focus group discussions to cover novel ideas was reduced, because discussions revolved around the interventions that were predetermined from the prior stakeholder workshop. To more fully understand the extent of this study's findings, a choice modelling study is planned. In this forthcoming research the relative values of various policy features will be assessed, as well as individual psychosocial factors and contextual influences for explaining policy acceptance and perceived effectiveness.

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Identifying the Drivers of Water Consumption: a Summary of Results from the South East Queensland Residential End Use Study

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Summary

The aim of the South East Queensland Residential End Use Study (SEQREUS) was to address the research gap on water end usage by way of generating a high resolution data registry of water end uses and using such a database to explore the relationships and influences of residential water consumption from a bottom up approach. Such data can be used to optimise future demand management strategies. Mains water end uses in a sample of 252 residential dwellings located within South East Queensland (SEQ) were measured using a combination of high resolution smart meters and data loggers and a parallel social survey design comprising a questionnaire, a stock (appliance) audit and self-reported water diary for each household. An array of detailed analyses were conducted from the subsequent data registry based on three separate two-week monitoring periods (2 x winter and 1 x summer). Impacts on water consumption from water-efficient technology (eg, star rated washing machines, flow regulated taps and showers), socio-demographics (household composition, income and education and perceptions and attitudes towards of water conservation on household, per capita, diurnal and peak demand water consumption are presented, including the variation in water end uses on a daily and seasonal basis. We conclude with some policy considerations that evolved from our data analysis and that may assist in optimising future demand management strategies. For example, it is recommended to target households with large families with young children or teenagers, as these homes are often associated with high shower and tap usage.

Keywords

Water end uses, micro-components, water consumption behaviors, water-efficient technology, demand management, rebound effects, water restrictions, intervention strategies.

Background

In an attempt to improve water security, many government authorities in Australia have imposed water restrictions and water saving measures to manage demand and ensure the mindful use of water across the residential, commercial and industrial sectors. The combination of enforced water restrictions and State and local government rebate programmes for water efficient fixtures and rainwater tanks have resulted in a large reduction in household water use in South East Queensland (SEQ) (Walton and Hume, 2011). Residential water end use data can facilitate the identification of correlations between water behaviours and key demographical subsets within a population (eg, income, age, gender and family composition). Currently, there is a scarcity of measured water end use data for SEQ and Australia in general. The overarching aim of the SEQ Residential End Use Study (SEQREUS) was to address the research gap on water end usage by way of generating a high resolution data registry of water end uses and using such a database to explore the relationships and influences of residential water consumption from a bottom up approach. This paper provides a summary of the key findings arising from the SEQREUS which focussed on four local council authorities (Brisbane City, Gold Coast City, Ipswich City and Sunshine Coast Regional Council) located in the south-eastern corner of Queensland, Australia.

Methodology

The total number of homes monitored was 252 (winter 2010), 219 (summer 2010-11) and 110 (winter 2011). The breakdown of homes per region and some general characteristics of the participating households within each region are shown in Table 2. The SEQREUS used a mixed method, advanced water end use measurement approach to capture and analyse water use data. Upon completion of household recruitment, standard council residential water meters were replaced with modified Actaris CTS-5 water meters. These 'smart' meters measure flow to a resolution of 72 pulses/L or a pulse every 0.014 L. The meters were connected to Aegis Data Cell series R-CZ21002 data loggers.

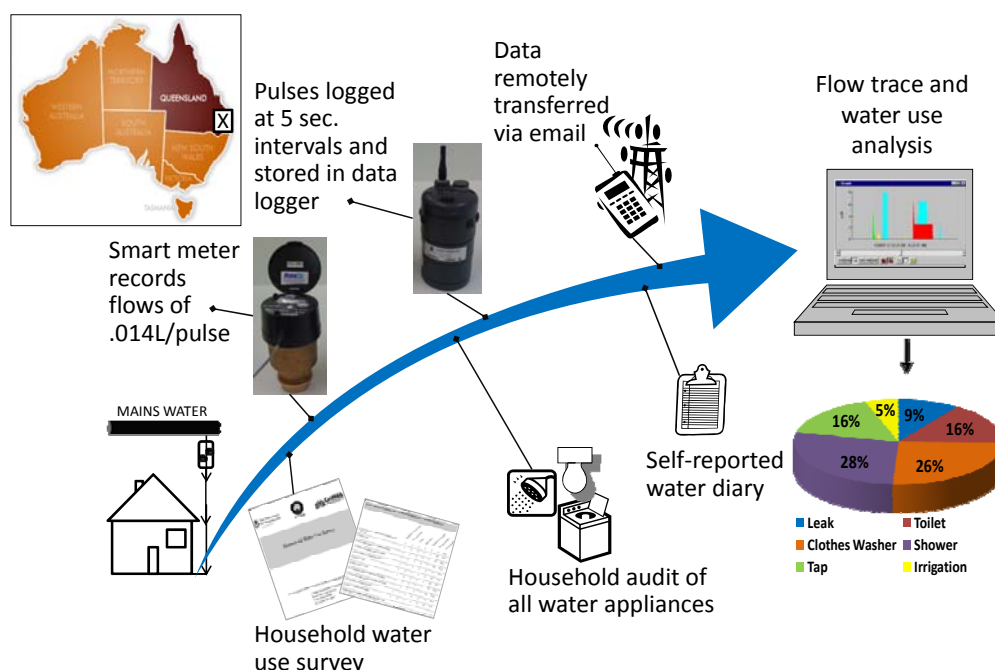


Figure 1. Mixed method approach used in the SEQREUS.

The loggers were programmed to record pulse counts at five second intervals. Data was wirelessly transferred to a central computer and stored in a database for subsequent analysis (Figure 1). A representative sample of received data was extracted from the database and disaggregated into all end use events associated with the sampled residential households using the Trace Wizard® software (Aquacraft, 2010). Concomitantly with meter and logger installation, a water fixture/appliance stock survey and 7-day self-reported water use diary was conducted at each participating home in order to investigate how householders interact with such stock. Additionally, each homeowner completed the Household Water Use Survey. The aim of the survey was to capture attitudes and behaviours toward household water conservation. A detailed discussion on the research methods is provided in Beal *et al.*, (2011) and Beal and Stewart (2012).

Table 1. Selected characteristics of households in the SEQREUS sample.

Sample Characteristics ¹	Gold Coast	Brisbane	Ipswich	Sunshine Coast	SEQ Combined
Household occupancy	2.6	2.6	2.7	2.5	2.6
No of people	230	164	96	171	661
No. homes	65	61	37	67	230
% Households with ≤ 2 people	58	41	51	69	55
% Households pensioners/retired	36	16	32	45	32
% Households with children (aged ≤ 17)	34	30	21	25	28
Average age of children (years)	8.8	2.7	4.4	10	6.5
Average household income (\$AUD) ²	73,290	81,630	87,900	60,070	75,722

Notes: ¹ data presented are averages; ²Estimated from taking the average of the household income category that each respondents selected (Gregory and Di Leo, 2003), where categories were: 1 = <\$30,000, 2 = \$30,000 – \$59,000, 3 = \$60,000 – \$89,999, 4 = \$90,000 – \$119,999, 5 = \$120,000 – \$149,999, 6 ≥ \$150,000.

Results and Discussion

Water End Use Breakdowns

The water end use breakdown on a per capita basis indicated that, on average, shower, tap and clothes washer comprised the bulk of the water consumption. This trend was consistent both temporally and spatially throughout the SEQREUS with almost 70% (approximately 100 L/p/d) and 74% (approximately 106 L/p/d) of total consumption was attributed to these three activities in winter 2010 and winter 2011, respectively. Of note, irrigation made up less than 5% of average total consumption. Typically, the homes that used the most water had a disproportionately high contribution from irrigation. Dishwasher and bathtub use was also over-represented by a small number of homes,

with the latter end use being generally associated with young families. Although the sample size for the winter 2011 data was less than half that of the previous winter, the average total water consumption was very similar at 145.1 L/p/d, and compared well with the QWC reported per capita water use of 148 L/p/d for the same period. The larger household water consumption in winter 2011 is likely to be associated with the different household compositions for this smaller sample ie, the higher percentage of families with children in the winter 2011 dataset would effectively increase the household consumption, but reduce the per capita consumption. This is explored in more detail in later sections of the paper.

Table 2. Water end use breakdowns for SEQREUS sample.

End Use	Winter ¹ 2010	Summer ² 2010-11	Winter ³ 2011	Winter 2010	Summer 2010-11	Winter 2011	Winter 2010	Summer 2010-11	Winter 2011
	L/p/d			L/hh/d			% of Total		
Leak	9	4	3.1	17.7	8.6	9.08	6.2	3.2	2.1
Toilet	23.7	23	24.4	58.6	56.2	70.1	16.3	18.3	16.9
Clothes washer	31	26.5	31.8	82.7	65.1	89.2	21.4	21.1	21.9
Shower	42.8	36.2	49.9	115.0	94.3	149.1	29.4	28.8	34.4
Dishwasher	2.5	1.9	2.2	6.5	5.0	6.1	1.7	1.5	1.5
Tap	27.5	27.4	25.1	68.6	67.1	70.1	19	21.9	17.3
Bathtub	1.8	1.5	1.9	5.8	4.6	4.9	1.2	1.4	1.3
Irrigation	7	4.8	6.7	15.7	10.3	16.9	4.8	3.8	4.6
Total	145.3	125.3	145.1	370.7	311.3	415.6	100	100	100

Notes: ¹ n= 252 households, ² n= 219 households, ³ n= 110 households.

There are several explanations for the 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.*, 2011a). Secondly, 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 SEQREUS 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. Due to these factors, greater care needs to be taken when applying the summer end use results towards water practice and policy actions.

Given the largest sample size and average rainfall and temperatures during the winter 2010 period of analysis, this dataset was considered the best representation of SEQ households with a strong mix of family types, income categories and household occupancies. Additionally, results suggest that the data obtained from this study compares well with other estimations of household consumption (ie, weekly reports from QWC). Consequently, detailed analysis presented is based on this winter 2010 dataset.

Regional Breakdowns

Looking at the regional breakdowns (Figure 2), it is clear that Gold Coast and Brisbane were mainly responsible for the upward shift in shower end use consumption. Collectively, for the winter 2011 analysis, households in these two regions had the highest percentage of families with children ≤ 17 years old. This may be reflected in the higher shower use and slightly higher bath end use also. Average end use breakdowns for Ipswich are very stable across the seasons, despite the reduced sample sizes, particularly for the final winter 2011 read (n=12). As there is very little irrigation for many of the Ipswich households, essentially it is the indoor end uses that are being compared in Figure 2. The Ipswich sample provides a good example of a low level of fluctuation between indoor end uses which have been reported elsewhere (eg, Willis *et al.*, 2011a; Jacobs and Haarhoff, 2004). The homogeneity in indoor end uses further emphasises the strong influence that irrigation can have on total household demand and thus the value of supplementing irrigation water sources such as rainwater tanks and greywater systems.

The Sunshine Coast residents consumed the highest volumes of water for all regions, with the biggest indoor water usages attributed to showers, clothes washers and toilets. The Sunshine Coast sample had the oldest average age for children (10 years) and highest percentage of pensioners and retired residents (Table 1). There was more likely to be greater occupancy during the day in this region than compared to regions that had a lower daytime occupancy rate (eg, Brisbane demographic are more likely to be working and attending school). During the analysis, it was regularly observed that the homes that were occupied by older residents tended to use more water for showers and toilets. This

is confirmed by the high shower usage and elevated toilet usage observed in Figure 2. Water loss attributed to leaks was the highest of all the regions at 14.1 L/p/d in winter 2010, although after alerting these households, leakage was reduced substantially.

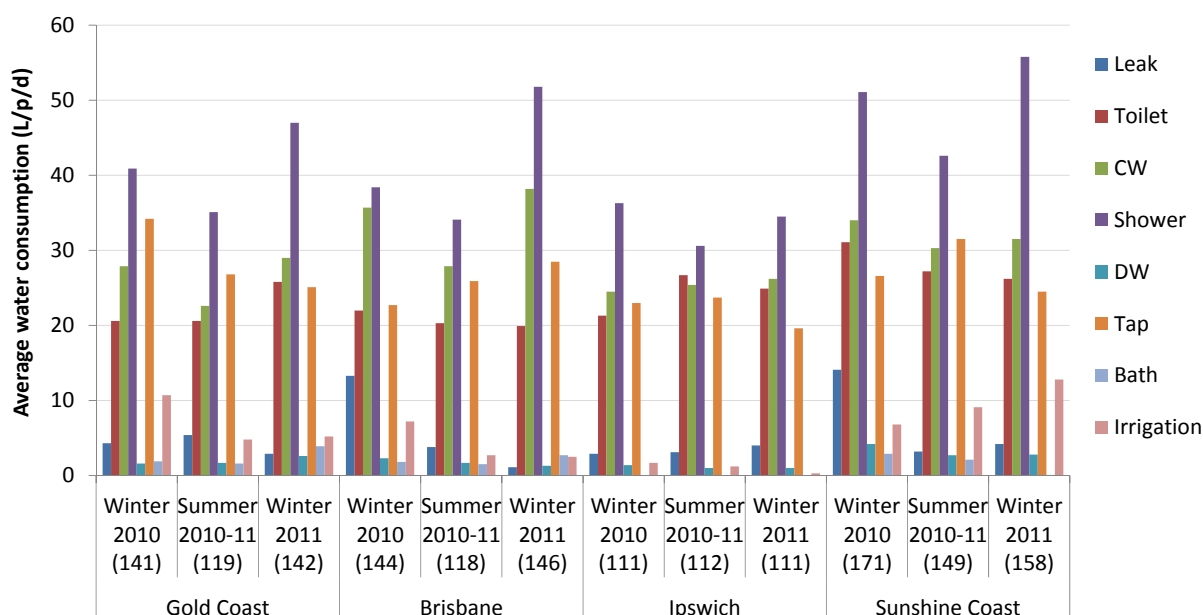


Figure 2. Regional water end use breakdown for winter and summer analyses. (Note: Total daily per capita use in parenthesis).

Average Daily Diurnal Patterns

The winter average daily diurnal pattern for 2010 is shown in Figure 3. For each of the read periods, there were twin consumption peaks in the morning and afternoon water use events. Shower, clothes washer and taps contributed the bulk of the water use activity at these peak times. The morning peaks, were typically higher than evening peaks for both the winter and summer reads, although the summer peak use was more prolonged or ‘flattened’, particularly in the afternoon. Irrigation use appeared to occur throughout the day across both seasons, demonstrating a conflict with current water restrictions and awareness messages that recommend outdoor watering in early morning and late afternoon. As a result of the leak intervention programme after winter 2010, leaks have reduced significantly in all regions and were consistently low throughout the day, showing little diurnal variation.

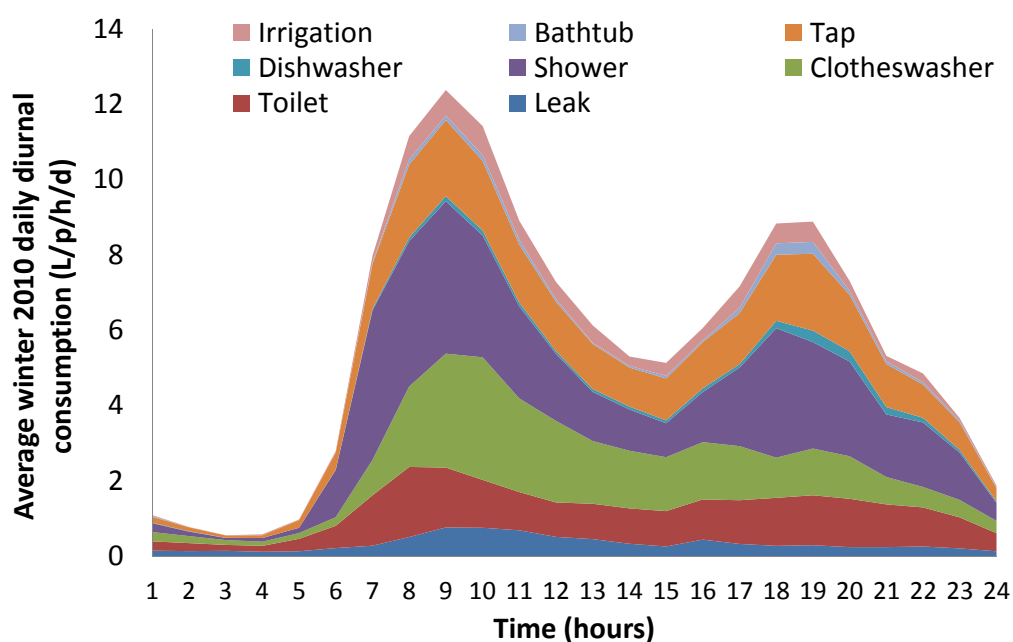


Figure 3. Average daily diurnal pattern analysis.

Impacts of Household Stock Efficiency on Water Consumption

The impact of water efficient stock on water consumption was examined on the winter 2010 dataset. For clothes washing machines there was a clear trend for higher star rating and front loading machines to use less water and in terms of star ratings; ≥ 4 star machines used significantly less ($p < 0.05$) water than ≤ 2 star machines (Figure 4). This equated to a potential savings of 8.8 kL/hh (or 29%) per year. The penetration of front loaders is likely to have increased sharply in the last three to five years due to the rebates offered in Queensland to install water efficient (typically front loading) machines. There was a significant reduction ($p < 0.05$) in shower water demand from high (AAA star) efficiency heads compared to low (A star) or poor (standard/old) efficiency clusters (Figure 5). Replacing the old style showerhead with any star rated shower head would significantly ($p < 0.05$) reduce water consumption by a minimum of 28 kL (or 75%) per year.

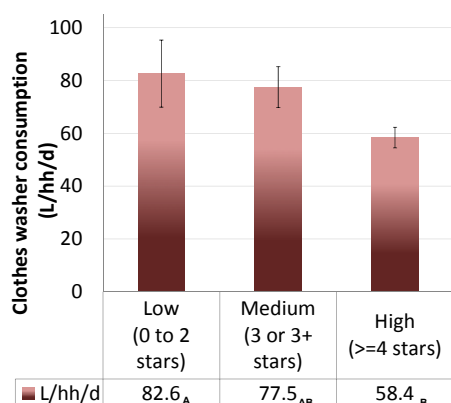


Figure 4. Daily clothes washer efficiency comparisons.

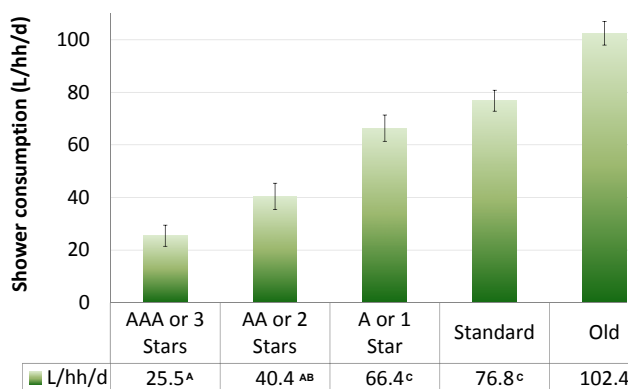


Figure 5. Daily shower efficient cluster comparisons.

There were significant differences ($p < 0.05$) between all three tap efficiency clusters, and replacing an old style tap with a ≥ 3 star tap fitting can save 12.9 kL/hh or 65% annually. Efficient dishwashers (eg, 3.5+ star rating) used significantly less ($p < 0.05$) water at a mean of 4.4 L/hh/d, compared to the average 9.2 L/hh/d from the inefficient dishwasher cluster. Homes with a rainwater tank (RWT) that was used exclusively for external events such as irrigation were found to have a statistically lower total household consumption than those homes without a RWT (Figure 6). Note that there were no homes that had internally plumbed RWT included in the study.

The study demonstrated that highly efficient water appliances and fixtures not only contribute to reduced use of potable water supplies but also lower the average day peak hour demand from which water supply infrastructure is designed. Data in Figure 7 compares the average daily diurnal patterns for the 50 least efficient and 50 most efficient homes in the sample. Water-efficient homes were found to have a reduced average peak hourly consumption of between 2.47 L/p/h/d (18.6%) and 3.52 L/p/h/d (19.3%). Both of these water demand reductions were statistically significant at $p < 0.01$.

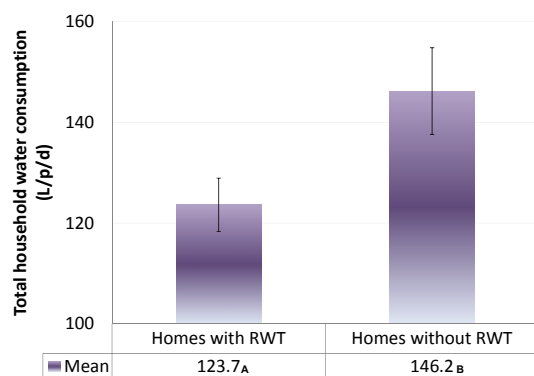


Figure 6. Water use for homes with and without a RWT.

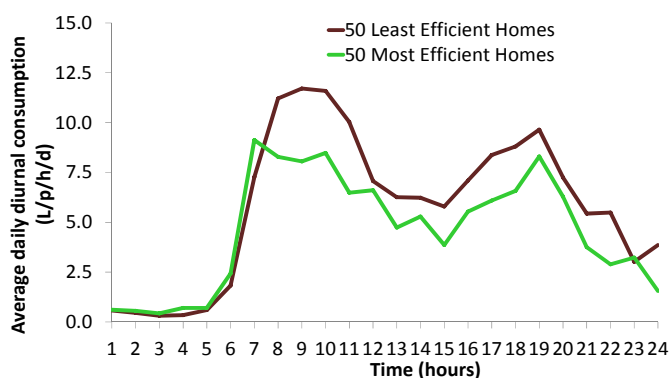


Figure 7. Average day diurnal pattern efficiency clusters.

Impacts of Household Socio-Demographics on Water Consumption

Higher income households consumed more water on average per day than lower income homes (Figure 8). The end uses that contributed most to the increased consumption were shower, clothes washer, dishwasher and bath. There was a trend for households with small families, with an older average age of residents and no children to consume less water per household on average. At an average total of 354 L/hh/d, households with either full and/or part-time residents consumed significantly more ($p>0.05$) water than those homes with retired and/or pensioned residents (253 L/hh/d) (data not shown). Typically, water consumption will be higher for large homes with large families as the demand for water is obviously greater and there are a higher number of water fixtures and appliances, however, larger families are typically more water efficient on a per capita basis than single families (Figure 9). Homes with one or more teenagers consumed significantly more water for shower events compared to homes without teenagers (Figure 10). In terms of perceived water use clusters, a clear pattern emerged from the results which showed that self-reported high water users typically consumed less (130 L/p/d) than both the self reported medium (156 L/p/d) and low (142 L/p/d) water users on a per capita basis (Figure 11). Further discussion of this can be found in Beal *et al.*, (2011b). Results also indicate a trend that higher income, larger, younger and more educated households tend to install efficiency appliances which may not always be sufficient in reducing water consumption if curtailment actions are not present. Therefore water consumption behaviours, as well as technology needs to be considered as part of a successful demand management strategy.

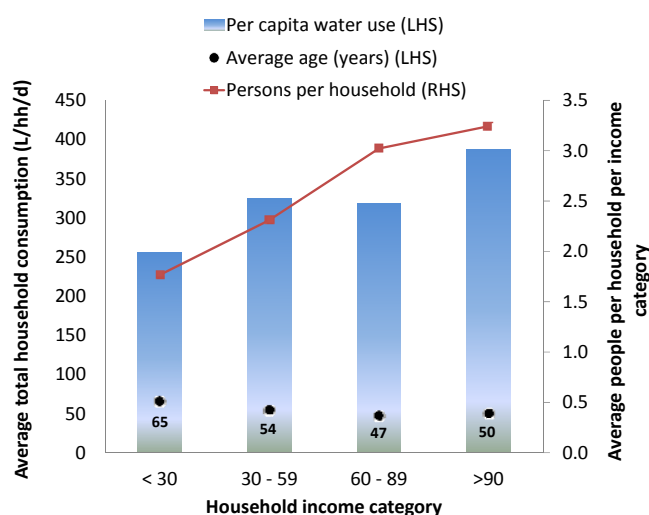


Figure 8. Income category, age and occupancy.

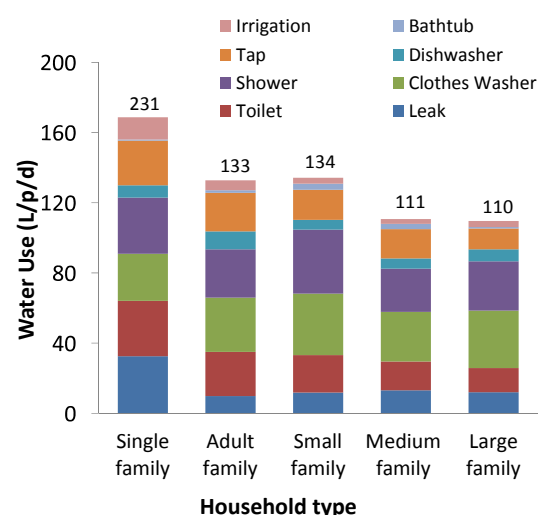


Figure 9. Family size and end use breakdown.

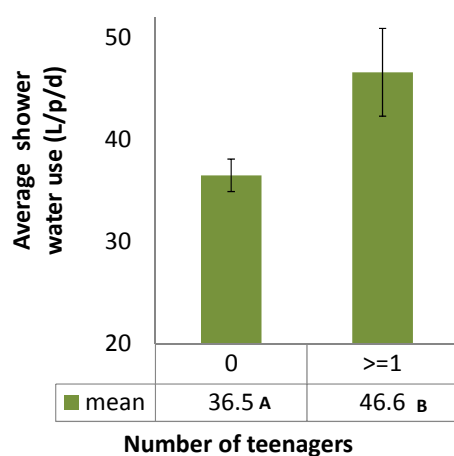


Figure 10. Teenagers and shower usage.

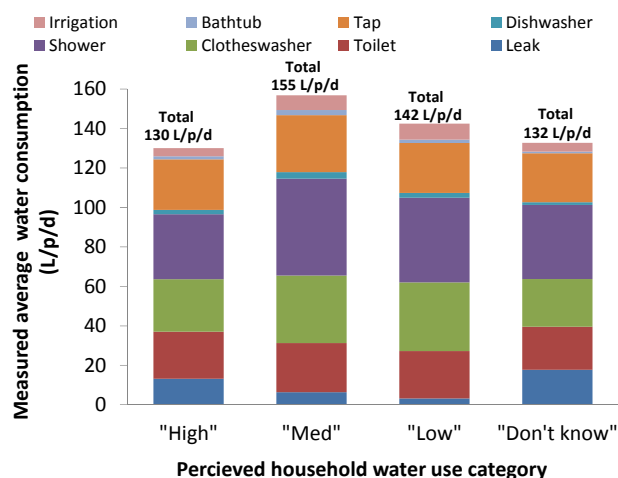


Figure 11. Perceived and actual household water use.

Conclusions

Some conclusions and water demand management key points for stakeholders have been drawn from the findings of the SEQREUS:

- There is still some degree of non-compliant irrigation during the 10 am to 4 pm period, particularly for homes in the Sunshine and Gold Coasts.
- Leaking toilets were more widespread than previously reported, however intervention programmes can be very effective at reducing these leaks as was shown in the summer and winter 2011 monitoring. Rapid post-meter leakage management is one of the key benefits of smart metering systems.
- Water efficient fittings for showers and taps are an excellent water demand management option for conserving water, confirming previous studies.
- Installing water efficient shower heads and clothes washers are significant areas for reducing average day peak hour demand.
- Changing to efficient washing machines and low-flow shower heads significantly reduces household consumption. Diurnal patterns indicate that by encouraging a shift in clothes washer operation from morning to evening, like the existing habit for dishwashers, would substantially reduce the average morning peak demand.
- Results strongly suggest the high importance of a sustained targeting of water consumption behaviour, particularly shower and tap use, as well as encouraging installation of water-efficient measures.
- Families with young children or teenagers are high water consumers on a household basis and this can be a target area for water conservation managers to consider, especially as these homes that have water-efficient technology, may not necessarily have low water consumption behaviours. Single person households, while having a high per capita consumption, typically do not contribute to the peak day demand periods.

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Water End Use Feedback Produces Long-Term Reductions in Residential Water Demand

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Summary

This paper reports on a study of household water use, which trialled three different interventions designed to reduce consumption. Two hundred and twenty-one households in South East Queensland (SEQ) were divided into four groups – three different intervention types and one control group – and their ongoing daily water consumption was monitored before, during and after the intervention. In the *information only* condition, households received general advice about how they could save water. In the *descriptive norm* condition, households received information about water saving along with information about the numbers of other “low water use households” that used these same behaviours. In the *water end-use feedback* condition, households received water saving tips along with tailored specific information about where water was being used in their own household. Longitudinal analysis showed that the *information only* and *descriptive norm* conditions resulted in water savings quite rapidly, but these changes were lost over time, as households later returned to their pre-intervention water use levels. In the *water end-use* condition, by contrast, water use declined more slowly, but the effect of this intervention was longer-lived. Implications for future interventions in domestic water consumption are discussed.

Keywords

Household water use, behavioural intervention, longitudinal analysis.

Introduction

Despite the fundamental role that water plays in sustaining human life and the health of the planet’s ecosystems, there is evidence that almost 80% of the world’s population is exposed to high level threats to water security (Vörösmarty *et al.*, 2010). There is growing evidence that human activities are placing unsustainable demands on fresh water resources with ground water supplies over-extracted in many regions in the world (Postel, 1999) and many major river systems without adequate water flows (Postel, 1996). Water resources will be placed under further pressure in coming decades by population growth, urbanisation and economic development (Beck and Bernauer, 2011; Güneralp and Seto, 2008; United Nations, 2009; Vörösmarty *et al.*, 2000). Moreover, climate variability is likely to further exacerbate existing stressors on water supplies (Bates *et al.*, 2008) and to result in increased impacts on water resources in some parts of the world (Arnell, 2004). Policy makers therefore face a critical challenge that requires them to balance human demand for water while at the same time protecting fragile ecosystems.

Addressing future water security will require a range of adaptive approaches enacted at different levels (eg, individuals, communities, business, government, international bodies). In urban contexts these adaptive approaches could include drought contingency planning, sourcing alternative water supplies, increasing the productivity of existing water supplies, and promoting adoption of water conservation practices in the home (Postel, 2000; de Loë, Kreutzweiser, and Moraru, 2001). Managing demand is considered an essential element of future water security (Arbués *et al.*, 2003; Brooks, 2006; Jeffrey and Gearey, 2006). De Loë and colleagues (2001) propose that water conservation and demand management approaches are appropriate measures for dealing with threats to water security in the near term as they promote water efficiency, reduce stress on the environment, and they are consistent with existing initiatives by government agencies. Moreover, the IPCC has heralded demand management as a “no-regrets option” to cope with future vulnerability of water supplies in the face of climate change impacts (Bates *et al.*, 2008).

A comprehensive review by Inman and Jeffrey (2006) showed that demand side management programs could reduce residential water consumption by 10–20% over a 10 to 20 year period. They concluded that moderate reductions could be achieved through price increases and voluntary demand management tools but more substantial reductions require large price increases or stringent mandatory policies. Similarly, Renwick and colleagues (2000) in a comparison of the efficacy of demand management approaches also conclude that voluntary measures reduce residential water demand, although more stringent mandatory approaches, such as water restrictions or water use allocations, result in greater reductions. Pricing mechanisms and mandatory approaches have clear drawbacks though for utilities and government agencies; there are equity issues involved with their implementation, limits to the price elasticity of residential demand, and evidence that they do not necessarily result in long-term change (Alitckov and Kostova, 1996; Duke *et al.*, 2002; Espey *et al.*, 1997). In addition, they require political will to implement. In contrast, research suggests that voluntary approaches that involve behavioral change may be critical to promoting long-term cultural shifts in the way community members think about and use water, and that these shifts in values and behavior can complement other demand management approaches.

In light of this, the current study tests the efficacy of voluntary demand management strategies for producing long-term reductions in residential water consumption. Our aim is to identify voluntary strategies that can reduce water demand in residential settings and to assess whether these strategies are effective not just in the short term but also in the long-term. As noted above, given the potential for restrictive mandatory policies and pricing mechanisms to generate resistance from the community, and therefore a failure of political will to implement them, it is critical to develop voluntary tools that can positively influence water demand. We conduct an experimental field study that integrates behavior change theory with water engineering technology. In comparison to a control condition, we test the effectiveness of providing three different types of information: 1) information about how to save water; 2) information about what other people in the community are doing to save water (ie, descriptive norms); and 3) water end use information that describes how water is used in the specific household. Households in our study have 'smart' water meters installed that allow accurate tailored feedback to households about how much water they use on average for each water using activity (in the water end-use condition). In addition, the smart water meters allow accurate daily measures of household water use to be collected. This study is the first to use smart water metering technology as a tool for behavior change, as well as a way to test the effectiveness of water demand management interventions.

Methodology

The study was conducted within four local government areas in South East Queensland (SEQ): Brisbane, Gold Coast, Ipswich and the Sunshine Coast. Participants were recruited from a sample of householders who had taken part in a previous study (Household Water Use Survey; see Fielding *et al.*, under review) during 2009, and who indicated willingness to participate in further research. Due to the necessity of accessing household water consumption data, targeted households were owner-occupiers of dwellings connected to the central water supply with an individual water meter attached to the premises. A total of 400 consenting responses were received from the four study regions (127 from Brisbane, 102 from Gold Coast, 73 from Ipswich, and 98 from Sunshine Coast). Consenting households were contacted to arrange for installation of the smart meters and data loggers. Due to technical issues affecting the suitability of smart meter replacement, not all willing households were able to be fitted with the meters. Further details about the smart meter installation and the associated home water audits can be found in Beal *et al.*, 2010. A baseline water end use data read and analysis was conducted in June 2010. In total, there were 221 households for which baseline end use data analysis and Household Water Use Survey data were available. These households were included in the current study. It was critical to have both sets of data as the survey provided household occupancy data that allowed us to calculate per person water usage.

Study Design

The study consisted of a randomised controlled trial testing three information-based interventions to reduce household water consumption. A control condition in which participants received no information or intervention was included in the study design for comparative analysis. The overall study period is broken down into three components: 1) the pre-intervention period during which participants were recruited and baseline survey data and water use data were collected (September 2009 – June 2010); 2) the intervention period during which the interventions were implemented and outcome data collected (June 2010 – May 2011); and 3) the post-intervention period (January 2011 – July 2011) in which the long-term effectiveness of the interventions were assessed.

Prior to the start of the intervention period, all participants received an initial letter (sent on 6 September 2010). There were two versions of the initial letter: one for the first three intervention conditions and one for the control condition. These letters differed only in that participants in the intervention conditions were informed that they would receive information from the research team over the coming months. The intervention medium for the three experimental groups was the provision of different types of information via a four-part series of postcards. Each postcard provided information about ways to save water in a particular location of the house (bathroom, laundry, kitchen) or through checking and fixing leaks. The postcards also included a cartoon graphic that related to the focus of water conservation for that period. For example, the postcard aimed at reducing water use in the bathroom had a cartoon relating to taking short showers. One side of each postcard was identical for all three experimental groups, and provided a set of tips for how to save water in that particular part of the house (eg, bathroom, kitchen, etc). The overleaf side of the postcard differed according to the experimental condition as described below.

The *information only* condition was included in the study design to allow the effects of receiving information about saving water to be separated from the effects of the specific intervention conditions (eg, descriptive norms, water end-use feedback). Therefore, as with the postcards in all conditions, participants in this condition received information about how to save water in that particular part of the house. In this condition, the second side of the postcard was a larger version of the cartoon graphic used in all of the experimental conditions.

The *descriptive norm* intervention provided participants with information about the sorts of actions that low water using households that were similar to theirs in terms of composition (ie, single person, multiple adults, or family with children) did to save water. The second side of the descriptive norm condition postcards was thus comprised of two components: 1) a cartoon graphic; and 2) statements about the percentage of low water using households who reported always or almost always performing a certain water saving behaviour. For example, the bathroom postcards said: “Join South East Queenslanders like you in conserving water. Our research of 1000 SEQ households identified what people who are low water users do to save water. Single person households like yours (Households like yours with children/Households like yours with multiple adults) do the following: 78% (70%/84%) take shorter showers; 90% (92%/90%) turn off the tap when they are cleaning their teeth; 94% (93%/93%) use half flush or don’t flush the toilet every time”. These statements were derived from data collected from the Household Water Use Survey in combination with water use data obtained from the water utilities, and were therefore construed to be representative of households in SEQ. Each household allocated to the descriptive norm condition received the postcard that matched their circumstances.

The *water end-use* condition aimed to provide participants with information about how much water was used in their household overall and in various parts of their household. Participants allocated to this condition received individualised postcards with information derived from water-end use analysis of their home. For two of the postcards (#1 – Bathroom and #3 – Kitchen) this consisted of a pie chart showing a breakdown of household water along with the average daily per person usage. For the bathroom postcard a statement was added saying: “As you can see, a large proportion of water is used in the bathroom. Turn over this card to see how you can save water in the bathroom” (a similar statement was also made in relation to the kitchen). Due to the time burden associated with performing the end use analysis, it was not possible to provide this information at each time-point. Thus, the second postcard (corresponding to saving water in the laundry) reminded participants of the amount of water used in the laundry from the previous analysis, and the final postcard (corresponding to checking and fixing leaks) either reminded participants that a leak had been detected in their home, or that one may occur in the future if no leak had been previously detected.

Measures

The initial household water use survey provided demographic information about household occupancy and household composition that enabled accurate calculation of per person water usage and descriptive norm information to be tailored to the household’s specific composition. The smart water metering data provided daily household water use for the pre-intervention, intervention, and post-intervention period. It also provided the average per person per day water use data for the water end-use condition postcards #1 and #3. Analyses were conducted on an adjusted measure of water consumption: “trend daily water use per person”. This measure provides a simple seven-day moving average of daily water use per person: the value for any particular day is the average of the previous seven days. Thus, the measure provides a lagged daily trend in water use which smoothes daily fluctuations in water use, reducing noise in the data (eg, higher water use on weekends for some households). Per person estimates were derived from measures of total household water use divided by the estimated number of persons in each household.

Results

Table 1 shows that the mean daily trend water use per person was 135.37 litres over the entire study. During the intervention period, mean water use dropped by 10.29 litres per person per day and the median dropped by 7.6 litres. In the post-intervention period the mean water use returned to near pre-intervention levels, without considering different interventions, however the median water use increased only slightly.

Table 1. Trend daily water use for different periods (litres per person per day).

Study Period	Dates	Mean	Median	SD	Skewness	Min	Max
Pre-intervention	20/6/2010 to 12/9/2010	141.52	123.21	86.50	3.35	10.32	1404.90
Intervention	13/9/2010 to 10/1/2011	131.23	115.61	90.26	4.06	10.00	1617.97
Post-intervention	11/1/2011 to 31/7/2011	139.00	116.24	144.40	13.05	10.00	3625.60
All periods		135.37	115.99	121.86	12.28	10.00	3625.60

Table 2 shows that during the intervention period the median trend water increased somewhat in the post-intervention period except for the water end-use condition. The water end-use and information only interventions have the lowest levels of median trend water use across all periods; however, note that the information only condition also started with the lowest pre-intervention level.

Longitudinal growth curve modelling was employed to increase the power of data analyses; to model non-linear effects such as rebound effects and to better account for any days of missing water use data. One problem with longitudinal data is that observations of daily water use for a particular household are not independent (ie, they are correlated with each other). This violates an assumption of most regression analyses. However, growth curve modelling is a regression technique, which controls for inter-correlations or inter-dependence between daily observations of water use for each household by estimating a constant (or pre-intervention) level of water use for each household around which fluctuations in water use are modelled.

Table 2. Median trend daily water use by intervention group and period (litres per person per day).

Intervention Group	Pre-Intervention Period	Intervention Period	Post-Intervention Period	All Periods
Control	126.01	120.02	121.96	122.47
Water end-use	121.02	112.61	110.26	114.07
Descriptive norm	128.27	116.65	120.38	120.89
Information only	112.83	110.93	112.93	112.09
Total	123.28	115.61	116.24	117.85

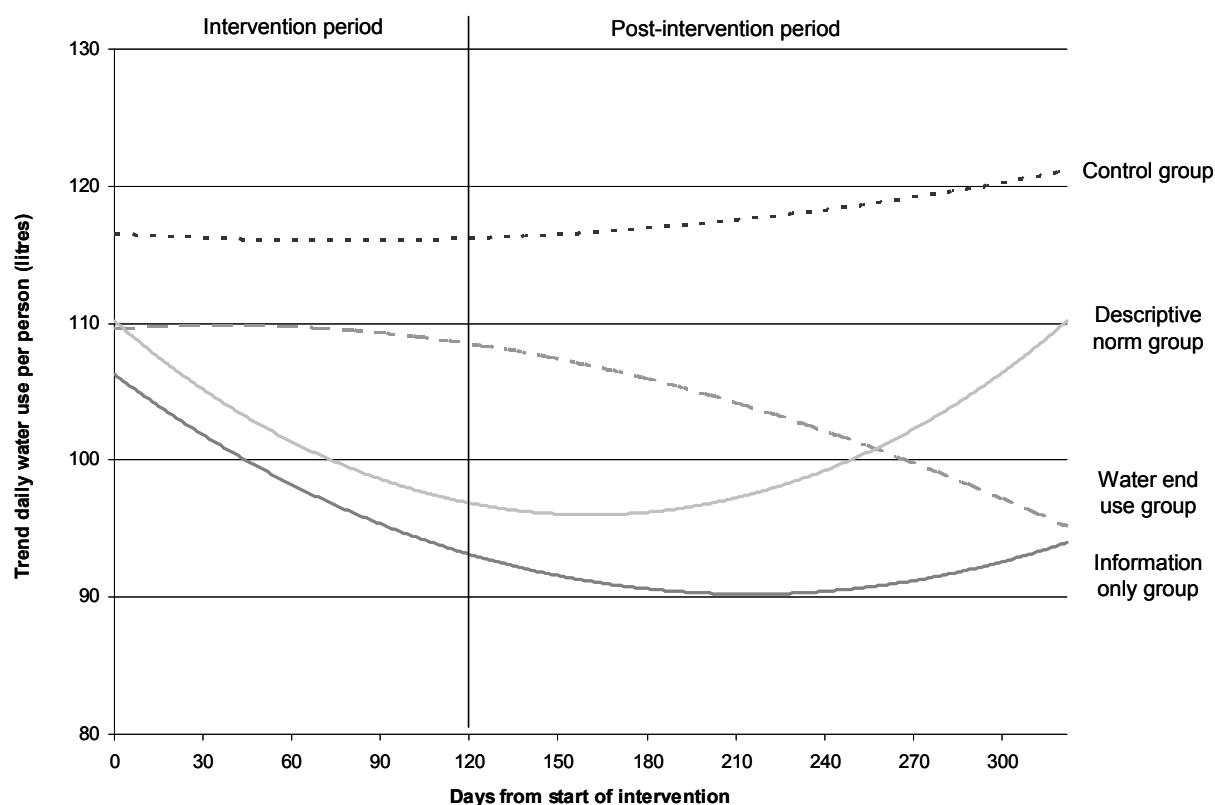
Table 1 shows wide variation and high skewness in daily water use, even though this was somewhat smoothed by measuring trend daily water use per person. Wide fluctuations and high skewness is problematic for modelling water use. Very high water use may occur during uncommon events like filling swimming pools and very low water use may occur when people are away (eg, on holidays). To better account for days with extremely high water use, the log of trend water use was modelled. Days with very low water use (zero or near zero water use) were treated as missing values because they are likely to represent household absences (and water leaks). Near zero water use was defined as less than 10 litres per person per day. These data preparation steps enable more accurate estimation of trend daily water use. To further enhance the quality of estimates for trend daily water use over time, households which did not have at least 10 weeks of daily water use data were excluded from the longitudinal analyses.

Longitudinal Modelling

The random intercept model estimates *logged* trend daily water use per person while also estimating an average level of water use per household, which varies between households. Estimates of this average water use (or y-intercept) for each household were enhanced by using the baseline water use measure as the first observation in the time series of water use for each household. This controls for an average level of water use associated with each household, along with other time constant attributes of households (eg, the specific appliances installed). Covariates were then included to model trend water use per person over time. A covariate for the number of days from the start of the intervention period was included to estimate long-run linear trends in water use over the entire intervention/post-intervention period, as well as including a covariate for days squared to estimate any non-linear change in long-run water use.

The main independent covariate of interest was the intervention group (control, descriptive norm, water end-use and information only groups). A dummy variable was added for the intervention group to estimate whether these interventions varied significantly in water use at the start of the intervention period. An interaction term between the intervention group and day number was added to estimate any linear changes in water use for each intervention group over time. Another interaction term was added between the intervention group and days squared to estimate any non-linear changes in water use over time for each intervention group. Lastly, a dummy variable for the four regions in this study was added to account for variations in water use that may exist between regions in SEQ.

Due to space constraints, results from the random intercepts model are reported only briefly in this paper. At the start of the intervention period, none of the intervention groups were significantly different from each other in water use. The coefficients for the different regions were also not significant, indicating that none of the regions differed significantly from Brisbane. Figure 1 shows that although the control group increased in trend daily water use over time, this was not statistically significant. The descriptive norm group declines significantly ($p < .05$) during the intervention period; however, shortly after the intervention period it increases again, losing all the gains made during the intervention period. In the information only group, there is a significant decline over the intervention period. However, there is an increase in water use again, approximately half way through the post-intervention period. The water end use group shows little decrease in water use initially. However, it declines significantly after the intervention period finishes (after day 120), and shows no rebound effect at 102 days after the intervention (ie, day 322).



Note: Due to transformation of logged data, Figure 1 reflects estimates of *median* water use.

Figure 1. Trend daily water use per person (litres) for intervention and post-intervention periods.

Discussion

Compared to the control condition, all three interventions were successful in reducing residential water demand, although they showed different profiles over time. Rebound effects, in which behavioural interventions show a positive initial effect that fades over time, were clearly identified for two of the three interventions. Such clear empirical demonstration of rebound effects is uncommon in the intervention literature. Although the water end-use condition did not show a rebound effect, it is unrealistic to expect that the households in this condition will continue to reduce consumption forever. Logically, they must reach some minimum consumption, and it is also likely that they too will show a rebound effect if enough time passes: it is unreasonable to expect that water savings will be maintained in the long term without some reinforcement or further intervention. Although it is probably impractical to continue this current project further to continue tracking the changes in water use profiles over time, we have demonstrated the high value to be gained from large-scale longitudinal research of household consumption: intervention studies with actual measures of the targeted behaviour clearly have a central role to play in furthering our understanding of what interventions are most effective.

The different impacts over time of the various interventions have important implications for the value of each intervention. It may be that more technology-intensive and expensive interventions, like the tailored feedback provided in our water end-use condition, may be more cost effective over time if they result in water savings that are more long lasting. Further analyses are planned to identify the costs of water savings for each of the interventions, so that this direct comparison of effectiveness can be made. Further research of this type is needed to replicate these results, and expand our understanding of when and why interventions of this type “wear off” over time. Understanding these effects will provide valuable knowledge for water utilities and other groups that are involved in demand management activities.

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Case Study - Occurrence of Non-Regulated Disinfection By-Products from the Capalaba Region's Distribution System

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Summary

A survey of disinfection by-product (DBP) occurrence was conducted at the Capalaba Water Treatment Plant (WTP) in Brisbane and at nineteen sampling points of the distribution system for six months through spring-summer 2011. In addition to currently regulated DBPs including *N*-nitrosodimethylamine (NDMA), DBPs that were reported as high priority due to potential toxicity were studied. These priority DBPs included five iodinated trihalomethanes (THMs), four haloacetonitriles (HANs), two haloketones (HK), chloronitromethane and chloral hydrate (CH). Results showed that all the measured regulated DBPs including CH were below the Australian Drinking Water Guidelines (ADWG) recommendations. THM speciation followed the order trichloromethane (TCM) > bromodichloromethane (BDCM) > dibromochloromethane (DBCM) > tribromomethane (TBM) in waters provided by the Capalaba WTP. The order was DBCM>BDCM>TBM>TCM for waters provided by the North Stradbroke Island (NSI) WTP as a result of different dissolved organic carbon/bromide (DOC/Br) ratios. HANs were measured at the WTP and across the distribution system at significant concentrations, but still lower than the World Health Organisation (WHO) recommendations in the case of dichloroacetonitrile and dibromoacetonitrile. Speciation of brominated and chlorinated HANs followed the trends observed for THMs. The formation of iodinated-THMs was low, in agreement with the iodide concentration measured. NDMA was not detected above the limit of detection (5 ng/L). NDMA formation potential of source water at the Capalaba WTP was 11.4 ± 3.4 ng/L, which is well below the ADWG (ie, 100 ng/L).

Keywords

Chlorine, distribution system, emerging DBPs.

Introduction

Naturally occurring organic matter, which is present in raw waters, reacts with disinfectants to form disinfection by-products (DBPs). Formation of DBPs in drinking water is of concern as some have been linked to potential health effects (Richardson *et al.*, 2008). Chlorine reacts with natural organic matter (NOM) and/or anthropogenic compounds to produce a mixture of DBPs such as trihalomethanes (THMs) and haloacetic acids (HAAs). Regulated THMs include trichloromethane, bromodichloromethane, dibromochloromethane and bromoform. Total THMs (tTHMs) refers to the sum of these four compounds. The USEPA regulates tTHMs at 80 µg/L (USEPA, 1998) while the value for the Australian Drinking Water Guidelines (ADWG) is 250 µg/L (NHMRC, 2011). There are 15 HAAs which can be formed in the presence of chlorine, bromide and iodide. The most common are dichloroacetic acid (Cl₂AA) and trichloroacetic acid (Cl₃AA). Other species, found generally at lower levels, are bromochloroacetic acid (BrClAA), dibromoacetic acid (Br₂AA), monochloroacetic acid (ClAA) and monobromoacetic acid (BrAA) as well as additional iodine-containing counterparts. In the USA, the sum of Cl₂AA, Cl₃AA, Br₂AA, ClAA and BrAA is commonly denoted as HAA5 and is currently regulated at 60 µg/L (USEPA, 1998). In Australia, only ClAA, Cl₂AA and Cl₃AA are regulated, at 150, 100 and 100 µg/L, respectively (NHMRC, 2011). While the US EPA drinking water standards for tTHMs and HAAs are numerically lower than the Australian drinking water guideline values, compliance in the US is defined on the basis of a running annual average of quarterly averages of all samples, whereas compliance is based on single exceedances in Australia. Recently, Australia has included *N*-nitrosodimethylamine (NDMA) in the ADWG at the World Health Organisation (WHO) guideline value of 100 ng/L (NHMRC, 2011; WHO, 2011). Other DBPs that may be generated at lower concentrations are haloacetonitriles (HANs), halogenated furanones, halonitromethanes (HNM), cyanogen halides, haloketones (HK), haloaldehydes and halogenated phenols, among others. However, the concentration of the halogenated DBPs identified to date account for less than 50% of the total organic halogens (TOX) present in a chlorinated water (Reckhow and Singer, 1984; Buffle *et al.*, 2004).

Nitrogen containing DBPs (N-DBPs) such as HANs and N-nitrosamines are suspected to be more toxic than carbon based DBPs (Plewa *et al.*, 2004; Muellner *et al.*, 2007; Plewa *et al.*, 2007). The higher toxicity of N-DBPs have already raised concerns, with regulators in Australia including NDMA at 100 ng/L, which is three orders of magnitude lower than the values used for more conventional DBPs such as tTHMs (tTHMs 250 µg/L). HANs are currently not included in the ADWG, however, some of them are already included in the recycling water guidelines,

with values as low as 0.7 µg/L for bromochloroacetonitrile (QPC, 2005). Moreover, N-DBP precursors are less amenable to conventional drinking water treatment processes than most precursors for carbon based DBPs, due to their hydrophilic characteristics (Bond *et al.*, 2011). Conventional water treatment systems are designed to treat potentially contaminated source water, in order to prevent the potential spread of waterborne disease-causing micro-organisms and reduce potable water problems related to NOM, such as undesirable colour, odour, taste and formation of DBPs. Such treatment systems typically consist of source water intake/screening, coagulation, flocculation, sedimentation, rapid sand filtration and disinfection processes (Kameya *et al.*, 1997). Coagulation changes the composition of organic matter by preferentially removing more oxidised NOM compounds, leaving compounds with a higher tendency to form chlorine-generated N-DBPs such as HANs (Xiao *et al.*, 2010). Nitrogen containing DBPs have been measured mainly in the United States (US), Canada and Europe (Richardson *et al.*, 2007). Simpson and Hayes sampled 16 drinking waters from around Australia and measured the sum of four HANs expressed as 4HAN, chloropicrin and cyanogen chloride (Simpson and Hayes, 1998). They found values up to 36 µg/L of 4HAN in different regions of Australia compared to the median and maximum levels of 3 and 14 µg/L found in a 2000-2002 US survey (Weinberg *et al.*, 2002; Krasner *et al.*, 2006). In another 2006-2007 US survey, median values for the sum of 4HAN was slightly higher at 4 µg/L (Mitch *et al.*, 2009).

In our previous work, we measured the DBP formation potential at three WTPs in Brisbane and we found the highest DBP formation from Capalaba source water (Farré *et al.*, 2011). Therefore, we selected this region for further evaluation. In the present paper we present the concentration of DBPs at Capalaba WTP (Brisbane) and its distribution system which also includes water from North Stradbroke Island (NSI).

Methodology

Samples were taken at Capalaba WTP and from the distribution system (Figure 1) during five sampling events during spring-summer 2011.

The sampling dates were 5/9/2011, 3/10/2011, 7/11/2011, 21/11/2011 and 6/12/2011. Samples were taken headspace free, in acid washed glass containers with Teflon lids. Ascorbic acid was used to quench the chlorine and to protect the present DBPs. Samples were shipped to The University of Queensland (UQ), with volatile DBPs analysed within 24 hours. The following DBPs were extracted by liquid-liquid microextraction and analysed by gas chromatography with electron capture detection (GC/ECD): trichloromethane (TCM); bromodichloromethane (BDCM); dibromochloromethane (DBCM); tribromomethane (TBM); dichloriodomethane (DCIM); bromochloriodomethane (BCIM); dibromiodomethane (DBIM); chlorodiiodomethane (CDIM); bromodiiodomethane (BDIM); trichloroacetonitrile (TCAN); dichloroacetonitrile (DCAN); bromochloroacetonitrile (BCAN); dibromoacetonitrile (DBAN); CH; trichloronitromethane (TCNM); 1,1-dichloropropanone (1,1-DCP); and 1,1,1-trichloropropanone (1,1,1-TCP). Additional analyses were done such as total organic carbon (TOC), SUVA, bromide and iodide. NDMA was extracted at UQ by means of solid phase extraction and concentration under nitrogen and samples were taken to QHFSS for analysis by gas chromatography coupled to mass spectrometry (GC/MS) with chemical ionization with ammonia gas.

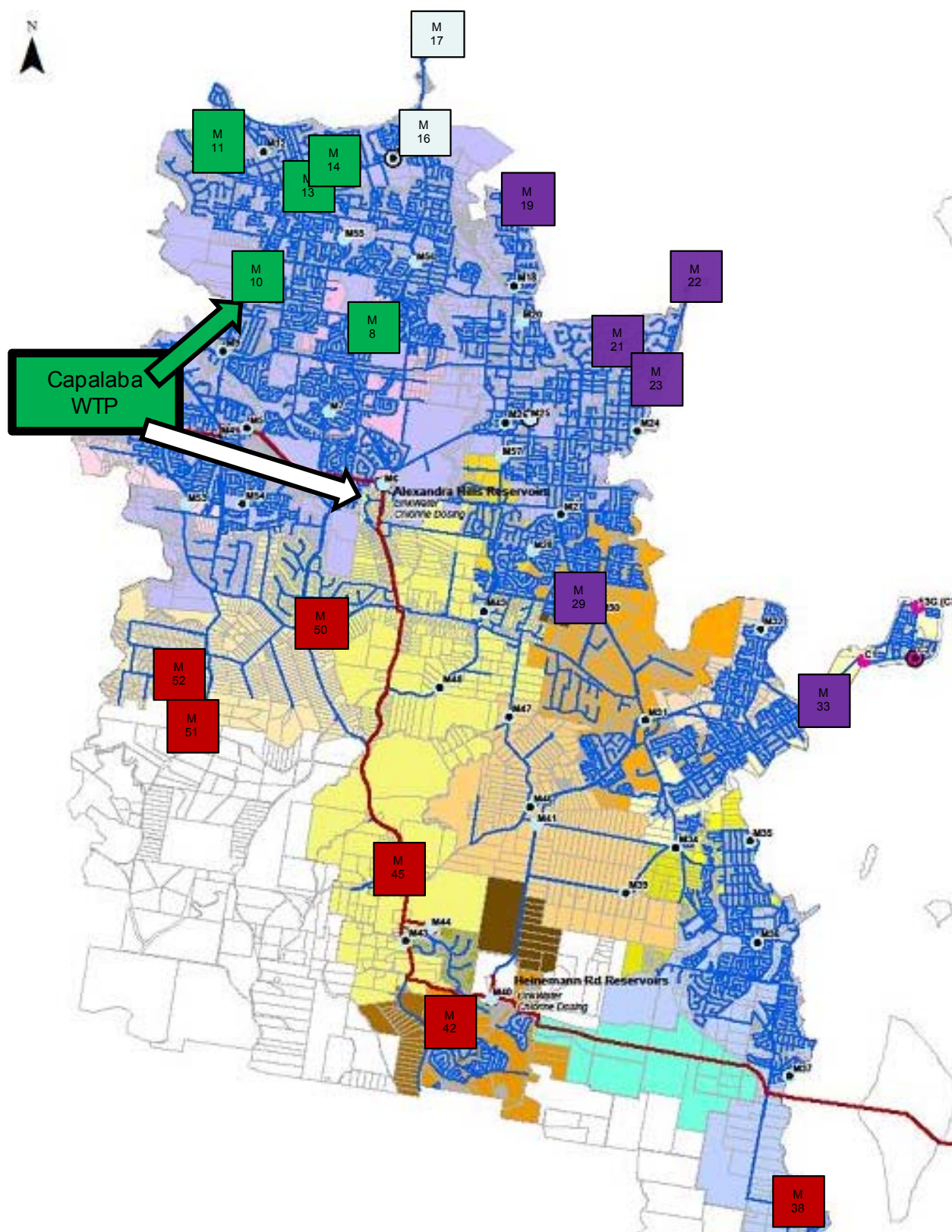


Figure 1. Sampling points selected for the study. NSI= North Stradbroke Island. Green=Capalaba WTP source water, red=NSI WTP source water, purple=Alexandra Hill Reservoir (Capalaba + NSI), blue=Capalaba + Alexandra Hill Reservoir. White arrow represents water from Capalaba to Alexandra Hill Reservoir (Courtesy of Allconnex Water).

Results

Table 1 shows TOC, bromide, dissolved and organic nitrogen (DON) values and the standard deviation for the sampling points selected for this study and Capalaba WTP. Samples have been divided according to the source water used. Samples M16 and M17 provide water from Capalaba WTP and Alexandra Hill Reservoir, which is a mix of Capalaba and NSI water. Hence, a higher proportion of Capalaba water is expected for these sites in comparison to sites M19-M33, which supply water from Alexandra Hill Reservoir without further blending. Maximum average TOC values were around 5-6 mg/L and were found in sampling points with a higher percentage of water from Capalaba WTP, while lower values were found in sampling points providing water from NSI. Bromide concentration ranged between 0.025 and 0.07 mg/L across the samples. DON was clearly higher in Capalaba water in comparison to NSI.

Table 1. Water characterisation during the sampling campaign in the Capalaba distribution system. n=number of samples, TOC=total organic carbon, DON=dissolved organic nitrogen, SUVA=specific UV absorbance, Cap=Capalaba WTP, NSI=North Stradbroke Island WTP.

Sampling point	n	Source water	TOC mg/L	St dev	Br- mg/L	St dev	DON mg/L	St dev
M8	5	Cap	3.83	0.79	0.025	0.004	0.29	0.29
M10	5	Cap	2.31	2.07	0.036	0.007	0.21	0.08
M11	4	Cap	4.93	0.72	0.051	0.006	0.29	0.15
M13	5	Cap	4.49	1.18	0.038	0.006	0.36	0.04
M14	4	Cap	4.07	1.42	0.040	0.005	0.29	0.08
M16	5	Cap + NSI (via Alexandra Hills res)	4.92	2.25	0.041	0.007	0.33	0.26
M17	4	Cap + NSI (via Alexandra Hills res)	2.93	0.64	0.060	0.008	0.27	0.13
M19	5	Alexandra Hills res (Cap + NSI)	1.67	0.63	0.030	0.005	0.15	0.12
M21	4	Alexandra Hills res (Cap + NSI)	2.46	1.81	0.030	0.005	0.11	0.10
M22	2	Alexandra Hills res (Cap + NSI)	1.07	0.34	0.027	0.005	0.14	0.07
M23	4	Alexandra Hills res (Cap + NSI)	2.10	2.48	0.043	0.007	0.20	0.18
M29	4	Alexandra Hills res (Cap + NSI)	1.00	1.07	0.066	0.003	0.11	0.04
M33	2	Alexandra Hills res (Cap + NSI)	0.55	0.34	0.060	0.002	0.11	0.01
M38	3	NSI	1.97	2.18	0.065	0.009	0.10	0.03
M42	3	NSI	1.89	2.56	0.070	0.003	0.09	0.12
M45	3	NSI	0.64	0.50	0.066	0.001	0.14	0.11
M50	4	NSI	0.46	0.42	0.064	0.004	0.08	0.10
M51	5	NSI	0.75	0.47	0.059	0.006	0.15	0.09
M52	3	NSI	1.21	0.34	0.061	0.002	0.13	0.11
Cap	4		5.99	0.97	0.053	0.007	0.32	0.11

THMs in conjunction with HAAs are the most prevalent DBPs in drinking water and are formed as a result of the reaction between chlorine and NOM. Figure 2 shows the mean concentrations of tTHMs found in Capalaba WTP and the distribution system.

In all instances, tTHMs values were lower than the ADWG value of 250 µg/L. Elevated values were found in M8-M17 which correspond to sampling points providing water from Capalaba WTP. On the other hand, the remaining sampling points provided water with a high contribution of NSI treated water. Average concentrations of tTHMs at Capalaba WTP were lower than at M8, M11, M13, M14 and M16, which evidenced the ability of THMs to increase during distribution within the system, mainly as a result of hydrolysis of other DBPs (Nikolaou *et al.*, 2001). Sampling points providing water from Capalaba showed a distribution of TCM>BDCM>DBCM>TBM, which is common speciation in drinking water with high concentration of organic carbon and low concentration of bromide. The speciation of THMs measured in sampling points providing water from NSI was DBCM>BCDM>TBM>TCM as a result of the presence of bromide in low organic carbon waters (< 2 mg/L). The rate constant of bromide with HOCl to generate HOBr is 1.5×10^3 1/M-s (Kumar and Margerum, 1987) and the rate constant of THM formation is in the range of 0.01 and 0.03 1/M-s (Gallard and Von Gunten, 2002). It is known that once formed, hypobromous acid reacts about 10 times faster than chlorine with NOM. The reason is that the activities of electrophilic substitution for electron release to stabilise a carbocation are more favourable for the Br atom due to its higher electron density and smaller bond strength relative to the Cl atom (Westerhoff *et al.*, 2004). Hence, the formation of Br-DBPs is limited by the initial Br concentration whereas the Cl-DBPs would be limited by the organic matter concentration. Figure 2 shows the average concentration of THMs in the waters with a high contribution from NSI.

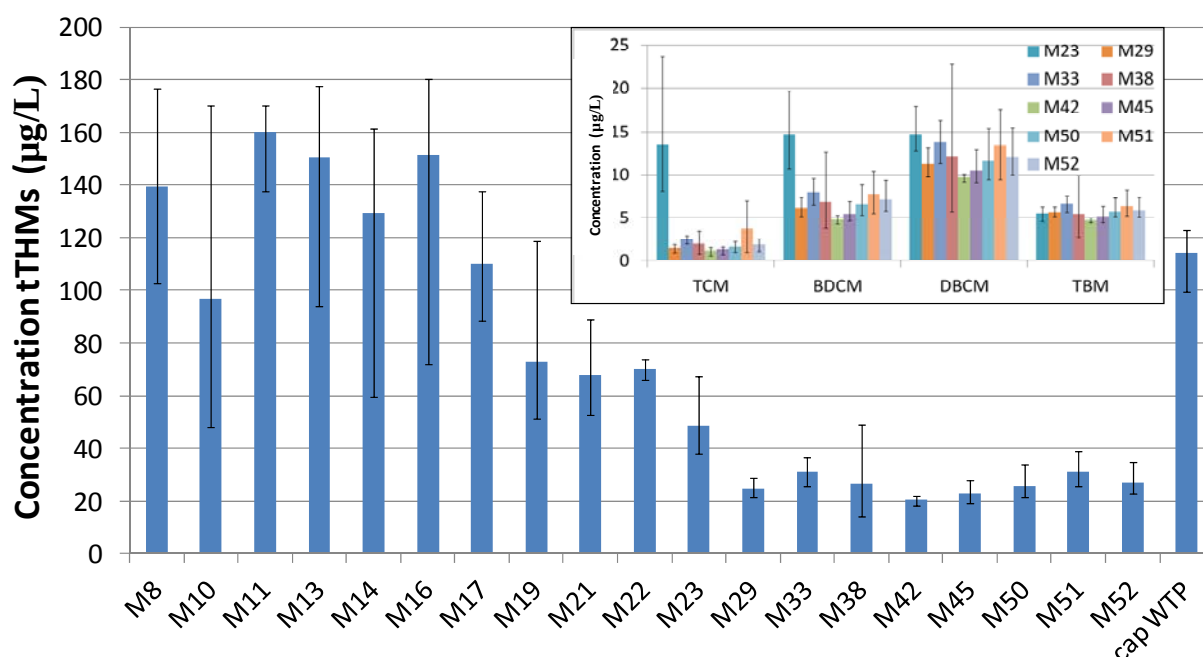


Figure 2. Average concentration and range of tTHMs in Capalaba WTP and distribution system. Inset shows the THMs speciation in waters from NSI.

Figure 3 shows the average concentration of the four analysed HANs. In agreement with Bougeard and co-authors (2010), all four analysed HANs were detected in all waters and their concentrations were typically an order of magnitude lower than the concentration of THMs. Also, the bromine/chlorine speciation was different between the sampling points providing water mainly from Capalaba WTP or NSI WTP as a result of the different concentration of TOC. Maximum concentrations measured corresponded to DCAN. 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, which is the likely explanation as to why it was rarely detected in this sampling campaign, as the pH of all the samples was 7 ± 0.4 (Croue and Reckhow, 1989). Although HANs are not included in the ADWG, the WHO has recommended the concentration of DCAN and DBAN to be 20 and 70 µg/L, respectively (WHO, 2011).

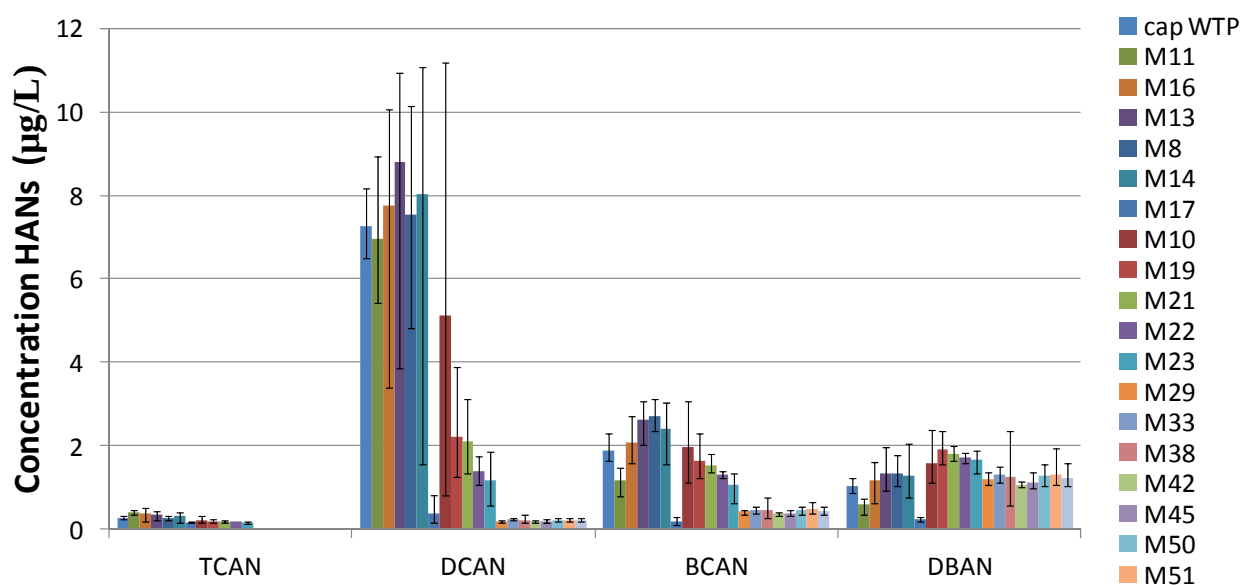


Figure 3. Average concentration and range of HANs in Capalaba WTP and distribution system.

Besides THMs and HAAs, CH is the next most prevalent DBP in chlorinated drinking water. Figure 4 shows the average concentration of CH also in conjunction with TCNM and two HKs.

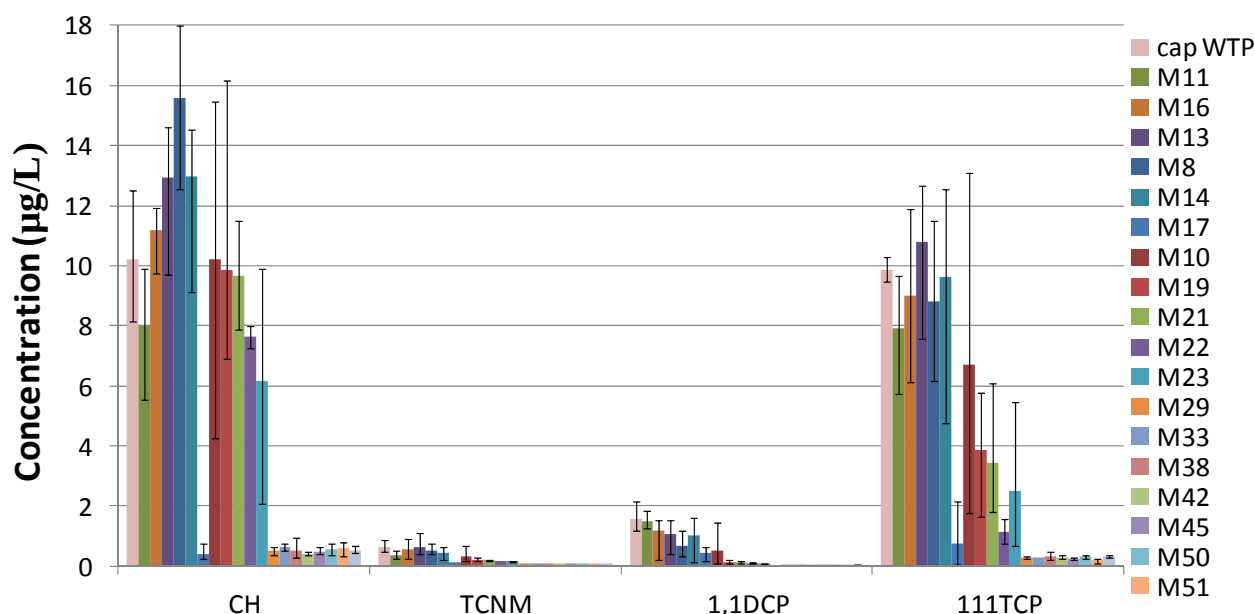


Figure 4. Average concentration and range of CH, TCNM, 1,1-DCP and 1,1,1-TCP in Capalaba WTP and distribution system.

At almost all the sampling points providing water from Capalaba WTP, concentration of CH was measured above 10 µg/L. 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 due to the different precursors involved in these reactions. Thus, the concentration of CH may continuously increase in the water supply system. CH has been previously studied, but the appearance of CH in drinking water is not well understood and causes many controversies (Goslan *et al.*, 2009). Trehy *et al.* (1986) reported that amino acids are potential precursors of CH and suggested that the precursors for TCM and CH are different. According to Ueno *et al.* (1995), nitrogen compounds and amino acids produced CH in the chlorination process.

TCNM and 1,1-DCP concentrations were low in all instances, while 1,1,1-TCP was found at concentrations up to 12 µg/L in the waters coming from Capalaba WTP. The differences in the formation of 1,1-DCP and 1,1,1-TCP can be partially explained by a simplified model developed by Reckhow and Singer (1984). In their model, chlorination of fulvic acid solutions led to the formation of intermediate by-products, such as 1,1-DCP, that could be further oxidised by chlorine to 1,1,1-TCP. This model reveals that further chlorine attack and hydrolysis are essential for the formation of 1,1,1-TCP.

Five iodinated-THMs were also measured in this study. We did not find I-THMs above 3.5 µg/L, which is in concurrence with the measurements of iodide in the samples that were, in all instances, below the 0.01 mg/L limit of detection (LOD). Similarly, NDMA was not detected above the 5 ng/L LOD across the distribution system. NDMA formation potential of source water at the Capalaba WTP was 11.4 ± 3.4 ng/L ($n=3$), which is also well below the ADWG value (ie, 100 ng/L).

Conclusions

All regulated DBPs were measured below ADWG values in all analysed samples across the Capalaba region.

- THM speciation followed the order TCM>BDCM>DBCM>TBM in sampling points providing water from Capalaba and DBCM>BCDM>TBM>TCM in waters blended with NSI water as a result of different DOC/Br ratios.
- HANs were measured at relatively high concentrations for locations serviced primarily by Capalaba WTP. Even though they were measured below WHO limits, we recommend investigating possibilities to control the formation of HANs at the drinking WTP as they are suspected to be more toxic than carbon-based regulated DBPs.

Acknowledgements

This research is being conducted as part of the Urban Water Security Research Alliance (UWSRA), a collaboration between the Queensland Government, CSIRO, The University of Queensland and Griffith University. The authors want to acknowledge Seqwater and Allconnex Water for giving access to the plants and assisting with the sampling.

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Disinfection By-Product Minimisation by Organic and Inorganic Precursor Removal

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Summary

A matrix of 18 synthetic waters with variable water quality parameters (alkalinity, natural organic matter (NOM), and halide concentration) was prepared. The DBP formation potential of these 18 samples was examined both before treatment, after enhanced coagulation (EC) and after a sequential treatment using EC followed by either powdered activated carbon (PAC), granular activated carbon (GAC), silver-impregnated activated carbon (SIAC), or MIEX® resin. The study shows that natural organic matter (NOM) concentration is greatly reduced with EC, and further reduced by each secondary treatment. Halide adsorption was not possible with EC or PAC, however, MIEX® and GAC had some halide adsorption capacity. SIAC exhibited the greatest halide removal (average 99% adsorption). Total DBP formation was reduced by EC, and further reduced by each secondary treatment process, however, specific highly brominated DBPs increased in concentration with each successive treatment step in all cases except after SIAC treatment. The more highly brominated DBPs may be of a greater public health concern than their chlorinated counterparts, therefore, in salinity-impacted waters, halide removal is desirable.

Keywords

Bromide, natural organic matter, chlorination, DBP formation potential, activated carbon.

Introduction

Understanding the formation and minimisation of disinfection by-products (DBPs) (emerging and regulated) in drinking water is important for the continued protection of public health (Richardson *et al.*, 2007). DBPs are primarily formed by the reaction of natural organic matter (NOM) and halides (bromide and iodide) with disinfectants such as chlorine and monochloramine. Some DBPs may be associated with detrimental health effects after long-term exposure at sufficiently high concentrations, therefore their concentrations must be kept to a minimum in drinking water (Bove *et al.*, 2002; Kargalioglu *et al.*, 2002). Chlorination is widely used globally in drinking water due to its excellent efficiency in removing pathogenic organisms, thereby protecting the public from waterborne disease. Each disinfection method produces its own unique suite of DBPs, including both regulated and unregulated compounds, however, DBPs from chlorination have received the most attention so far, with about 50% of the total organic halogen (TOX) in chlorinated water having been identified to date (Richardson *et al.*, 2007).

DBP minimisation strategies can be broadly categorised into three groups (Zwiener, 2006): controlling disinfection parameters to minimise DBP formation (eg, disinfectant dose/disinfection method management); removing DBPs after disinfection (eg, air-stripping to reduce trihalomethanes (THMs)); and removing DBP precursors from the source water prior to disinfection. Precursor removal prior to disinfection is an attractive strategy for DBP minimisation, in that it can potentially minimise all DBPs (known and unknown) rather than targeting a known, measureable suite of DBPs, which may omit toxicologically important compounds. Iodinated and brominated DBPs are often of greater concern from a public health perspective than their chlorinated analogues, and these may be formed in disinfected waters in which these halides are present (eg, salinity-impacted waters (Nielson, 2003)) (Agus, Voutchkov and Sedlak, 2009). Bromide is oxidised to hypobromous acid by hypochlorate, and this reacts with NOM to form Br-DBPs. This reaction is favoured under conditions of low NOM (high Br⁻/DOC ratio), since NOM and bromide compete for oxidant. That is, removing NOM allows bromide to compete more effectively for oxidant (chlorine), thereby allowing formation of the reactive form of bromide (hypobromous acid), and ultimately Br-DBPs. Therefore, organic DBP precursor removal techniques applied during water treatment have the potential to form increased concentrations of Br-DBPs. So, although traditional DBP precursor removal techniques focus on the removal of organic matter from water sources, the removal of halide precursors is also important. DBP precursor removal strategies can generally be classified as; membrane techniques, electrochemical techniques, and adsorptive techniques (Watson, 2012). Membrane techniques such as reverse osmosis are very efficient in removing both organic and inorganic DBP precursors, however, this technology is relatively expensive compared to, for example, adsorptive techniques such as activated carbon filters.

NOM consists of a complex mixture of many different organic compounds, primarily arising from plant detritus, and comprised of humic acids, fulvic acids, amino acids and carbohydrates (Chow *et al.*, 2008). Coagulation, which is widely used in water treatment, is able to remove the high molecular weight, high hydrophobicity NOM compounds efficiently, however it is not optimal for removing low molecular weight, high hydrophilicity NOM compounds, which may be important for DBP formation. Adsorptive techniques such as activated carbons and ion exchange resins can allow greater NOM removals to be achieved than what could be achieved with coagulation alone, however many of these techniques do not adsorb halides, particularly in the presence of competing ions such as bicarbonate (Kristiana *et al.*, 2011).

The significance of this study is in the development of an understanding of how the interrelationships between NOM and halide DBP precursors impact on DBPs formation and speciation upon disinfection, and how selected adsorptive DBP precursor removal strategies can be best utilised in the control of DBP formation. The pre-treatment strategies studied enabled lower concentrations of disinfectants to be used during water treatment, as well as lowering the formation of many DBP species.

Methodology

Chemicals

All DBP standards were purchased from Accustandard, excluding the iodinated trihalomethanes (THMs), which were purchased from Orchid Cellmark (Canada). Sodium sulfate (anhydrous) was purchased from Mallinckrodt chemicals, and phosphate buffers (Na_2HPO_4 and KH_2PO_4) were purchased from Acros Organics. Commercial DPD test kits (HACH) were used for the analysis of free and total chlorine. All other chemicals were purchased from Sigma-Aldrich.

Synthetic Water Matrix

Matrices of 18 synthetic waters of differing characteristics were utilised for all DBP formation potential and DBP precursor removal experiments (Table 1). The experimental matrix was developed based on a face-centred central composite design, with three variables, namely, NOM concentration, halide precursor concentration (combined Br^- and I^-) and alkalinity. Synthetic waters were made by dosing 1 L samples of laboratory purified water with sodium bromide standard (to give final concentrations of 100 $\mu\text{g/L}$, 450 $\mu\text{g/L}$, or 800 $\mu\text{g/L}$ as Br^-), potassium iodide standard (to give final concentrations of 4 $\mu\text{g/L}$, 18 $\mu\text{g/L}$, or 32 $\mu\text{g/L}$ as I^-) and NOM isolate (Suwannee river NOM) to give a final dissolved organic carbon (DOC) concentration of approximately 3 mg/L, 7.5 mg/L, or 12 mg/L. Alkalinity concentrations were 38 mg/L, 138 mg/L, or 238 mg/L as CaCO_3 . Sodium chloride was added to give a Br^-/Cl^- ratio of 1:300 by weight, to mimic a natural water (Magazinovic *et al.*, 2004). Calcium sulfate and magnesium sulfate were also added, in proportion to alkalinity, as described (Eaton, 2005). After making the waters, the pH was adjusted to 7 with dilute HCl. Each 1 L sample was stored in an amber glass bottle until use (within 24 hrs).

Table 1. Concentrations of components of the synthetic water matrix are shown.

Sample Number	Alkalinity (mg/L NaHCO_3)	$\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$ (mg/L)	MgSO_4 (mg/L)	Br^- ($\mu\text{g/L}$)	I^- ($\mu\text{g/L}$)	Cl^- (mg/L)	TOC (mg/L)
1	32	20	20	100	4	30	3
2	32	20	20	100	4	30	12
3	32	20	20	800	32	240	3
4	32	20	20	800	32	240	12
5	200	125	125	100	4	30	3
6	200	125	125	100	4	30	12
7	200	125	125	800	32	240	3
8	200	125	125	800	32	240	12
9	116	72.5	72.5	450	18	135	3
10	116	72.5	72.5	450	18	135	12
11	116	72.5	72.5	100	4	30	7.5
12	116	72.5	72.5	800	32	240	7.5
13	32	20	20	450	18	135	7.5
14	200	125	125	450	18	135	7.5
15	116	72.5	72.5	450	18	135	7.5
16	116	72.5	72.5	450	18	135	7.5
17	116	72.5	72.5	450	18	135	7.5
BLANK	116	72.5	72.5	0	0	135	0

DBP Formation Potential Tests

DBP formation potential tests were performed on 225 mL samples buffered to pH 7 with phosphate buffer (20 mM). Samples were individually dosed with sodium hypochlorite equal to the chlorine demand (calculated according to Method 5710, Standard Methods for the Examination of Water and Wastewater (Eaton, 2005)), plus 2 mg/L chlorine (as Cl_2), to ensure a final residual in the range 2 ± 1.5 mg/L after the 72 hrs contact time. Each formation potential test was then allowed to react in headspace free amber glass bottles at constant temperature (25°C) for 72 hrs. After this time, chlorine residual was measured using the *N,N*-diethyl-p-phenylenediamine (DPD) method (Eaton, 2005) then quenched with ascorbic acid and stored, headspace free, at 4°C until extraction (within 24 hrs).

DBP Analysis by Gas Chromatography with Electron Capture Detection (GC-ECD)

Samples were prepared for DBP analysis by liquid-liquid salted extraction, and analysed by GC with ECD (Krasner *et al.*, 2006). All samples were extracted in duplicate. The sample pH was adjusted to pH 3.5 with 0.2 N sulfuric acid then dosed with 3 mL of methyl *tert*-butyl ether (MtBE) containing 200 µg/L of 1,2-dibromopropane as an internal standard. Approximately 10 g of pre-baked sodium sulfate was then added, and the samples were shaken for 1 minute and left to settle for at least 5 min. Aliquots of the MtBE layer were then transferred into amber glass vials for analysis. The room temperature was maintained at 21-24°C, since at higher temperatures low recoveries of volatile DBPs were experienced. The analysis was carried out on an Agilent 7890A GC-ECD, and the chromatographic separation was performed on both a DB-5 Agilent column (30 m length $\times$ 0.25 mm inner diameter $\times$ 1.0 µm film thickness) and on a DB-1 Agilent column (30 m length $\times$ 0.25 mm inner diameter $\times$ 1.0 µm film thickness). Pulsed splitless injection was used at a temperature of 200°C. The temperature program was as follows: 40°C for 25 min, ramp to 145°C at 5°C/min and hold for 2 min and then ramp to 260°C at 20°C/min and hold for 10 min. The temperature of the ECD detector was set at 290°C. The relative percent differences of the analysis of duplicate samples were in all instances less than 10%. The reporting limit for all DBPs was 0.1 µg/L and the recovery for all analytes was between 70% and 130%. The DBPs analysed for were: trichloromethane (TCM), bromodichloromethane (BDCM), dibromochloromethane (DBCM) and bromoform (TBM), dichloroacetonitrile (DCAN), trichloroacetonitrile (TCAN), bromochloroacetonitrile (BCAN), dibromoacetonitrile (DBAN), 1,1-dichloropropanone (1,1DCP), 1,1,1-trichloropropanone (1,1,1TCP), trichloronitromethane (TCNM), chloral hydrate (CH), dichloroiodomethane (DCIM), chlorodiiodomethane (CDIM), bromochloroiodomethane (BCIM), dibromoiodomethane (DBIM) and bromodiiodomethane (BDIM).

Other Analyses (Bromide, DOC, UV₂₅₄)

Bromide analysis was conducted using a Dionex ICS-5000 ion chromatography system, fitted with an IonPac AS19 analytical column (4 mm internal diameter $\times$ 250 mm length). Injections of 50 µL enabled bromide detection from 20 µg/L when running the instrument in 1-dimensional mode. The gradient employed was as follows: the first 10 minutes held KOH concentration at 10 mM, then a gradient was applied which ramped to 58 mM KOH over 30 minutes. This concentration was held for 5 minutes before returning the KOH concentration to 10 mM over 3 minutes.

DOC concentration was analysed on a Shimadzu TOC-V_{CSH} TOC analyser using a high temperature catalytic oxidation method (Standard Method 5310A) (Eaton, 2005). Samples were filtered through 0.45 µm syringe filters prior to analysis and a blank (filtered milliQ) was also included in the analysis to determine the background DOC of the system. Certified reference materials of known DOC concentration were included in each analytical run, and the concentrations calculated were always within 10% of the reported value.

UV₂₅₄ was measured using a Bio-Rad SmartSpec Plus spectrophotometer. All samples were filtered through 0.45 µm syringe filters prior to analysis.

Enhanced Coagulation (EC)

Bench-scale enhanced coagulation jar tests were performed in accordance to US EPA's Enhanced Coagulation and Enhanced Precipitative Softening Guidance Manual (USEPA, 1999), using a Platypus Jar Tester, equipped with four overhead stirrers and 1 L jars. The synthetic waters were treated using individual coagulant doses optimised to give greatest NOM removal and a final pH in the range 5.5 – 6.5 for each sample (from 30 to 120 mg/L Alum (as $\text{Al}_2(\text{SO}_4)_3 \cdot 18\text{H}_2\text{O}$) depending on the sample alkalinity and starting NOM concentration). Following the addition of the coagulant, the samples were subjected to rapid mixing for 1.5 min at 200 rpm, flocculation for 13.5 min at 30 rpm, then settled for 60 min before vacuum filtering through Whatman No. 1 filter papers to simulate sand filtration at the WTP (Hudson, 1981).

Secondary Treatment

Powdered Activated Carbon (PAC) and MIEX Treatment

After EC, samples were dosed with either 60 mg/L of Norit W35 PAC, or 10 ml/L regenerated, settled MIEX resin, then jar tested for 30 min at 100 rpm, settled for 5 minutes, then vacuum filtered through Whatman No. 1 filter papers.

Granular Activated Carbon (GAC) and Silver-Impregnated Activated Carbon (SIAC) Treatment

After EC, samples were passed through either Norit 1240 GAC or Norit 1840, 0.1% Ag SIAC columns (treated sample approximately 120 bed volumes). The activated carbons were washed with purified water prior to loading into activated carbon columns. Column aspect ratio was 8.1.

Results

NOM Removal

EC was found to precipitate approximately 56% of the organic carbon from the matrix of synthetic waters. High removals were achieved in high NOM, high alkalinity waters (average $59 \pm 5\%$) and lower removals were achieved in low alkalinity, low NOM waters (average 36%). PAC removed an average of $20 \pm 6\%$ of the DOC remaining after EC, with highest removals occurring in high NOM waters (Table 2). MIEX® resin adsorbed only approximately 13% of the DOC remaining after EC, however DOC adsorption was highly dependent on both DOC concentration and alkalinity, with DOC removals of approximately 36% under low alkalinity, high NOM conditions. This reflects competition for binding sites on MIEX® resin between bicarbonate and anionic NOM. Activated carbon beds (GAC and SIAC) both exhibited additional DOC removal beyond what was achieved by EC, with average removals of $25 \pm 10\%$ by GAC and $44 \pm 12\%$ by SIAC.

Halide Removal

EC and EC combined with PAC had no capacity to adsorb bromide (Table 2). GAC and MIEX®, however, had a moderate affinity for bromide ions. MIEX® was able to achieve approximately 49% bromide removal under conditions of low alkalinity (ie, 38 mg/L), but this was reduced to approximately 10% under high alkalinity conditions (ie, 238 mg/L). This reflects competition for binding sites on MIEX® between bicarbonate and bromide. Bromide adsorption by GAC was less affected by the presence of competing ions, with an average of $29 \pm 11\%$ bromide adsorption capacity under low alkalinity conditions and $21 \pm 11\%$ in high alkalinity waters. SIAC exhibited excellent bromide removal capacity irrespective of alkalinity or starting NOM and bromide concentration. Waters as high as 800 µg/L in bromide had bromide concentration reduced to below detection limit (10 µg/L).

Table 2. The average bromide and NOM (as DOC) removal for EC, four secondary treatments, and combined EC/secondary treatment is summarised.

EC Treatment	Average Br ⁻ Removal (%)	Average DOC Removal (%)	Secondary Treatment	Average Br ⁻ Removal (%)	Average DOC Removal (%)	Combined EC/Secondary Treatment	Total Average Br ⁻ Removal (%)	Total Average DOC Removal (%)
Alum Coagulation	0	56	PAC (Norit W35)	0	19	EC/PAC (Norit W35)	0	75
			GAC (Norit 1240)	21	25	EC/GAC (Norit 1240)	21	82
			MIEX resin	32	13	EC/MIEX resin	32	66
			SIAC, 0.1% Ag, Norit 1840	99	44	EC/SIAC, 0.1% Ag, Norit 1840	99	90

DBP Formation

The change in concentration of the individual DBPs studied throughout each treatment process for the centrepont samples (medium alkalinity, starting halide concentration, and starting NOM concentration) of the experimental matrix is shown (Figures 1-3). All treatment strategies studied led to increases in concentrations of TBM and DBAN, except EC combined with SIAC, since this treatment was the most efficient for concurrent DOC and bromide removal (Figures 1 and 2). Although GAC and MIEX® had some bromide adsorbing ability, this was not sufficient

to stop a significant increase in the Br⁻/DOC ratio upon treatment, thereby increasing the production of Br-DBPs. DBCM was also increased by some treatment methods, with GAC and SIAC being the only treatment strategies that were able to lower the formation of this DBP.

The formation of chlorinated DBPs, however, was very effectively mitigated by all of the treatment methods studied (Figures 1-3), with approximately 93% less TCM being formed after EC, and further reductions in formation after each of the secondary treatments. This suggests that the combined treatment methods would be excellent for DBP management in waters in which bromide concentrations are low, not only forming lower levels of DBPs, but requiring much lower doses of disinfectant. On the other hand, in the presence of bromide (eg, salinity-impacted waters) all treatments except EC combined with SIAC will preferentially form Br-DBPs, which is undesirable.

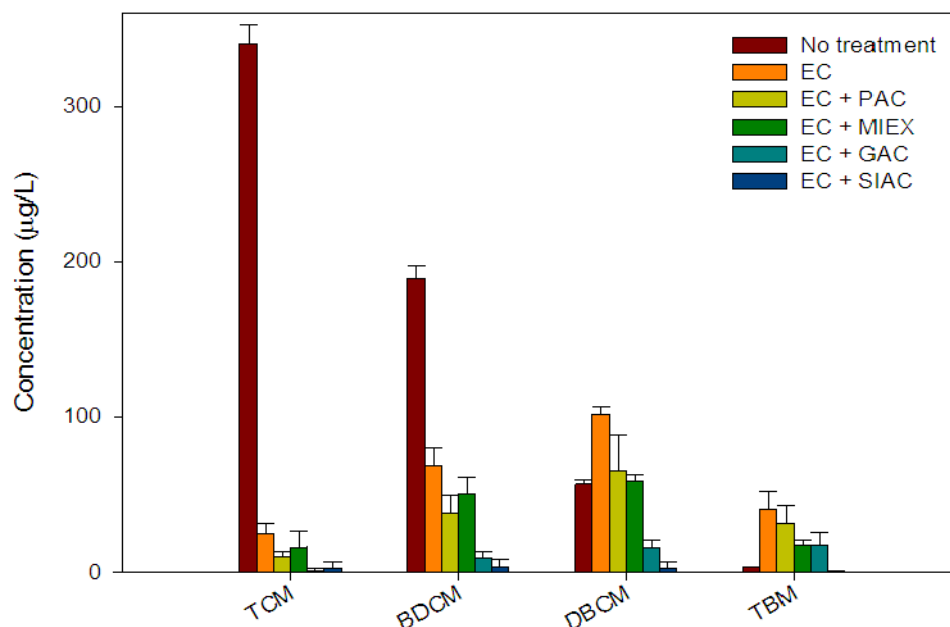


Figure 1. The speciation of individual THMs after each treatment is shown, and compared to THMs formed without any DBP precursor removal treatment. Average of triplicate analyses of centrepont samples for each treatment method is shown. Starting water quality parameters were as follows: bromide concentration 450 µg/L, DOC 6 mg/L, alkalinity 83 mg/L as CaCO₃ mg/L.

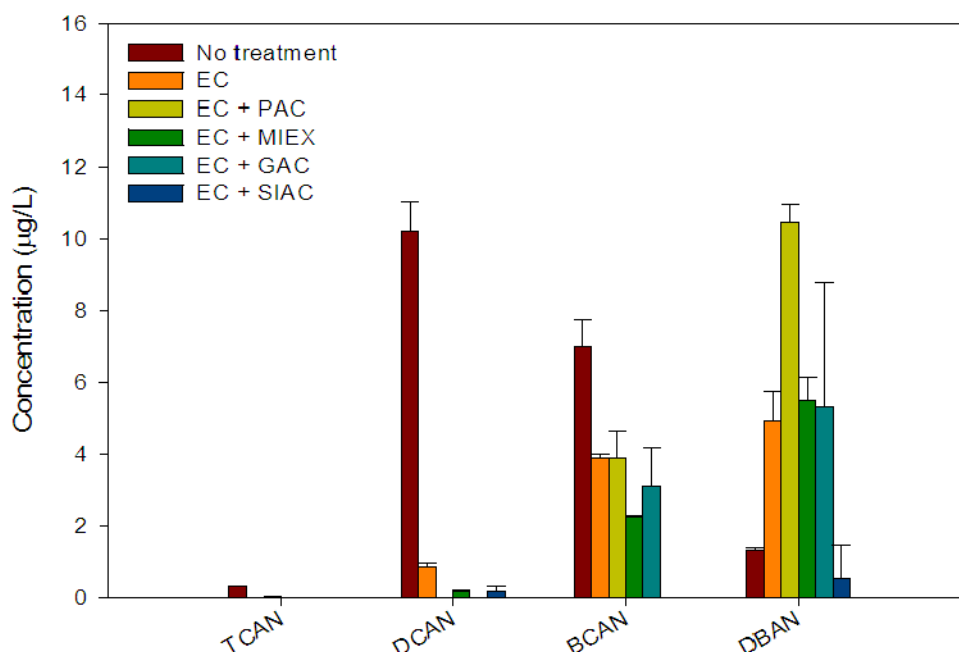


Figure 2. The speciation of individual HANs after each treatment is shown, and compared to HANs formed without any DBP precursor removal treatment. Average of triplicate analyses of centrepont samples for each treatment method is shown. Starting water quality parameters were as follows: bromide concentration 450 µg/L, DOC 6 mg/L, alkalinity 83 mg/L as CaCO₃ mg/L.

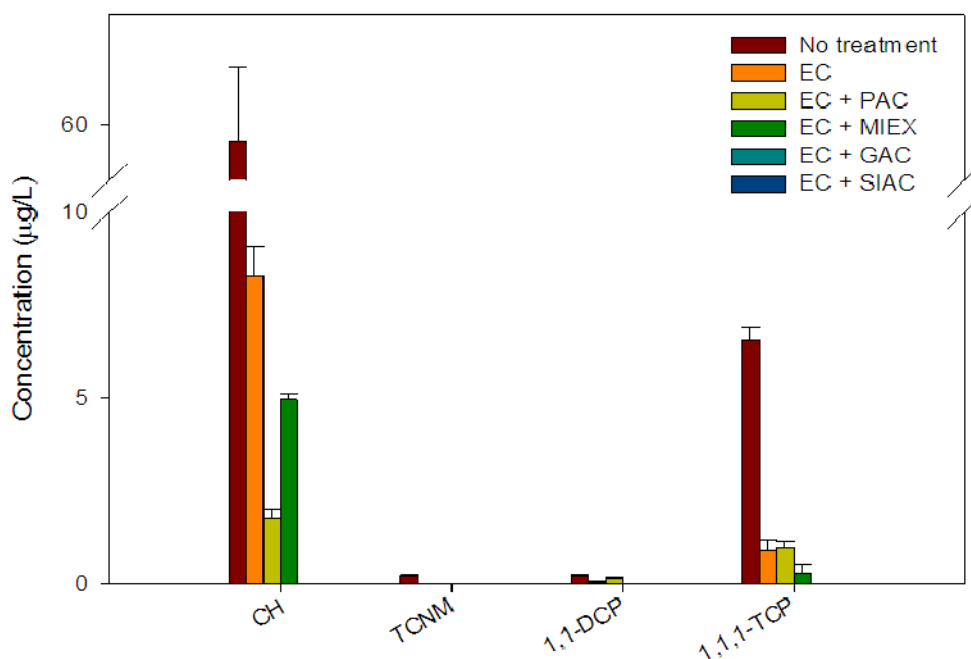


Figure 3. The speciation of specific chlorinated DBPs after each treatment is shown, and compared to DBPs formed without any DBP precursor removal treatment. Average of triplicate analyses of centrepont samples for each treatment method is shown. Starting water quality parameters were as follows: bromide concentration 450 µg/L, DOC 6 mg/L, alkalinity 83 mg/L as CaCO₃ mg/L.

Total THMs and tHAnS are also shown (Figure 4), and illustrate the overall decrease in these parameters with EC, and further decrease with secondary treatment in most cases. It is important to note that a parameter such as tTHM does not illustrate the changes in THM speciation that occur during treatment, and the concurrent change in toxicological profile of the mixture.

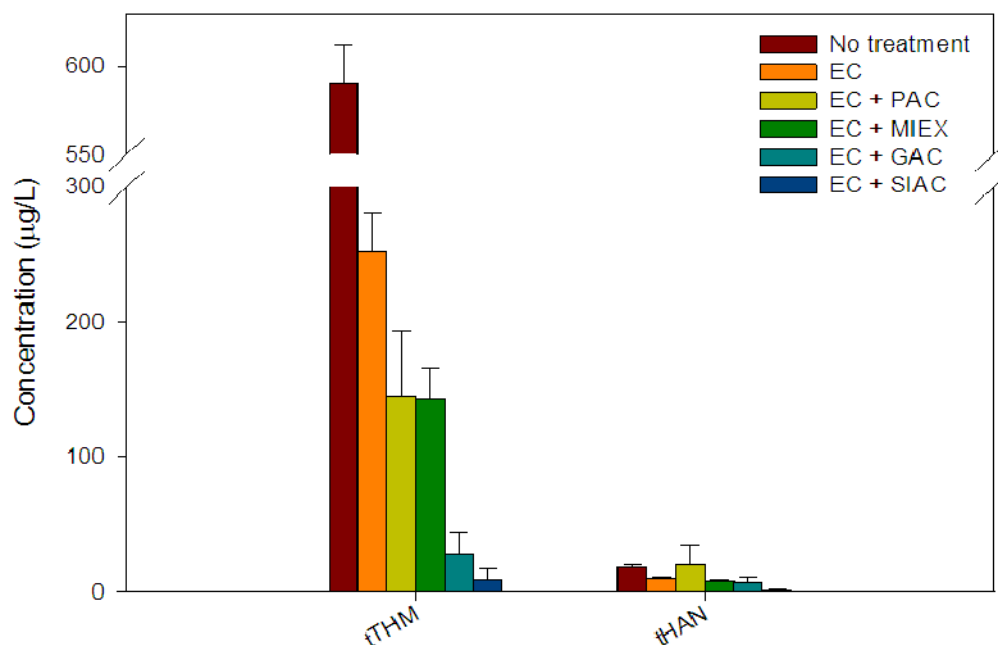


Figure 4. Total THMs and total HANs after each treatment is shown, and compared to concentrations formed without any DBP precursor removal treatment. Average of triplicate analyses of centrepont samples for each treatment method is shown. Starting water quality parameters were as follows: bromide concentration 450 µg/L, DOC 6 mg/L, alkalinity 83 mg/L as CaCO₃ mg/L.

Conclusions

- The combination of EC with SIAC beds was excellent at removing both organic and inorganic DBP precursors from waters of variable alkalinity, therefore, this is the optimal method of those studied for DBP precursor removal from salinity-impacted waters.
- All secondary treatments improved removal of organic precursors, beyond EC alone, thus low chlorinated DBP formation was achieved in all cases.
- All secondary treatments except SIAC caused an increase in the formation of specific, highly brominated DBPs due to the increased Br⁻/DOC ratio created by the treatment.
- MIEX® and GAC had some halide adsorption capacity, but it was insufficient to stop increased formation of some Br-DBPs compared to 'no treatment' samples.
- Brominated THMs may indicate formation of other brominated species.

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Bioanalytical Assessment of the Formation Potential of Disinfection By-Products during Drinking Water Treatment

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Summary

Disinfection by-products (DBP) formed from natural organic matter and disinfectants like chlorine and chloramine may cause adverse health effects. For monitoring purposes and to complement chemical analysis, we propose to use *in vitro* bioassays. *In vitro* bioassays allow us to assess the hazard potential of DBPs early in the chain of cellular events, when the DBPs react with their molecular target(s) and activate stress response and defence mechanisms. A suite of reactive endpoints including genotoxicity, protein damage and oxidative stress were evaluated on water samples disinfected with chlorine and chloramine. Cytotoxicity increased after disinfection. Reactive mechanisms were mostly below the limit of detection in the source water taken from different drinking water plants and showed significant effects after chlorination and chloramination in the laboratory. Especially, source water with high organic matter content induced DNA damage repair, glutathione conjugation and the Nrf2 oxidative stress response pathway.

Keywords

In vitro bioassay, cytotoxicity, reactive modes of toxic action, genotoxicity, oxidative stress.

Introduction

Disinfection of drinking water is the most successful measure to reduce water-borne diseases and protect health. However, disinfection by-products formed from natural organic matter and disinfectants like chlorine and chloramine may cause hazardous effects (Richardson *et al.*, 2007). For monitoring purposes, *in vivo* toxicology is too costly and too slow. Also *in vivo* effects manifest themselves only after the body has succumbed despite repair and compensation mechanisms. A more precautionary approach would be to assess the hazard potential of DBPs early in the chain of cellular events, when the DBP reacts with its molecular target and activates stress response and defence mechanisms. *In vitro* cell-based bioassays are adequate monitoring tools for that purpose. They give an account of the mixture effects and, if distinct modes of toxic action are targeted, also an idea about the characteristics of the toxic chemicals in water samples. Most *in vitro* bioassays on individual DBPs and disinfected water samples have targeted genotoxicity and mutagenicity, which is evident given the link to cancer development. DBPs are primarily formed by oxidative processes and many DBPs contain reactive moieties. Since many DBPs are reactive chemicals, we cannot expect that their reactive toxicity is limited to DNA damage but rather that a suite of reactive mechanisms is relevant. This paper examines the reactive mechanisms, level of cytotoxicity and stress response pathways in the source water taken from different drinking water plants and following chlorination and chloramination in the laboratory.

Methods

Source water from four drinking water treatment plants was sampled after the coagulation stage in acid washed amber bottles. The samples were protected from light and shipped cool to the Advanced Water Management Centre (AWMC) laboratory at the University of Queensland (UQ). After filtering through a 11µm filter, lab based disinfection experiments started within 24 h. The chlorine dose was chosen as chlorine demand plus 2-3 mg/L Cl₂ and contact time for reaction was three days at 23°C. After chlorination with hypochlorous acid HOCl or chloramination with monochloramine (MCA), the organic compounds in disinfected water were extracted by solid phase extraction (Macova *et al.*, 2010) and submitted to the bioassays listed in Table 1 run according to the protocols in the cited literature references.

Table 1. Applied bioassays.

Mode of Action	Assay	Targeted Chemicals (reference for method)
Cytotoxicity	Bioluminescence inhibition assay with <i>Vibrio fischeri</i>	All chemicals (Macova <i>et al.</i> , 2011)
Cytotoxicity	AREc32 cell viability w/ MTS	All chemicals (publication in preparation)
Genotoxicity	<i>umuC</i> (genotox)	Aromatic amines, polycyclic aromatic hydrocarbons (PAH), hard electrophiles (eg, methyl methanesulfonate (MMS)) (Macova <i>et al.</i> , 2011)
Protein damage	<i>E.coli</i> GSH±	Soft electrophiles (eg, Seanine) (Tang <i>et al.</i> , 2012)
Oxidative stress	Induction of Nrf2 in AREc32	Quinones, reactive oxygen species (Wang <i>et al.</i> , 2006; Escher <i>et al.</i> , 2012)

Results

Of the two cytotoxicity assays only the bacterial bioluminescence inhibition assay with *Vibrio fischeri* showed a dose-dependent response for the various water samples tested (Figure 1), while the cell line AREc32, which was derived from a mammalian breast cancer cell line, was insensitive to the water sample extracts. Figure 1 expresses the cytotoxicity as baseline toxic equivalent concentrations (TEQ) as defined in (Escher *et al.*, 2008), with a large bar meaning a higher effect. The effect in *Vibrio fischeri* was higher for chlorination than for chloramination and effects were higher if the source water had a higher organic matter content (expressed as dissolved organic carbon, DOC).

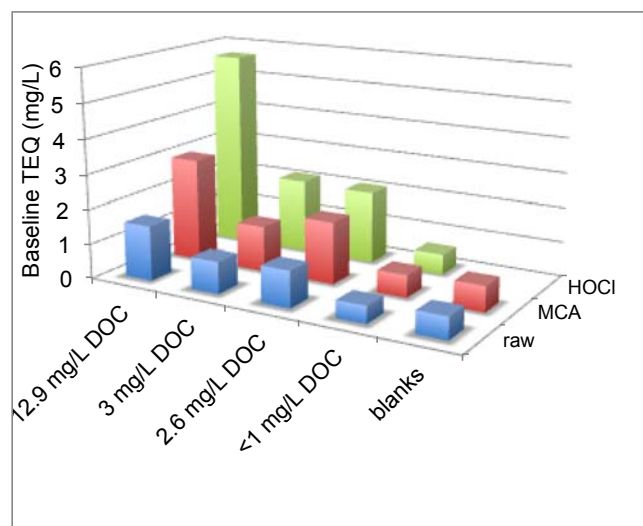


Figure 1. Toxic equivalent concentrations of the water samples in the bioluminescence inhibition assay with *Vibrio fischeri*. DOC refers to dissolved organic carbon.

The umuC assay, which is indicative for direct genotoxicity and can also be run in the presence of a liver enzyme cocktail to assess genotoxicity after metabolic activation, only showed a response for the source water with the highest DOC content (Figure 2). The results in Figure 2 are again expressed as toxic equivalent concentrations. 4-Nitroquinoline-N-oxide (4-NQO) served as a reference compound for direct genotoxicity and 2-aminoanthracene (2AA) as a reference compound for genotoxicity after metabolic activation with a liver enzyme cocktail. Chlorination induced more direct genotoxicity, expressed as 4-nitroquinoline equivalent concentrations, while chloramination produced DBPs that were only genotoxic after metabolic activation.

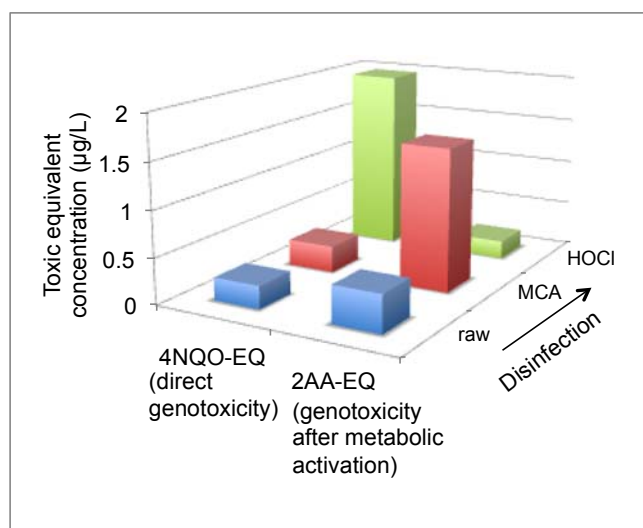


Figure 2. Toxic equivalent concentrations of the water samples of the source water that contains 12.9 mg/L DOC. The umuC assay is indicative of genotoxicity.

In the *E. coli* assay, indicative of conjugation with glutathione and indirectly indicative of protein damage, only chlorinated samples showed an effect. Oxidative stress was induced by all source water types with a DOC content higher than 1 mg/L (Figure 3). Again higher effects, expressed as t-butylhydroquinone equivalents (tBHQ-EQ), were observed for chlorination than for chloramination (Figure 3).

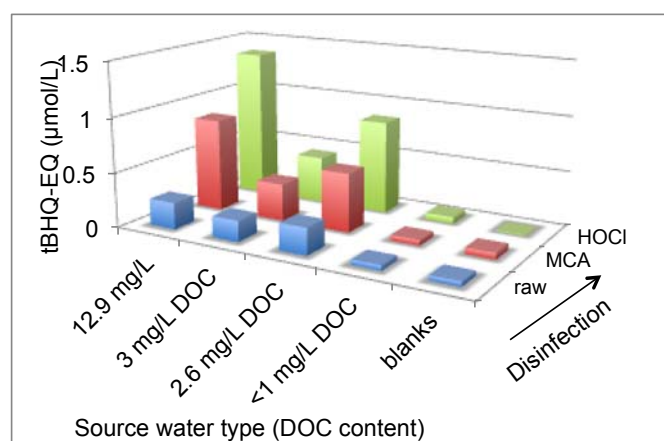


Figure 3. Toxic equivalent concentrations (t-butyl hydroquinone equivalents, tBHQ-EQ) of the water samples in the AREc32 assay indicative for response to oxidative stress.

Overall, the results of the bioassays were in qualitative agreement with chemical analysis, with higher DBP production of the source water with higher DOC despite the fact that, at this stage, the bioassays are limited to non-volatile DBPs.

Conclusions

In this study we assessed several reactive endpoints on water samples disinfected with chlorine and chloramine, among them the reactive toxicity towards proteins and peptides and the activation of the oxidative stress response. All reactive modes of action were induced after disinfection of source water, more in chlorinated than in chloraminated water and the effect was higher if the source water had a higher organic matter content (Table 2). This is a preliminary study, which was limited to non-volatile DBPs but results are encouraging and we plan to expand the battery to more reactive mechanisms and adapt the bioassays for dosing of volatile compounds.

Table 2. Summary of results.

Mode of Action	Assay	Observation
Baseline toxicity	Bioluminescence inhibition assay	All samples showed an effect. Treated water from source water with the highest content of dissolved organic carbon had the highest effects.
General cytotoxicity	AREc32 MTS	All samples below detection limit.
Genotoxicity	<i>umuC</i> (genotox)	Only source water with the highest content of dissolved organic carbon showed effects after disinfection. Higher effects after chlorination than chloramination.
Direct reactive toxicity towards proteins	<i>E.coli</i> GSH±	Only source water with the highest content of dissolved organic carbon showed (strong) effects only after chlorination.
Oxidative stress	Induction of Nrf2 in AREc32	Very low levels across all samples, no effects in disinfected blank and disinfected source water from a desalination plant.

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Cost of Pollution and Options Evaluation – Case Study of Moreton Bay Regional Council Total Water Cycle Management Plan

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Summary

This paper provides a case study of the use of pollutant costs to extend cost effectiveness analysis for Total Water Cycle Management Planning. This approach can enhance option evaluation by including both water quantity and quality measures. Dollar values for pollution were developed using Marginal Abatement Cost Curves and pollution reduction targets. The costs and benefit of pollution mitigation were compared for achieving a ‘no worsening’ of the waterways over the planning period. The pollution cost for Total Suspended Solids, Total Nitrogen, Total Phosphorus and Greenhouse Gas emissions was then added to the capital and operating cost of a particular water supply option. The pollution cost contributed up to 50% of the total cost of some water supply options and changed the relative ranking of options by cost. However, in most cases the pollutant cost was less than 20% of the total cost and pollutants such as greenhouse gas emissions contributed only a couple of percent to the total cost. Final results incorporating uncertainty and sensitivity analysis will be completed over the coming months and will provide guidance on the application of extended cost effectiveness for Total Water Cycle Planning.

Keywords

Total Water Cycle Management Plan, Extended Cost Effectiveness Analysis, Marginal Abatement Cost Curves, Cost Benefit Analysis, pollution, water quality, nutrients, sediments, greenhouse gas emissions.

Introduction

Cost Effectiveness Analysis (CEA) is an established economic method for evaluating the cost of an option to achieve an objective (Pearce *et al.*, 2006). The application of cost effectiveness analysis in assessing water quality interventions in South East Queensland (SEQ) has also recently been reviewed (Alam *et al.*, 2008; Hall, 2012). Cost effectiveness analysis can be used for evaluating both pollution abatement options as well as water supply options. In this case, pollution abatement costs were reviewed to extend the cost effectiveness analysis of water supply options. This method was noted as being suitable for capturing sustainability issues of sub regional Total Water Cycle Management Plans (TWCMP) for water supply conservation and water supply augmentation (Hurikino *et al.*, 2010, p12; Fane *et al.*, 2010, p12). This method also supports National Water Initiative pricing principles for including full cost recovery, including recovery of environmental externalities (DEWHA, 2010). This study considered pollution abatement costs for greenhouse gases, nutrients and sediments to extend the cost effectiveness analysis. Equation 1 expresses the cost effectiveness calculation including pollution costs. Figure 1 illustrates the comparison of extended costs for two projects and shows that the most cost effective option may change when pollution costs are considered.

$$Y = C_p + O_p + \sum_j^m P_j \cdot W_j \quad \text{Equation 1}$$

Where:

Y = extended cost effectiveness

C_p = capital cost of a project

O_p = operating cost of a project over the period of analysis

P_j = pollution emitted by the project

W_j = value of pollution for a defined pollution reduction target

j = first pollutant considered

m = last pollutant considered

Pollutant costs were developed using pollutant targets and Marginal Abatement Cost Curves (Hall, 2012). The case study drew upon pollution costs and quantities calculated for the Total Water Cycle Management Plan for Moreton Bay Regional Council (MBRC) (BMT-WBM, 2012; BMT-WBM, 2010; BMT-WBM, 2011). All costs were expressed in 2011 values using a discount rate of 6%.

Project A

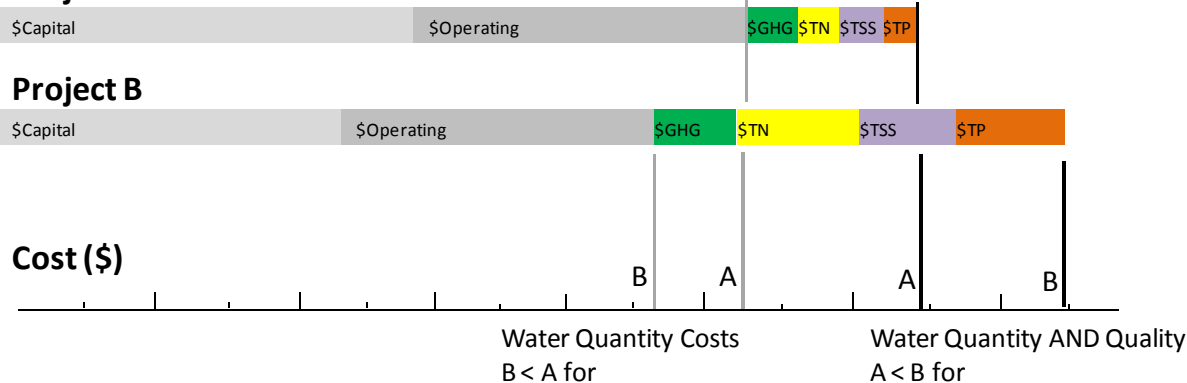


Figure 1. Options evaluation with costs extended for pollution.

Draft Results

Note: The following results are illustrative and have not been fully checked or reviewed. Final results with uncertainty and sensitivity analysis will be available in June 2012.

The pollution reduction objective adopted for the following results was a ‘no worsening’ of pollutant loads to receiving waters over the 20 year planning period. A pollution reduction objective of achieving the Environmental Protection Policy (Water) Water Quality Objectives (WQO) (QG, 2009) was not pursued based upon limitations for existing modelling results which were only calculated for a dry year and gave very low loads to meet WQO concentrations (BMT-WBM, 2011). This meant that the sustainable load *reductions* required over the next 20 years based on the dry year modelling were actually lower than the reductions required to achieve the less stringent ‘no worsening’ objective based on average loads.

Table 1 provides a summary of the load reduction required for a ‘no worsening’ of pollutant loads based upon annual average rates and projected increases (BMT-WBM, 2010). Benefits for pollution abatement were calculated from the willingness to pay for water quality improvement per household per year (Binney, 2010) and the population projection for MBRC. Benefits for achieving ‘no worsening’ were much higher than marginal benefits to achieve EPP (Water) WQO. This might reflect the expected diminishing returns for marginal benefits for pollution abatement (Aldrich, 1996). As noted in *Managing What Matters* this suggests that the community has a strong economic preference for investing to stop decline rather than rehabilitating later on (Binney, 2010). However, this may also be partly explained by Prospect Theory and the psychological effect of valuing losses more than gains (Kahneman, 2011). Kahneman (2011) notes that the willingness to pay depends on the reference point and a loss has a greater value than a gain of the same amount (Kahneman, 2011 p293, 283). The allocation of costs and benefits was based upon a distance-to-target approach using the sustainable loads calculations for waterways (BMT-WBM, 2011). This approach captures the effect and relative importance of current and future pollutant loads on water waterway health by using the sustainable load as the target. The allocation based upon distance to sustainable loads targets was similar to an allocation based upon Water Sensitive Urban Design (WSUD) minimum reductions in pollutant loads for urban storm water (DERM, 2009; Hall, 2012). The benefit allocated to each pollutant to achieve ‘no worsening’ of the waterways was then coupled together with the load reduction required to provide a unit benefit for pollution reduction.

Table 1. Pollution Reduction Targets, Load Reduction and Benefits.

	Total Suspended Solids	Total Nitrogen	Total Phosphorus	Total
Allocation of Costs and Benefits Based on Distance to Target	0.462	0.153	0.384	1
Marginal benefit for achieving ‘no worsening’ (Present Value, Million\$AUD2011)	153	51	127	330
Marginal benefit for achieving sustainable loads targets (Present Value Million\$AUD2011)	64	21	53	138
Load reduction to achieving ‘no worsening’ over analysis period (tonnes)	85,126	2,002	369	-
Marginal benefit for achieving ‘no worsening’ per tonne of abatement (Present Value \$AUD2011/tonne)	1,794	25,327	344,630	-

Figure 2 provides the Marginal Abatement Cost Curve for Total Phosphorus (TP) based upon the Draft TWCMP for Moreton Bay Regional Council (BMT-WBM, 2012). Each abatement option is represented as a box with the cost effectiveness captured by the height and the amount of abatement captured by the width. Some abatement options, namely ‘Recycled Water for Public Open Space irrigation’ had a negative abatement cost which can be interpreted as a cost saving. This cost reflects that the option provides water for less cost than the bulk water supply and the cost saving was allocated to pollution abatement. However, the volume of pollution abated using recycled water was relatively small compared to the load reduction target. Consequently, the weighted average cost of abatement, shown as a red dotted line, reflects the Purified Recycled Water (PRW) and WSUD retrofit abatement costs. The weighted average abatement cost for achieving the load reduction required for a ‘no worsening’ target was approximately \$220 per kilogram TP. This compares to the benefit of pollution abatement of about \$344 per kilogram calculated from the willingness to pay for ‘no worsening’ of water quality.

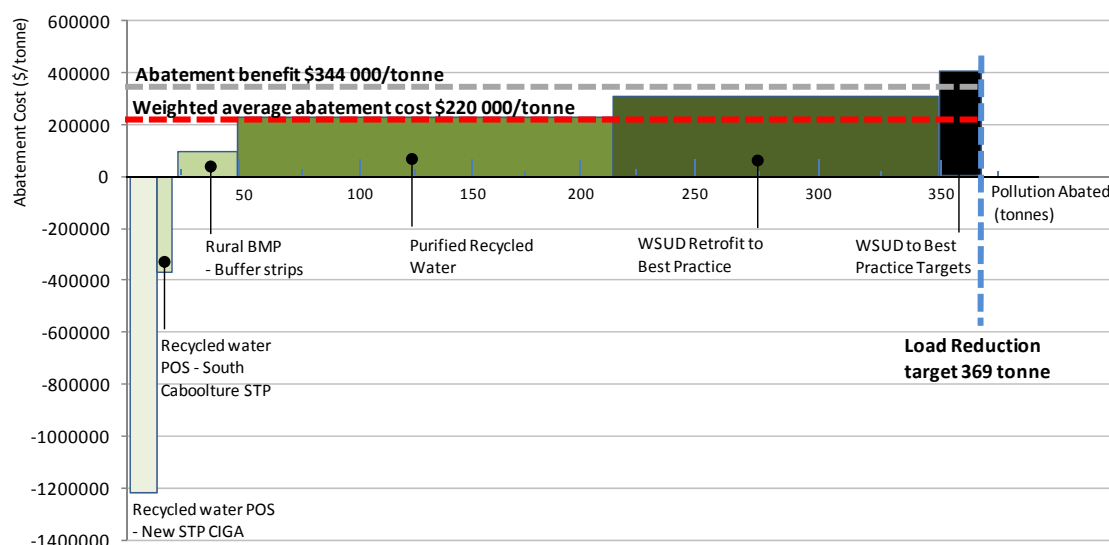


Figure 2. Total Phosphorus Marginal Abatement Cost Curve for Moreton Bay Regional Council using TWCMP data.

In a similar manner, weighted average abatement costs were calculated for Total Suspended Solids (TSS) and Total Nitrogen (TN). Table 2 provides a summary of the weighted average abatement costs for TSS, TN and TP as well as the assumed cost of carbon pollution based upon the Clean Energy Future policy from the Commonwealth Government.

Table 2. Summary of Pollution Costs for Moreton Bay Regional Council TWCMP.

Pollutant	Cost per Tonne (\$/t)
Total Suspended Solids	213
Total Nitrogen	273,000
Total Phosphorus	220,000
Carbon Dioxide	23

Figure 3 presents a comparison of the water supply option capital and operating costs (project costs) with the extended cost effectiveness (project costs and pollution mitigation costs) for a unit of water supply for Caboolture and Caboolture Identified Growth Area CIGA catchment options from the Draft Total Water Cycle Management Plan (note the abbreviation at the beginning of the option name gives the catchment and scenario number eg cab1 refers to Caboolture catchment Scenario 1). There was a relatively large difference in the project cost and extended cost effectiveness for recycled water due to the water pollution abatement. In this case the abatement provided by the solution was a cost saving and the extended cost effectiveness was lower than the project cost. This affects the ranking of some options especially in comparison to grid water supplies. The cost of carbon pollution was a relatively small addition to the project cost and did not affect the relative ranking of options. However, if the marginal supply of bulk water is desalination, then it could be argued that greenhouse gas emission may have a greater effect on the extended cost.

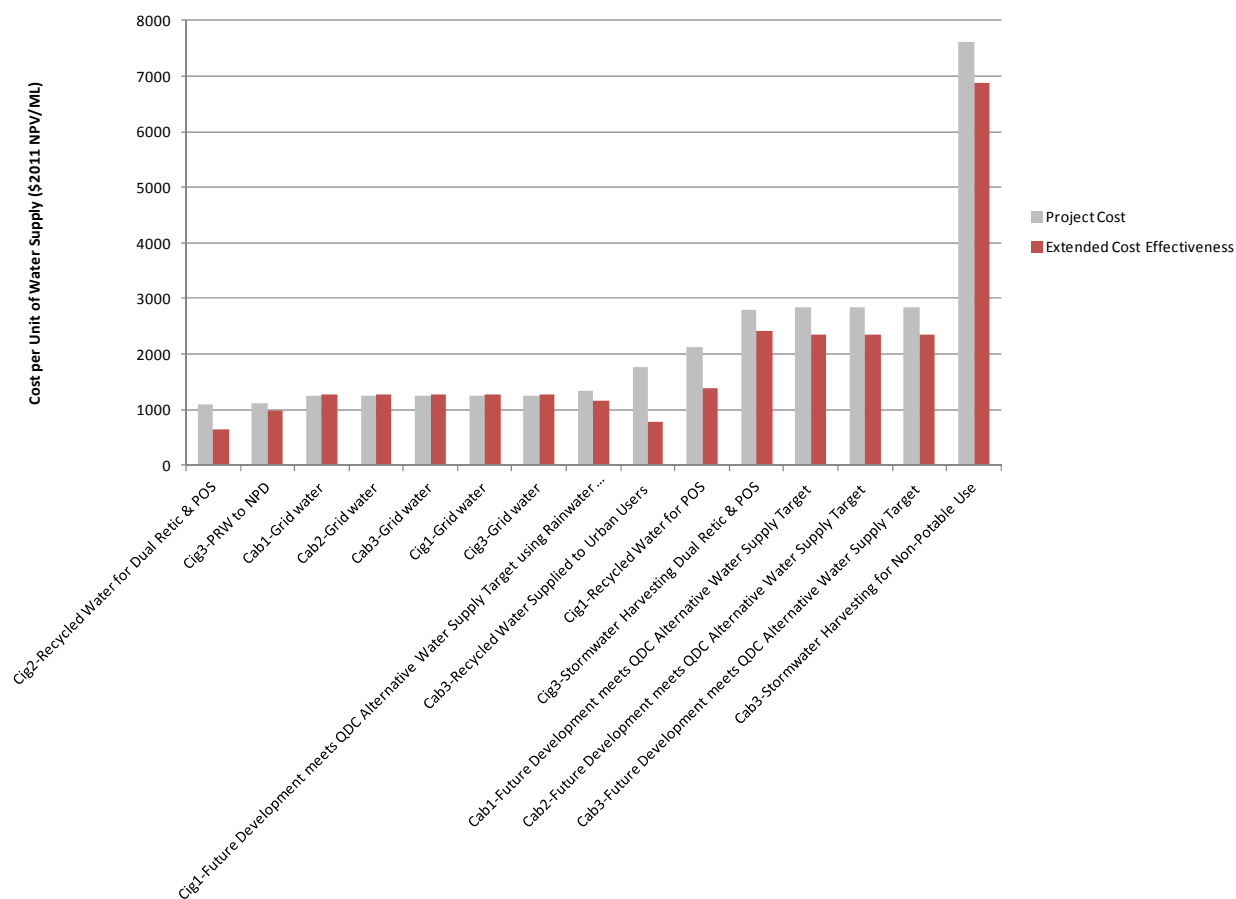


Figure 3. Comparison of Project Cost and Extended Cost Effectiveness for a unit of water supply for Caboolture and CIGA Catchment Options.

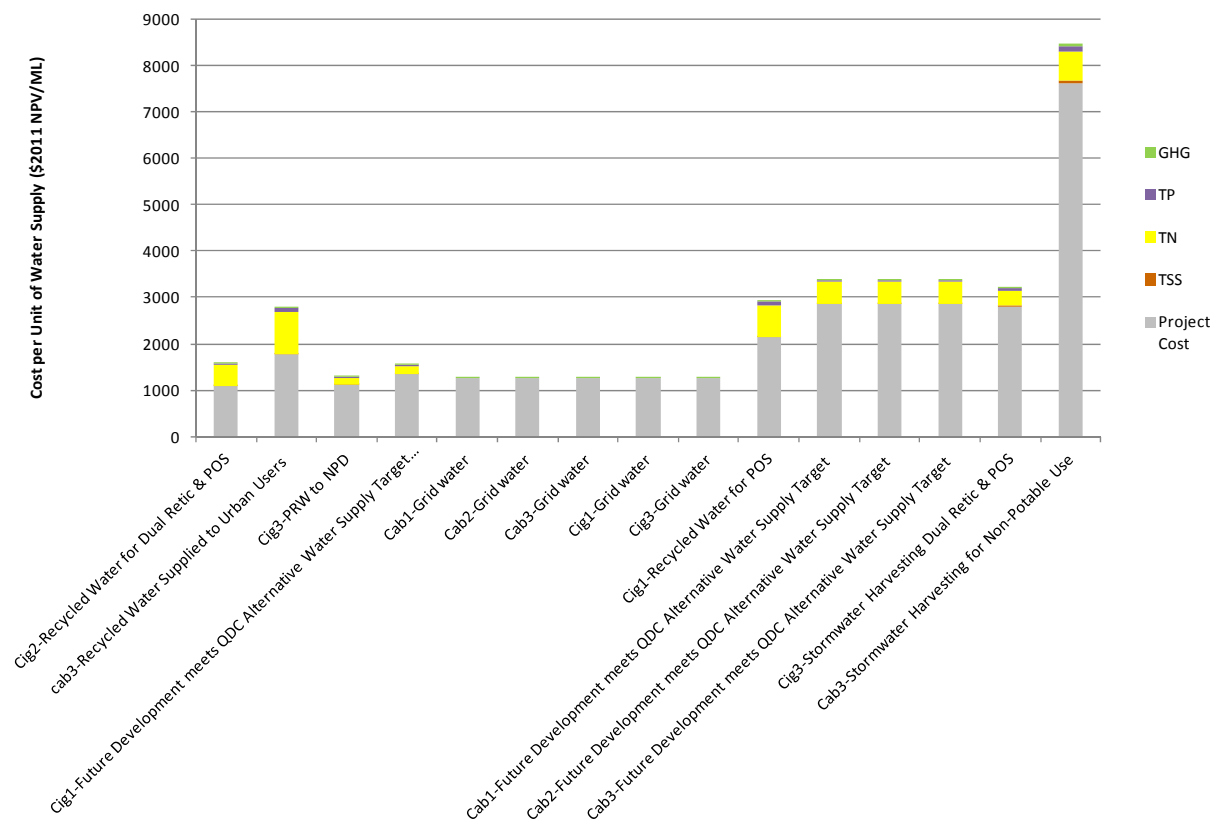


Figure 4. Contribution of Pollutant Costs to the Extended Cost Effectiveness.

Figure 4 illustrates the contribution of pollutants to the extended cost. The pollutant values were expressed as absolute numbers (the total extended cost adds pollution costs and subtracts abatement costs). Total nitrogen abatement appears to have the largest effect of the pollutants upon the extended cost. This reflects the relatively high abatement of nitrogen compared to phosphorus for options such as rainwater tanks and recycled water. This also suggests that the water supply options by themselves are unlikely to achieve the load reduction objectives and additional abatement is required, especially for TSS and TP. The Marginal Abatement Cost Curves provides guidance on the most cost effective means of reducing pollutant loads. This does not affect options evaluation for water supply but will add to the budget for achieving the objectives of the TWCMP. For example, Figure 2 defines the priority abatement options and the amount of implementation required for Total Phosphorus. In this case, the cheapest additional abatement measure is Rural Best Management Practices followed by WSUD to achieve a ‘no worsening’ of phosphorus pollution over the 20 year planning period. These abatement costs need to be added to the budget for achieving the objectives of a TWCMP.

Uncertainty

The uncertainty in the results is yet to be fully analysed. Figure 5 provides results from a review of abatement costs for total phosphorus based upon literature (note the log scale) (Hall, 2012). The benefit per unit of abatement based on the willingness to pay was shown by the grey dotted line. In particular, bioretention costs were in the range of \$300,000 to \$3,500,000 per tonne based upon Water by Design case studies for SEQ. This abatement cost was significantly different to the abatement costs based on the Draft TWCMP. In this case, the WSUD abatement cost was also greater than the benefit for abatement suggesting that it is not a viable abatement option. This difference is also important because the cost for TP assumed in Table 2 is sensitive to the bioretention abatement cost due to its large contribution to the load reduction target. This will have a large effect on the TWCMP budget to achieve a ‘no worsening’ of pollution loads. However, the ranking of options by extended cost effectiveness is likely to be less sensitive because of the relatively small amounts of TP that each water supply option abates.

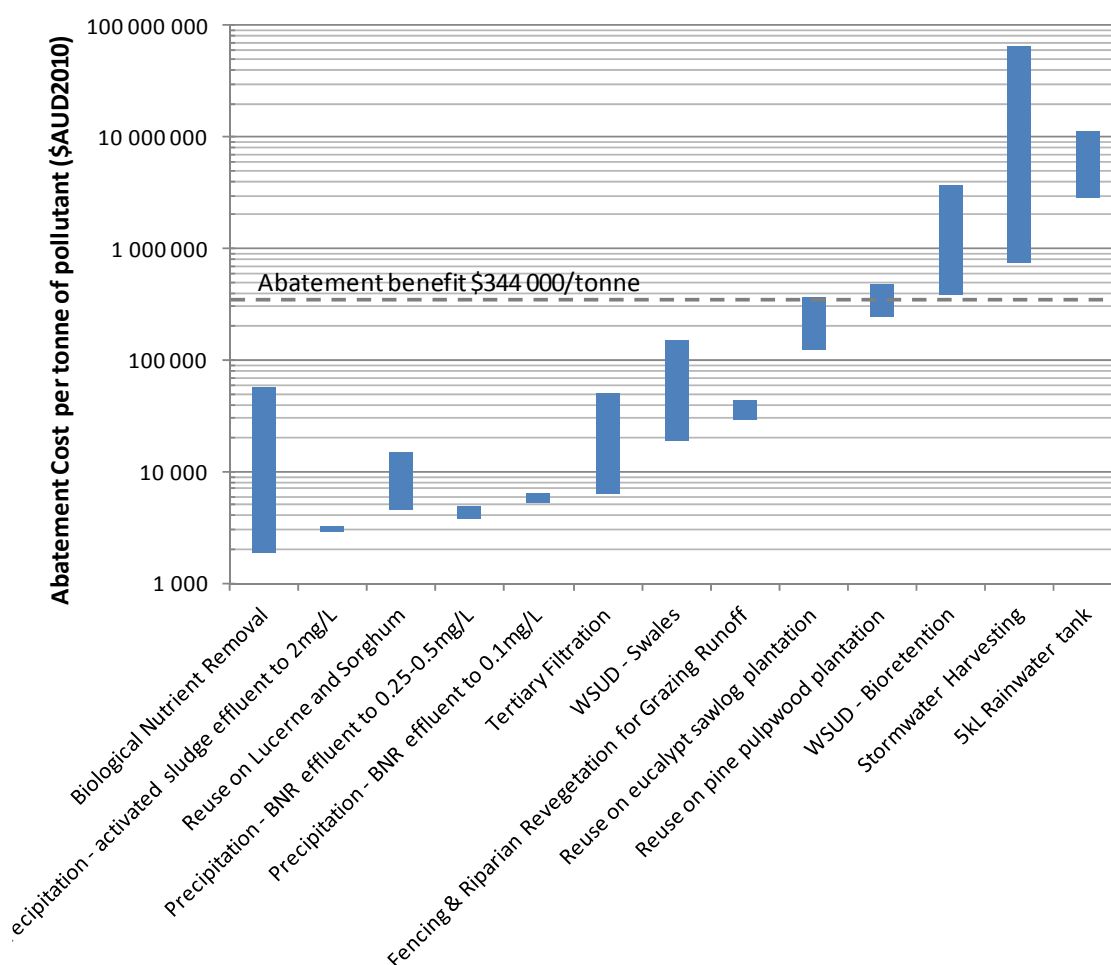


Figure 5. Total Phosphorus Abatement Costs from Literature with Benefit per Unit of Abatement.

Other abatement options are also likely to be sensitive to assumptions about the value of water. The Queensland Water Commission (QWC) bulk water price path for MBRC was assumed for water produced from rainwater tanks and recycled water for urban uses (QWC, <http://www.qwc.qld.gov.au/reform/bulkwaterprices.html>). The value of recycled water for agriculture was based on Schedule 14 Water Charges of the *Water Regulation 2002* (QG, 2011).

Conclusions

Initial results suggest that water quantity and a number of pollution impacts can be captured in a single dollar metric and used for cost effectiveness analysis for Total Water Cycle Management Planning (TWCMP). This simplifies options analysis for water supply and environmental objectives for waterway health and global warming. Marginal Abatement Cost Curves can also be used to define the most cost effective way to meet waterway health objectives and provide input to the budget to meet all TWCMP objectives (as well as input to policy measures such as pollution trading). It is also possible that additional externalities, such as social costs and benefits, can be added to the pollution and water quality costs to further extend the cost effectiveness analysis.

The draft results suggest that pollutant costs can be up to 50% of the total cost for some options such as recycled water. This can change the relative ranking of options by cost. However, in most cases the pollutant costs are less than 20% of the total cost and greenhouse gas emissions contribute only a couple of percent of the total cost. Final results with uncertainty and sensitivity analysis will be completed over the coming months and will provide guidance on the application of extended costs to TWCMP in SEQ.

Acknowledgements

Special thanks to Lavanya Susarla from Moreton Bay Regional Council and Nicole Ramilo from BMT WBM for data and support of the research.

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Life Cycle Perspectives for Total Water Cycle Planning

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Summary

The Life Cycle Assessment (LCA) methodology has been applied to a case study from the Total Water Cycle Management Plan (TWCMP) that has recently been developed for the Moreton Bay Regional Council. A sample of the results are used to illustrate how the life cycle perspective, and the data collected during this research, can be used to improve the urban water planning process.

Keywords

Total Water Cycle Management, Life Cycle Assessment, Moreton Bay Regional Council.

Introduction

The Total Water Cycle Planning (TWCP) process aspires to an integrated planning approach for water supply, wastewater treatment, and stormwater harvesting; in a manner that optimises social and environmental benefits and minimises costs (WBD 2010). This approach seems well suited to South East Queensland (SEQ), given the enormous challenges involved in meeting current constraints on water use and pollution loads, at the same time as planning for a possible doubling in population by 2050 (QWC 2010).

TWCP is being implemented on a local government basis in SEQ, and two key questions warrant further consideration. Firstly, how will these local TWCP processes account for the integrated nature of the SEQ's water supply grid? Secondly, in focusing on locally specific environmental issues, how will (and should) the TWCP process account for environmental externalities that might not currently be on the radar of urban water system planners. While the water industry has extended its interest to greenhouse gas (GHG) accounting, many studies have shown that GHG might not be considered a good proxy for the range of environmental issues of relevance to urban water systems (de Haas *et al.* 2011; Huijbregts *et al.* 2010; Lane and Lant 2012; Laurent *et al.* 2012).

The Life Cycle Assessment (LCA) methodology (Bauman and Tillman 2004; Guinee *et al.* 2011) is potentially a useful tool for addressing these two questions. Firstly, because the protocols laid out in ISO14040 (ISO 2006) require rigorous attention in setting appropriate system boundaries, thereby ensuring that all relevant parts of the system are included in the analysis. Secondly, because it quantifies a broad range of environmental impacts resulting from direct system operations, and indirectly from the manufacture, supply and disposal of all materials and energy used by the system in question. The ultimate goal of LCA is to reduce the risk that something important gets overlooked in the decision making process. That is not to say that it can provide definitive answers, as the scope of LCA excludes certain issues (eg, financial and social impacts) that will also be important considerations in urban water systems planning.

The objective for this research is to investigate how LCA might be used to inform the Total Water Cycle Planning (TWCP) process in SEQ. This paper uses a case study for the Moreton Bay Regional Council, specifically for the Caboolture catchment, to demonstrate the perspectives that LCA can provide. It is expected that this research will deliver benefits in three ways: (1) by providing more robust data for estimating externalities associated with the technologies under consideration in the SEQ urban water system; (2) by identifying key knowledge gaps that should be addressed if decision making is to provide environmentally optimal outcomes; and (3) to extend the horizons of urban water planners, giving them some insight into which environmental externalities might one day come onto their management radar.

Methodology

System Boundary

Life Cycle Assessment (LCA) was used to replicate and extend the analysis undertaken as part of developing the Total Water Cycle Management Plan (TWCMP) for the Moreton Bay Regional Council (MBRC). The results presented in this paper were based on the preliminary TWCMP modelling work as of September 2011, at which point the TWCMP draft recommendations were under review. Analysis reflecting more recent developments in the TWCMP will be incorporated in the final reporting for this project. The LCA analysis was restricted to the Caboolture River catchment of the MBRC area, excluding the Caboolture Identified Growth Area.

The TWCMP study identified three scenarios under consideration for the Caboolture River catchment area (Table 1). These represent an increasing level of sophistication/effort being invested into waterway and infrastructure management – Scenario 2 incorporates the elements of Scenario 1 with additional management actions; Scenario 3 incorporates the elements of Scenario 2 with additional and more sophisticated infrastructure. The planning study assumed that the current upgrade (in progress) of the South Caboolture Sewage Treatment Plant (STP) is implemented and will accommodate any future population growth in its sewage catchment. No variations on the level of this secondary wastewater treatment were considered. The scenarios differed in their approach to wastewater reuse, stormwater treatment and reuse, rainwater tank implementation, and measures to reduce catchment sourced pollutant generation.

Table 1. Scenarios under consideration for the Caboolture catchment of the Moreton Bay Regional Council total water cycle management planning study. Scenario 2 builds on and extends the management options included in Scenario 1. Scenario 3 builds on and extends the management options included in Scenario 2.

Scenario	Catchment	Stormwater	Water Supply	Wastewater
1 (minimum requirements for new development)	--	TSS, TP, TN reduced by 80/60/45% (for new developments)	Rainwater Tanks to toilet, laundry and outside use (for new developments)	Linear growth (with population increase) in STP throughput and wastewater discharge loads
2 (easier advances beyond minimum requirements)	Revegetation and farming best-management practices (BMPs)	as per Scenario 1	as per Scenario 1	Linear STP growth and treated wastewater loads
				Wastewater recycling to agriculture
3 (more ambitious objectives)	as per Scenario 2	as per Scenario 1 + enhanced WSUD for existing areas	Rainwater Tanks to laundry use (for new developments)	Linear STP growth and treated wastewater loads
		Stormwater harvesting and reuse to urban centres		
			Class A+ wastewater reuse to urban areas	

We used these same three scenarios. As our starting point, we adopted the overall household predictions, and water and nutrient balances of the catchment, developed for each scenario in the TWCMP study. Important modifications to the data used are summarised in the next section. A selection of the LCA impact assessment results are presented in this paper.

Data Collection

Where possible, the best available empirical data, from local studies, was used to model operations of the systems employed by each scenario. Where such data was not available, best estimates were informed by the latest science, and a similar study undertaken for the Gold Coast area of SEQ (Lane *et al.*, 2011). That Gold Coast study also informed our materials inventories for infrastructure construction.

Key differences from the data used in the TWCMP study were:

- *Household and commercial water use* – Assumptions for external use were adjusted to account for insight from recent SEQ end-use studies (Beal and Stewart 2011) and the limited available data on the net implications of rainwater tanks and Class A+ water being used for outdoor use (Stewart 2011).
- *Rainwater tanks* – Average tank yields were estimated in the preliminary stochastic analysis of Coultas *et al.* (2012), accounting for non-linear scale up effects from simplified tank yield models (such as that used in the TWCMP study). Tank energy use estimates were based on results from the laboratory characterisation of rainwater pump performance (Tjandraatmadja *et al.*, 2011).
- *Mains water supply* – Modelling of sea water desalination is based on our previous research in SEQ (Lane *et al.*, 2011).
- *Wastewater treatment and reuse* – Detailed operating models developed for operation of the South Caboolture Sewage Treatment Plant (STP) and Water Reclamation Plant (WRP) after the STP upgrade has been completed. These models incorporated: (a) plant specific data for energy and chemicals use, sludge generation and wastewater quality; and (b) best available estimates for fugitive gas emissions.
- *Stormwater harvesting and reuse* – Assumptions for chemicals use and energy demand for water reticulation followed those used for the wastewater recycling technologies.

LCA Impact Models

The selection of LCA impact models is a subset of those chosen for use, and described more fully, in a previous study (Lane *et al.*, 2011). The only modification for this study was that we incorporated a preliminary estimate of a *Minerals Depletion* impact factor for phosphorus resources, so as to better account for the implications of wastewater nutrient recovery.

Results and Discussion

Data Quality and System Boundary

In this section, we highlight the potential implications of improving the data and system boundary used for modelling externalities such as greenhouse gases. Figure 1 shows a selection of indicators used to compare across the scenarios, but with three different sets of data used in the analysis. A simplified, hypothetical, situation is envisaged in which these three indicators are the only criteria for ranking the options under consideration. Nonetheless, the simplified results still highlight how significant the changes might be if they affected an issue that was influential to the decision making process.

The first point to note is that including the improved energy use and fugitive GHG emissions data could change the rankings of the scenarios. The first example (a), which uses basic energy data typical of that appearing regularly in SEQ planning studies, would suggest little downside in moving from Scenario 2 to the greater nutrient removal of Scenario 3. However the second example (b), using the best-available fugitive emissions and energy data, and including full life cycle (scope 3) GHG emissions, provides a contrary story. In this case, the relatively high energy intensity of wastewater distribution to residential households creates a notable downside to the benefits of recycling.

Figure 1. Comparison of the scenarios using three different sets of data: Fig 1(a) uses the default inventory data collected for the draft TWCMP; Fig 1(b) uses alternative datasets collected through this study, but with the same system boundary; Fig 1(c) uses the same data sources as for Fig 1(b), but extends the system boundary to include offsets for avoided mains water production.

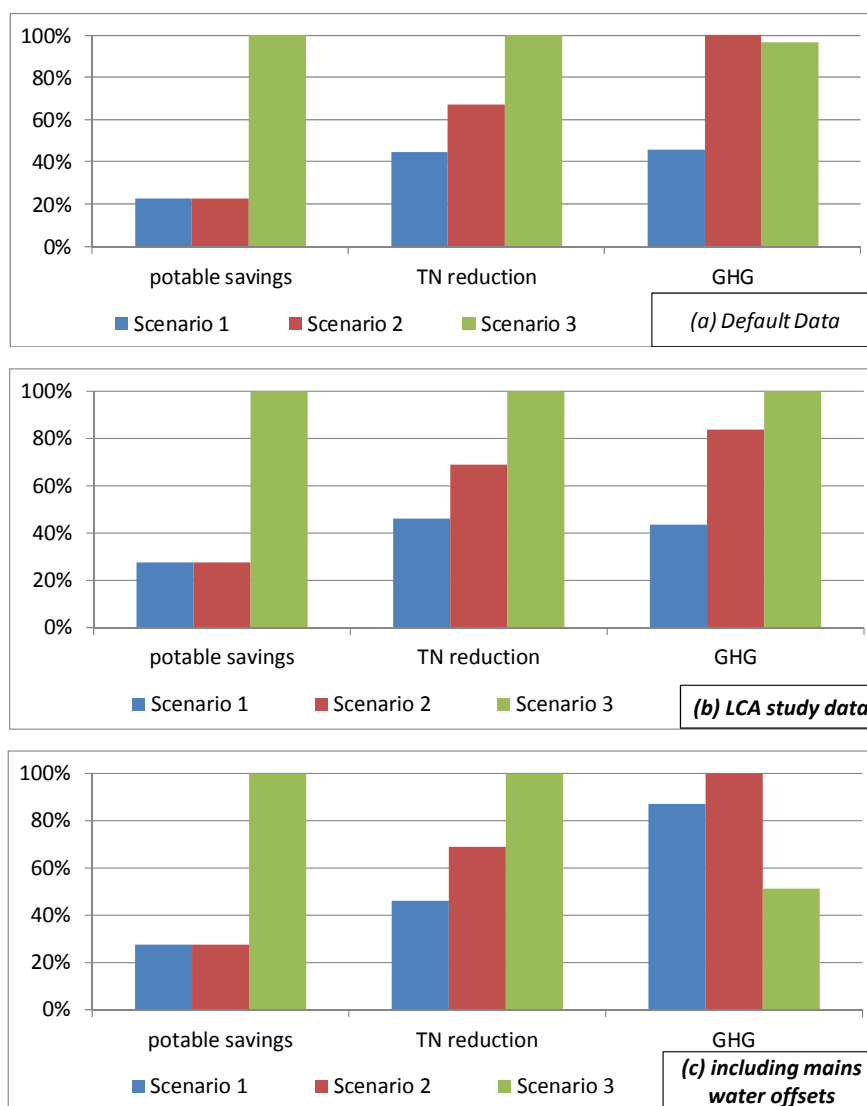
A comparison of the three graphs shows that improved data (Fig 1b) would identify a downside (increased GHG) to the benefits (decreased mains water use and nitrogen discharge) of Scenario 3 that was not apparent with the default data (Fig 1a); but that if the reduced mains supply requirement is included in the GHG analysis (Fig 1c), Scenario 3 has the best performance across all three criteria.

The scenarios are identical in each case:

Scenario 1 (rainwater tanks and best practice WSUD for new development);

Scenario 2 (rainwater tanks and best practice WSUD for new development; catchment management measures; wastewater recycling to agriculture);

Scenario 3 (rainwater tanks for new development; enhanced WSUD + stormwater harvesting and reuse; catchment measures; wastewater reuse to urban areas).



The second concern is that the system boundary used for the analysis in versions (a) and (b) of Figure 1 is flawed. In constraining the analysis to local systems (water supply, wastewater and stormwater) only, the system boundary does match the institutional boundaries and scope. However, it fails to account for the implications that a choice between these options would have on the operations of the SEQ water grid. Implementation of Scenario 3 would deliver the lowest demand on the water supply grid, and therefore require the least amount of mains water to be produced. If one assumes that the long-term marginal supplier of mains water in SEQ is seawater desalination, then the wastewater recycling actually delivers a net positive GHG effect (Figure 1c). In this case, Scenario 3 becomes a clear winner in the options comparison.

It is encouraging to see that mains water offsets have been quantitatively included in the GHG analysis used in the most recent version of the Moreton Bay Regional Council TWCMP; however there remains a substantial hurdle to appropriate implementation of this concept. The problem is that there is no clear cut way to choose which water supply technology (or mix of technologies) should be modelled for the mains water offsets. There has been a tendency in urban water systems planning to benchmark against the status quo, whether that be the traditional dam based supplies, or now the current SEQ grid mix. But in the context of growing population and growing overall demand, the status quo provides no insight into what will actually change as a result of decisions that lead to more or less mains water demand.

What should be used in the analysis is the supply source that will change its production rate in the event of a marginal increase or decrease in demand. Given the complexity of the SEQ water grid, determining the short-term or long-term marginal supply sources is not a trivial exercise, and will be beyond the capacity of individual planning studies based on specific planning areas. Long-term grid supply cost forecasts are unlikely to be helpful if they involve the averaging of contributions from all infrastructures in place at any point in time, and will be heavily influenced by capital expenditure for infrastructure construction. To minimise the GHG implications of the choices they make, urban water planners will need access to a range of grid operations modelling forecasts. These forecasts will need to reflect the operational rules constraining the SEQ Water Grid Manager, and consider the possibility of significant changes in policy direction over time.

Environmental Scope

The other purpose for which LCA was considered was to extend the environmental scope of the decision making process. The objective of this paper is not to pick a winner (from the three Scenarios) based on the full gamut of environmental implications, but rather to illustrate the challenges involved in doing so. We have included a subset of the available LCA impact categories (Figure 2). Other impact categories have been shown to have relevance to urban water systems studies (Lane *et al.*, 2011), but were excluded because of data constraints in this particular case study.

In Figure 2, the Figure 1 management issues “Potable water savings” and “TN reduction” have been recast as the environmental measures “*Freshwater Extraction*” and “*Eutrophication Potential*” (respectively). As we have adopted seawater desalination as the marginal mains water supply, the potable water savings in fact deliver no reductions in *Freshwater Extraction*. The benefit shown for Scenario 2 is from wastewater reuse on agricultural lands, which was assumed to offset direct extraction of local stream flows.

The direct comparison (Figure 2a) shows that there are a number of tradeoffs involved, and that picking a clear winner from the three scenarios is not possible. Scenario 2 provides benefits in *Freshwater Extraction* and *Minerals Depletion* from the wastewater reuse to agricultural lands. Scenario 3 has a greater *Eutrophication Potential* benefit because of reduced wastewater and stormwater discharge, and lower downsides (*Global Warming Potential*, *Ozone Depletion Potential*, *Fossil Fuel Depletion*) as a result of its greater mains water savings.

Stakeholder feedback from previous LCA case study application in SEQ (Lane *et al.*, 2011) illustrated clearly that, for the planning process to make use of such broad spectrum quantitative information, the decision makers will require some perspective on the relative significance of the scale of the different impacts being caused. There are a number of ways to provide such perspective, but the ultimate choice will always be a matter of preference for those involved in each planning process. We therefore highlight two possible approaches.

Firstly, the comparative results are benchmarked against an estimate of the total LCA environmental impacts (for the same impact categories) of the entire urban water system in the Moreton Bay Regional Council area (Figure 2b). The benchmark dataset was created using a similar approach to that done for the Gold Coast in a previous study (Lane *et al.*, 2011), adjusting key parameters (infrastructure scale and type) to account for important differences in the Moreton Bay region. Each result effectively indicates the percent change that would be made (by each particular scenario) to the overall impacts caused by the MBRC urban water system. The changes in *Freshwater Extraction* and *Ozone Depletion Potential* now look very minor, as the whole-of-system impacts for these two are dominated by contributors (dam water use and secondary wastewater treatment, respectively) that are not affected by this particular set of scenarios. From the perspective of reducing (or constraining) the overall environmental burden of the regional water system, the important tradeoffs in this case study might be narrowed down to those associated with nutrient

discharge (Eutrophication Potential) and energy use (*Global Warming Potential* and *Fossil Fuel Depletion*). Other issues might also be notable if the environmental scope was extended to include the full range of LCA impact categories.

Different issues look important change if the results are benchmarked against the total impacts associated with the Australian economy (Figure 2c). This latter dataset is described further in Lane *et al.* (2011). From this national perspective, the *Freshwater Extraction* and *Eutrophication Potential* implications look of a similar scale, with the others being more minor. Using this approach could clearly affect the options selection process, as any downsides from Scenario 2, when compared to Scenario 1, now look insignificant. However, caution is required when using this approach, as the benchmarking step involves significant potential for hidden bias to be introduced (Heijungs *et al.*, 2007). Certainly, important gaps in the available Australian dataset are well recognised (Foley and Lant, 2009; Lundie *et al.*, 2007). There might also be conceptual problems with using production-based benchmarking data when exports play an important role in the national economy (Breedveld *et al.*, 1999; Jungbluth *et al.*, 2011).

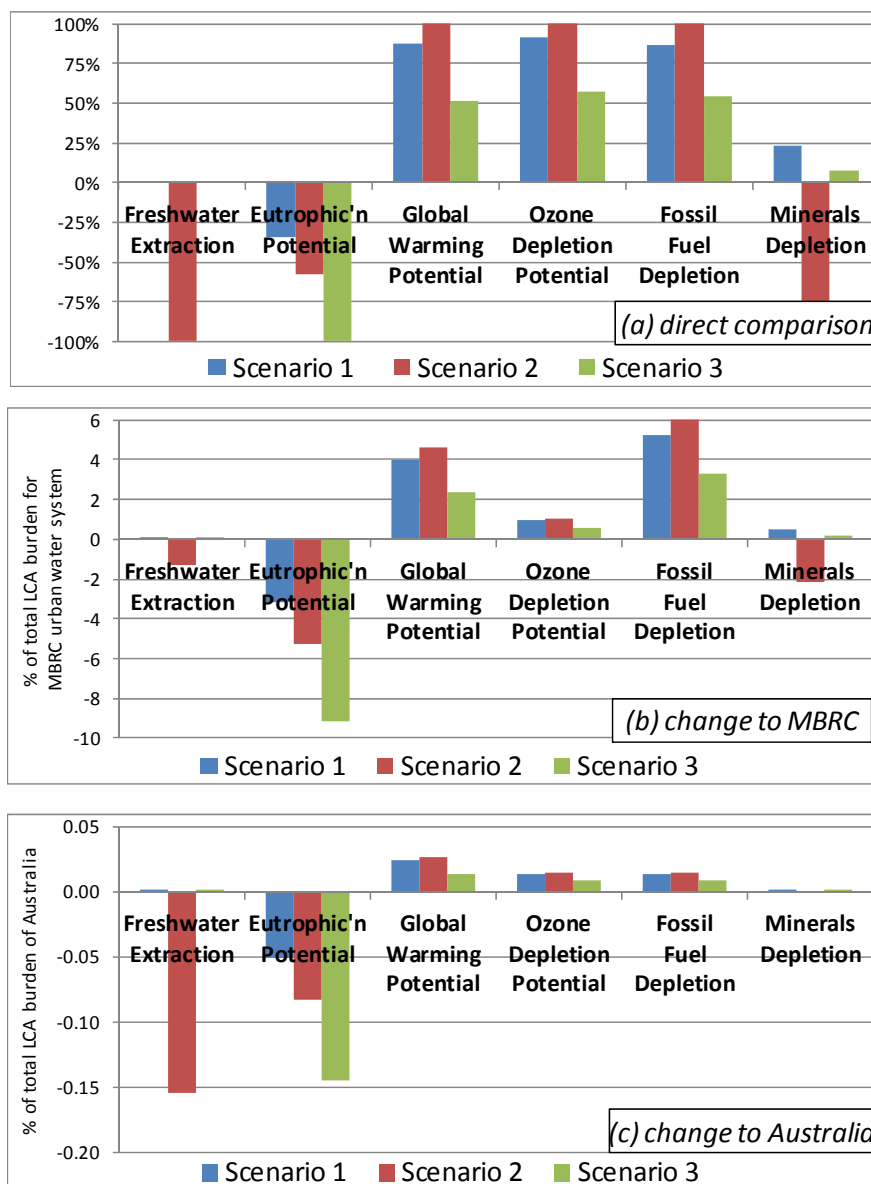
Figure 2. Three different perspectives on the comparison of scenarios across a selection of local, global and resource impact categories from LCA literature. The complex tradeoffs shown in the direct comparison (a) illustrates the difficulty of including broad spectrum LCA in the decision making process. Two possible means to provide perspective on these tradeoffs are to benchmark the results at the institutional level (b) or at the national level (c). These two perspectives suggest different priorities, showing that decision makers will need to consider a range of perspectives if they are to best utilise LCA impact assessment results.

The scenarios are identical in each case:

Scenario 1 (rainwater tanks and best practice WSUD for new development);

Scenario 2 (rainwater tanks and best practice WSUD for new development; catchment management measures; wastewater recycling to agriculture);

Scenario 3 (rainwater tanks for new development; enhanced WSUD + stormwater harvesting and reuse; catchment measures; wastewater reuse to urban areas).



Conclusions

The UWSRA Life Cycle Assessment (LCA) research has collected significant amounts of data, and developed protocols for its inclusion in options assessment. These could be used to improve the Total Water Cycle Planning (TWCP) process. Many of the important datasets are extremely uncertain due to a lack of quality empirical data, and these will be highlighted further in the final project reports. The LCA approach provides a framework for incorporating a broad range of environmental issues into multi-criteria decision support systems that are typically used in urban water systems decision making. It also provides robust and transparent bio-physical accounts that can be used to underpin financial and economic assessment of environmental externalities.

Where non-grid supplies are being considered under the TWCP process, reduced mains water supply operations should be included in the analysis of options. However, there will be two challenges with implementing this effectively. Further modelling work is required before water systems planners will have realistic insight into what assumptions to use for the marginal mains supply source – the status quo (the current water grid supply mix) is of little relevance where there is growth in population and overall demand for water. Secondly, the inclusion of mains water offsets will make the analysis more sensitive to the considerable uncertainties surrounding the effective supply contributions made by alternative supply sources (rainwater tanks, wastewater reuse, stormwater reuse). Further research should be targeted at better characterisation of how much these technologies can reduce mains water demand.

This paper did not use a sufficiently broad set of LCA impact categories to assess whether energy use, or GHG emissions, could be used as proxy for the environmental externalities in the TWCP process. However, previous studies would suggest that this is not the case (Lane *et al.*, 2011; Lane and Lant, 2012). We did, however, demonstrate that normalising the case study results against a number of different benchmarks can provide a variety of useful perspectives. However, further research is required to improve the available benchmarking datasets, consider alternative perspectives, and understand key uncertainties so that the results can be interpreted appropriately.

Finally, while not the objective of this paper, the results did suggest that adopting wastewater and stormwater reuse schemes could deliver a range of local and global environmental benefits, with little or no downside if the default mains supply is energy-intensive seawater desalination. In situations where this is the case, environmental analysis is likely to support the case for planners to move beyond the minimum standards required for servicing new development.

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Assessing Urban Water Strategies for Total Water Cycle Management

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Summary

This paper reports on a study undertaken as part of the Urban Water Security Research Alliance to explore the applicability of a number of emerging integrated assessment methods to Total Water Cycle Management planning. The decision making context is elaborated on, and then the assessment methods are briefly described. The paper concludes with a discussion on how these integration assessment activities can help in Total Water Cycle Management planning in South East Queensland (SEQ). The discussion is undertaken with the target context of Moreton Bay, but with no explicit case study being described.

Keywords

Integrated assessment, Total Water Cycle Management, optimisation, uncertainty analysis, cost effectiveness.

Introduction

Total Water Cycle Management (TWCM) is described as a management philosophy based on systems thinking that recognises that all elements of the water cycle are interdependent (Water by Design, 2010), and has been applied to decrease water demand, reduce stormwater run-off and improve pollutant wash-off from urban catchments by adopting sustainable water management practices (Chanan and Woods, 2006; van der Sterren *et al.*, 2009). The approach has been incorporated into water planning and management practices in a number of Australian contexts (Arbon and Ireland, 2003; Chanan and Woods, 2006; Najia and Lustig, 2006; van der Sterren *et al.*, 2009). It is a value-driven philosophy with holistic aspirations of managing the full total water cycle to achieve desirable environmental outcomes. As such, it is similar in aspirations and values to other concepts such as Water Sensitive Urban Design (Wong, 2006), Sustainable Urban Water Management (Larsen and Gujer, 1997), and Integrated Urban Water Management (Burn *et al.*, 2012; Maheepala *et al.*, 2010). Activities that are typically associated with TWCM are greywater recycling, sewer mining, rainwater harvesting or stormwater harvesting; or Water Sensitive Urban Design (Arbon and Ireland, 2003; Chanan and Woods, 2006; Najia and Lustig, 2006; van der Sterren *et al.*, 2009).

In South East Queensland (SEQ), a recent history of water scarcity has prompted major investments into infrastructure, with “the Water Grid” being a key part of the strategy to link water supplies across SEQ that include new dams, desalination facilities and water recycling infrastructure (Brown *et al.*, 2009). The response to water scarcity also included water restrictions and mandated water conservation efforts for new developments (Brown *et al.*, 2009; Queensland Government, 2008). Brown and colleagues argue, perhaps unfairly, that SEQ missed the opportunity to adopt modern concepts of urban water management (such as TWCM) as a means to deal with this crisis because they did not develop contingency plans for alternative, future climatic scenarios and overlooked the existing suite of decentralised technological approaches available (Brown *et al.*, 2009). However, Total Water Cycle Management plans are now required for all councils in the region aiming to achieve a larger SEQ Healthy Waterways Vision as part of the wider SEQ Water Strategy developed by the Queensland Water Commission (QWC) (Harman and Wallington, 2010). Key elements in the QWC’s commitment to TWCM are the consideration of all water sources (including wastewater and stormwater), sustainable use of water, equitable allocation of water, and the consideration of natural water processes (Harman and Wallington, 2010). TWCM is currently being implemented in SEQ at the regional, sub-regional and local scales, including a number of projects, programs and initiatives.

To support decision makers and managers in SEQ, Dutra and colleagues (2010) have developed a Management Strategy Evaluation (MSE) tool to support decision makers in SEQ to assess different strategic options. Decision makers in this case are managers from local councils and natural resource management bodies. It is important to note that the management strategies that are considered involve not only those related to urban water management, but also those that relate to farming practices, vegetation, channel erosion, rural stormwater, wastewater disposal, urban stormwater and urban design. The evaluation of strategies is done on the basis of indicators of water quality (total nitrogen, total phosphorous, turbidity, chlorophyll, light penetration and dissolved oxygen) which are used to assess health of waterways. Social perceptions of the value of the health of waterways are also considered as well as economic assessments of management options. The process of designing management strategies for evaluation is done through a MSE computer model and results are simply displayed to show outcomes for Water Quality Scorecards and cost approximations.

This paper reports on a study undertaken as part of the Urban Water Security Research Alliance to explore the applicability of a number of emerging integrated assessment methods to TWCM planning. Secondly we will discuss how the emerging integrated assessment methods can be applied to develop evidence based TWCM plans in SEQ. The study backdrop is the Moreton Bay Regional Council local government area where a TWCM plan is currently being developed by the Moreton Bay Regional Council. Methods are being developed to support TWCM in this location.

The Decision Problem

To support decision makers in assessment activities, it is imperative to first understand the decision making context. On the surface, the task seems relatively simple, ie, a) define a number of possible strategies, b) evaluate which strategy that will best achieve goals according to defined criteria, and c) choose and implement the strategy that appears to be the most appropriate. Obviously, reality does not quite oblige to allow this simple process, for a number of reasons. These seven points, inspired by Rittel and Webber (1973), are difficulties that decision making tools/frameworks should ideally consider and/or address, including difficulties relating to:

1. *Goal formulation*: this is not well-defined and there are many possible goals that one may want to achieve, and the choice of assessment metrics is a value-driven process.
2. *Limited scope*: strategies are not always within the control of decision makers: they will only be able to influence a sub-set of those factors that have an impact on the desired outcomes.
3. *Unlimited option space*: it is virtually impossible to define an exhaustive list of conceivable strategy options, and the formulation of such a list is a process that requires creativity and analysis, in combination with some kind of filtering out of solutions that are not, for various reasons, appropriate.
4. *Assessment difficulties*: the nature of the TWCM problem is difficult to describe in a way that *easily* helps us to evaluate the effectiveness of strategies (in achieving goals). Systems are usually not very well understood, and there are serious limitations in terms of data, limiting the scope of assessments.
5. *Limited planning capacity*: the amount of effort spent on the task of finding and choosing strategies is more constrained by issues like the availability of money, time or understanding, rather than the sense of “being sure of having found a good solution”.
6. *One-off operations*: every time this exercise is undertaken there are unique factors, such as local weather patterns, land use patterns and geography, that can’t be ignored, and there are therefore limited opportunities to learn by trial-and-error. Incorporating judgments on the importance of such factors would be important.
7. *High stakes*: every TWCM decision will have (sometimes serious) impacts on the community, and the planner is in some ways to be found responsible for the outcomes of his recommendations.

Considering these issues, it is perhaps not surprising that there is a complex set of classes of interacting of factors that contribute to good decision making in the TWCM context (Dutra *et al.*, 2010): leadership (presence of transactional leaders or not, etc), information (availability and quality), regulatory framework (existence or not, conflicting or not), conditions (favourable or not), assumptions (robust or not), communication (effective or not; good or bad quality), and logistics (implementation and timing of strategies, etc). The relative importance of each of these factors for a perceived successful strategy choice and implementation depends on the application. Modelling primarily concerns information and assumptions, but could also support communication, realisation, logistics and even leadership. The function of modelling activities in a decision making contexts like this needs to be viewed in a holistic sense and not just in terms of the validity of models. The paper describes four activities in TWCM planning (as per Figure 1 but excluding the evaluation), and subsequently presents how integration assessment activities in the study context of Moreton Bay can contribute to this process in a way that hopefully deals with some of the properties of the decision problem described above.

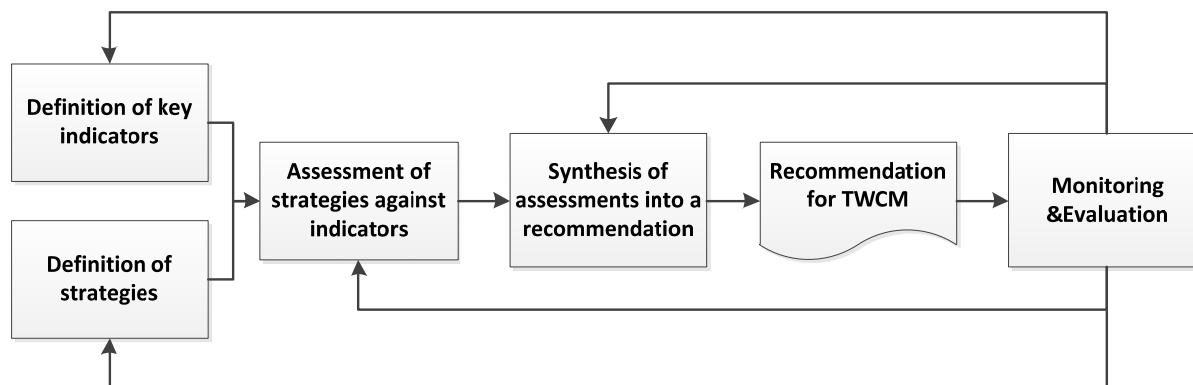


Figure 1. TWCM Planning Process Sketch.

Activities in TWCM Planning

The process of TWCM planning involves a number of steps, as per Figure 1, and the outcome of the planning process is a recommendation. This is a similar process to that of decision making in general, and in fact similar to the (structured) process of deciding which car to purchase. When purchasing a car, one may consider what the assessment metrics are (fuel efficiency, reliability, price, safety, aesthetics, etc), and would identify some viable options (Honda Jazz, Toyota Yaris, etc), then collect information about each of the options against each of the assessment metrics and finally synthesise all this and come to a recommendation. TWCM planning is similar, where a large set of assessment metrics are broadly grouped in three major groups (economic, environmental and social). However, other categories like health and system reliability are now being added as main assessment metrics. In the case of TWCM planning, problem definition is not as clear.

Definition of Key Indicators

There are many potential goals that could be considered in TWCM, and an important step in planning processes is to identify which assessment metrics should be the basis for analysis. For example, the assessment metrics used in the MSE (Dutra *et al.*, 2010) are different to those explored by Sharma and colleagues (2009), which in turn are different to those that are explored in our context of Moreton Bay. Choosing the “right” set of assessment metrics is a strategic decision that is value-driven and where stakeholder input is essential; and this is a step that is outside of the scope of traditional modelling activities. The choice of assessment metrics should not however be dictated fully by what is “easy to evaluate”. To only assess that which is easy would be like deciding which car to purchase on the basis of what it looks like alone. Further, relatively straight-forward research may reveal critical aspects, such as its fuel efficiency, safety, reliability and its general condition. Further assessments of the car, such as inspection of the detailed condition of the car may be considered crucial, but will come at a significant cost. Such more detailed assessment would only be undertaken once you are “almost certain” that you’d like to make the purchase. This example illustrates how assessments typically need to be undertaken at different levels of detail at different stages of the assessment process.

Definition of Strategies

TWCM strategies include, but are not limited to, a range of water supply options, farming practices, riparian re vegetation, channel erosion and bank stabilisation, rural stormwater, wastewater disposal, urban stormwater and urban design. Different approaches to urban water management that include rainwater harvesting are also considered. Strategies are also both spatial (ie, different actions at different locations) and temporal (ie, different actions at different times). As such, even when the individual actions are within a limited set, the total number of possible strategies is very large. Also, not all strategic actions are available for decision makers, and this shows that there is a need to coordinate actions across a range of stakeholders. Furthermore, the reality of the TWCM process is that the definition of strategies is an on-going and adaptive process, with feedback from assessment of strategies and synthesis into the definition of strategies as in Figure 1.

Assessing Strategies against Indicators

When evaluating different strategies in TWCM, there are a range of aspects to consider. These are often categorised as social, economic and environmental aspects, in a triple bottom line framework as Baldwin and Uhlmann (2010) claim is needed for water planning in SEQ. This prompts the following questions, relating to each strategy’s range of impacts (ie, its assessment metrics):

- What are the environmental impacts, locally and globally? In the target context of Moreton Bay, such questions are answered by means of Life Cycle Assessments (LCA) with the evaluation of a number of assessment metrics: freshwater extraction, eutrophication potential, global warming potential, ozone depletion potential, etc.
- What are the social impacts, locally and globally? In the target context of Moreton Bay, Externalities assessments help answer such questions.
- What are the economic implications? In the target context of Moreton Bay, this would be assessed in terms of costs and cost-effectiveness of strategies.

It is very common that some assessment metrics, such as community acceptance or logistic feasibility, cannot be assessed (or are too costly to assess) by means of analytic approaches and in such cases it is possible to rely on expert knowledge, or other judgments that are considered to be reliable.

Synthesising Assessment Results into a Coherent Recommendation

To reach recommendations for TWCM, assessments of a range of strategies must be synthesised into one single analysis. This is problematic from the point of view that one typically has to balance the achievement of one goal against the achievement of another. It is a rare situation when the maximum achievement of all assessment metrics occurs for a single solution; and this raises the need for considering the priorities of stakeholders such as the public, business interests and government. Furthermore, one may at this point have to consider uncertainty in outcomes, for example the cost for one solution may vary considerably depending on what the future brings (as for example would be the case if the cost relied on the price of petrol), whilst another strategy carries a completely certain cost. The synthesis would also have to consider the adequacy of the information that is available, and perhaps whether better information is required before a recommendation can be made.

Integration Assessment Activities

There are three integration assessment-related methods that could support decision making in the TWCM context either by themselves individually, or in a combined framework. These are briefly described below.

Multi-Objective Optimisation

Multi-objective optimisation (MOO) is a process that can be undertaken on mathematically formulated problems where the goals are clearly defined using one or a number of functions. It is a process that searches through a multitude of possible decisions and evaluates them based on how they perform against a set of assessment metrics. For the Moreton Bay case study (Grant *et al.*, 2010), the decisions include which infrastructure to develop, and which policies to implement, for example whether to supply recycled water to urban users, or retrofit rainwater tanks to existing dwellings, etcetera, there being 35 solutions in all. With many such proposed strategies, the possible number of combinations grows exponentially, so an automated approach to searching through them may be warranted. After the optimisation is completed, a Pareto front of the optimal solutions is produced. Such a graph shows the trade-off between the objectives. This can be useful information to allow planners to remove those alternatives from consideration that are sub-optimal regardless of value judgments. This leaves the planner with a much smaller subset of possible strategies to deal with. The planner may subsequently select the most expensive, most effective solution, or compromise on price to achieve less benefit, while keeping in mind any objectives unable to be represented easily by software (such as public acceptability). By creating a hypothetical model of Moreton Bay incorporating the many different proposed solutions, the aim is to determine the circumstances under which multi-objective optimisation approach is useful to TWCM. The main limiting factor for its usefulness will be the accuracy of the Moreton Bay model itself – optimising a model to a fine degree can be counter-productive if the model itself is a coarse representation of the real world system (Cunge, 2003). In such cases, analysing a few different scenarios by hand after a panel of experts has performed a Multi-criteria Analysis (MCA), the approach taken in Moreton Bay (Grant *et al.*, 2010), may well be superior.

Cost-Benefit Analysis

Cost-benefit analysis calculates improvement in assessment metrics on a per invested dollar basis. This is a critical tool to allow for comparison of actions to improve assessment metrics (such as pollution of phosphorous into Moreton Bay) within the scope of the study context (for example urban stormwater harvesting) with those that are outside the scope of the Moreton Bay case study (for example changed agricultural practices in the catchment). This provides a way for policy makers to identify target areas in order to reach policy targets at minimum cost. The approach adopted in this study in Moreton Bay was to calculate an extended cost effectiveness (Hall, 2012). Pollution abatement costs within the catchment, when applied to the Moreton Bay study context, were calculated to reach a range of water quality objectives. The cost of abating pollution from the Moreton Bay study area was then added to the capital and operating costs of the strategy option. This provides optimisation for two objectives, namely water supply and water quality and the results are expressed in dollars. Additional pollutant costs such as greenhouse gas can be expressed in dollars and added to the costs. Similarly, benefits such as aesthetic and recreational values can also be considered if monetised (Daniels and Porter, 2011).

Uncertainty Analysis

Bayesian Networks and Subjective Logic can be used in a framework to combine a number of assessments into one single assessment; either within a sustainability assessment framework (Moglia *et al.*, 2011; Moglia *et al.*, 2012b) that can be extended into a Multi-Criteria Decision Assessment framework that incorporates subjective value judgments by stakeholders (Hajkowicz and Collins, 2007), or in a Bayesian Network framework that calculates the combined likelihood of achieving all the separate goals defined in terms of thresholds (Moglia *et al.*, 2012a). Furthermore, incorporating Subjective Logic into the framework adds the ability to assign a “reliability” judgment on all the probabilistic statements (ie, sub-assessments), and hence allows for assessing the overall “reliability” of the

integrated assessment, as well as the uncertainty in the overall outcomes. If monetary value judgments are assigned to different outcomes, it is also possible to estimate the aggregated value of a strategy outcome, both in terms of the statistical expectation (monetary) value as well as the standard deviation, describing the amount of expected uncertainty around this expected value outcome. This method is also particularly useful for incorporating different types of assessments and judgments into a coherent whole, and hence this allows for combine qualitative judgments made by experts with the more robust quantitative assessments made by analysts. It is thought that this type of analysis is particularly useful in the process of synthesising a range of assessments into a single recommendation.

Discussion

In this paper, the process of TWCM has been described both in terms of its process as well as some of its dilemmas. Furthermore, three types of integration assessment activities have been described, and now we intend to discuss how those activities can support the TWCM process, and where particular methods can bring value throughout the process of narrowing down options, see Table 1 describing their sequence.

Table 1. Issues and implications relating to integration assessments.

Process	Issues and Implications Relating to Integration Assessment
1. Define Key Indicators	The definition of assessment metrics remains outside of the scope of integration assessment, but provides critical input into the process. The critical issue is that the subsequent feasibility of assessment should not limit the choice of metrics. Choose an integration framework that supports the incorporation of both quantitative assessments and expert judgments.
2. Definition of Strategies	Multi-criteria optimisation, if the problem can be translated into mathematical formulation, can help narrow down the number of interesting strategies into a smaller set that is manageable from the subsequent analysis point of view. Interestingly however, the optimisation approaches requires automated assessments, although the process of undertaking assessments is a subsequent step. Hence, the automated assessments are usually limited to only a sub-set of assessment metrics and to approximations, and therefore, this approach has to be seen as a first attempt at reducing the complexity of the decision problem rather than a tool that provides recommendations.
3. Assessments of Strategies Against Indicators	Assessments need to be carried out for all assessment metrics, even though some of these are based on rigorous and quantitative analysis; whilst other assessments are most likely based on expert judgments. To be incorporated into subsequent uncertainty analysis, assessments need to evaluate uncertainty as well judgments on the reliability of each sub-assessment.
4. Synthesis of Assessments into Recommendations	This can be done using Multi-Criteria Assessment, Cost-benefit analysis, or Uncertainty Analysis. Uncertainty analysis using Bayesian Networks and/or Subjective Logic provides a way to incorporate all the assessments into a single recommendation in a way that makes transparent the uncertainty in outcomes as well as the reliability of assessment (ie, whether you can trust it). This allows the planner to formally describe his reasons for why further analysis and/or data collection is needed. It also provides a way to formally undertake risk management. Given the high stakes of the problem, it is prudent that the planner makes sure that he/she does not put the community at risk of severe unwarranted consequences. Sometimes a certain (ie, low risk) but not optimal solution is better than an optimal but risky solution (on average good but small chance of very poor performance). This is likely to reduce the cognitive load for the planner and is hoped to reduce any cognitive biases.
5. Recommendations for TWCM	Cost-benefit analysis provides a way to compare different assessment outcomes, both within the decision makers range of influence, as well as outside of this range; and hence provides an excellent approach for communicating results. It is thought that recommendations have to be provided to government as well as to stakeholders, and that estimated cost-effectiveness provides a way to justify expenditure on strategies.

Other considerations in the process of synthesising a number of assessments into a single recommendation relate to the nature of the decision making problem (see the 7 dilemmas previously described). Some of these dilemmas have already been addressed in Table 1 (ie, relating to goal formulation, definition of strategies, limitations on data and the need to incorporate expert judgments). Perhaps the most important dilemma relates to decision makers limited availability of time and funds in order to come to a recommendation; and the fact that data and analysis is not necessarily the only pathway towards good decision making. This critical factor, and the complex nature of the problem, means that a straightforward approach (in terms of the decision makers' process, not in terms of underlying algorithm) such as the MSE (Dutra *et al.*, 2010) would seem appropriate. This approach allows for easy formulation and assessment of strategies against a range of criteria, which means that the decision maker can quickly learn about the nature of the TWCM problem. The MSE would appear to be, at the very least, an appropriate tool in the initial stages of TWCM planning. The question is whether more in depth and detailed analysis is warranted? The answer to this question depends on the level of risk taking that is appropriate. If there is a need for more detailed analysis of TWCM strategies (as would be indicated by dilemma 6 and 7 above) then uncertainty analysis may provide a good framework for the incorporating detailed analysis, based on initial Life Cycle Analysis and incorporating externalities.

Finally in terms of strengths of different approaches, it is thought that the cost-benefit analysis provides an excellent means for communicating results both within and outside of the planners' organisation. It provides a way to benchmark the actions (for example in relation to reducing pollution loads into Moreton Bay) against other plausible actions by others or oneself. This addressed the concern that any action that is recommended by the planner depends on what is considered to be within the scope of the planner; whilst ignoring potential other strategies that may be much more cost-effective means of achieving the same goals. To conclude, we believe that the integration assessment activities within the context of Moreton Bay can definitely support the TWCM planning process, but that the practical testing of these activities to real cases would need to be undertaken to ensure that this is indeed the case.

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Ozonation and Biological Activated Carbon Filtration of Wastewater Treatment Plant Effluents

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Summary

The project investigated the fate of trace organic chemicals (TrOCs) in three full-scale reclamation plants using ozonation followed by biological activated carbon (BAC) filtration to treat wastewater treatment plant effluents. Chemical analysis was used to quantify a wide range of TrOCs and combined with bioanalytical tools to assess non-specific toxicity (Microtox assay) and estrogenicity (E-SCREEN assay). It was found that the combination of ozonation and BAC filtration can achieve removals of 50% for DOC and more than 90% for a wide range of TrOCs as well as a reduction of 70% of non-specific toxicity and more than 95% of estrogenicity. This process combination is therefore suggested as an effective barrier to reduce the discharge of TrOCs into the environment or their presence in water recycling schemes.

Keywords

Ozonation, biological activated carbon, micropollutants, pharmaceuticals, bioassays, baseline toxicity equivalent, reuse.

Introduction

The presence of a large variety of anthropogenic organic compounds at trace levels (typically $\mu\text{g L}^{-1}$ and below) in domestic wastewater has been reported worldwide (Ternes *et al.*, 1999; Kim *et al.*, 2007; Hollender *et al.*, 2009; Reungoat *et al.*, 2010). These contaminants of emerging concern are commonly designated as trace organic chemicals (TrOCs). Among them, pharmaceuticals have received particular attention since they have been designed to be bioactive. While some are effectively removed by conventional biological treatments (eg, ibuprofen, paracetamol), others (eg, carbamazepine, diclofenac) are barely affected (Onesios *et al.*, 2009). As a result, pharmaceuticals are released into surface water via wastewater treatment plant (WWTP) effluents and this situation is of concern as the effects of low-level but long-term exposure on aquatic life are still largely unknown. Moreover, even though there is no sound evidence of impact on human health, the precautionary principle should be applied in the case of planned or unplanned indirect potable reuse. Therefore, additional steps have to be considered for the advanced treatment of WWTP effluents to reduce the discharge load of TrOCs into sensitive receiving waters.

Several technologies have proven to be effective in removing TrOCs from water of various qualities: activated carbon adsorption (Westerhoff *et al.*, 2005; Snyder *et al.*, 2007), ozonation and advanced oxidation processes (Esplugas *et al.*, 2007; Hollender *et al.*, 2009; Reungoat *et al.*, 2010) and membrane filtration (Kimura *et al.*, 2004; Snyder *et al.*, 2007). Activated carbon adsorption and ozonation are considered economically feasible options for advanced treatment of WWTP effluents (Joss *et al.*, 2008). However, ozonation is known to lead to the formation of transformation products largely not identified to date, which raises concerns regarding their potential impact on the environment and human health (Benner and Ternes, 2009; Radjenovic *et al.*, 2009; Dodd *et al.*, 2010; Stalter *et al.*, 2010; Stalter *et al.*, 2011). Activated carbon adsorption following ozonation has proven to be very effective in further removing TrOCs and decreasing non-specific and specific toxicity but the adsorbent has to be renewed or regenerated regularly to maintain its adsorption capacity (Reungoat *et al.*, 2010).

Biological sand filtration following ozonation has the potential to further remove some organic compounds present at trace levels and reduce non-specific toxicity (Göbel *et al.*, 2007; Reungoat *et al.*, 2011; Stalter *et al.*, 2011). Activated carbon is also a commonly used media to support biological activity but has received little attention so far for the advanced treatment of wastewater despite the fact that it is potentially more efficient than sand (Gerrity *et al.*, 2011; Reungoat *et al.*, 2011). The so called biological activated carbon (BAC) filtration has been used for many years in drinking water treatment, usually after ozonation, and has proven to be able to significantly remove natural organic matter, ozonation transformation products, disinfection by-product precursors as well as taste and odour compounds, eg, geosmin and 2-methylisoborneol (Simpson, 2008).

Three full-scale reclamation plants using ozonation followed by BAC filtration were investigated to: (i) determine the overall effectiveness of the combined processes for the removal of selected TrOCs and the reduction of non-specific toxicity and estrogenicity; (ii) determine whether ozonation leads to a mixture of transformation products with lower toxicity compared to the non-ozonated effluent; (iii) determine whether BAC filtration is capable of removing TrOCs and reducing the toxicity of the transformation products mixture formed by ozonation; and (iv) assess the influence of ozone dose and empty bed contact time (EBCT) on the performance of ozonation and BAC

filtration respectively. For that purpose, chemical analysis for the quantification of TrOCs was combined with *in-vitro* bioanalytical tools, as they complement each other to evaluate water quality (Macova *et al.*, 2010; Reungoat *et al.*, 2010). The decrease of non-specific toxicity was quantified with the Microtox assay, which is based on the bioluminescence inhibition of the marine bacterium *Vibrio fischeri*. The Microtox assay was selected to obtain a measure of the sum of all TrOCs in a water sample, as it responds rather non-specifically to all chemicals, and to estimate which fraction of the observed toxicity was elicited by the quantified compounds. In addition, estrogenic activity was quantified with the E-SCREEN assay to assess a receptor mediated mode of toxic action that is a critical end point relevant for WWTP effluents (Leusch *et al.*, 2010).

Materials and Methods

Full-Scale Reclamation Plants

Samples were collected from three full-scale wastewater reclamation plants located in Australia (Figure 1). All the plants receive treated effluent from WWTPs with biological nutrient removal. After various pre-treatment stages, they all use ozonation followed by BAC filtration before final disinfection using various technologies. However, the ozone dose and empty bed contact time (EBCT) in the BAC filters differ from one plant to another, providing different operating conditions. The specific ozone doses at the time of sampling were 0.2-0.3, 0.4-0.5 and 0.6-0.8 mgO₃ mgDOC⁻¹ for Landsborough, Gerringong and Caboolture respectively (Table 1). The granular activated carbon used in the BAC filters was from various origins. At Caboolture, the filtering media had been replaced in March 2008 and the samples were collected in July 2010, by which time approximately 68,000 bed volumes had passed through the filter. The BAC filters were commissioned in 2003 at Landsborough and the media had not been renewed since, leading to more than 350,000 bed volumes filtered at the time of sampling (March to June 2010). Finally, at Gerringong, the four BAC filters were commissioned in 2002 and the media of two filters was renewed in August 2009. Therefore, at the time of the sampling campaign in September 2010, half of the media had filtered approximately 95,000 bed volumes and the other half had filtered about 13,000 bed volumes. Given the large numbers of bed volumes filtered in each plant, it is reasonable to assume that all the filters have passed the breakthrough of organic matter and adsorption is negligible. Dissolved oxygen concentrations measured before and after filtration through the BAC showed a decrease confirming that they were biologically active.

Sample Collection

Three sets of grab samples were collected from each plant at the sampling points indicated on Figure 1. Grab samples were collected as opposed to composite samples since the study focuses on treatment process efficiency and not on pollutant loads. Moreover, the balance tanks allow a steady flow rate along the advanced treatment train and variations of water quality were not expected to occur while sampling.

Chemical Analysis Methods

Dissolved organic carbon (DOC) was measured after filtration of the sample through a 0.45 µm nylon membrane. For each sample, two to three replicates were measured, giving a relative standard deviation of less than 3%. Forty-one TrOCs were quantified by a method consisting of solid phase extraction, elution, concentration, and analysis of the extract by liquid chromatography coupled with tandem mass spectrometry.

Bioassays

The E-SCREEN assay was applied to quantify the estrogenic activity of the sample and the estradiol equivalent concentrations (EEQ) in a sample. The assay was performed as described in Macova *et al.* (2010). Many previous studies have shown a good agreement between EEQ_{bio} (determined by the bioassay) and EEQ_{chem} (calculated from the estrogenic compounds concentrations and their relative estrogenic potential) and therefore estrogenic chemicals were exclusively quantified by bioanalysis. The limits of quantification and detection are 0.03 and 0.01 ng L⁻¹ respectively (Macova *et al.*, 2010).

The bioluminescence inhibition test with *Vibrio fischeri*, was used to quantify non-specific toxicity integrating the potency scaled contribution of all micropollutants present in a sample. Water samples were cleaned and enriched by solid phase extraction (Macova *et al.*, 2010; Reungoat *et al.*, 2010). While SPE does not capture very hydrophilic/polar and volatile compounds, those are not expected to contribute substantially to the mixture toxicity (Escher *et al.*, 2008). Bioassays were expressed as baseline-toxicity equivalent concentrations (baseline-TEQ_{bio}) according to Escher *et al.* (2008), and compared to the predicted toxicity of the quantified TrOCs (baseline-TEQ_{chem}). The later was calculated from the measured concentrations and the relative potencies of the chemicals using a hydrophobicity-based prediction model (Vermeirssen *et al.*, 2010).

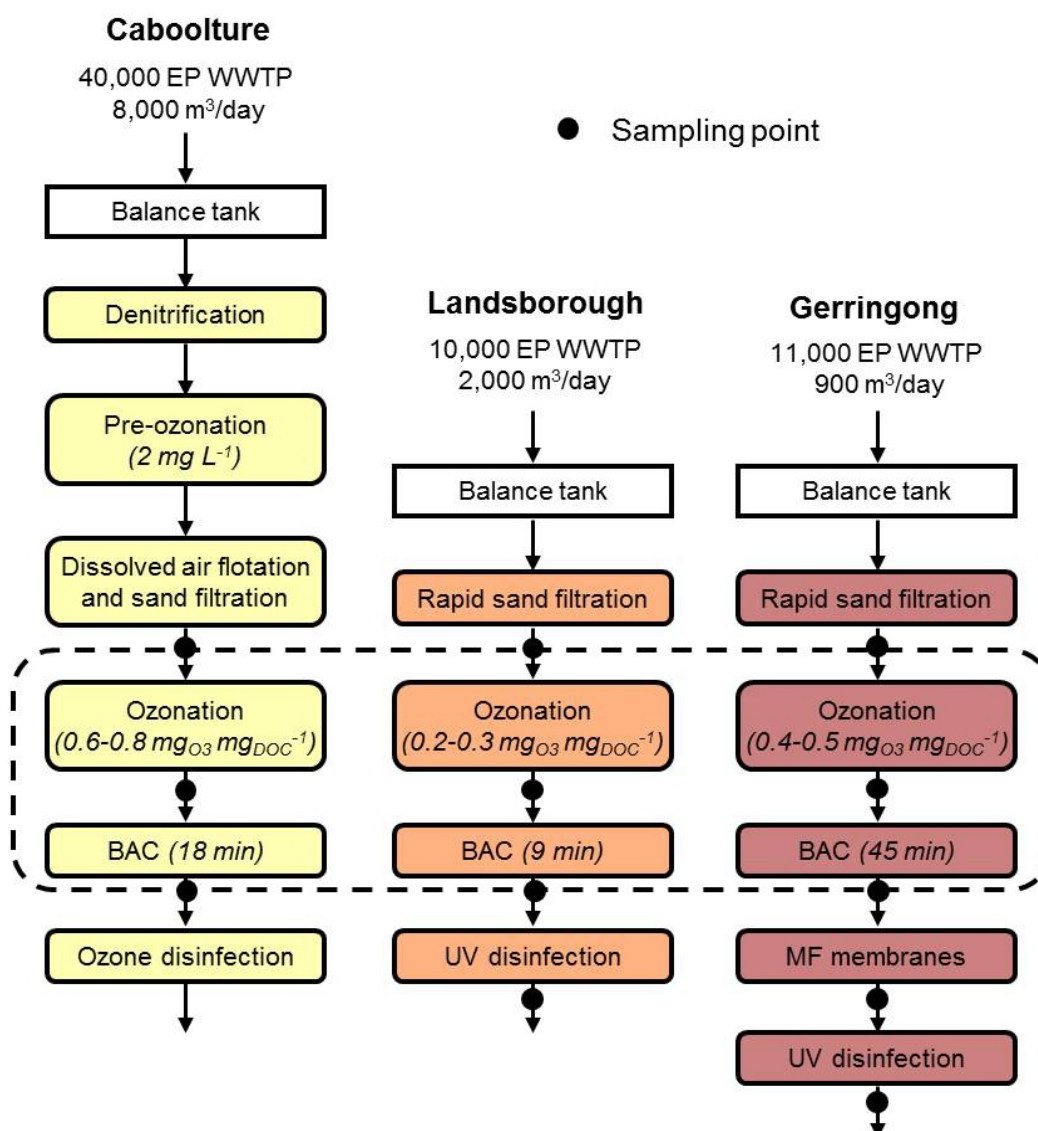


Figure 1. Treatment trains of the three investigated full scale reclamation plants (EP = equivalent population; WWTP = wastewater treatment plant; BAC = biological activated carbon, MF = microfiltration).

Results and Discussion

A previous study of Caboolture's reclamation plant showed that the treatment stages upstream of the main ozonation (ie, denitrification, pre-ozonation and dissolved air flotation/sand filtration) have a limited impact on the TrOCs concentrations, with removals of less than 20% generally observed (Reungoat *et al.*, 2010). The samples collected at Landsborough also showed that no removal occurred in the rapid sand filtration. The removal of TrOCs in the disinfection treatment stages downstream of the BAC filters was difficult to evaluate due to the low concentrations reached at this stage. However, the results showed that ozone disinfection, UV disinfection and microfiltration led to limited removal of the compounds present. This is not surprising given the fact that the ozone and UV doses used for disinfection are typically much lower than the doses required to observe TrOCs degradation. The microfiltration membranes used at Gerringong have a 0.2 µm pore size and are therefore not expected to remove small organic molecules. The ozonation and BAC filtration stages were therefore identified as the treatment steps responsible for TrOCs removal in these reclamation plants and are further discussed hereafter.

Water Quality before Ozonation

The quality of the treated effluents before the ozonation stage was similar in all the plants (Table 1). The DOC and nutrients levels were low, showing the efficacy of the WWTPs in removing these compounds. However, most of the quantified TrOCs were detected before ozonation with concentrations varying from low ng L⁻¹ up to µg L⁻¹ levels, showing their incomplete removal in the WWTPs. It is interesting to note that the concentrations of most of the compounds remained in the same order of magnitude across the three plants despite the different locations and sampling times. This shows how ubiquitous these compounds are in treated effluents as well as a regular consumption pattern within Australia.

Table 1. Water quality parameters in the reclamation plants before the ozonation stage (N/D = not determined).

	Caboolture	Landsborough	Gerringong
T (°C)	22.0	22.6 – 28.5	N/D
pH	6.6 – 6.7	6.7 – 7.1	6.7 – 6.9
Conductivity ($\mu\text{S cm}^{-1}$)	879 – 910	392 – 507	520 – 563
DOC (mgC L^{-1})	6.5 – 8.1	5.8 – 6.6	4.2 – 5.8
PO_4^{3-} (mgP L^{-1})	≤ 0.02	0.22 – 2.00	< 0.02
NH_4^+ (mgN L^{-1})	< 0.03	0.22 – 0.45	0.18 – 1.36
NO_2^- (mgN L^{-1})	< 0.02	0.03 – 0.06	< 0.02 – 0.04
NO_3^- (mgN L^{-1})	< 0.02 – 0.95	0.18 – 0.47	0.39 – 1.14
Baseline-TEQ _{bio} (mg L^{-1})	1.83 – 2.72	1.50 – 2.01	1.10 – 1.84
Baseline-TEQ _{chem} ($\mu\text{g L}^{-1}$)	1.74 – 2.62	3.31 – 5.81	2.77 – 2.97
Baseline-TEQ _{chem} / Baseline-TEQ _{bio}	0.10 – 0.11%	0.19 – 0.29%	0.15 – 0.26%
EEQ (ng L^{-1})	0.98 – 1.73	1.13 – 1.44	0.57 – 1.53

Ozonation

Dissolved Organic Carbon

In Caboolture, which uses the highest ozone dose, a slight removal of 5 to 10 % of DOC was observed, but in the other plants DOC was not affected by the ozonation stage (Figure 2). At the doses employed, ozonation leads to limited mineralisation.

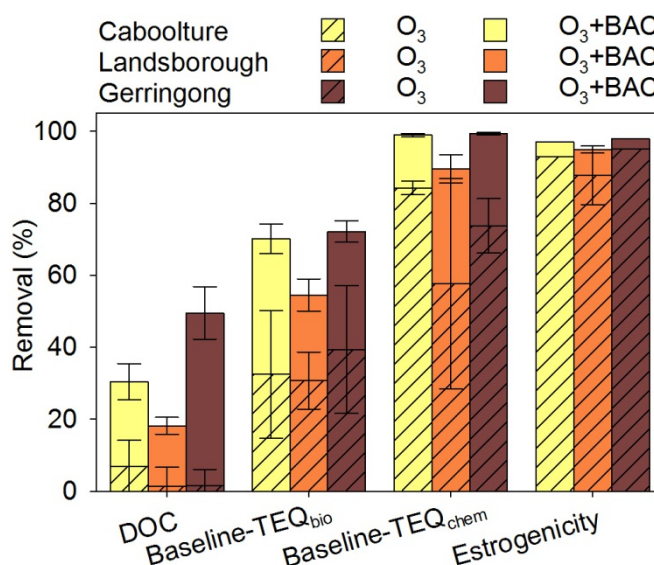


Figure 2. Removal of dissolved organic carbon (DOC), baseline-toxicity equivalent concentrations (baseline-TEQ_{bio} and baseline-TEQ_{chem}) and estradiol equivalent concentration (EEQ) compared to levels before ozonation (average of 3 independent samples $\pm$ standard deviation). No error bar means the values were below the limit of quantification, the removal is therefore a minimum observed.

Trace Organic Chemicals

In the three plants, ozonation achieved TrOCs removal to a degree depending on the compounds and the ozone dose. Some compounds were effectively removed in all plants regardless of the ozone dose while the removal of others was lower and generally depended on the ozone dose (Figure 3). It is clear that increasing the specific ozone dose leads to increasing removal, particularly for compounds that show lower removal. In ozonation processes, organic compounds can be oxidised via two mechanisms: reaction with molecular ozone (direct pathway) and reaction with hydroxyl radicals generated by ozone decomposition in water (indirect pathway). Molecular ozone reacts selectively with organic compounds and reaction rates vary over several orders of magnitude. On the contrary, hydroxyl radicals are not selective and reaction rates are typically $>10^9 \text{ M}^{-1} \text{ s}^{-1}$. However, due to $[\text{HO}^\bullet]/[\text{O}_3]$ typically in the range of 10^{-9} to 10^{-7} (von Gunten, 2003; Buffle *et al.*, 2006b), the indirect pathway is not always the dominant one. The compounds that were highly removed independently of the ozone dose (ie, diclofenac, sulfamethoxazole,

trimethoprim, propranolol, naproxen, carbamazepine, roxithromycin, erythromycin) have direct reaction rates with molecular ozone $>10^4 \text{ M}^{-1} \text{ s}^{-1}$ and/or have been previously shown to be easily removed from treated effluents even at low ozone dosage. These compounds have electron rich functional groups that are highly reactive with molecular ozone such as aniline (diclofenac, sulfamethoxazole), pyrimidine (trimethoprim), naphthalene (propranolol, naproxen), aromatic rings and double bonds (carbamazepine) and tertiary amines (roxithromycin, erythromycin). Oxidation of these compounds occurs almost exclusively via direct reaction with molecular ozone (Buffle *et al.*, 2006b; Hollender *et al.*, 2009).

Among the compounds that showed lower removal and/or dependency on the ozone dose, metoprolol, diuron, 2,4-D, atenolol, hydrochlorothiazide and caffeine have direct reaction rates with ozone $<10^3 \text{ M}^{-1} \text{ s}^{-1}$. Compounds with low direct reaction rates require exposition to higher ozone doses to allow their effective oxidation (Hollender *et al.*, 2009; Wert *et al.*, 2009). When the direct reaction rate constant with ozone decreases, the relative importance of oxidation by hydroxyl radicals increases and for values $<10^2 \text{ M}^{-1} \text{ s}^{-1}$, oxidation occurs almost exclusively via the indirect pathway (Buffle *et al.*, 2006b; Hollender *et al.*, 2009). During the initial phase of ozonation, ozone decomposes rapidly while reacting with the effluent organic matter and generates high amounts of hydroxyl radicals (Buffle *et al.*, 2006a; Buffle *et al.*, 2006b). Therefore, even at low ozone doses, some removal of compounds refractory to ozone can be observed (eg, 2,4-D, diuron, caffeine).

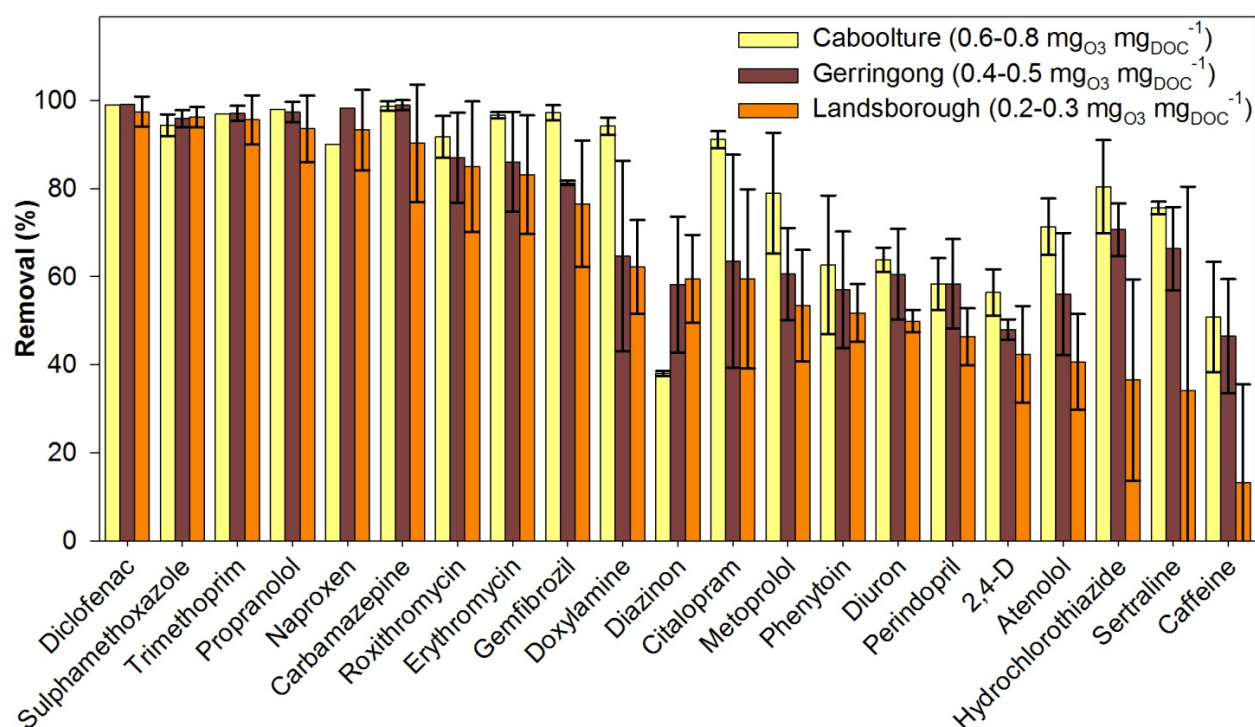


Figure 3. Removal of selected trace organic chemicals by ozonation (average of 3 independent samples $\pm$ standard deviation). No error bar means the values were below the limit of quantification, the removal is therefore a minimum observed.

Estrogenicity

More than 87% reduction of estrogenicity expressed as EEQ was observed in the ozonation stage of all the reclamation plants. Even at the lowest dose of 0.2 to 0.3 $\text{mg}_\text{O}_3 \text{ mg}_\text{DOC}^{-1}$, high removal of estrogenicity was achieved. This is consistent with previous findings showing that ozonation is a very effective treatment for the reduction of estrogenic activity of treated wastewater even at relatively low ozone doses (Snyder *et al.*, 2006; Escher *et al.*, 2009; Reungoat *et al.*, 2010). Indeed, several estrogenic compounds are very reactive with molecular ozone (Deborde *et al.*, 2005) and it has been suggested that the transformation products lose most of their estrogenic potential (Huber *et al.*, 2004). This finding can be rationalised by the fact that receptor mediated effects require a good steric fit between the ligand (TrOC or natural) and the receptor. Oxidation leads to a dramatic decrease in this interaction and thus to a decrease or complete loss of estrogenic potency (Lee *et al.*, 2008).

Non-Specific Toxicity

A decrease of baseline- TEQ_bio between 31 and 39% was observed after the ozonation stage in all three plants (Figure 2). This indicates that the mixture of TrOCs, their transformation products and possible formed oxidation by-products has a lower non-specific toxicity compared to the mixture of parent compounds. Therefore, there should be no concern regarding a possible increase in non-specific toxicity due to the generation of oxidation by-products.

during the ozonation of treated effluents. However, this assay does not take into account the formation of transformation products with specific and reactive modes of toxic action that could still present a hazard to the environment and human health. Specific toxicity is usually receptor mediated and oxidation leads to transformation products that typically have much lower affinity to receptors as shown for estrogenicity. In contrast, reactive intermediates can be formed and there is not enough knowledge on their effect.

The reduction of baseline-TEQ_{bio} was similar in the three plants and, contrary to what was observed for TrOCs, there was no trend following the ozone dose. The observed reductions were also in a similar range as previous findings on a Swiss WWTP, which indicates that these case studies allow some degree of generalisation (Escher *et al.*, 2009). Contrary to what was observed here, in the Swiss study ozone doses from 0.3 to 1 mg_{DOC}⁻¹ resulted in an increased reduction in non-specific toxicity from 25% to approximately 70%. It must be noted though, that the reduction of baseline-TEQ_{bio} was quite variable in the Swiss study, which is to be expected because of not only the ozone dose but also other determinants, such as the temperature and the type of TrOCs.

The baseline-TEQ_{chem} in the samples taken before the ozonation step, which were calculated from the relative potencies and concentrations of the TrOCs concentrations, were approximately three orders of magnitude lower than the baseline-TEQ_{bio} measured with the bioassays (Table 1). Thus, the quantified TrOCs explain less than 0.3% of the non-specific toxicity. This implies that more than 99.7% of the measured non-specific toxicity is contributed by other compounds present in the water. After ozonation, the fraction of toxicity explained by chemical analysis decreases by a factor of two to four, indicating that either the quantified chemicals were more degradable than the ones not quantified or that the chemicals are just transformed and their toxicity is reduced but not fully eliminated.

Previous studies on the ozonation of effluent organic matter showed that ozone reacts preferentially with its most hydrophobic fraction, leading to the formation of more hydrophilic compounds (Gong *et al.*, 2008; Rosario-Ortiz *et al.*, 2008; Domenjoud *et al.*, 2011) that have a lower non-specific toxicity. This is also evidenced by the Quantitative Structure Activity Relationship (QSAR) used to determine the non-specific toxic potential of individual compounds, which shows that it is strongly dependent on the compounds' hydrophobicity. Indeed, a ten-fold decrease in hydrophobicity, as would occur if for example a hydroxyl group is introduced into a molecule, would also lead to an approximately ten-fold reduction of toxicity of the transformation product (Escher and Fenner, 2011).

Gong *et al.* (2008) showed that ozonation had limited effect on the more hydrophilic fractions of effluent organic matter and therefore it is unlikely that toxic oxidation by-products are formed. It is generally assumed that effluent organic matter is too large to be bioavailable and thus does not cause any effects in the bioassays. However, smaller breakdown products and assimilable organic carbon are likely to be bioavailable and they will contribute to the baseline-TEQ_{bio}, provided they are also extracted with solid phase extraction.

Biological Activated Carbon Filtration

Dissolved Organic Carbon

Contrary to the ozonation, BAC filtration significantly removed DOC in the three plants (Figure 2). The removal increased with increasing EBCT, from around 20% in Landsborough (9 minutes) to reached almost 50% at Gerringong (45 minutes). The life of BAC filters can be divided in three phases (Simpson, 2008). During the first phase, organic matter is mainly removed by adsorption onto granular activated carbon. This phase is usually characterised by a high removal of organic matter. Rapidly, bacteria attach to the media and start growing, feeding on the organic matter and nutrients present in the water being filtered. In the meantime, the adsorption efficiency starts to decrease as the activated carbon capacity becomes exhausted. During this second phase, the removal of organic matter typically decreases with time. Eventually, the biomass is fully established in the filter and adsorption sites are exhausted. In this last phase, the removal of organic matter observed is predominantly due to biodegradation by the bacteria and is typically much lower than the removal observed in the initial phase. This third phase can last for several years because the granular activated carbon needs to be renewed only due to the attrition that occurs during backwashes.

In this study, the BAC filters investigated have been in use for several years and have filtered tens of thousands of bed volumes. The bacteria therefore had ample time to establish, which was confirmed by the reduction of dissolved oxygen concentration observed across the filters in Caboolture and Landsborough, 3.7 to 4.1 and 3.0 to 3.8 mg L⁻¹ respectively. Dissolved oxygen was not measured in Gerringong but it is reasonable to assume that bacteria have developed in these filters as well. A longer contact time allows the bacteria to degrade more organic matter as shown in previous studies on BACs (Seredynska-Sobecka *et al.*, 2006) and simulated soil filtration (Rauch and Drewes, 2004; Maeng *et al.*, 2008). However, the DOC removal in the BACs alone did not increase linearly with the contact time and a higher removal rate was observed for short EBCT (17±2%, 25±6% and 48±10% for 9, 18 and 45 minutes respectively). The easily (rapidly) biodegradable organic matter is likely to be removed first (ie, at short contact time) and the biodegradability of the remaining fraction decreases inducing lower biodegradation rates. Consistently, previous simulations of soil filtration showed a faster removal of organic matter in the first stages of the filtration (Rauch and Drewes, 2004; Maeng *et al.*, 2008).

Trace Organic Chemicals

Filtration through biological activated carbon was able to further remove all the remaining compounds after ozonation except perindopril in Landsborough (Figure 4). Removal varied from 0% to more than 99% depending on the compound and the plant. One of the parameters influencing the removal was the EBCT, the removal was higher for the filters with 18 and 45 minutes compared to 9 minutes. However, there was no clear increase between 18 and 45 minutes EBCT. The observed removal of DOC (Figure 2) suggests that the filters are in the third phase of their life in which organic matter is mainly removed by biodegradation. However, most of the compounds known to be poorly or moderately removed in WWTP were significantly removed in the filters, even with an EBCT as short as 9 minutes. Reungoat *et al.* (2011) observed high removal of pharmaceuticals over a period of two years in biological activated carbon filters treating non-ozonated and ozonated wastewater. These observations suggest that the bacterial community adapts to the biodegradation of compounds refractory in WWTP as it has been shown in simulated aquifer recharge (Rauch-Williams *et al.*, 2010). But even though it is hypothesised that the adsorption capacity of the activated carbon in the filters is largely exhausted, the removal of specific TrOCs is not correlated with the removal of bulk organic matter; TrOCs breakthrough can be observed much later than DOC breakthrough (Wang *et al.*, 2007). Moreover, TrOCs with different properties can show varying breakthrough times separated by tens of thousands of bed volumes (Snyder *et al.*, 2007).

Adsorption onto activated carbon is difficult to predict as the mechanism involves several types of interactions such as electrostatic interactions (between a charged compound and the activated carbon surface charges), van der Waals interactions and hydrogen bonding (Moreno-Castilla, 2004). Westerhoff *et al.* (2005) showed that removal efficiencies of TrOCs by powdered activated carbon tend to increase with increasing octanol-water partition coefficient ($\log K_{ow}$), but some protonated bases and deprotonated acids did not follow this general trend. This is partially due to the fact that charged compounds are more hydrophilic than their neutral forms. Therefore, the octanol-water distribution coefficient obtained at a given pH ($\log D_{ow}$) might be a better way to estimate adsorption potential of charged compounds. The $\log D_{ow}$ (pH 7) of selected compounds were calculated from their respective $\log K_{ow}$ and pKa according to the equations proposed by Scherrer and Howard (1977). In Figure 4, compounds are presented according to increasing $\log D_{ow}$ (pH 7) from left to right; no trend of increasing removal can be seen. The removal mechanisms of TrOCs in biological activated carbon filters remain unclear at this stage and could be a combination of adsorption and biodegradation, depending on the compounds.

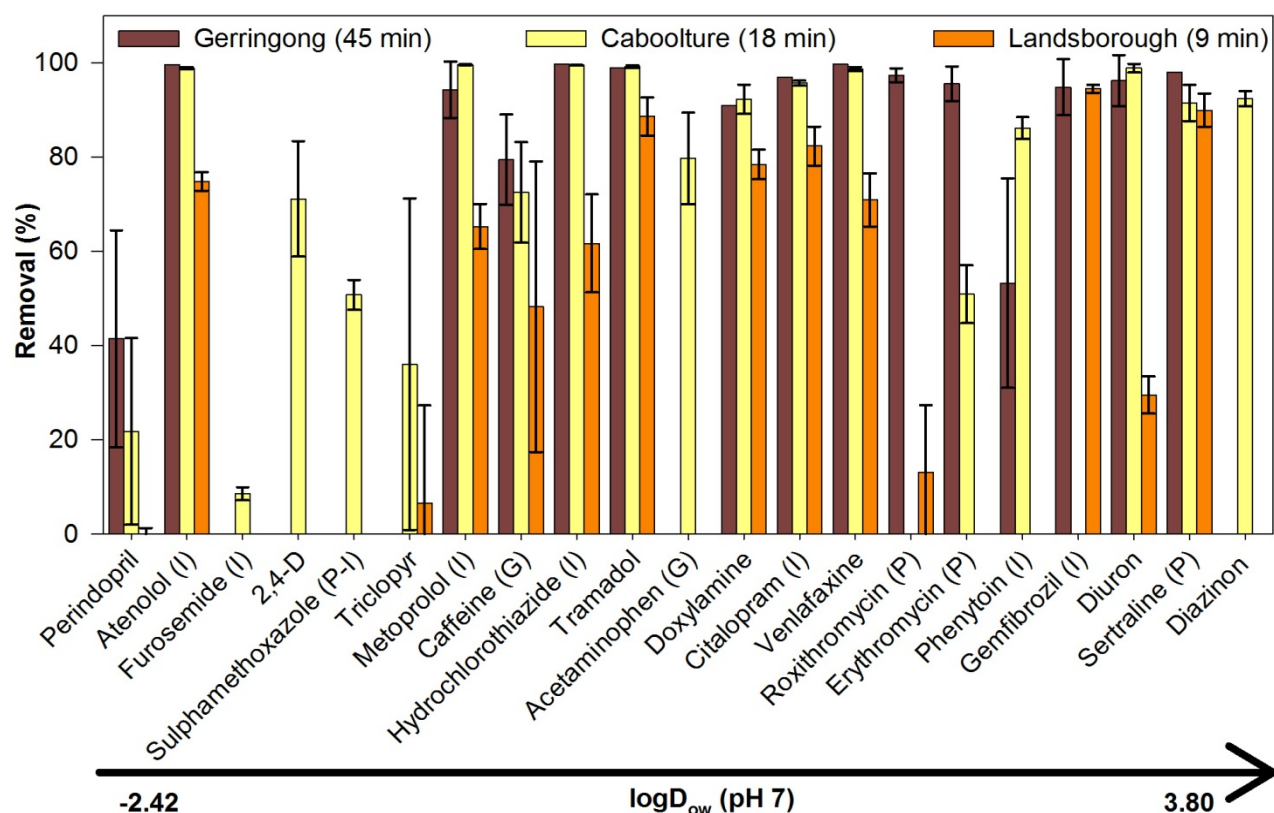


Figure 4. Removal of selected trace organic chemicals by biological activated carbon filters, empty bed contact time is indicated in the legend (average of 3 independent values $\pm$ standard deviation). No bar means a removal could not be calculated because concentrations were either too low or below the LOQ. Letters in brackets indicate removal generally observed in WWTP estimated from Onesios *et al.* (2009): P=poor (<20%); I=intermediate (20-80%); G=good (>80%).

Estrogenicity

Biological activated carbon filtration further reduced estrogenicity in Landsborough but it is difficult to assess its efficiency as the levels were already very low after ozonation. In the other two plants, the levels were even lower before BAC filtration and close to or below the quantification limit (0.03 ng L^{-1}) after it. In the samples that were above the LOQ before and after BAC filtration, the estrogenicity was only reduced by a factor of two to three. This indicates that the residual estrogenic compounds that were left after ozonation are not well biodegradable and are likely to be xenoestrogens and/or ethinylestradiol as those are less biodegradable than the natural estrogens (Liu *et al.*, 2009).

Non-Specific Toxicity

Biological activated carbon filtration significantly reduced the baseline- TEQ_{bio} after ozonation by $33 \pm 13\%$, $51 \pm 15\%$ and $54 \pm 13\%$ in Landsborough, Gerringong and Caboolture, respectively. In parallel, the DOC was reduced by $17 \pm 3\%$, $48 \pm 10\%$ and $24 \pm 6\%$ respectively, indicating that compounds contributing to the non-specific toxicity are preferentially removed or transformed to metabolites with lower toxic potential. In regards to specific toxicity and chemical analysis, ozonation as a single step would be sufficient for the removal of TrOCs. However, the non-specific toxicity shows us the interest of subsequent BAC filtration because this bioassay integrates the effect of all TrOCs present in the sample. Most transformation products cannot yet be quantified with chemical analysis and, as discussed above, will only marginally contribute to estrogenicity, but can still substantially contribute to non-specific toxicity. This is an important point and justifies the parallel application of bioassays when investigating the removal of TrOCs in various wastewater treatment processes. Similarly to TrOCs, the reduction of toxicity increased when EBCT increased from 9 to 18 minutes but not when it was increased to 45 minutes.

Conclusions

The investigation of three full-scale reclamation plants using ozonation followed by biological activated carbon filtration showed that:

- (i) the combination of chemical and biological treatment processes can improve treated effluents quality by removing DOC up to 50% and a wide range of TrOCs by more than 90%. It can also reduce non-specific toxicity by up to 70% and estrogenicity by more than 95%;
- (i) the non-specific toxicity of the ozonation by-products mixture was 30 to 40% lower than the parent compounds mixture suggesting that the by-products have a lower toxic potential;
- (ii) the BAC filtration is capable of further removing some of the TrOCs remaining after ozonation by up to 99% and also reducing the non-specific toxicity of the by-products mixture by up to 54%;
- (iii) increasing the ozone dose and filtration EBCT generally have a positive influence on the removal of DOC and TrOCs as well as on the reduction of non-specific toxicity, but there is no direct linear relationship. Therefore, increasing the ozone dose and EBCT further will not necessarily lead to substantive gains in water quality.

This study also confirms the benefits of using a combination of chemical analysis and bioanalytical tools to assess process efficiency since both do not always lead to the same conclusion; so their combined application gives a more detailed picture. Based on these results, it can be concluded that the combination of ozonation and biological activated carbon filtration could be employed to upgrade wastewater treatment plants for environmental protection or in water recycling schemes. However, the removal mechanisms of TrOCs in biological activated carbon filtration are still unclear. Further research is needed in this area to determine to what extent these compounds are adsorbed and/or biodegraded and if transformation products are formed.

Acknowledgements

This work was funded by the Urban Water Security Research Alliance under the Enhanced Treatment Project. The National Research Centre for Environmental Toxicology (Entox) is a joint venture of the University of Queensland and Queensland Health Forensic and Scientific Services. The authors acknowledge the following institutions and individuals who contributed to this study: Unitywater; Veolia Water Australia; Sydney Water Corporation and their staff for providing access to their reclamation plants and help with sampling; Yvan Poussade for his help to gain access to the Gerringong plant; Beatrice Keller and Jelena Radjenovic for their help in establishing the analytical method at the AWMC.

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Electrochemical Treatment of Reverse Osmosis Concentrate: Strategies to Minimise the Formation of Halogenated By-Products

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Summary

The capability of advance oxidation processes (AOPs) including electrochemical oxidation in degrading the organic matters in chloride-containing wastewaters, such as reverse osmosis concentrate (ROC), has been demonstrated in several studies. While the electrochemical processes benefit from higher chloride concentrations, the formation of chlorinated by-products has raised concerns due to their increased toxicity of the treatment effluent. This paper reports on two alternatives for minimising the formation of chlorinated by-products during electrochemical oxidation of ROC at boron doped diamond (BDD) anode which capable of generating $\text{OH}^\bullet$ in-situ.

Since the competition between $\text{OH}^\bullet$ and chlorine species (HOCl/OCl^-) will depend on the operating pH, electrochemical oxidation of ROC was performed at pH 2 and pH 6-7 in the first alternative. A complete removal of chemical oxygen demand (COD) was observed to be faster at acidic pH, ie, after 5.2 Ah L^{-1} , due to a more intense electro-hypochlorination. However, enhanced participation of $\text{OH}^\bullet$ seemed to improve mineralisation of organic matter at pH 6-7, although the formation of adsorbable organic halogen (AOX) and low molecular weight by-products such as trihalomethanes (THMs) and haloacetic acids (HAAs) was still observed. At the end of the experiments, the total concentrations of THMs and HAAs were decreased to 0.13 and 1.8 mg L^{-1} , and 0.6 and 3.1 mg L^{-1} at acidic and circumneutral pH, respectively.

In the second alternative studied, the contribution of HOCl , $\text{OH}^\bullet$ and peroxodisulfate ($\text{S}_2\text{O}_8^{2-}$) to electrochemical oxidation was investigated using electrodialysed ROC with added Cl^- , NO_3^- , and SO_4^{2-} ion mediators, respectively at circumneutral pH. The results confirmed that enhanced COD removal is obtained in the presence of HOCl/OCl^- as the dominant oxidant species. However, this led to an abundant formation of THMs and HAAs. On the other hand, in the presence of much lower Cl^- ions concentration less HOCl is generated at the anode, and improved dissolved organic carbon (DOC) removal is obtained due to the enhanced participation of $\text{OH}^\bullet$ and $\text{S}_2\text{O}_8^{2-}$. Furthermore, the concentrations of formed THMs and HAAs were significantly lowered. Therefore, the second alternative of combined electrodialysis and electrochemical oxidation, with or without the addition of SO_4^{2-} ions, may be more favourable for environmental application of electrochemical oxidation process.

Keywords

Electrochemical oxidation, reverse osmosis concentrate, electro-hypochlorination, halogenated organic by-products.

Introduction

Reverse osmosis (RO) membranes are widely applied in municipal wastewater reclamation as they represent an excellent barrier for organic and inorganic substances, bacteria and viruses. However, it also generates a waste stream, reverse osmosis concentrate (ROC), which normally represents 15-25% of the feed and contains almost of all the original dissolved salts and recalcitrant organics such as pesticides and pharmaceuticals (Bellona *et al.*, 2004). Direct and indirect disposal of these streams has raised concerns regarding its potential effects on the receiving water bodies (Khan *et al.*, 2009). Several alternatives for the treatment of ROC are being actively investigated, such as coagulation, adsorption, and advanced oxidation processes (AOPs) (Dialynas *et al.*, 2008, Perez-Gonzalez *et al.*, 2011). The latter has shown to be feasible and promising among other common technologies to degrade the organic matter, and it is based on the production of reactive hydroxyl radicals ($\text{OH}^\bullet$), which generally consume high amount of energy and/or chemicals.

On the other hand, in electrochemical oxidation, $\text{OH}^\bullet$ and other reactive oxygen species (ROS), eg, H_2O_2 , O_3 are formed in-situ at the electrode surface by the electrolysis of water. Thus, electrochemical oxidation is a chemical-free, robust and versatile process, capable of dealing with different types of wastewater at ambient temperature and pressure (Anglada *et al.*, 2009). Furthermore, this method is beneficial for the treatment of refractory waste streams containing high concentrations of Cl^- ions such as landfill leachate and ROC, since high concentration of Cl^- ions ($[\text{Cl}^-]$) lowers the ohmic resistance of the system, making it more energy-efficient (Anglada *et al.*, 2011, Perez *et al.*, 2010, Van Hege *et al.*, 2004).

Among the existing materials, the highest formation of $\text{OH}^\bullet$ has been reported for novel boron-doped diamond (BDD) electrode, characterised by the high O_2 overpotential electrode. However, although BDD electrodes are expected to generate more $\text{OH}^\bullet$ than conventional mixed-metal oxide (MMO) electrodes, competition between electro-chlorination and advanced oxidation by $\text{OH}^\bullet$ can be expected (Boudreau *et al.*, 2010). Generation of undesired chlorinated by-products such as trihalomethanes (THMs) and haloacetic acids (HAAs) has been reported in electrochemical oxidation of ROC at both BDD (Perez *et al.*, 2010) and MMO anodes (Bagastyo *et al.*, 2011).

In this paper, two alternative strategies were evaluated in attempts to minimise halogenated by-products formed during electrochemical oxidation of ROC on BDD anode by: (i) varying the operating pH; and (ii) separating chloride ions prior to electrochemical oxidation. The operating pH of the bulk liquid can be expected to affect the outcome of electrochemical oxidation since: (i) the equilibrium of active chlorine species (ie, $\text{Cl}_2/\text{HOCl}/\text{ClO}^-$) and thus mechanisms of electro-chlorination are pH dependable, with hypochlorous acid (HClO) becoming the main species at $\text{pH} < 7.5$; and (ii) scavenging of $\text{OH}^\bullet$ by HOCl/ClO^- and Cl^- ions and thus conversion of $\text{OH}^\bullet$ into less reactive inorganic radicals (eg, $\text{ClO}^\bullet$, $\text{Cl}_2^\bullet$, $\text{Cl}^\bullet$) is expected to be enhanced at acidic pH (De Laat *et al.*, 2004, Gonzalez *et al.*, 2011).

Therefore, in the first study electrochemical oxidation of ROC was investigated at acidic (ie, $\text{pH} \leq 2$) and circumneutral pH (pH 6-7). The second study focused on the evaluation of electrochemical oxidation in the presence of significantly lower concentration of Cl^- ions. In the case of low $[\text{Cl}^-]$, oxidation will rely more on the participation of $\text{OH}^\bullet$ as well as other electro-generated, such as peroxodisulfate ($\text{S}_2\text{O}_8^{2-}$) (Zhu *et al.*, 2008).

Electrodialysis was conducted as a pre-treatment of ROC prior to electrochemical oxidation, in order to lower the $[\text{Cl}^-]$. Therefore, in order to investigate the role of HClO, $\text{OH}^\bullet$, and $\text{S}_2\text{O}_8^{2-}$, electrochemical oxidation of electrodialysed ROC was performed in the presence of chloride, nitrate and sulfate ions, respectively. The performance of electrochemical oxidation was evaluated based on the removal of chemical oxygen demand (COD) and dissolved organic carbon (DOC), whereas the generation of halogenated by-products was determined as individual THMs and HAAs species, and adsorbable organic chlorine, bromine, and iodine (AOCl, AOBr, and AOI, respectively).

Experimental Methods

Two different ROC streams from advanced water treatment plant (AWTP) in Bundamba were used (ie, ROC collected with eight months time in between) for conducting the two studies, and were marked as ROC-1 and ROC-2, respectively, while the electrodialysed ROC from ROC-2 is marked as ROC-2_{ED}. Although the objective of electrodialysis was to separate Cl^- contained in the ROC, COD and DOC of ROC-2_{ED} were also lowered compared to the original ROC-2. This is due to the electrodialysis of charged and low molecular weight (MW) organics, which can particularly be expected in the case of non-selective ion exchange membranes (Zhang *et al.*, 2009). The main characteristics of these ROCs are shown in Table 1.

Table 1. Main characteristics of the two ROC and the ROC-2_{ED} used for in the experiments.

Measures, Unit	COD, mg L ⁻¹	DOC, mg L ⁻¹	pH	Conductivity, mS cm ⁻¹	$[\text{Cl}^-]$, mg L ⁻¹
ROC-1	136	42	7.7	6.1	1,386*
ROC-2	175	53	7.8	5.2	1,526
ROC-2 _{ED} **	145	46	6.8	1.4	142

* The concentration of bromide and iodide ions in ROC-1 was 1.6 mg L⁻¹ and 0.5 mg L⁻¹, respectively.

** The concentration of SO_4^{2-} and NO_3^- in ROC-2_{ED} was 90 and 2.1 mg L⁻¹, respectively.

The electrochemical cell consisted of two rectangular frames (inner dimensions of $20 \times 5 \times 2$ cm, effective volume = 190 mL), separated by a cation exchange membrane, and bolted together by two plates. The BDD anode was purchased from Adamant Tech., Switzerland (dimension: $4.8 \times 8.5 \times 0.2$ cm), while home-made stainless steel electrode of the same dimensions was used as cathode. In all experiments, electro-oxidation was operated in a well-mixed batch mode at room temperature ($25 \pm 1^\circ\text{C}$), with a galvanostatic control at high anodic current density, $J = 12.5 \text{ mA cm}^{-2}$ to ensure anodic potential (E_{AN}) higher than the threshold value for electro-generation of $\text{OH}^\bullet$, ie, >2.3 V vs. standard hydrogen electrode (SHE). The cathode medium was a 0.5 M H_2SO_4 solution. In the first part of the study, 5 L of ROC-1 were continuously recirculated during 96 h at a rate of 120 mL min^{-1} , and electrochemically oxidised at induced-acidic pH with no pH adjustment (ie, $\text{pH} \leq 2$) and controlled circumneutral operating pH (pH 6-7) by automatic dosing of 3 M NaOH.

In the second experiment, 1 L of ROC-2_{ED} was electrochemically oxidised during 10 h with the same recirculation rate, and at controlled pH 6-7. The supporting ion mediators added to ROC-2_{ED} were 0.05 M NaCl, NaNO_3 , and Na_2SO_4 . All samples were quenched by adding specific amounts of Na_2SO_3 solutions in order to eliminate further reaction of free available chlorine (FAC) before chemical analysis. COD, DOC, THMs (ie, trichloromethane (TCM), bromodichloromethane (BDCM), dibromochloromethane (DBCM), and tribromomethane (TBM)), and HAAs (ie, monochloroacetic acid (MCAA), dichloroacetic acid (DCAA), trichloroacetic acid (TCAA), bromochloroacetic acid (BCAA), monobromoacetic acid (MBAA), and dibromoacetic acid (DBAA)) were determined as described elsewhere in (Bagastyo *et al.*, 2011), while AOCl, AOBr and AOI (expressed as mg L⁻¹ of their corresponding halide ions) were analysed by a method described in Kristiana *et al.* (2009).

Results

Effects of Operating pH on Electrochemical Oxidation of ROC

Removal of COD and DOC

Under galvanostatic condition ($I = 510$ mA), constant E_{AN} was observed (ie, 3.4 - 3.7 V vs. SHE), and thus electro-generation of $\text{OH}^\bullet$ was ensured at both pH 2 and pH 6-7. Figure 1 depicts the profile of COD and DOC removal during the electrochemical oxidation in both pH conditions. An enhanced COD removal was observed at pH 2 compared to pH 6-7, resulting in a faster complete removal, ie, after 5.2 Ah L^{-1} . This was likely a consequence of more intense electro-chlorination and HOCl being the dominant species at pH 2, since the chemical reactivity of chlorine species decreases with the pH increase due to the dissociation of HOCl to less reactive OCl^- (Deborde and Von Gunten, 2008). However, although complete removal was already observed at 5.2 and 6.6 Ah L^{-1} , the corresponding DOC removal was only 48% and 59% at acidic and circumneutral pH, respectively.

At the final point ($Q = 10.9$ Ah L^{-1}), the remaining DOC was 15 and 13 mg L^{-1} , which was not oxidisable by the dichromates ($\text{K}_2\text{Cr}_2\text{O}_7$) in the COD test kit. While more intense electro-chlorination at acidic pH results in the enhanced overall oxidation of organics, it does not improve the mineralisation of the organic matter. A complex scenario of oxidation mechanisms can be expected since the participation of $\text{OH}^\bullet$ in the oxidation of organic matter is lowered, particularly at acidic pH. The formed HOCl/OCl^- and halide ions (mostly Cl^-) present in the bulk act as efficient scavengers of $\text{OH}^\bullet$ generated at the BDD anode, yielding $\text{Cl}_2^{\bullet-}$ species with $\text{HOCl}^\bullet$ and $\text{Cl}^\bullet$ as intermediates (De Laat *et al.*, 2004, Grebel *et al.*, 2010). Therefore, the less reactive radical and non-radical reactive halogen species (RHS) were responsible for the higher COD removal, but were less capable of breaking the organic bonds than $\text{OH}^\bullet$. On the other hand, participation of other oxidants, eg, $\text{S}_2\text{O}_8^{2-}$ can also be expected to enhance DOC removal in circumneutral pH (Zhu *et al.*, 2008).

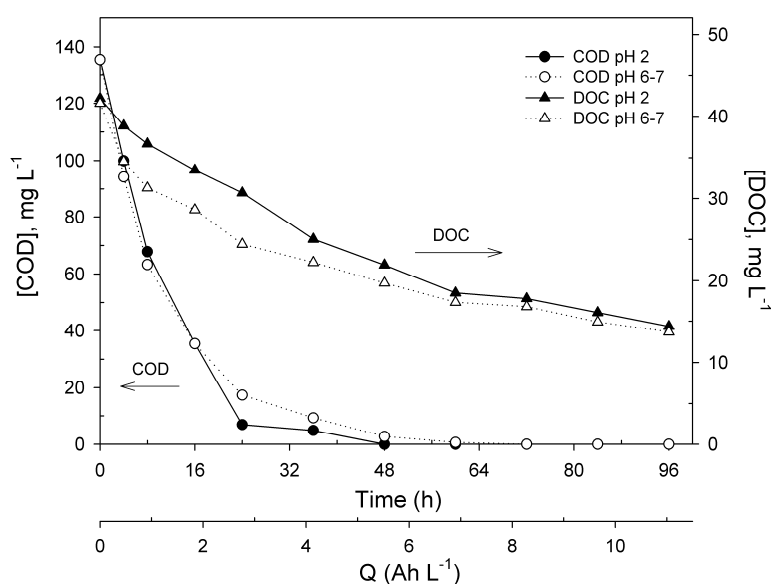


Figure 1. Removal of: (●○) COD and (▲△) DOC vs. supplied specific electrical charge (Q , Ah L^{-1}) during electrochemical oxidation of ROC at pH 2 (black) and pH 6-7 (empty).

Formation of AOCl, AOBr, and AOI

The formation of AOCl and AOBr is shown in Figure 2. AOCl was a major species contributing 80% to the AOX. AOCl was increased from the initial 0.9 mg L^{-1} to 30.7 and 28.7 mg L^{-1} at the end of electrolysis at acidic and circumneutral pH, respectively. Considering the ratio between the remaining DOC and formed AOCl, increased toxicity of organic matter can be expected by the presence of (poly)chlorinated by-products. Further degradation is possible by prolonging the oxidation time until no residual chlorine is present. Nevertheless, such operation would not be economically viable due to the very high electrical charge applied. On the other hand, although $[\text{Br}^-]$ and $[\text{I}^-]$ in the initial ROC were much lower than $[\text{Cl}^-]$ (Table 1), AOBr formation was still observed since electro-generated $\text{HOBr}/\text{OBr}^\bullet$ or $\text{HOI}/\text{OI}^\bullet$ species are usually more reactive than chlorine, especially with phenolic compounds (Gallard *et al.*, 2003). However, brominated organics are more susceptible to oxidation than their chlorinated analogues, and although AOBr was increased from the initial 0.2 to 1.9-2 mg L^{-1} after 0.8 Ah L^{-1} , it was further decreased to 1.2 mg L^{-1} at the end of oxidation (ie, $Q = 10.9$ Ah L^{-1}). In the case of AOI, their formation was only detected in the first two samples, ie, 0.3 mg L^{-1} .

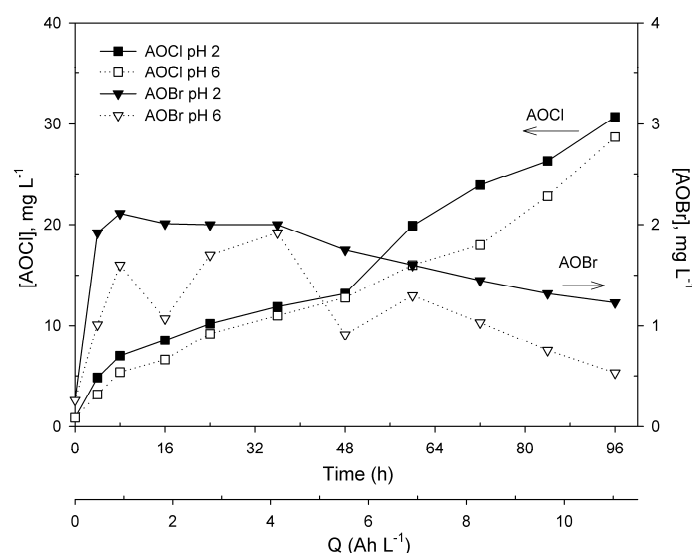


Figure 2. Formation of: (■□) AOCI and (▼▼) AOBBr vs. passed specific electrical charge (Q , Ah L^{-1}) during electrochemical oxidation of ROC at pH 2 (black) and pH 6-7 (empty).

Formation of THMs and HAAs

Figure 3 shows the formation of THMs and HAAs with their corresponding individual species after $Q = 5.2$ and 10.9 Ah L^{-1} at acidic and circumneutral pH. In the untreated ROC, the measured concentration of total THMs (tTHMs) and total HAAs (tHAAs) was already 0.13 and 0.16 mg L^{-1} , respectively. At pH 2, their concentrations were further increased to 0.4 and 3.2 mg L^{-1} , respectively after 48 h, and then degraded to 0.14 and 1.8 mg L^{-1} after 96 h of electro-oxidation (Figure 3a and 3c). Similarly, at pH 6-7, their concentrations were measured at 1.6 and 4.1 mg L^{-1} for THMs and HAAs, respectively. The higher concentration of THMs and HAAs at pH 6-7 is due to their increased formation by hydrolysis of other DBPs which were not measured in this study, eg, haloacetonitriles (HANs) and haloacetaldehydes (HAs) as studied by Chen (2011).

Further degradation of these by-products was observed, reaching 0.5 and 3.1 mg L^{-1} at the final point (Figure 3b and 3d). In all cases, TCM was a dominant species ($> 70\%$ of tTHMs), followed by BDCM, DBCM, and TBM. Similarly, polychlorinated HAAs (ie, TCAA and DCAA – comprising of 60-80% of tHAAs) were prominent among the HAAs measured in this study, while brominated HAAs, ie, BCAA, MBAA and DBAA were detected in lower concentrations. Previously, a continued formation of THMs and HANs in electro-oxidation of ROC at BDD electrode was reported (Anglada *et al.*, 2011). However, degradation of THMs and HAAs was eventually obtained in this study after 96 h ($Q = 10.9 \text{ Ah L}^{-1}$) at both pH. The oxidative degradation of brominated compounds will be faster than the chlorinated species (Tang and Tassos, 1997). Nevertheless, the enhanced degradation by volatilisation is possible for both chlorine and bromine species, particularly at acidic pH.

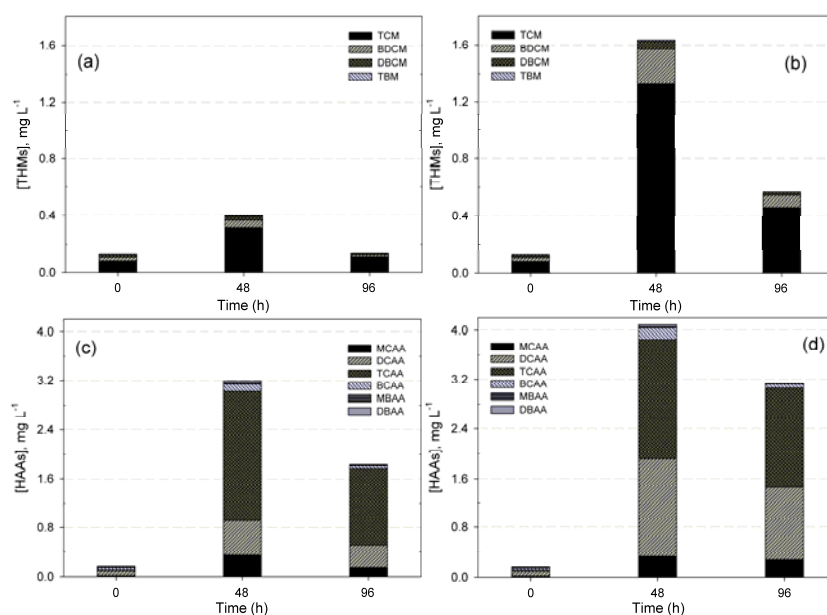


Figure 3. Formation of: (a) THMs at pH 2, (b) THMs at pH 6-7, (c) HAAs at pH 2, (d) HAAs at pH 6-7 vs. passed specific electrical charge (Q , Ah L^{-1}) during electrochemical oxidation of ROC.

Effects of Cl^- , NO_3^- , and SO_4^{2-} on Electrochemical Oxidation of ROC

Removal of COD and DOC

In the second part of the study, electrochemical oxidation of ROC-2_{ED} on BDD was performed at pH 6-7 in the presence of Cl^- , NO_3^- , and SO_4^{2-} in order to investigate the role of HOCl , $\text{OH}^\bullet$, and $\text{S}_2\text{O}_8^{2-}$, respectively. As seen in Figure 4, faster COD removal was obtained in the presence of Cl^- , with complete removal observed after 6 h ($Q = 3.3 \text{ Ah L}^{-1}$). On the other hand, the remaining COD was 80 and 40 mg L^{-1} in the presence of NO_3^- and SO_4^{2-} in ROC-2_{ED}, respectively. This indicates that removal of COD by indirect oxidation via electro-chlorination ($\text{HOCl}/\text{OCl}^\bullet$) is more efficient than via oxidation by $\text{OH}^\bullet$ and $\text{S}_2\text{O}_8^{2-}$. However, DOC removal was still incomplete.

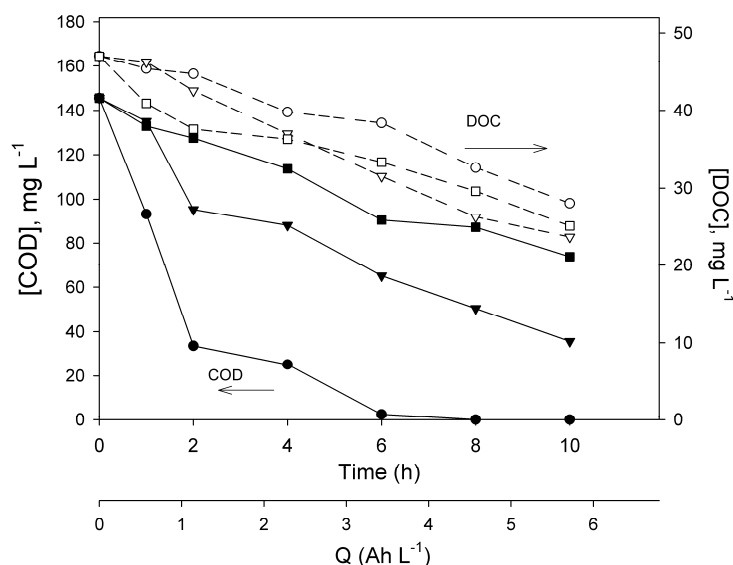


Figure 4. Removal of: COD (solid line) and DOC (dashed line) vs. passed specific electrical charge (Q , Ah L^{-1}) during electrochemical oxidation of ROC-2_{ED} at pH 6-7 in the presence of (●○) Cl^- , (■□) NO_3^- , and (▼▽) SO_4^{2-} .

Although the most efficient COD removal was obtained in the presence of Cl^- , higher DOC removal was achieved in the case of NO_3^- and SO_4^{2-} , ie, from 46 to approximately 25 mg L^{-1} , with faster removal was observed in NO_3^- in the first 4 h. The presence of SO_4^{2-} will lead to the generation of long-lived persulfate ions ($\text{S}_2\text{O}_8^{2-}$) (Zhu *et al.*, 2008), very strong oxidants capable of diffusing away from the electrode surface into the bulk liquid where they can further degrade the organics. In the case of NO_3^- , no electro-generation of long-lived oxidants would be expected. However, the observed mineralisation could be explained by the existence of $\text{OH}^\bullet$ at the vicinity of the BDD surface, whereas the scavenging effect from inorganic compounds is expected to be minimal in ROC-2_{ED}. These results are in agreement with previous studies on electrochemical oxidation using BDD for oxidative degradation of bisphenol-A (Murugananthan *et al.*, 2008) and propanil herbicides (Ozcan *et al.*, 2008).

Considering the higher ratio of active volume (V_{ACT}) versus total volume (V_{TOT}) in this second alternative investigated compared to the first one, faster removal of COD and DOC should be achieved. However, the organic matter contained in the ROC-2_{ED} seemed to be less amenable to oxidation compared to the original untreated ROC, indicating that 15% of organics that electrodesalysed together with Cl^- were likely low molecular weight compounds prone to oxidative degradation.

Formation of THMs and HAAs

Figure 5 shows the formation of THMs and HAAs at the end of electrochemical oxidation of ROC-2_{ED} in the presence of Cl^- , NO_3^- , and SO_4^{2-} ($Q = 5.6 \text{ Ah L}^{-1}$). The formation of THMs and HAAs was significantly decreased when $[\text{Cl}^-]$ was lowered from 1526 to 142 mg L^{-1} . In the presence of Cl^- , high concentrations of tTHMs and tHAAs were observed, ie, 1.1 and 9 mg L^{-1} , respectively, and several (poly)chlorinated by-products were measured, ie, TCM, TCAA, DCAA. On the other hand, although $[\text{Cl}^-]$ in the initial ROC-2_{ED} was low, the tTHMs and tHAAs was slightly higher in the presence of SO_4^{2-} than in NO_3^- , ie, 0.23 and 1.79 mg L^{-1} . This could be explained by the oxidation of Cl^- to HOCl species in the bulk by $\text{S}_2\text{O}_8^{2-}$, which then react with the organics.

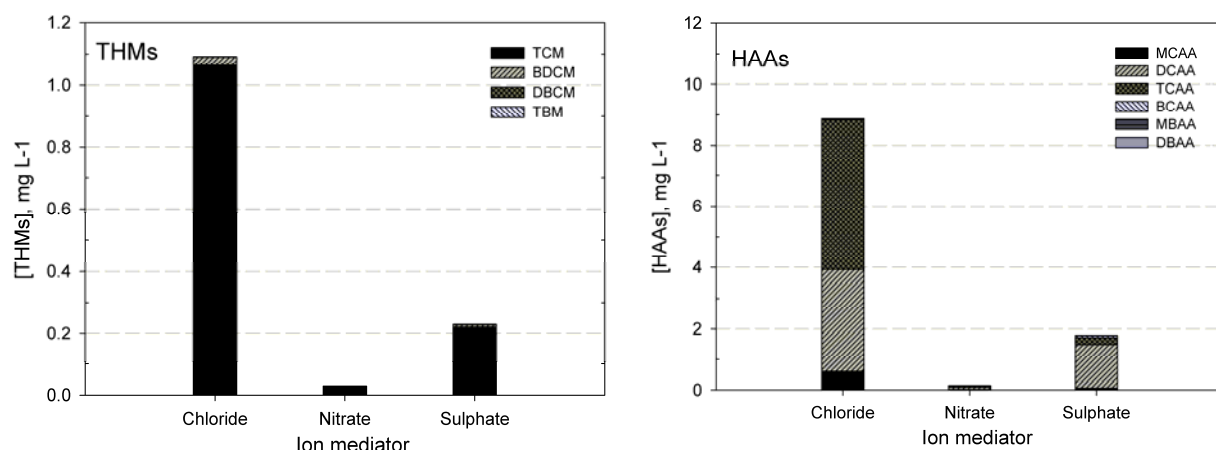


Figure 5. Formation of THMs (*left*) and HAAs (*right*) in electrochemical oxidation of ROC-2_{ED} in the presence of Cl⁻, NO₃⁻, and SO₄²⁻ measured at the end of the experiment ($Q = 5.6 \text{ Ah L}^{-1}$).

Conclusions

Electrochemical oxidation at BDD anode was investigated for the treatment of ROC and electro dialysed ROC at high applied current ($I = 510 \text{ mA}$). In the first study, although faster oxidation of organic matter was observed at acidic pH due to the enhanced participation of HOCl species, increased contribution of OH[•] and other ROS at circumneutral pH lead to the higher DOC removal. Based on the measured AOX, THMs and HAAs, it can be concluded that (poly)chlorinated by-products, ie, TCM, TCAA and DBAA were the dominant species contributing to the AOCI. However, both pH levels had no significant impact on the amount of AOCI formed, and AOB_r and AOI formation was limited by the initial levels of their corresponding halide ions. Considering an enhanced DOC removal observed, circumneutral pH may be more favourable for electrochemical oxidation process, although electro-hypochlorination of organic matter can still be expected.

In the second study, higher mineralisation was achieved for lower concentration of Cl⁻ ions, particularly in the presence of SO₄²⁻ ions which were oxidised at the electrode to reactive, long-lived S₂O₈²⁻ species. When NO₃⁻ ions were added to ROC-2_{ED}, enhanced participation of OH[•] was obtained. Furthermore, the formation of THMs and HAAs was significantly lowered for ROC-2_{ED}, with the concentration of tTHMs and tHAAs of 0.03-0.23 and 0.15-1.83 mg L⁻¹, respectively, at the end of the oxidation of ROC-2_{ED} in the presence of SO₄²⁻ and NO₃⁻ ions.

The separation of Cl⁻ (eg, by electrodialysis) prior to electrochemical oxidation on BDD anode will not only avoid or minimise the formation of THMs and HAAs, but also improve the degradation of DOC. Electrodialysis and electrochemical oxidation at BDD electrode is therefore suggested as an alternative strategy to the other common AOPs, eg, UV-based AOPs, for the treatment of ROC. The estimated energy consumption for applying both electrodialysis and electrochemical processes in this study was 51 kWh m⁻³ (with the electrochemical oxidation alone was between 34-37 kWh m⁻³). Further research and development on combining both electrodialysis followed by electrochemical oxidation in continuous processes may eventually become a viable option for treating chloride-containing wastewaters, such as ROC.

Acknowledgements

This research was supported by the Australian Research Council (grant LP0989159), Veolia Water Australia, Seqwater, Queensland Health Forensic and Scientific Service, and the Urban Water Security Research Alliance. The authors would like to thank Dr Ina Kristiana and A/Prof. Cynthia Joll of Curtin University for the collaboration regarding AOX analyses.

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Optimising Micropollutants Extraction for Analysis of Water Samples: Comparison of Different Solid Phase Materials and Liquid-Liquid Extraction

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Summary

Bioanalytical tools are widely applied for water quality monitoring. Typically, the water sample must be enriched prior to application in a bioassay. The aim of this study was to determine the extraction recovery of a wide variety of common environmental micropollutants (eg, pesticides, pharmaceuticals, hormones and industrial compounds) using a variety of solid phase extraction (SPE) materials. Commonly used liquid-liquid extraction (LLE) techniques were also compared. The results show that hydrophilic-lipophilic balanced (HLB) SPE cartridges in particular provide good recovery of a wide range of compounds. A method combining an Oasis HLB cartridge with a Supelco coconut charcoal cartridge produced an average extraction efficiency of 89-93% with more than 92-95% of the compounds recovered from spiked drinking and river water samples. The method is a useful wide-spectrum method to extract micropollutants from water samples for bioanalytical screening.

Keywords

Gas Chromatography-Mass Spectrometry (GC-MS), Hydrophilic-Lipophilic Balanced (HLB), Liquid Chromatography Tandem Mass Spectrometry (LC-MS/MS), Liquid-Liquid Extraction (LLE), Solid Phase Extraction (SPE).

Introduction

The increasing contamination of freshwater with thousands of natural and man-made chemical compounds is one of the key environmental issues facing humanity in the 21st century (Schwarzenbach *et al.*, 2006). One of the key challenges in addressing the issue of micropollutants in water is to develop the tools necessary to assess the presence and impact of these compounds on aquatic life and human health, a challenge that current chemical analytical methods can only meet to a limited extent. Combining chemical analysis with bioanalytical screening is fast becoming a well recognised approach to overcome some of the limitations of chemical analysis alone (Escher and Leusch, 2012). There are several advantages to bioassay analysis including the detection of non-target biologically active compounds and integration (to a certain extent) of mixture toxicity.

Bioanalytical tools can detect all biologically active compounds in a water sample, but only if those compounds are successfully extracted from the water phase and recovered during sample concentration. Previous studies have looked at the extraction efficiency of different methods, but are usually focused on specific classes of compounds (such as pharmaceuticals, Escher *et al.*, 2005) or a particular bioassay endpoint (such as estrogenic activity, Leusch *et al.*, 2006).

In this study, we looked at the recovery efficiency of various solid phase extraction material and liquid-liquid extraction for a wide range of micropollutants. This understanding is critical to our appreciation of bioanalytical results.

Materials and Methods

The project was carried out in two stages: stage 1 was designed to compare the recovery of different solid phase and liquid-liquid extraction techniques to allow selection of an optimal method; and stage 2 was designed to test the influence of a natural matrix (in this case river water) on the extraction efficiency of the selected method. The water samples were spiked with a wide variety of pesticides, pharmaceuticals, hormones and industrial compounds to determine the recovery efficiency of the method for compounds with a wide range of physico-chemical properties (Figure 1).

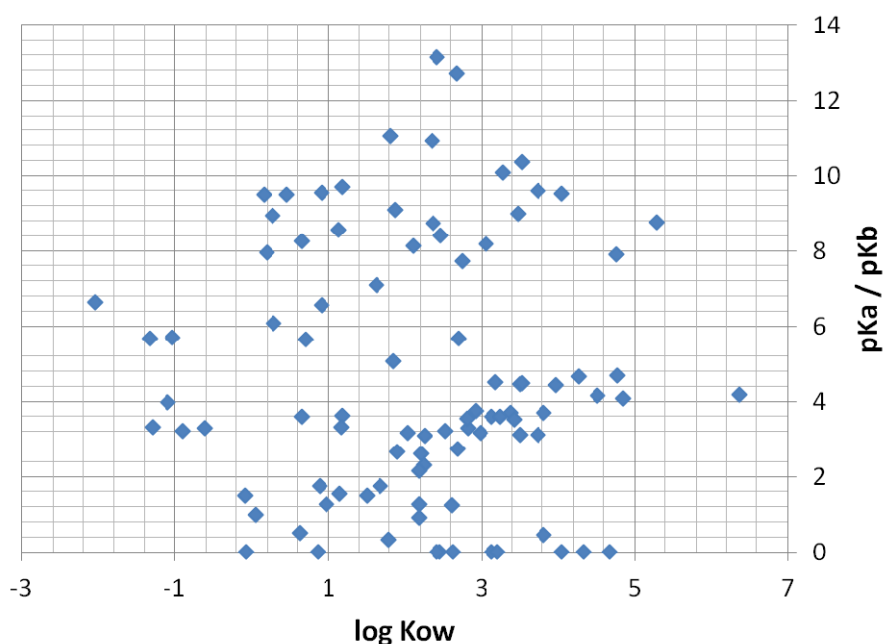


Figure 1. Acid dissociation constant (pKa) vs. octanol-water partition coefficient (Kow) for the pesticides spiked in this study, showing the wide range of physico-chemical properties of the spiked compounds.

Stage 1 – Comparison of Different Material and Extraction Methods with Pure Water

In the first stage, ultrapure (reverse osmosis) laboratory water was spiked with 179 pesticides at 1 µg/L and 84 pharmaceuticals and herbicides at 20 ng/L. The pH of the spiked water was adjusted to pH 2 or pH 7, and the samples were extracted in duplicates using eight different extraction methods, six solid phase extraction (Table 1) and two liquid-liquid extractions (LLE).

Table 1. Solid-phase extraction (SPE) cartridges used in this study.

Cartridge	Size (sorbent / cartridge)	Distributor	Catalogue Number
Oasis HLB	200 mg / 6 cc	Waters Corporation, NSW, Australia	WAT106202
Supelco SupelSelect HLB	200 mg / 6 cc	Sigma-Aldrich, NSW, Australia	54183-U
Varian Bond Elut PPL	500 mg / 6 cc	Agilent Technologies, VIC, Australia	12255001
Strata X	500 mg / 6 cc	Phenomenex, NSW, Australia	8B-S100-HCH
Supelco Supelclean Coconut Charcoal	2 g / 6 cc	Sigma-Aldrich, NSW, Australia	57144-U
Varian Bond Elut Carbon	500 mg / 6cc	Agilent Technologies, VIC, Australia	12252201

For the solid phase extraction, the SPE cartridges were pre-conditioned by passing 2 × 5 mL of acetone:hexane 50:50, 2 × 5 mL of methanol, and 2 × 5 mL of ultrapure water by gravity. One L of the spiked water was then passed by vacuum (up to 2.6 kPa) through the 6 mL cartridges (Table 1). After passing the full one L, the cartridges were air-dried on the manifold for a minimum of 30 min, until visibly dry, and stored at 4°C until ready for the next step. The cartridges were eluted with 2 × 5 mL methanol and 2 × 5 mL acetone:hexane 50:50, allowing the solvent to pass through the sorbent bed by gravity first and finished by applying vacuum to pull all the solvent off the cartridge. The 20 mL eluate was pooled and evaporated to dryness at 40°C under gentle nitrogen stream, reconstituted in 2.5 mL of methanol, and split into three aliquots for the different analysis methods: 1 mL for liquid chromatography – tandem mass spectrometry (LC-MS/MS) analysis; 1 mL (solvent-exchanged into dichloromethane) for gas chromatography – tandem mass spectrometry GC-MS/MS analysis; and 0.5 mL for archiving.

For the liquid-liquid extraction, 500 mL of the spiked water was mixed with 200 mL of either ethyl acetate (EthA) or methyl tert-butyl ether (MTBE) on a shaker for 30 min. The solvent was recovered using a separatory funnel, and the operation repeated twice more with 50 mL of solvent. The pooled 300 mL solvent was evaporated to dryness in a rotary evaporator, reconstituted in 2.5 mL of methanol, and split into three aliquots for the different analysis methods: 1 mL for LC-MS/MS analysis; 1 mL (solvent-exchanged into dichloromethane) for gas chromatography – mass spectrometry (GC-MS) analysis; and 0.5 mL for archiving.

Stage 2 – Performance of the Selected Method with Spiked Drinking and River Water

After selecting a combination of Waters Oasis HLB and Supelco Supelclean coconut charcoal SPE methods based on the results of Stage 1 and those from a previous project (NWC, 2011), the performance of the established method was tested in more relevant environmental matrices such as drinking and river water.

Metropolitan tap water and river water samples were collected. The river water was filtered (Millipore AP20 filter) and half the samples were spiked with 12 endocrine disrupting compounds (hormones and industrial xeno-estrogens) at 50 ng/L, 215 pesticides at 0.8 µg/L, and 88 pharmaceuticals and herbicides at 30 ng/L. The pH of the river water samples was adjusted to pH 2 or pH 7 (only pH 7 for the drinking water), and the samples were extracted in duplicates using the following SPE method.

The SPE cartridges were pre-conditioned separately by passing 2 × 5 mL of acetone:hexane 50:50, 2 × 5 mL of methanol, and 2 × 5 mL of ultrapure water by gravity. The cartridges were then stacked, with a Waters Oasis HLB cartridge on top of a Supelco Supelclean coconut charcoal cartridge. One L of the spiked and unspiked water samples were passed by vacuum (up to 2.6 kPa) through two cartridges in series. After passing the full one L, the cartridges were separated and air-dried on the manifold for a minimum of 30 min, until visibly dry, and stored at 4°C until ready for the next step. The cartridges were eluted with 2 × 5 mL methanol and 2 × 5 mL acetone:hexane 50:50, allowing the solvent to pass through the sorbent bed by gravity first and finished by applying vacuum to pull all the solvent off the cartridge. The two 20 mL eluates were pooled and evaporated to dryness at 40°C under gentle nitrogen stream, reconstituted in 3 mL of methanol, and split into three aliquots for the different analysis methods: 1 mL for LC-MS/MS analysis; 0.5 mL (solvent-exchanged into dichloromethane) for pesticides GC-MS analysis and 0.5 mL (solvent-exchanged into dichloromethane) for endocrine disrupting compound GC-MS analysis; and 1 mL for archiving.

Chemical Analysis

All samples from Stages 1 and 2 were analysed using standard methods at the NATA-accredited Queensland Health Forensic and Scientific Services (QHFSS) laboratory.

Pesticides were analysed by GC-MS for multi-screening of organochlorine, organophosphorus, synthetic pyrethroid pesticides and some herbicides using a standard protocol (QHFSS Document No 16315: Organochlorine, Organophosphorus and Synthetic Pyrethroid Pesticides, Urea and Triazine Herbicides and PCBs in Water).

Pharmaceuticals and herbicides were analysed by LC-MS/MS using a standard protocol (QHFSS Document No 27701: PPCP in Water, Preparation and Analysis by SPE and LCMSMS).

Endocrine disrupting compounds (only spiked in stage 2) were derivatised with N,N-bis(trimethylsilyl)trifluoroacetamide (BSTFA) + 1% trimethylchlorosilane (TMCS) and analysed by GC-MS using a standard protocol (QHFSS Document No 25391: Determination of Endocrine Disrupting Compounds in Effluent, River and Recycled Water).

Total (and dissolved) organic carbon was measured using a Shimadzu TOC-V CSH total organic carbon analyser at the Smart Water Research Centre.

Results and Discussion

Stage 1 – Comparison of Different Materials and Extraction Methods with Pure Water

The results show that most compounds are well recovered by most of the SPE materials selected for comparison, and confirm the wide retention spectrum of HLB sorbent (Figure 2). The Supelco Supelclean coconut charcoal cartridge retained the least number of compounds and had the lowest median recovery, but had been previously shown to be relatively effective at capturing amines such as NDMA (NWC, 2011). For this reason, we chose to combine an Oasis HLB cartridge with the Supelco coconut charcoal cartridge in Stage 2.

Lowering the pH to 2 resulted in a minor improvement in both median extraction recovery and the number of compounds recovered (Figure 2).

Both LLE techniques yielded an average extraction recovery for the compounds selected in this study that was similar to that of the SPE methods (Figure 2), however the recovery efficiency was significantly more variable between different compounds (as indicated by the larger standard error with the LLE samples, Figure 2, left) and twice as many compounds were not recovered with LLE compared with SPE methods (Figure 2, right). The LLE methods also used significantly more solvent than the SPE methods (600 mL / 1 L vs. 20 mL / 1 L) and left an insoluble residue after evaporation. Liquid-liquid extraction can also create emulsions at the interface between the solvent and the water, which can make extraction of some environmental water samples difficult (Wells, 2002).

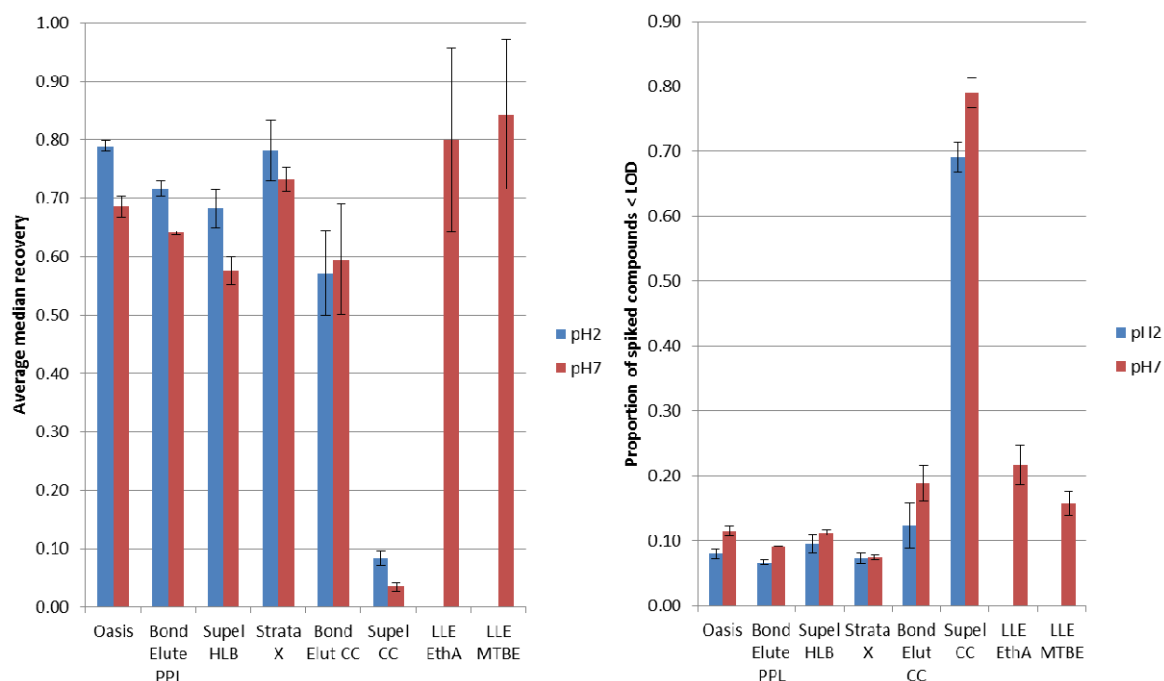


Figure 2. Average median recovery (left) and proportion of spiked compounds below the limit of detection (right) $\pm$ standard error for the different solid phase media (Oasis HLB, Bond Elute PPL, Supelco HLB, Strata X, Bond Elute CC and Supelco CC) and liquid extraction methods at pH 2 (blue) and pH 7 (red).

Stage 2 – Performance of the Selected Method with Spiked Drinking and River Water

A few pesticides and pharmaceuticals were detected at low ng/L concentrations in the river water sample (data not shown), and the spike recovery is therefore calculated as (spiked – unspiked) / spiked concentration.

The recovery of the combined Oasis HLB / Supelco CC method was very good, with an average recovery of 89-93% in both river and drinking water (Figure 3). The more complex river water sample did not affect the recovery efficiency, suggesting that the extraction method is sufficiently robust to deal with a moderate level of organic matter (Table 2).

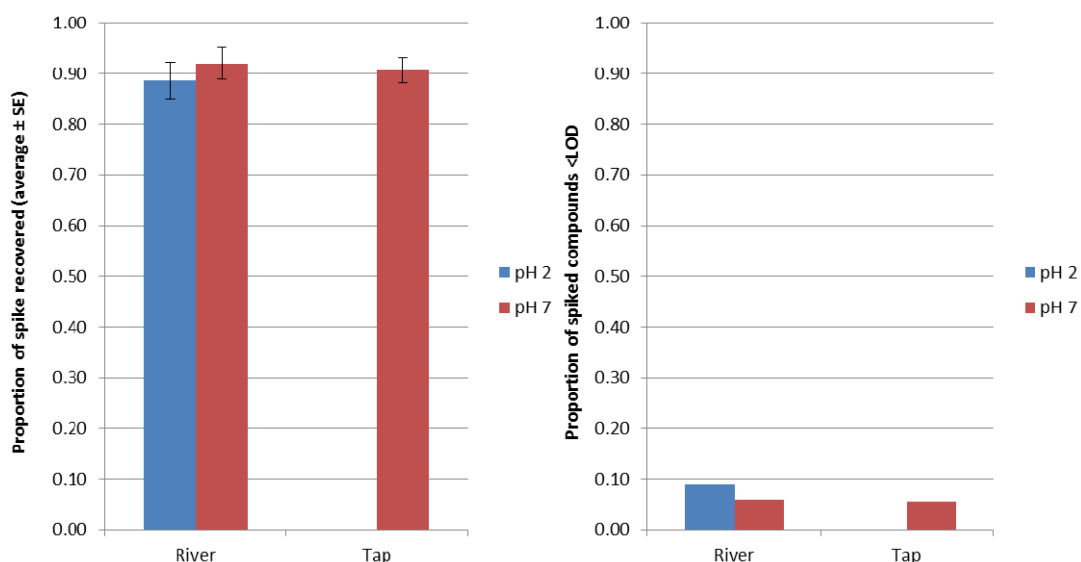


Figure 3. Average recovery $\pm$ standard error (left) and proportion of spiked compounds below the limit of detection (right) for compounds spiked in river and tap water at pH 2 (blue) and pH 7 (red) using a tandem Oasis HLB / Supelco CC solid phase extraction cartridge.

Table 2. Total organic carbon in the water samples used in this study.

Sample Type	Total Organic Carbon (TOC)
Ultrapure laboratory water	0.15 mg/L
Tap water	2.05 mg/L
River water (filtered)	8.31 mg/L

Conclusions

The use of a combined Oasis HLB and Supelco coconut charcoal cartridges in series results in good recoveries of a wide spectrum of micropollutants even in environmental water samples. This extraction technique provides a sound method for extraction and concentration of water samples for bioassay analysis. Further work will develop an empirical-based model to predict the recovery of compounds of different chemistry in the various SPE sorbents tested in this study.

Acknowledgments

This project was supported by the Urban Water Security Research Alliance. We thank Dr Nicole Knight, Steve Carter, Vince Alberts for laboratory support.

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Implications of Resource-Efficient Technology on Peak Water Demand and Water-Related Energy Demand

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Summary

This paper uses a detailed water end use data registry generated from the South East Queensland Residential End Use Study (SEQREUS) to examine (a) peak hourly and daily demand, and (b) water-related energy demand. The impact of water-efficient fixtures and appliances, as required by current building codes in Queensland, is also explored. The four peak demand days selected had increasing peak day factors of 1.3, 1.5, 1.6 and 1.7. The range of these values is slightly lower than those used in the Department of Environment and Resource Management (DERM) guidelines where the peak day factors range from 1.5 to 2.3, suggesting that the frequency and volume of peaking factors may be lower than those that are currently being used for network distribution design; due likely to the high penetration of water-efficient technology and growing water conservation awareness by consumers. This type of knowledge can facilitate the optimisation of infrastructure design and sizing and inform the subsequent deferral of such assets. In terms of water-related energy demand, it was found that the hot water components of showers, and to a lesser extent, taps and clothes washers were the most influential. Shower energy consumption for electric cylinder hot water systems (HWS) and solar electric-boosted HWS was 810 and 351 kWh/p/year, respectively. The type of hot water system was significant in dictating the volume of energy-related carbon emissions, with the results confirming the significant impact that electric storage water heating has on total household energy consumption and related carbon emissions. Substantial savings can be achieved by substituting water (eg, high star rating clothes washers and shower heads) and energy (eg, solar hot water system) efficient appliances in the home. Knowledge on the end uses that are influencing peak water and energy demand can: 1) facilitate the optimisation of infrastructure design and sizing and inform the decision process regarding subsequent deferral of these expensive assets; and 2) underpin future sustainability codes for new buildings.

Keywords

Water end use consumption, water micro-components, water demand management, peak demand, urban water supply design.

Introduction

Accurate and up-to-date peak demand data is essential to ensure that future mains water supply networks reflect current usage patterns and are designed efficiently from an engineering, environmental and economic perspective. Similarly, there is a lack of measured data on simultaneous water and energy consumption from residential end uses such as shower, washing machines and taps. Knowledge on peak demand and water-related energy use is particularly relevant in jurisdictions where all new developments require the installation of energy and water-efficient fixtures to achieve a more sustainable water and carbon footprint. Given this, the aim of this paper was to identify the water end uses which drive a) peak hour and day demand and b) energy demand using over 18 months of water consumption data obtained from high resolution smart meters installed in 252 residential properties across South East Queensland (SEQ), Australia. A further aim was to determine the peak day diurnal demand patterns at an end use level of resolution and the carbon emissions arising from water-related activities in the home. Finally, the impact on peak and energy demand by water-efficient fixtures and appliances, as now required by building codes in Queensland, is also explored.

Methodology

The data for this paper was generated from the Beal and Stewart (2011) South East Queensland Residential End Use Study (SEQREUS) located in the south-eastern corner of Queensland, Australia. A total of 252 households were used, providing a good representation of SEQ households with a varying range of household occupancies, family composition and household income categories. Further discussion on the research methods for acquiring the end use data is provided in Beal *et al.*, (2011) and Beal and Stewart (2011).

Average day (AD) diurnal demand patterns, at an end use level, were generated using a software application specifically developed for this study. The Smart Water Information Portal, or SMIP, is a tool that enables total water use to be extracted and interrogated from all homes being metered for the study (as distinct from end uses from individual homes). The SMIP tool was used to obtain a complete timeline of average daily total water consumption sourced from 567 days of continuous logging from up to 230 homes per day, equating to over 93,000 data points. From this, peak demand (PD) days and associated PD/AD ratios were determined.

Energy demand was determined by multiply published energy intensity values (in Wh/L) for each end use by the known (measured) average water volume (L) of each end use for each home. Using published greenhouse gas (GHG) emission factors and methods presented in the Australian National Greenhouse Accounts report (Australian Government, 2011) the energy use values were converted into GHG emissions. The GHG emission factors for electric cylinders (coal-fired power stations), gas and solar (electric boosted) were 1, 0.197 and 0.138, respectively. A number of scenarios were devised to determine the impact on carbon emission reductions from various water and energy-efficient technologies (Table 1). Percentage savings from the base case scenario (worst case scenario of no efficient strategies and electric HWS) were calculated when comparing to a range of sequentially applied water and energy efficiency intervention strategies.

Table 1. Scenario description and key assumptions for energy demand reduction interventions.

Scenario Number	Intervention Scenario	Assumptions
S1	Conversion to energy-efficient solar HWS	a) Solar panels with electric-boosted storage system; b) direct replacement of electric HWS; c) long term average solar radiation data taken from Brisbane airport and assuming same characteristics across SEQ d) 38 days or 10% of year with insufficient insolation.
S2	Water-efficient shower heads	a) Substitute high flow shower head with low flow shower head of flow rate at 0.09 L/s; b) co-efficient of 1.2 applied to compensate for increased duration due to lower flows.
S3	S2 + Water-efficient clothes washer	a) CW internally heats cold water; b) front load only; c) cold water connection only; d) directly substituting dual connected front or top load CW.
S4	S3 + Tap aerators	a) Tap flow rate fixed value of 0.08 L/s (Australian Government 2011).
S5	S4 + Shower temperature reduced to average of 37 °C	a) Original shower temperature set at 40 °C (Flower 2009); b) existing shower head efficiencies (eg, low or high flow roses) remain.
S6	S5 + Energy-efficient dishwashers (DW)	a) > 3 star rated machines considered 'energy-efficient'; b) two efficiency clusters generated from SEQREUS data: ≤ 3 star and >3 star rated.

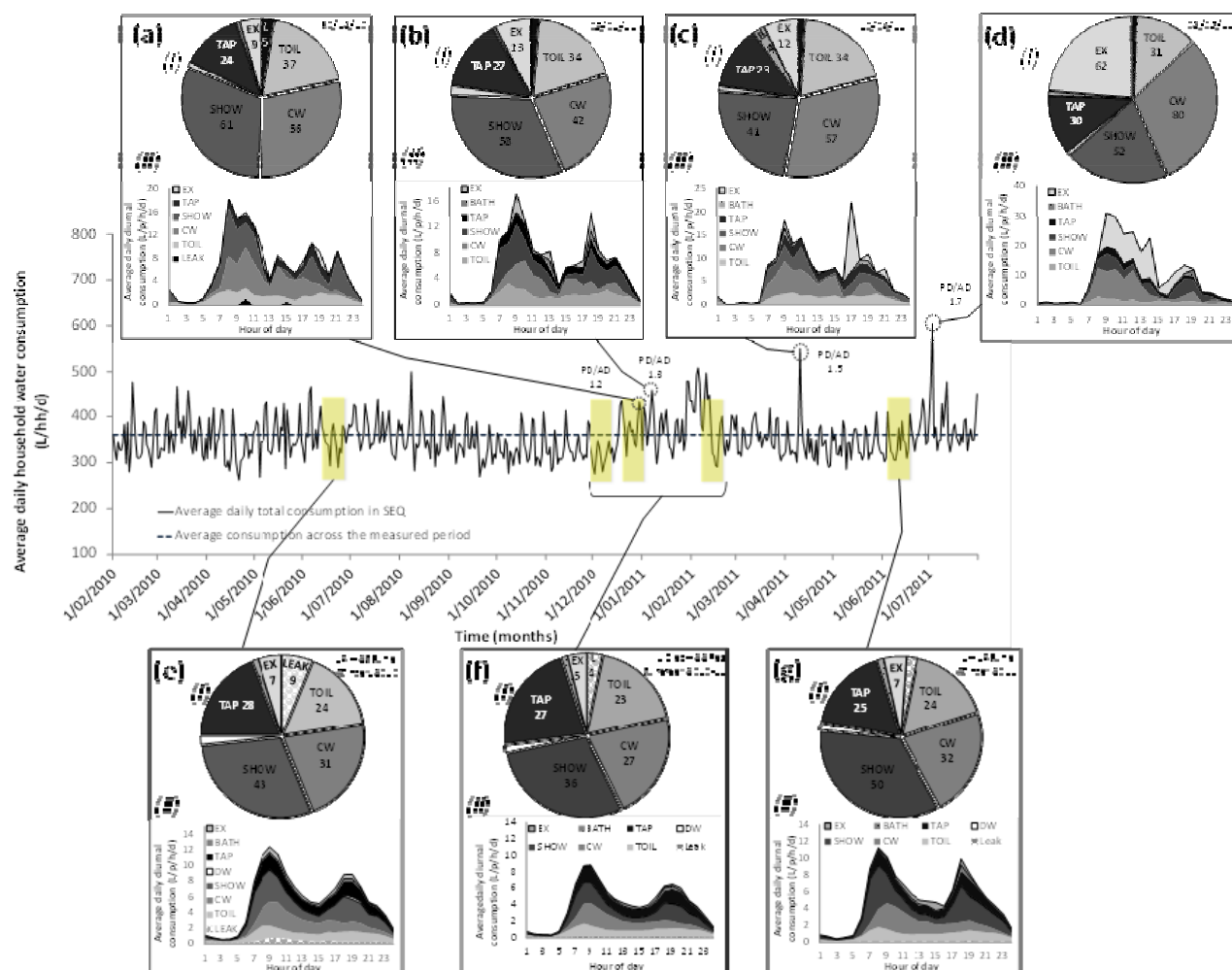
Results and Discussion

Peak Hour and Day Demand

The timeline of daily total household water consumption (in L/hh/d) recorded from all functioning data loggers is presented as a combined SEQ average (Figure 1); the corresponding climate data is presented for the combined SEQ average in Figure 2.

In general, water usage increased in the summer months (December to February) and decreased during the winter months (June to August). The insets (a) to (d) presented in Figure 1 provide per capita water end use breakdowns (L/p/d) for a 24-hour period for four days of above average water consumption: 31/12/2010 (Figure 1a), 07/01/11 (Figure 1b), 10/04/2011 (Figure 1c), and 02/07/11 (Figure 1d). Also shown is a reproduction of the winter 2010 end use pie chart (Beal *et al.*, 2011) to offer a 'baseline' dataset for comparison with the end use breakdowns from peak demand days (Figure 1e). Note that the pie charts and diurnal usage patterns are generated from a smaller sample size (average $n = 25$) due to the human resource requirements to undertake the Trace Wizard analysis which generates end use breakdowns. As such, the pie charts and diurnal usage patterns have been included to provide an indication or snapshot only of the type of end use activities that typically contribute to peak hour and peak day demand.

Data presented in the pie charts shown in insets (a), (b) and (c) in Figure 1 indicate that while external use is above the average (Figure 1e-g), demand was largely driven by increased indoor water usage, from clothes washers (CW) and showers (SHOW). There was little variation in tap usage across all pie chart snapshots, suggesting that this is not likely to be an end use that would drive peak day demand, although it may be attributable to peak hour demand especially if they are external tap fixtures. Note that the 'tap' category in this paper refers to indoor tap use or low external tap use. Large tap events have been allocated to the external water use category as this is typically the location of such large flows from taps (Mayer and DeOreo, 1999).



Notes: EX = external, TOIL = toilet, CW = clothes washer, SHOW = shower, DW = dishwasher, L = leak.

Figure 1. Timeline for total water consumption showing (i) water use breakdown in L/p/d and (ii) average daily diurnal water use (L/hh/d) for the selected peak demand days of (a) 30/12/10, (b) 07/01/11, (c) 10/04/11, and (d) 02/07/11 and (e) baseline data for winter 2010.

Diurnal Breakdown of Peak Day and Hour Demand

The uniform, twin peak periods occurring in the morning and afternoon which are typically seen in average day demand diurnal patterns (Figure 1e-g) are not so evident for the four peak day demand diurnal patterns shown in Figure 1a-d. The peak day diurnal patterns exhibited a frequent occurrence of peak events throughout the day, particularly for the external water usage. Large outdoor usage events are more likely to drive peak hour demand, relative to the overall peak day demand. The latter is more likely to be driven by indoor water uses such as shower and clothes washer as seen in the 30/12/10 and 07/01/11 pie and diurnal charts (Figure 1a,b).

Others have also drawn similar conclusions on the different end uses driving peak hour versus peak day demand (Cole and Stewart, 2012; Polebitski *et al.*, 2011; Lucas *et al.*, 2010). One other important characteristic of the peak day diurnal patterns is the timing of the external water use activities. In SEQ where the data was sourced from, there is a current restriction on irrigation between 10 am and 4 pm. Results demonstrate a degree of non-compliance with water restrictions during this timeframe and further, this practice appeared to have increased rather than decreased over the 18 month monitoring period judging by the consumption timeline shown in Figure 2.

Influence of Climate on Total Household Consumption

A timeline of average temperature, rainfall and daily total consumption for each month of the study is presented in Figure 2. The four peak days selected for analysis are indicated by the hatched triangles on the water consumption curve. There is a weak relationship between increased temperature and peak demand days, although this is not consistent across the timeline. There is a stronger association between temperature and total household water consumption for the warmer months eg, January to April 2010 and November to April 2011. Cole and Stewart (in review) report a strong correlation between temperature and bulk water demand for the Hervey Bay region of SEQ. Others have also observed this relationship between temperature and residential water consumption (Water Corporation, 2011; Willis *et al.*, 2011b; Adamowski, 2008). It should be noted that the high rainfall events occurring in December 2010 and January 2011 contributed to major flooding across SEQ, effectively eliminating the need for irrigation during the period that typically characterises peak day demand in this sub-tropical region of Australia (Beal and Stewart, 2011; Willis *et al.*, 2011b; Beal *et al.*, 2011).

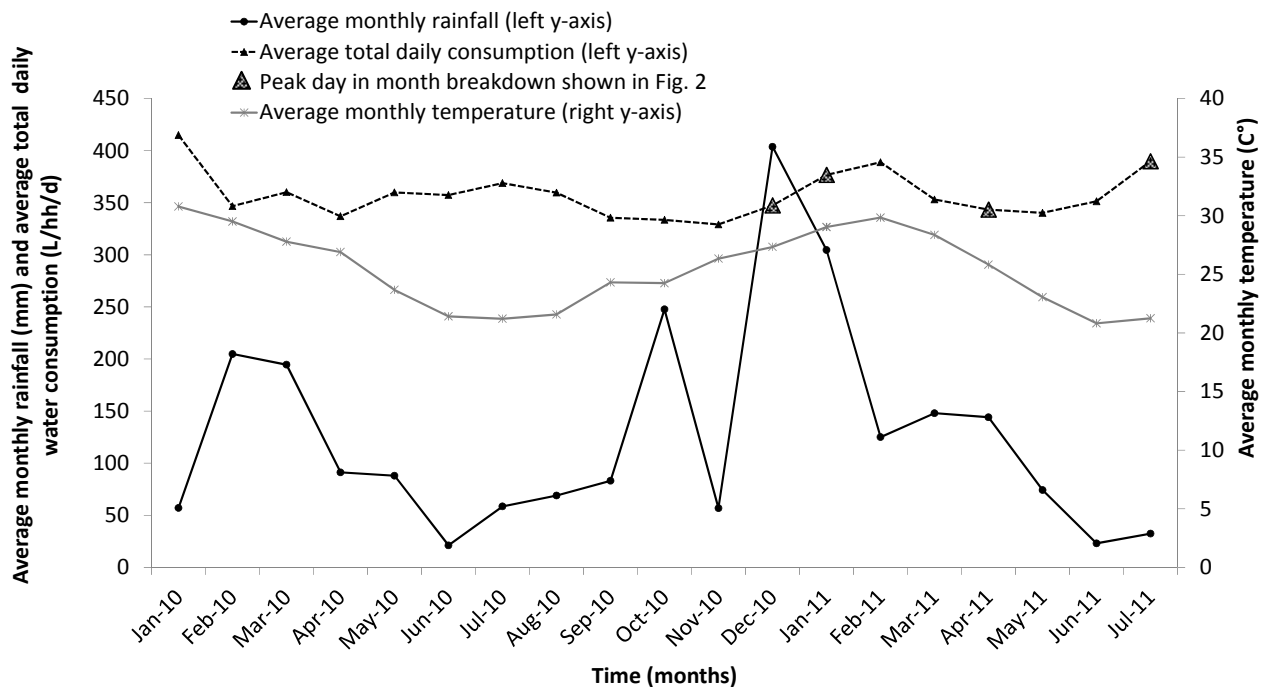


Figure 2. Timeline of average monthly climate data and daily household water consumption.

The greatest peak demand day of 261 L/p/d on Saturday 02/07/11 is shown in Figure 1d and is clearly driven by external water use where the peak hour and peak day demand was 13.4 L/h/p/d and 61 L/p/d, respectively. Prior to this peak water usage day was a period of approximately two weeks where negligible rainfall and low relative humidity occurred in the SEQ region. It is postulated that these conditions were likely to have prompted the observed sudden increase in weekend irrigation events. Further analysis of the correlation between climate pattern and peak end usage is currently underway.

Peaking Factors and End Use Analysis

The peaking factor (PF) is the ratio of the maximum flow to the average daily flow in a water system. Peaking factors for peak day (eg, PD/AD) and peak hour (eg, PHPD/PHAD) are the basis for designing mains water supply infrastructure. A peaking factor over 1 indicates high water volumes, with potential capacity constraints if the supply pipes are not of sufficient diameter.

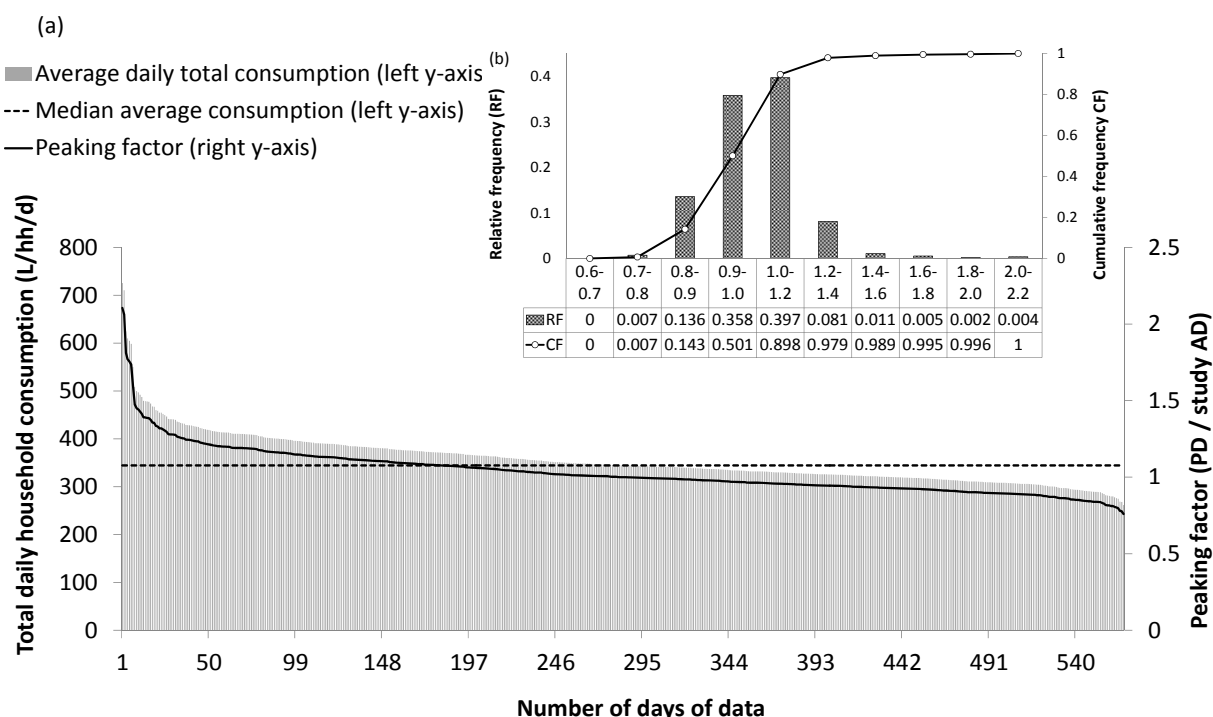


Figure 3. Breakdown of (a) average daily total water consumption (LHS) and PD:AD ratio (RHS) and (b) frequency distributions for combined SEQ sample peaking factors.

A breakdown of average daily total water consumption showing the peaking factor trend for combined SEQ sample illustrates that less than a third of the data had peaking factors over 1, thus extreme usage from a small number of days, primarily driven by irrigation/external use, is likely to dictate the average peaking factor of any given region (Figure 2). The relative frequency distribution of PD/AD of each month is shown in Figure 3b (inset) where it can be observed that PD/AD factors between 1.0 and 1.2 occurred at the greatest frequency.

The four peak demand days selected had increasing peak day factors of 1.3, 1.5, 1.6 and 1.7 as shown in Figure 1. The range of these values is slightly lower than those used in the Department of Environment and Resource Management (DERM) guidelines where the peak day factors range from 1.5 to 2.3 (DERM, 2010). Although individual end use peaking factors have not been presented here, based on data in Figure 1, the contribution of external use to average total consumption is higher for the peak days observed along the timeline. This suggests that external water use events are probably driving peak day (and peak hour) use. This is consistent with other findings such as Willis *et al.*, (2011b) who found a strong association with irrigation and peak morning and afternoon diurnal patterns. Similar results were reported by Roberts (2005) and Heinrich (2007). Shower use has also been strongly associated with peak hour and peak day demand (Beal *et al.*, 2011, Willis *et al.*, 2011b).

Future Trends in Peak Demand

While peak day demand in a SEQ context has been shown to be consistently influenced primarily by external (irrigation) and shower events, this may not necessarily remain the case in the future, due to the high penetration of water efficient technology and a shift toward more frugal consumer behaviour. Peaking factors may well be lower now and into the future than say a decade ago where water consumption was an average of 300 L/p/d in SEQ. Tsang (2010) found that using the same SEQREUS data, peak flow rates were around 15% less than currently estimated values for bulk trunk mains in Gold Coast, Queensland; translating into potential lower diameter trunk mains in future infrastructure planning. Irrigation has shown to be dramatically reduced in the post-drought and post water restriction environment in SEQ (Willis *et al.*, 2011b; Beal *et al.*, 2011). Further, three years post drought, the expected rebound back to higher (eg, >200 L/p/d) consumption is still not yet evident, despite the many months of above average rainfall which one would expect to encourage less frugal water use. In terms of shower use contributing to the peak demand, one would expect an eventual reduction in this end use by the installation of water efficient devices such as shower heads and monitors (Willis *et al.*, 2010a) and ongoing conservation messages to consumers (Walton and Hume 2011; Russell and Fielding 2010). The effect of water efficient technology on daily diurnal patterns and peak flow is discussed by Carragher *et al.*, (2012) as shown in Figure 4. They concluded strongly that the likely reductions in AD peak hour water demand is essentially inevitable given the mandate for all new dwellings to incorporate water efficiency stock (eg, low flow showers, water-efficient clothes washers and tap fixtures). Reduced peak day and peak hour demand due to residential water stock efficiency measures has implications for optimising pipe network modelling and capital infrastructure eg, deferral or reduction in water distribution infrastructure.

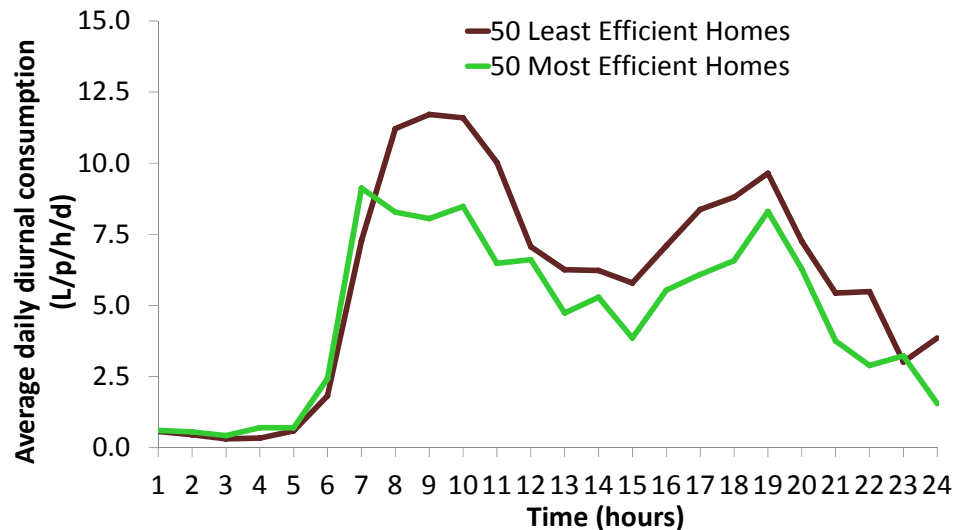


Figure 4. Average day diurnal pattern efficiency clusters.

Water-Related Energy Study

Water-Related Energy Demand and Carbon Emissions

Descriptive statistics for energy consumption for dishwasher and the hot water components of the shower and tap usage for the four main HWS types of electric cylinder (EC), gas cylinder (GC), instant gas (IG) and electric-boosted solar (SEB) are presented in Figure 5. Results demonstrate that SEB HWS require substantially less grid energy than conventional electric systems. Specifically, energy demand can be reduced by an average of 460 kWh/p (or 56%) for shower use and 220 kWh/p (or 57%) for tap use annually if an EC was replaced with an SEB system. This degree of energy demand reduction from solar HWS is consistent with other findings (Tsilingiridis and Martinopoulos, 2010; Perry *et al.*, 2008; Crawford *et al.*, 2003). Even greater energy savings is indicated using a GC system due to its lower energy intensity than an EC. A comparison of energy intensities (EIs) is useful as it provides gauge of the relative energy efficiency for each water end use (Figure 6). End uses which rely on externally heating water clearly have higher EIs than those that internally heat water, with the exception of dishwashers which had an EI of 55 (SD± 11) kWh/kL. This suggest that the typical dishwasher is not overly energy-efficient in relation to the amount of water it requires, however due to the low water demand, around 2-3% total average household water consumption (Beal and Stewart 2011), its overall energy demand is reduced.

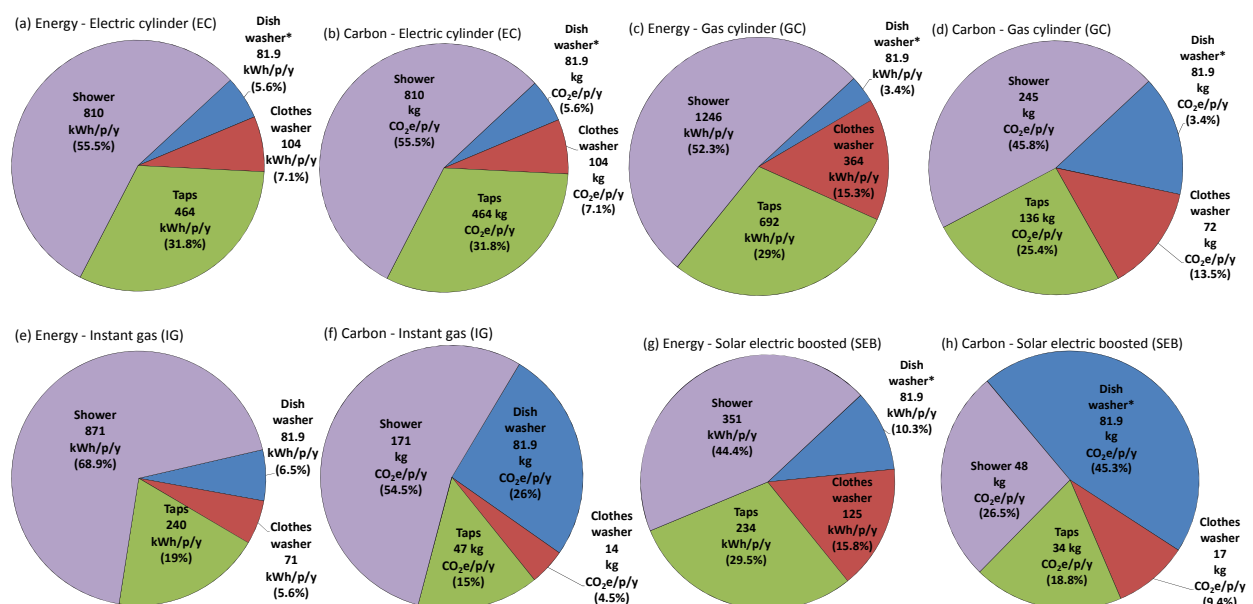


Figure 5. Average annual energy and carbon emissions end use breakdown for water-related energy based on hot water system. Note: *Dishwasher energy sourced solely from electricity grid using coal-fired power therefore energy demand and carbon emissions are constant for the four HWS scenarios.

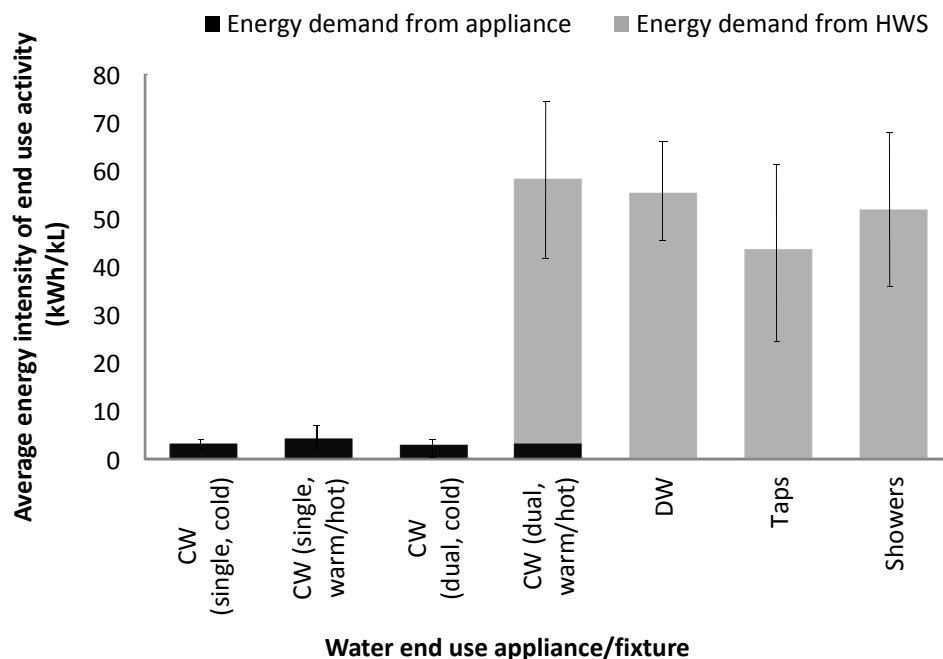


Figure 6. Average energy intensities for water end use appliance/fixture.

Although average values for clothes washer are provided in Figure 5, estimating the energy demand from clothes washers is quite complex and requires consideration of the various configurations of tap connections, HWS types and temperature wash cycles. Many, but not all, of the later model machines are horizontal axis (or front loading) machines that only have a single, cold water tap connection to the machine and thus source hot water from internal heating. Older models tend to be vertical axis (or top loading) machines that have a larger capacity and have dual water connections (eg, hot and cold tap connection to the machine), where hot water is sourced from the external hot water service. Fortunately, the data registry available from the SEQREUS allowed a high level of precision in clustering each of the configurations, although sample size was quite low or absent for some categories. In this study, the highest EI was 58.4 kWh/kL for dual connected warm/hot cycle comprising 55.1 (SD± 16) and 3.3 (SD± 1) kWh/kL for external heating and operation, respectively (Figure 6). Conversely, for the same warm/hot wash cycle, the single connected clothes washers had an average EI of 4.2 (SD± 3) kWh/kL. That is, the additional energy required to heat the water internally is lower than the energy required to heat the water via the HWS which heats larger volumes of water to a great temperature. The majority of these single connected systems were front loading (or horizontal axis) machines. The results emphasise the importance of knowing details on clothes washer configurations in order to produce a representative dataset.

Impact of Intervention Scenarios on Energy and Carbon Emissions

A number of resource-efficient intervention scenarios were modelled to quantify reductions in energy and carbon emissions compared with base (worst) case scenario for a household with no water-efficient appliances/fixtures and an electric cylinder HWS. Individual volumetric and percentage savings for each resource-efficient scenario were also determined (Table 2). Installing a solar HWS (electric boosted) was the most energy-efficient scenario reducing total household energy savings of around 46% at an equivalent volumetric savings of 737 kWh/p/y and 102 kg CO₂-e/p/y. Results from other studies suggest that this may be a conservative estimation with reductions of up to 60% (Kenway *et al.*, 2008) and 75% (Flower, 2009), however there are many factors that influence the efficacy of solar HWS which must be considered when comparing energy reductions and subsequent carbon emissions savings. These include climate, type and location of solar cells and storage systems, and method of booster (gas or electric).

Table 2. Individual savings from various resource-efficient scenarios.

Resource-Efficient Scenario	Individual Savings – Volumetric			Individual Savings – Percentage	
	Water consumption (kL/p/y)	Energy consumption (kWh/p/y)	Carbon emissions ¹ (kgCO ₂ -e/p/y)	Water consumption (%)	Energy consumption ² (%)
Solar HWS (EB)	-	737	102	-	46
Water-efficient shower head	5.2	217	30	37	63
Water-efficient clothes washer	2.4	183	25	27	87
Tap aerators	2.6	93	13	27	38
Shower reduced to 37°C	-	16	6	-	13
Energy-efficient dish washer	-	23	3	-	28

Notes: ¹ applicable for conversion to solar HWS (electric-boosted) only as current carbon emissions factor for electricity generated from coal-fired power stations in Queensland is 1, therefore carbon emission savings if no conversion from EB HWS to SEB HWS will equal the values in the energy consumption volumetric savings column; ² carbon emission percentage savings is equivalent to energy consumption percentage savings.

The most optimal solution for reducing both water and energy saving is the installation of a low-flow shower rose. This resulted in a potential total savings of 37% of annual total household water consumption and 63% energy savings (Table 2). This is the best water/energy saving combination of all scenarios tested, and also one of the cheapest. Other studies have shown the substantial reductions to total household water and energy use from low-flow shower heads (Beal *et al.*, 2011; Mayer *et al.*, 2004). Locally however, the margin for savings may not be as great as other resource-efficient strategies, such as water-efficient clothes washers, due to the already high penetration of low-flow shower heads in Australian homes. Savings of around 183 kWh/p/y and 25 kg CO₂-e/p/y (or 27%) were found by installing a water-efficient single connected clothes washers. Reducing the temperature of the hot water from 40 to 37°C also resulted in notable energy savings of about 13% (Table 2). Replacement of standard dishwashers with low energy use dishwashers reduced energy demand by 23 kWh/p/y, which is an annual savings of about 28% of carbon emissions per person.

Calculations show that by replacing a conventional electric HWS with SEB HWS, the annual carbon emissions can potentially decrease from an average of 1,618 to 882 kg CO₂e/kWh/p/y. However, this is a slightly simplistic argument, particularly in regard to economic savings – replacing an old electric with a new solar HWS can be expensive. Calculated payback periods for solar HWS and low-flow shower heads were estimated at 9.6 years and 1.1 years, respectively. This aligns well with other reported values by where a payback period for water-efficient shower devices is between 1 to 1.5 years (Willis *et al.*, 2010) compared to installing a solar HWS which may be around 10 years (Crawford *et al.*, 2003). Retrofitting old shower heads with low flow roses is regarded as a cheap and relatively easy to install solution to reducing both energy and water consumption.

Conclusions and Implications

Peak day and peak hour demand was examined for four peak days identified from 18 months of empirical household water consumption data. Peak day demand that yielded peaking factors between 1 and 1.5 were observed to be primarily driven by clothes washer and shower use. However, as the peak day and peak hour demand rose well above a PD/AD of 1.5, demand was strongly driven by external water usage. Internal peak demand from shower and clothes washers also translate into peak water-related energy use, particularly so if the hot water is sourced from electricity sourced from coal-fired power stations. Results showed that shower and certain configurations of clothes washers can have very high energy demands which may translate into substantial greenhouse gas outputs, depending on the hot water service.

The increase in water-efficient technology has been shown to markedly reduce peak morning and afternoon demand. This is especially pertinent in jurisdictions where factors influencing future water demand (thus water-related energy demand) are evident, such as the wide incorporation of water efficient stock and permanent shifts in water conservation behaviours. Further, results demonstrate that significant reductions in both peak water demand and energy demand use can be achieved by various water efficient fixtures and appliances. Knowledge of the end uses that are influencing peak water and energy demand can: 1) facilitate the optimisation of infrastructure design and sizing and inform the decision process for the subsequent deferral of these expensive assets; and 2) underpin future sustainability codes for new buildings.

Acknowledgements

The authors would like to acknowledge the Urban Water Security Research Alliance for funding the SEQREUS from which much of this data was based.

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Mathematical Material Flow Analysis of Water-Related Energy in a Brisbane Household Quantifies the System and Potential for Savings

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Summary

Household energy use, including private transport, accounts for about one-third of primary energy use in industrialised countries. Households are important drivers of energy use and related greenhouse gas emissions. In order to understand water-related energy use in households, a detailed mathematical material flow analysis of water, energy, CO₂ emissions and costs was developed. This was applied to a specific family household in Brisbane, Australia, and calibrated with water, energy and natural gas use records over three years (2007-2009). A detailed scenario investigation determined the impact of (i) potential and (ii) realistic reduction values for all relevant (a) behavioural and (b) technical parameters, including a shift from gas to a solar hot-water system. The reduction potential for water use ranged from 4-77% depending on the measures. Similar reduction potential was observed for greenhouse gas emissions (4-85%), water-related energy consumption (15-93%), water costs (1-31%) and water-related energy costs (13-90%). The study showed that for this household, technical improvements alone, without changing to a solar hot-water system, result in less than a 15% change in terms of energy and greenhouse gas emissions. In contrast, combined behavioural and technical changes have a much higher reduction potential. Understanding water-related energy could provide significant scope for water managers to influence residential energy use and associated greenhouse gas emissions and monetary flows.

Keywords

Water, energy, greenhouse gas emissions, material flow analysis, modelling.

Introduction

Primary energy consumption in Australia was about 8 kW¹ per person in 2007-08. This is around 68,000 kWh/p/year or 1,500,000 GWh/year of total domestic primary energy use (ABS 2011; IEA 2011). It is four times higher than the global average of 2 kW per person (IEA, 2010). Greenhouse gas emissions (from energy consumption) for Australia is about 26,500 kg CO₂-equivalent per person per year (576 x 10⁶ t per year (AGO, 2010)).

Australia aims to reduce greenhouse gas emissions by 60% below the year 2000 levels by 2050 (Wong, 2009). In order to reduce greenhouse gas emissions, an understanding of current energy use is crucial. Private households in Australia consumed about 19,700 kWh/p/year of primary energy; about 2.2 kW/person. This amount includes private transport, which accounts for about 11,000 kWh/p/year (1.3 kW/person) (ABS 2011). Industry consumed about 32,600 kWh/p/year of primary energy (3.7 kW/person), including industry-related transport (around 6,000 kWh/p/year). Commercial and public services, agriculture and forestry, fishing and non-energy use account for the rest (1.8 kW/person) (Figure 1).

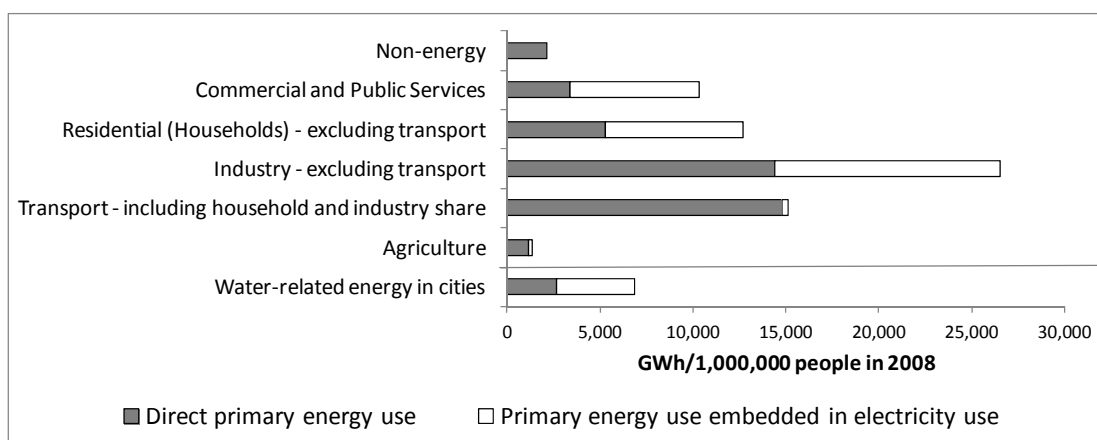


Figure 1. Energy consumption (direct and embedded in electricity) in Australia*. Note this totals to Australia's total primary energy use excluding water-related energy use in cities** which is cross-sectoral, and shown for context.*

*Based on data from (IEA 2011); ** Source: (Kenway *et al.*, 2011).

¹ kW continuous energy use every hour for one year represents 8,766 kWh.

Unlike the sector-based estimates above, water-related energy is not systematically accounted; it is a cross-sectoral issue. Water-related energy in Australian cities has been estimated at 6,800 kWh/person (0.78 kW/person) measured as total primary energy (Kenway *et al.*, 2011a). Of this, households account for about 30% of the influence – ie, about 2,040 kWh/p/year or 0.23 kW/person of primary energy use. This is about 10% of household primary energy use including transport energy demands.

Private households are important key drivers since they directly consume energy (for mobility, heating/cooling, housing, etc). When households consume electricity, they also inadvertently consume primary energy (largely coal), needed to produce the electricity. Finally, households also consume grey energy via the amount, origin, quality and lifetime of everyday products. Unlike sectors such as mobility, heating/cooling and communications, water-related energy consumption in households has not been studied as substantially as individual sectors. The relatively few studies that have been conducted on water-related energy in households have tended to rely on “average” circumstances; existing analysis frameworks in this area do not yet deal with uncertainties or sensitivities of the “systems” (Conrad *et al.*, 2011), and relatively little modelling work has been validated or calibrated with empirical data (Kenway *et al.*, c 2012).

Consequently, this paper focuses water-related energy consumption, greenhouse gas emissions and related costs in a single household. It seeks to improve the knowledge of key factors, sensitivities and uncertainties by using a validated model at the single household scale.

Methods

In this study, we developed and applied a mathematical material flow analysis (MMFA) to quantify household flows of water and energy. The approach is an extension of the classical material flow analysis (MFA) developed in the economic sector in the 1950s (Leontieff and Stroud, 1963) and later adapted to regional investigations (Baccini and Brunner, 1991). More recently, it has been applied to solving diverse environmental problems (Huang *et al.*, 2007; Kwonpongsagoon *et al.*, 2007; Schaffner *et al.*, 2009). As pointed out by (Schaffner *et al.*, 2009), the key benefit of the method is its ability to provide an understanding of the system based on current knowledge using often scarcely available data rather than conducting large monitoring and data collection campaigns. The method further aims to identify the key parameters (driving forces) involved. This is crucial for discussing possible measures (scenarios) to reduce the flows. The MMFA comprised the following steps:

- (1) System analysis.
- (2) Mathematical model.
- (3) Data collection and calibration.
- (4) Simulation including uncertainty analysis, sensitivity analysis and scenario calculations.

The core of the model is comprised of the subsystems which rely on water, and/or energy, to provide the necessary functions within households. The sub-systems of demand include: shower, bath, washing machine, indoor taps, dishwasher, outdoor use, toilet, kettle, air-conditioning and “other energy use”. Energy and cold water may be directly supplied to each of these sub-systems. Energy and cold water are also provided to the hot water system in order to then supply each sub-system with hot water. Parameters are used to describe the probability distribution, mean, deviation and upper and lower bounds of (i) the overall household including the hot water system and (ii) each sub-system. For example the nineteen overall “household” parameters include: the number of adults; the number of children; the temperature of cold water; the temperature at the hot water system and the hot water system (thermostat) temperature. Parameters describe each sub-system such that the (a) water and resultant (a) energy consumption can be characterised. For example, parameters describing the shower sub-system include: (a) flow duration per shower for adults, flow rate per shower for adults, number of showers per adult per day and (b) temperature of showers for adults.

In order to test the model, a specific household was investigated. The study used a typical “Queenslander” five-bedroom house in Milton, Brisbane. The house was occupied by two adults and two children (aged between six and ten). Parameters were characterised according to the best available information for. A period of relatively uniform water use (2007-2009) was selected. Level 2-6 water restrictions were in place during this time. At Level 2, most outdoor water use (eg, irrigation) was banned. At Level 5, residents were encouraged to reduce water use to 140 l/p/d.

Interviews were used to determine water usage patterns and behaviours. Repeated measurements were performed for the shower and bath temperatures of adults and children, washing of dishes by hand and the water used for shaving. Appliance efficiency data was recorded from manuals or similar appliance data available on-line. Appliance plumbing to either the cold or both hot and cold water supply was confirmed by inspection. The natural gas hot-water storage was located outside. The front-loading washing machine and dishwasher are both plumbed to cold water only, and were seven and 12 years old respectively at the start of the period. Natural gas (37.7 J/m³) is used in a cooker top. A small gas heater is used for a total of approximately 40 hrs per year during winter. Electricity from

coal-fired plants meets all other energy demands including microwave and oven cooking, air heating and cooling. The household has no pool, spa, aquarium, or water-chilling devices.

Literature values were used to characterise the relatively small number of parameters which could not be determined by measurement or interview. For example, carbon dioxide equivalents were determined on the basis of current rates for electricity supplied from coal-fired power plants (1.04 kg CO₂-e/kWh) and natural gas (0.197 kg CO₂-e/kWh), which are the average estimates in Queensland for the full fuel cycle (Commonwealth of Australia 2008). The costs for water, electricity and gas were calculated according to the current tariff in Brisbane. All parameters of the model, the full method and system description are contained in Kenway *et al.* (c 2012).

The calibrated parameter values describe an “average day” in the 2007-2009 study period. Uncertainties were calculated for an “average day” as well as for a “single day”. Where data was available, standard deviations were calculated to obtain an estimate. In other cases, standard deviations were estimated after discussions among the authors based on their knowledge of the household, its occupants, technologies and prevailing conditions.

Water, electricity and natural gas for the house considered were obtained from the local water and energy providers to validate the model. Absolute and relative local sensitivity analysis was used to inform data collection towards parameters of greatest influence on the results. Collectively, the approach has enabled a world-first probability-distribution based characterisation of water-related energy at the scale of an individual household.

Results

Simulated water and electricity use were in good agreement with measured data (Table 1). The simulated gas use is 30% higher than measured use. This is possibly due to slight overestimation of shower flow rates. Compared to the average Australian household, the Milton household uses significantly less water and energy, namely 50% less water and 60% less energy on a per person basis. One reason may be that the average Australian household typically includes 2.6 persons whereas the Milton one comprises four persons including two children who consume less. Another reason could be that Brisbane had stricter levels of water restrictions than many other Australian areas during the period of survey. The household members are also generally proactive with regard to water and energy conservation.

Table 1. Modelled and measured water, electricity and natural gas use.

	Unit	Source	Mean	STDV-Average-Day	Remarks
Total water use	l/(hh·day)	Model	464	33	Milton 4-person household
	l/(hh·day)	QUU ^a	451	74	Milton 4-person household
	l/(hh·day)	Australian average (National Water Commission 2010)	548		2.6 persons per household
Total gas use	kWh/(hh·day)	Model	9.1	2	Milton 4-person household
	kWh/(hh·day)	AGL ^b	7.4	2.9	Milton 4-person household
	kWh/(hh·day)	Australian average ^c (ABS 2011)	17.7		2.6 persons per household
Total losses alone	kWh/(hh·day)	Model ^d	1.4	1.3	Milton 4-person household
Total electricity use	kWh/(hh·day)	Model	12.1	1.0	Milton 4-person household
	kWh/(hh·day)	Energex ^e	12.3	1.2	Milton 4-person household
	kWh/(cap·day)	Australian average (ABS 2011)	7.4		
Hot water system (gas)	kWh/(hh·day)	Model	7.6	2	Milton 4-person household
Total water-related greenhouse gas emissions	kg CO ₂ -e / (hh·day)	Model	5.4	0.5	Milton 4-person household

a Data from Queensland Urban Utilities covering 12 quarters from 16/01/2007 to 16/01/2010. b AGL data is based on 10 bills from 22/05/2007 to 9/04/2010. c Data for Australian average includes LPG usage in addition to natural gas. Data based on 8,235,000 households and 21,498,500 persons. d Includes heat losses from the (gas) hot water system and pipes. e Energex data is from 13 records from 12/02/2007 to 9/02/2010.

The shower sub-system used the most water-related energy (3.3 +/- 0.7 kWh/hh.d of natural gas). This was followed by the clothes-washer (1.8 +/- 0.3 kWh/hh.d electricity use), bath (1.62 +/- 0.3 kWh/hh.d gas) and tap (1.4 +/- 0.3 kWh/hh.d gas) sub-systems. However, for water-related greenhouse gas emissions, the clothes-washer, followed by dishwasher and electric kettle, were the most dominant sub-systems. This change in hierarchy is because the specific CO₂ emissions per kWh for electricity (from coal-fired power plants) and gas differ by a factor of 5.3. The contributions of the gas-operated shower, bath and taps are consequently much lower than of the electrically operated clothes washing machine, dishwasher and kettle. This would obviously change if power plants with lower carbon emissions were used for the electricity.

Energy costs account for more than half the cost (\$/hh.d) of shower, bath, clothes-washer, dishwasher, tap and kettle water use.

Uncertainty, Sensitivity and Scenario Analysis

The uncertainty of three parameters was responsible for more than 80% of the uncertainty for the average day. These were: heat co-efficient of the hot water storage, flow-rate per shower for adults, temperature of shower for adults. In contrast six parameters influenced 80% of the uncertainty for the single day case: number of cycles of warm front loader clothes-washer use per day; number of adults per household; temperature of cold water; heat co-efficient of hot water storage; number of showers of adults per day; and number of cycles of hot front-loader clothes-washer per day. Improved data collection (eg, time-series data) for these parameters would significantly reduce uncertainty in the results.

A first insight into possible reduction measures of water and energy use is provided by a local sensitivity analysis showing the local change of the variables for small changes in the parameters. This analysis allows those parameters with the greatest influence on key variables to be identified and is basic for a careful scenario analysis yielding possible reduction measures (Schaffner *et al.*, 2009)). However, the local sensitivity analysis works only for continuous parameters but not for discrete ones such as a switch between standard (gas/electric) and solar hot-water systems.

Scenario analysis considered the influence of realistic and potential changes for “technical” parameters such as those relating to appliances and their efficiency. The corresponding values for the “behavioural” parameters such as shower duration and frequency were estimated by discussions among the household occupants (Refer to Kenway *et al.*, 2012 for details). The basic assumption for the scenarios was that the “level of service” remains at least on the current level. Examples include that personal hygiene remains on an accepted level, clothes are still washed, cooling / heating is still possible and watching TV remains enjoyable (same screen size, quality). The difference between potential and realistic reductions is that the latter mean no change in comfort and service (whereas potential reductions may cause a slight decrease in comfort but not in service). For example, the temperature of showers for adults is currently 41°C, whereas the realistic value was seen as 38°C and the potential one as 35°C.

The reduction potential for water, CO₂ emissions, water-related energy, water costs and water-related energy costs are 4-77%, 14-85%, 15-93%, 1-31% and 13-90% respectively depending on the measures taken (for detail refer to Kenway *et al.*, 2012). Technical measures alone, ie, without changing to a solar hot water system, led to the smallest reduction whereas the combined scenario, namely technical + behavioural + solar hot water system + connection of washer/dishwasher to the hot water system led to the highest reduction (Figure 2). Technical changes with behavioural assistance (eg, shower flow-rate, volume per toilet flush) have the potential to save a lot of water, namely up to 30%. Behavioural changes alone are very effective. In fact, they account for up to 75% of the whole possible reduction of water and water costs, two-thirds of CO₂ emissions and water-related energy and just over one-third of the water-related energy costs.

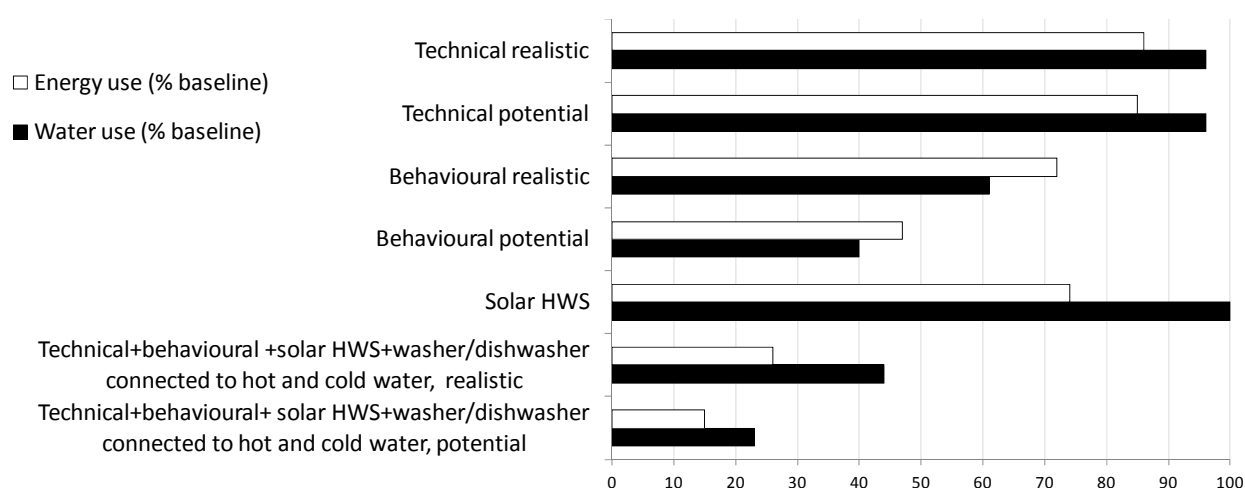


Figure 2. Scenario influence on water and energy use (summary).*

*Note shower flow rate is considered a behavioural parameter for this analysis.

Conclusions

The study showed that technical improvements alone without changing to a solar hot water system in the Milton household result in a less than about 15% reduction in energy and CO₂ emissions. This is because this is already equipped at a high technical level. The technical improvements required for other types of household might be much greater. Technical improvements with behavioural assistance offer a large potential for water saving.

Behavioural changes have the potential to reduce 50% or even more water-related energy and CO₂ emissions. Hence, such changes can be very effective. The major advantages of behavioural changes are: a) no additional costs are involved; b) no change in installations or infrastructure is needed; c) they can be applied immediately; and d) each individual can apply them independently.

The most effective technical measure would be the installation of a solar hot water system combined with the connection of clothes washing machines and dishwashers to a hot water source (rather than using coal-fired electricity to heat water within the machine). This would lead to a reduction of about 70% in water-related energy and about 25% in CO₂ emissions. However, this measure would require changes in installations and associated costs.

It should be pointed out that replacing a washing machine connected to a natural-gas operated hot water system to a water-efficient washing machine plumbed for cold water intake, results in an increase in greenhouse gas emissions, even though energy use is reduced. This is because washing machines with a cold water intake use coal-fired electricity to heat the water internally, rather than drawing on the natural gas heating system. Greenhouse gas emissions consequently increase because coal-fired electricity has around five times the carbon-dioxide equivalent intensity of natural gas per kWh of energy delivered.

This study has focussed on a single, specific household. Consequently the results cannot be widely generalised. What is unique about this analysis is that it provides a systematic method to understand water-related energy in households. The method could be applied to other household types.

The analysis presented in this paper underlines the importance of detailed household analyses in order to understand key factors determining the water, water-related energy and CO₂ emissions. Since households are the building block of cities, acquiring a high understanding of them is crucial to understanding the cities themselves. Therefore the next step is to extend the analysis to the different types of households in a city, which is currently in progress. The developed model has been designed so that it can be applied to a whole city comprised of various household types.

Acknowledgements

The authors gratefully acknowledge the support given by the Urban Water Security Research Alliance, including the Queensland Government, CSIRO, The University of Queensland (including the School of Chemical Engineering and Advanced Water Management Centre) and Griffith University, and The Swiss Federal Institute for Environmental Science and Technology (Eawag). The support of the Australian-American Fulbright Commission is also appreciated. We are also appreciative of the data provided by the Bureau of Meteorology, Seqwater, Queensland Urban Utilities, AGL and Energex.

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Decentralised Wastewater Systems: Robustness and Carbon Footprint

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Summary

In Australia, the demand on the traditional water supply from the water grid is escalating in conjunction with issues of climate change, rapid industrialisation and booming commercial activities in most densely urbanised regions. This is further exacerbated by rapid population growth, which could have a direct impact on reliable water supply, wastewater disposal and treatment. In South East Queensland (SEQ) alone, it has been projected that a population growth of 1.5 million is possible by 2031, which might be a challenge to the local water utilities. In order to provide water and wastewater services, a number of water recycling infrastructure and reuse schemes have been implemented. This study investigates the potential of small decentralised wastewater treatment systems to deliver a solution to address: (1) the robustness of the small treatment systems in handling the daily variations in water flow and pollutant loads; and (2) the carbon footprint of such systems. Two contrasting decentralised wastewater systems - one with a predominant membrane bioreactor (MBR) technology and one with an aerobic bio-filtration system - were monitored and modelled in this study.

Modelling results show that the decentralised MBR wastewater system at Capo de Monte was relatively robust in terms of its treatment capacity where the probability of exceedances on Total Nitrogen and Total Phosphorus concentrations against the limits approved by the Environmental Protection Agency was only 2% and 3%, respectively. Subsequently, through the validated BioWin[®] model (with periodical wastewater sampling), it was found that the decentralised MBR system is quite robust and can withstand up to 50% of additional wastewater flow rate and 30% of nitrogen loading. As for the carbon impacts, the direct carbon footprint (energy consumption) of the MBR system was significantly higher than the aerobic bio-filtration system. However, when the overall carbon footprint of the systems (direct and fugitive emissions) was accounted for, both systems are quite comparable. With the water balance and robustness modelling, decentralised wastewater systems certainly provide a “*fit-for-purpose*” solution to accommodate the population growth issue. However, more research needs to be done to optimize the energy efficiency of the systems, together with validation of fugitive greenhouse gas emissions, which might affect the sustainable selection criteria for these systems.

Keywords

Water recycling and reuse, energy efficiency, water-energy nexus, greenhouse gas emissions.

Introduction

Demands on traditional water supply from the water grid in Australia are escalating in conjunction with issues of climate change, rapid industrialisation and booming commercial activities in most of the densely urbanised regions. This is further exacerbated by rapid population growth, which has a direct impact on reliable water supply, wastewater disposal and treatment. In South East Queensland (SEQ) alone, it was projected that a population growth of 1.5 million is possible by 2031, which might be a challenge for the local water utilities (DIP, 2009). In the traditional “*end-of-pipe*” paradigm for water servicing, such an increase in urban water supply and sewage handling could be easily resolved by expanding the existing centralised water and wastewater collection, conveyance and treatment systems. This might, however, incur high capital costs to upgrade or construct new water and wastewater infrastructures and associated recurring pumping energy cost for transportation over long distances.

Decentralised water and wastewater systems are regarded as important elements in the urban water cycle where they can act as transitional solutions to accommodate urban population growth by providing location specific water servicing (Tjandraatmadja *et al.*, 2009). The wastewater systems can collect and treat the wastewaters from local sources to Class A⁺ recycled water quality that can be utilised for non-potable applications surrounding the urban households. It was reported by Peter-Varbanets *et al.* (2009) that only a small fraction of mains water is used for domestic consumption (ie, drinking and cooking), whereas the majority is used for non-potable applications (ie, toilet flushing, washing machine, irrigation and other end uses). In the past, decentralised wastewater systems were largely viewed as an alternative to centralised systems for remote or rural developments. However, with the emergence of various advanced suites of treatment technologies, a greater flexibility in process selection and matching end uses allows the increased adoption of decentralised wastewater systems in urban contexts. At present, the major technical limitations to the wider uptake of decentralised systems are the lack of knowledge and information on technology selection and its effective design to enable stable, cost effective and sustainable operation.

In this paper, two different contrasting decentralised wastewater systems are compared in terms of their robustness and carbon footprints. In this context, robustness refers to ability to handle daily variations in water flow and pollutant loads while carbon footprint refers to the water-energy nexus and system specific fugitive gas emissions. The membrane bioreactor (MBR) and textile bio-filter decentralised systems are located at Capo di Monte (CDM, Mount Tamborine) and Currumbin EcoVillage (CEV, Currumbin Valley) respectively and were designed to produce Class A⁺ recycled water for toilet flushing and external irrigation use. A quantitative risk model was used to measure the potential risk associated with the treated effluent quality against the approved Environmental Protection Agency (EPA) limits. Diurnal wastewater quality sampling was conducted for the MBR at CDM in order to calibrate a commercially available activated sludge BioWin® model. Subsequently, the robustness of the MBR plant was tested in terms of varying hydraulic and nitrogen shock loads. In addition, the water-energy nexus was also monitored via smart meters. Finally, the total carbon impact was estimated via a combined monitoring and first principle mass balance approach.

Monitoring Sites and Systems Description

Capo di Monte

Capo di Monte (CDM) decentralised wastewater treatment system was built as part of a 4.3 ha urban residential development to provide wastewater servicing of 46 detached and semi-detached residential dwellings and a large community centre. The predominant reason for the adoption of this system was the absence of a centralised sewer system. The decentralised system was designed for an estimated peak hydraulic flow of 11,000 L/day. Figure 1 shows a schematic for the decentralised wastewater system at CDM. The system is made up of a raw sewage holding wet-well followed by a submerged flat sheet MBR (Kubota) that incorporates a raked screen, anoxic/aerobic treatment zones, alum dosing in aerobic zones (for phosphorus removal), UV disinfection and chlorination. The flat sheet membrane is positioned within the aerobic zone of the MBR. A submersible pump in the aerobic MBR zone allows for the return of the activated sludge (RAS) stream back to the anoxic zone. Excess activated sludge from the anoxic zone is pumped out on a fortnightly basis to a Gold Coast regional sewage treatment plant for further treatment. Class A⁺ recycled water is produced from this system and used for household toilet flushing and external irrigation via a dual reticulation system. A 6,000 m² vegetated buffer zone is available for land application of excess treated wastewater to prevent direct discharge into the local waterway.

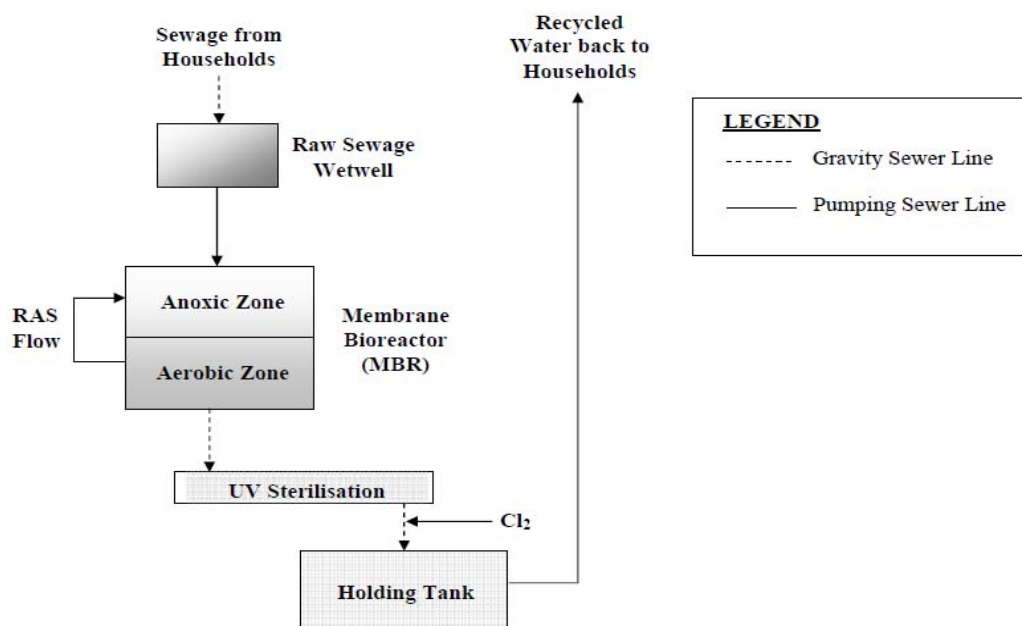


Figure 1. Schematic of decentralised wastewater system at Capo di Monte. RAS flow is the return activated sludge stream.

Currumbin EcoVillage

The Currumbin EcoVillage (CEV) is situated in the Currumbin Valley, Gold Coast, and comprises 110 residential lots that range from 400 to 1,600 m² with extensive proportions allocated for communal open areas (80:20 of open-to-living space). The main reason for the uptake of decentralised wastewater technologies was due to the inaccessibility to a centralised sewer network. The CEV sewage treatment plant (STP) has a design capacity of 51,000 L/day for raw sewage treatment. Figure 2 shows the schematic for the treatment processes, as well as its

wastewater flow lines. The wastewater is collected at each household and conveyed to the STP using a combination of gravity and sewer pumping. The initial anaerobic treatment is performed by three in-series septic tanks with a BioTube® filter installed in the last tank to remove carry-over solids. The sewage effluent is then treated in a secondary process of aerobic bio-filtration and denitrification. An Orenco Advantex® Textile Filter (AdvanTex AX100) is used for the simultaneous aerobic degradation and nitrification of carbonaceous and nitrogen compounds in the primary treated effluent. A proportion of the treated effluent from the textile bio-filters is recycled back to an anoxic/recirculation tank to allow a denitrification process to occur. This recycling ratio is a crucial process parameter and is currently set at a 5:1 ratio (Xavier, 2008). This recycling ratio means that only one sixth of the wastewater flow in a full pumping cycle is diverted for subsequent downstream treatment, whilst the remaining flow fraction is recycled continuously to ensure sufficient biological oxygen demand (BOD) reduction is achieved. The diverted effluent is treated to a Class A⁺ recycled water via microfiltration (with an effective pore size of 0.2 µm) followed by UV disinfection and chlorination. The Class A⁺ recycled water produced from the STP is stored in a large storage tank before being reticulated to the households for toilet flushing and external irrigation use.

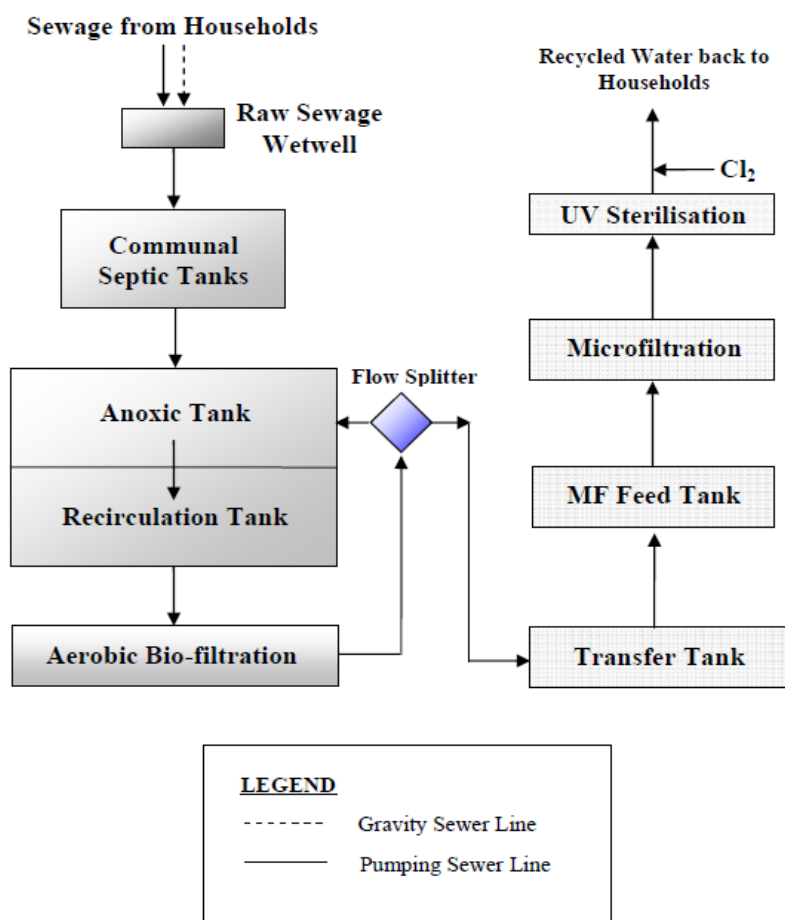
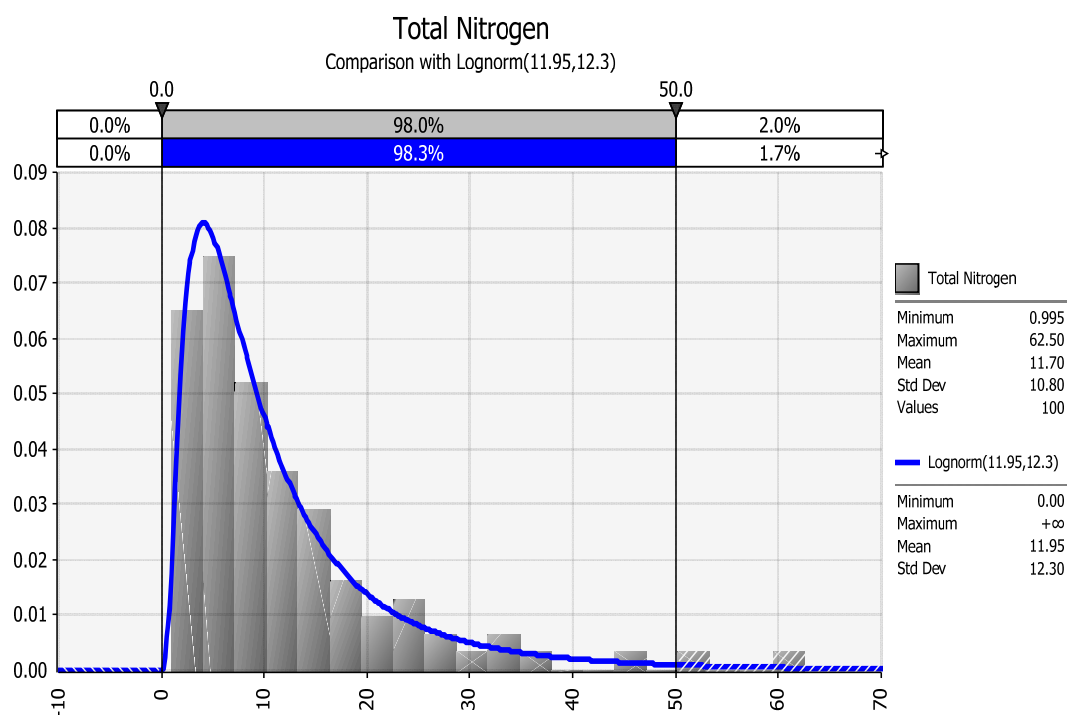


Figure 2. Schematic of the Currumbin EcoVillage decentralised wastewater system.

Results

Figures 3 (a) and (b) show the outcomes from the quantitative risk models of the sampled effluent qualities of total nitrogen (TN) and total phosphorus (TP) from CDM. The sampling was conducted from April 2008 to September 2010 on a weekly basis, so should provide a good basis for the quantitative risk modelling in terms of the reliability of its mean and standard deviation values. Results show that the decentralised wastewater system at CDM is relatively robust in terms of its treatment capacity where the probabilities of exceedances for TN and TP against the EPA approved limits of a maximum of 50 mg/L and 15 mg/L were only 2% and 3%, respectively. Table 1 also shows the probability of exceedances for other treated effluent qualities from the CDM system. It seems that the system is relatively stable for other treated effluent qualities of BOD, total suspended solids (TSS) and *Escherichia coli* (*E. coli*) where zero probability of exceedances were noted against the EPA approved limits.

(a)



(b)

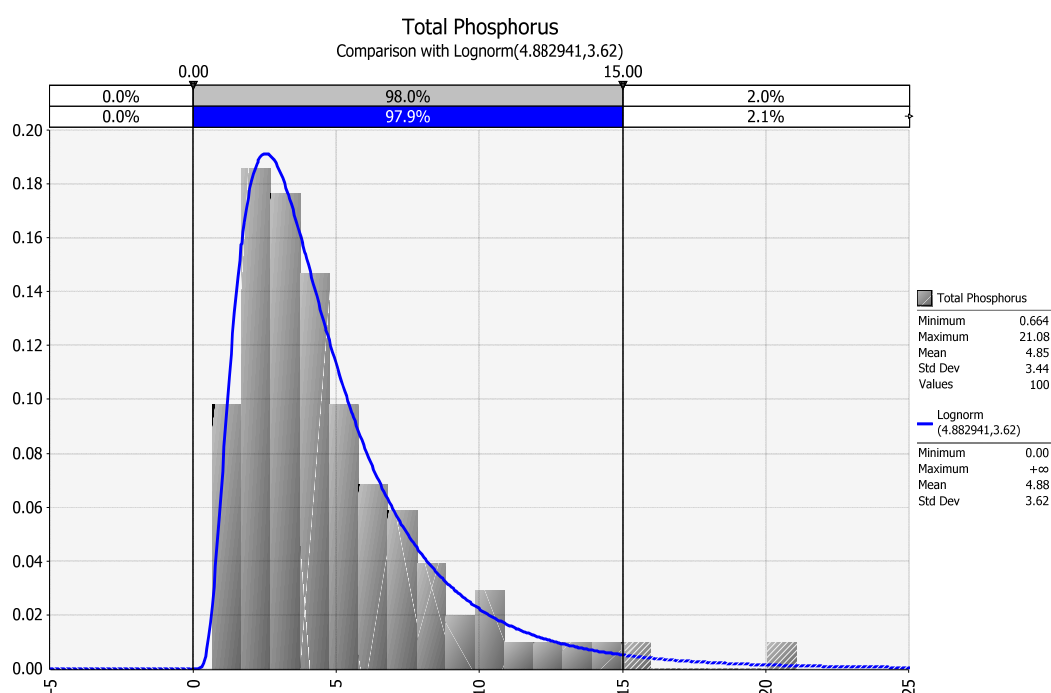


Figure 3. Probability of exceedances against the EPA approved treated effluent quality limits. (a) Total nitrogen (TN); (b) Total phosphorus (TP). Both Y-axes refer to the probability density function; whole X-axes refer to the nutrient concentrations. The grey bars refer to the experimental values; and blue line refers to the fitting with log-normal function.

Further to the quantitative risk models which reinstated the system stability at CDM in the current condition, a commercial activated sludge model, BioWin[®], was used to simulate the system robustness to varying shock loads of TN and TP. The model inputs were obtained from a combination of grab and composite sampling at CDM. Figure 4 shows the simplified process schematic setup in the BioWin[®] model. Simulation results reaffirmed that the MBR system can tolerate up to 1.5 times the design hydraulic flow. As for TN, when the nitrogen shock loads were increased stepwise up to 30% (~ 140 mg/L), it was found that the system was unbalanced and it took 12 hour to regain a steady-state condition (Chong *et al.*, 2011). The variation in TP was highly dependent on the alum dosage and thus, can be adjusted accordingly.

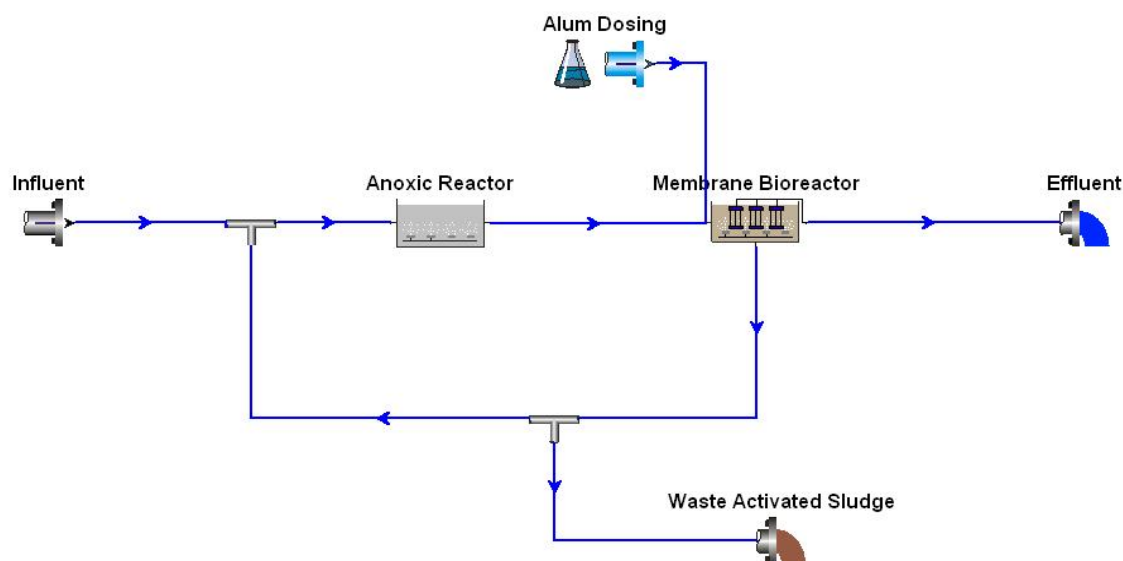


Figure 4. Simplified process schematic in BioWin® simulation model. WAS represents the waste activated sludge stream.

Table 1. A summary of the EPA approved treated effluent quality limits, mean and standard deviation of sampled effluent quality and the probability of exceedances for the decentralised wastewater system at CDM.

Parameters	EPA Approved Limits			Sampled Effluent Quality		Quantitative Risk Model				
	50th percentile	80th percentile	Max	Mean	Std Deviation	Min	Max	% Exceedances		
								50th percentile	80th percentile	Max
BOD (mg/L)	-	10	20	3.26	0.88	1.47	7.30	-	0%	0%
Total suspended solids (mg/L)	-	10	20	2.16	1.04	0.42	7.32	-	0%	0%
Total nitrogen (mg/L)	10	20	50	11.95	12.30	1.00	62.50	41%	16%	2%
Total phosphorus (mg/L)	7	10	15	4.88	3.62	0.66	21.08	19%	8%	3%
Escherichia coli (cfu/100mL)	10	-	-	1.16	1.48	0.07	7.89	1%	-	-

Traditionally, centralised water servicing is preferred because of “economies-of-scale” and operational ease, whereas cost and operating issues may be encountered with decentralised systems. Previously, a number of studies identified the drivers for decentralised systems and discussed that a direct comparison should not be made as the decentralised option is able to be tailored for location specific solutions, avoid costly augmentations to centralised systems and avoid the inherent financial risks for new large wastewater infrastructure (Tjandraatmadja *et al.*, 2009; Fane *et al.*, 2006). In addition, it is important to understand the implications and extent of the water-energy nexus for decentralised systems in comparison to other major water and wastewater servicing options.

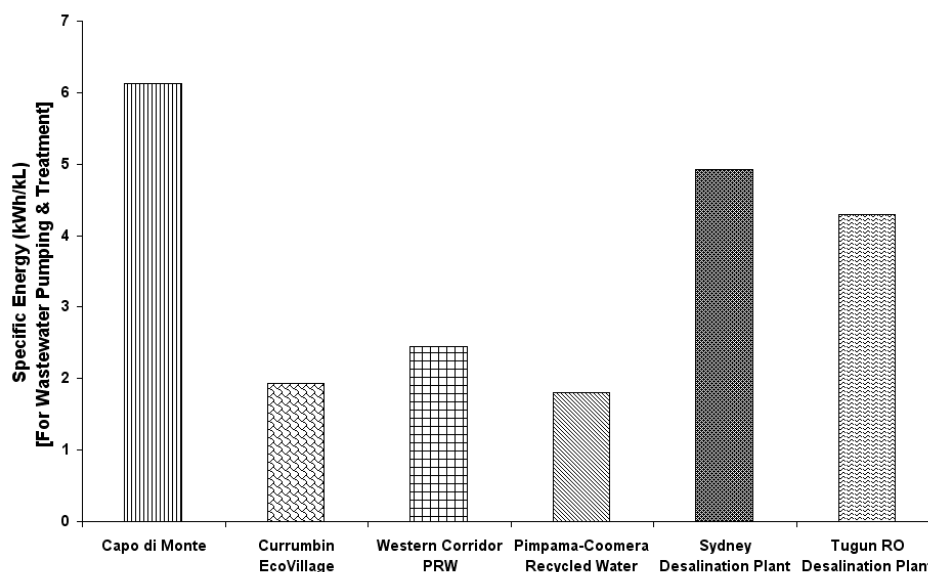


Figure 5. Comparison of specific energy use for wastewater pumping and treatment at our studied sites to other recycled water schemes in Australia.

Figure 5 shows the specific energy (in kilowatt-hours per kilolitre of treated effluent) for the two monitored decentralised wastewater systems. The CDM-STP was found to consume 6.1 kWh/kL whereas; the CEV-STP has a much lower total specific energy requirement of 1.9 kWh/kL. Results in Figure 5 also show the specific energy requirement for CDM is the highest of all the considered recycled water schemes, higher even than the two desalination plants considered. The energy requirements for the CEV-STP is similar to the centralised wastewater treatment facilities in Pimpama-Coomera (Gold Coast) and the Western Corridor purified recycled water (PRW) scheme (Hall *et al.*, 2009; Kenway *et al.*, 2008; Australia Institute Ltd., 2005). Such a comparison suggests that decentralised systems (ie, CEV-STP) have some potential to deliver alternative urban water resources at a better energy cost, if the systems can be properly selected, configured and operated.

When the fugitive greenhouse gases (GHG) were estimated using the first principle approach, the total carbon footprint from the two decentralised systems studied provides a different perspective (Table 2). In this instance, the fugitive GHG emissions of methane (CH₄) and nitrous oxide (N₂O) were estimated based on the Sasse (1998) and Foley (2009) models, respectively.

Table 2. Overall GHG emissions from the decentralised wastewater systems.

Components	Estimated GHG Emissions (kg CO ₂ -e per kL)	
	CDM	CEV
Current average daily wastewater flows (kL/d)	9.3	50.5
Energy related GHG emissions from imported electrical power	5.59	1.81
CH ₄ emissions from identified decentralised process	0	4.92
N ₂ O emissions from identified decentralised process	0.23	0.22
Landfill disposal of screens, grit and bio-solids	0.01	0
Effluent disposal for irrigation	0.02	0.03
Dissolved CH ₄ in raw sewage	0.08	0.08
Chemical and fuel consumption	0.03	0
Total GHG Emissions	5.96	7.06

Table 2 shows the overall GHG emissions from the two decentralised wastewater systems, including the measured energy-related GHG and the estimated fugitive GHG emission values from theoretical approaches. Results indicated that the CH₄ emission from the communal septic tanks in the CEV system significantly exceed the high energy-related GHG emissions measured for CDM. The overall GHG emissions from CEV are estimated at 7.06 kg CO₂-e per kL of treated wastewater compared with 5.96 kg CO₂-e per kL for CDM (ie, a reversal of magnitude when only energy-related GHG was considered).

Conclusion

This research has provided insight into the operational stability, energy use and estimated GHG impacts of two decentralised wastewater technologies. From the outcomes of this study, it can be concluded that MBR operated at a decentralised scale offers a good treatment option in terms of final treated effluent qualities (ie, meeting the license requirements), system robustness, and resistance to shock loadings. However, the utilisation of MBR at CDM comes at the expense of high specific energy use (kWh per kL of treated sewage), which disadvantages the MBR in terms of energy-related GHG emissions. In comparison, the decentralised CEV system provides an effective solution to treat the sewage effluent to Class A⁺ recycled water with much lower specific energy-related GHG emissions. However, when the fugitive GHG emissions from the communal septic tanks at CEV are included, the high CH₄ emission potential along with its uncertainties makes the CEV system operation relatively unattractive based on the theoretical estimation of fugitive gas emissions. Further field investigations are required to substantiate the above estimations. Overall, our findings have provided some useful technical insights into the selection of decentralised technologies to achieve sustainable operations in future urban developments.

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Reduction of Pharmaceutical loads In Municipal Wastewater: Would Onsite Treatment of Hospital Wastewater Be Effective?

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Summary

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. Hospital wastewater is often suspected of being a major source of pharmaceutical residues in municipal wastewater and, as such, in need of treatment prior to discharge. But are such suspicions truly founded? To answer this question, the Alliance Hospital Wastewater project investigated the contribution of six hospitals located in South East Queensland (SEQ) to the loads of 589 pharmaceuticals in municipal wastewater using a predictive consumption-based approach. The results suggest that the contribution of hospitals towards the total load of pharmaceuticals in the influent of sewage treatment plants (STP) is limited. For a majority of the pharmaceuticals consumed at each individual hospital, these hospitals were contributing less than 15% to pharmaceutical loads in municipal wastewater. In contrast, 153 distinct pharmaceuticals returned contributions between 97 and 100% across the six hospitals investigated. Therefore, the risks of these hospital-specific pharmaceuticals being present in concentrations that may present a hazard for human health were investigated. Overall, only 12 compounds of potential concern were identified as warranting more detailed investigation on environmental and human toxicity, biodegradation and source control options. In order to validate the outcomes of this predictive study, the contribution of a 296-bed hospital to a STP serving a population of 75,000 people will be investigated.

Keywords

Hospital wastewater, pharmaceutical loads, contribution, predictive approach, prioritisation, sampling.

Introduction

The continual improvement of analytical equipment over the past 20 years has allowed the detection of more and more pharmaceutical compounds such as antibiotics, analgesics, antidepressants, antineoplastics, beta-blockers and X-ray contrast media at increasingly lower concentrations (down to ng L⁻¹) in municipal wastewater, ground and surface waters and more recently drinking water (Sacher *et al.*, 2001; Fick *et al.*, 2009; Busetti *et al.*, 2010; Metcalfe *et al.*, 2010; Vuilliet *et al.*, 2011). Consequently, the emergence of such contaminants has raised concerns not only for the scientific community but also for the water industry and the general population as these substances are originally designed to be biologically active and could potentially cause adverse effects on aquatic life and human health (Monteiro and Boxall, 2010).

The presence of pharmaceutical residues in the aquatic environment primarily originates from the discharge of treated municipal wastewater. Sources of pharmaceuticals in municipal wastewater includes excretion from humans (Sanderson *et al.*, 2004) and improper disposal of unused medicines (Watts *et al.*, 2007). Agriculture and industries also contribute to pharmaceutical pollution in the environment (Kümmerer, 2004). Finally, in numerous countries including Australia, untreated hospital wastewater is discharged to STPs along with domestic wastewater. Due to their activities, hospitals release high concentrations of various pharmaceuticals and numerous pathogens in their effluent (Thomas *et al.*, 2007; Jury *et al.*, 2011). Therefore, it is not surprising that they are often perceived as major point sources of pharmaceuticals in municipal wastewater (Hawkshead, 2008). Although the decentralised treatment of hospital wastewater has often been proposed as a solution to reduce pharmaceutical inputs to STPs, recent studies concluded that it would be of limited impact on the reduction of pharmaceutical residues in municipal wastewater. In these studies on the characterisation of hospital wastewater in Europe and Australia, the contributions of hospitals were limited, with values under 15% (Thomas *et al.*, 2007, Ort *et al.*, 2010). Such studies are, however, based on results obtained for a limited number of experimentally measurable substances, while hundreds of various pharmaceuticals are consumed in hospitals and may be of greater concern than the one currently analysed for.

Prior to field measurements, it is necessary to identify and quantify pharmaceuticals of concern and to determine if pharmaceuticals exclusively used in hospital should receive priority attention when compared to pharmaceuticals used by the general population. For this purpose, a prioritisation tool based on pharmaceutical consumption data at six hospitals located in SEQ was developed. This tool allowed predicting the contribution of these hospitals to the loads of 589 pharmaceuticals in municipal wastewater, hence identifying compounds for which hospitals would be a major contributor. Pharmaceuticals exclusively used in hospitals (ie, returning a 100% contribution) were further screened by predicting their concentration in both hospital wastewater and influent of the corresponding STP to evaluate their possibility to be present at levels that may pose a risk for human health. To confirm the outcomes of

this study and validate our consumption based approach, the contribution of a 296-bed hospital to a STP serving a population of 75,000 people will be investigated experimentally.

Material and Methods

Hospitals Investigated

The six hospitals selected are public hospitals located in three distinct catchments with populations ranging from 45,000 inhabitants, in the catchment where the Caboolture hospital (CAB) is located, up to 572,000 inhabitants in the catchment including the Prince Charles (PC), Princess Alexandra (PA) and Royal Brisbane and Women's (RBWH) hospitals (Figure 1). QEII hospital is the smallest of the six hospitals with a total number of 132 beds. The general and teaching hospital, RBWH, is the largest hospital with 882 beds. The volumes of water consumed at these two hospitals in 2008 were $95 \text{ m}^3 \text{ day}^{-1}$ and $627 \text{ m}^3 \text{ day}^{-1}$ respectively. QEII discharges its effluent to the Oxley Creek STP, which in 2008 treated on average $55,336 \text{ m}^3 \text{ day}^{-1}$. RBWH discharges its effluent to the Luggage Point STP, with an average of $148,622 \text{ m}^3$ of wastewater treated per day in 2008.

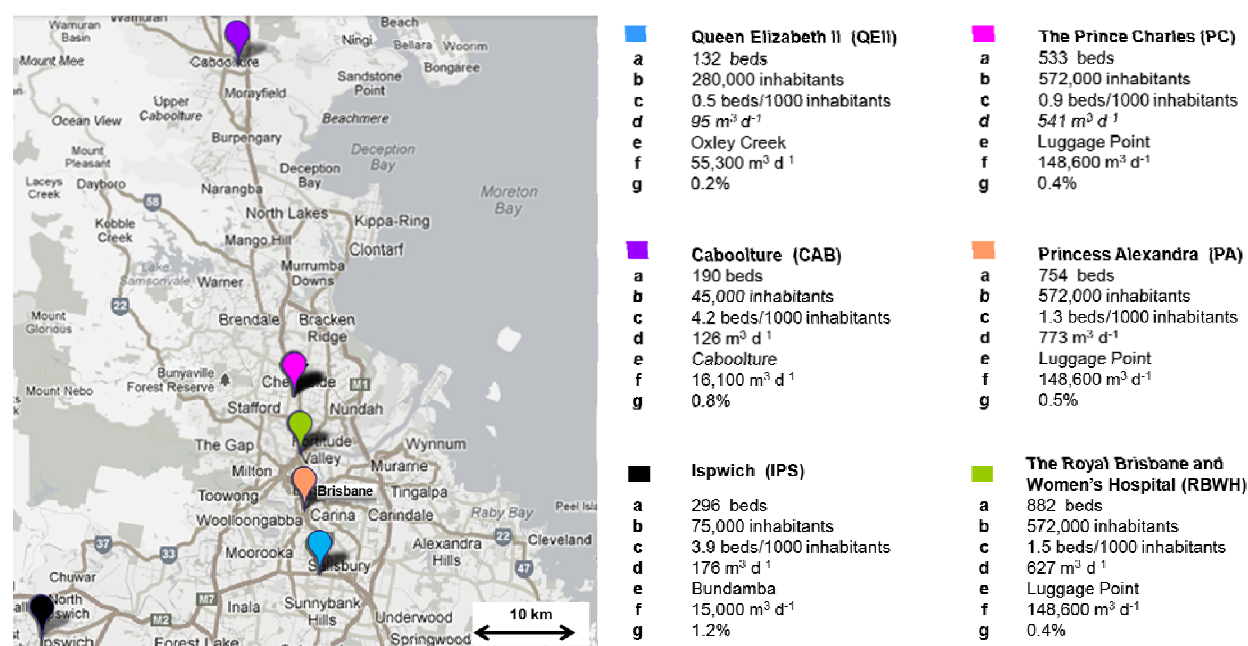


Figure 1. Location of the six hospitals investigated in SEQ and their characteristics (a - number of beds; b - population in catchment area; c - bed density; d - hospital water consumption; e - STP to which the hospital discharge its effluent; f - average raw influent flow rate to STP; proportion of STP influent wastewater coming from the hospital).

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

Annual pharmaceutical consumption audit data collected from public hospitals in Queensland (Source: Medication Services Queensland, Queensland Health, Clinical and Statewide Services Division, Queensland Government) were compared with pharmaceutical consumption by the general population (Source: Drug Utilisation Sub-Committee (DUSC, Department of Health and Ageing, Australian Government) to predict the contribution of a hospital to the loads of pharmaceutical in influent of the corresponding STP.

Overall, 589 active pharmaceutical ingredients (API) to evaluate were extracted from the hospital audit data base. These substances excluded naturally occurring substances such as hormones, sugars and enzymes as well as gaseous substances. Compounds available over the counter in Australia (TGA, 2011) were also excluded from the list since information on consumption for these substances are not available.

The consumption of a pharmaceutical in a catchment of a STP was estimated by calculating an average per capita consumption from the national consumption data multiplied by the number of inhabitants in the catchment. The consumption by in-patients in the hospital was added to the domestic consumption to obtain an estimate for the total STP influent load according to equations 1 and 2 below:

$$\text{Contribution}_{\text{Hospital}_{i,j}} = \left[\frac{\text{Consumption}_{\text{Hospital}_{i,j}} \cdot X_r}{\left(\text{Consumption}_{\text{Catchment}_j} \cdot X_r \right) + \left(\text{Consumption}_{\text{Hospital}_{i,j}} \cdot X_r \right)} \right] \quad (1)$$

where:

$\text{Consumption}_{\text{Hospital}_{i,j}}$ is the consumption of the hospital i in catchment j for an API [g y^{-1}];
 $\text{Consumption}_{\text{Catchment}_j}$ is the consumption of the same API by the general population in the catchment j where hospital i is located [g y^{-1}] (see equation 2);
 X_r is the excretion rate for a given API [-]. *Note: this parameter cancels out of equation 1 when calculating contributions of hospitals.*

$$\text{Consumption}_{\text{Catchment}_j} = \left[\frac{\text{Consumption}_{\text{Australia}}}{\text{Population}_{\text{Australia}}} \right] \cdot \text{Population}_{\text{Catchment}_j} \quad (2)$$

where:

$\text{Consumption}_{\text{Australia}}$ is the consumption data for a single API provided in the national consumption database [g y^{-1}];
 $\text{Population}_{\text{Australia}}$ is the number of inhabitants in Australia, here rounded to 20,000,000;
 $\text{Population}_{\text{Catchment}_j}$ is the number of inhabitants in the catchment j .

Hospital-Specific Compounds: Comparison of Predicted Concentrations with Effect Thresholds

In order to assess if hospital-specific compounds (ie, 100% contribution) may present a hazard for human health, estimated concentrations ($\mu\text{g L}^{-1}$) in both hospital wastewater (concentration eff.Hospital_i) and influent of the corresponding STP (concentration inf.STP_j) were compared to effect thresholds (ET) in the form of margin of exposure (MOE) according to equation 3.

$$\text{MOE} = \frac{\text{ET}}{\text{Concentration}_{\text{eff.Hospital}_i \text{ OR } \text{inf.STP}_j}} \quad (3)$$

ETs ($\mu\text{g L}^{-1}$) 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). This approach is highly conservative as it uses stringent guidelines determined for recycled water intended for (in)direct potable water supply to assess risk associated with wastewater prior to any treatment. For each hospital-specific compound, ET values were determined based on acceptable daily intakes (ADI, in $\mu\text{g kgBW}^{-1} \text{ d}^{-1}$). When not available, a substitute ADI (S-ADI, in $\mu\text{g kgBW}^{-1} \text{ d}^{-1}$), was determined using the Lowest Daily Therapeutic Doses (LDTD, in $\mu\text{g d}^{-1}$) available in the Australian Medical Information Management System (MIMS Australia, 2011).

Any MOE value above 1 indicates that predicted concentrations are below the calculated effect thresholds. To add another safety step to the approach as a precautionary principle, a conservative MOE of 100 has been chosen as the limit below which compounds would potentially cause concerns for human health. A $\text{MOE} > 100$ implies that the pharmaceutical concentration predicted in either wastewater type (hospital effluent or STP influent) is more than 100-fold below a “concentration of no concern”. This means that such a compound is unlikely to present a risk of reaching drinking water sources at elevated concentrations and to affect human health.

Results

Predictive Approach

Hospital Contributions

Out of 589 compounds investigated, the number of compounds returning a contribution value was 487 at QEII and CAB, 502 at IPS, 524 at PC, 541 at PA and 548 at RBWH. Overall, the six hospitals were found to contribute from 1% (in the catchment of QEII hospital) to 9% (in the catchment of PA, PC and RBWH hospitals) of the total pharmaceutical load at the corresponding STP (Table 1). Reducing pharmaceutical loads in municipal wastewater through onsite treatment of these hospitals effluent would therefore be of limited benefit.

Table 1. Overall contribution of hospitals to the total loads of pharmaceuticals in the corresponding STPs.

Hospital	QEII	CAB	IPS	PC	PA	RBWH
Catchment	Oxley	CAB	IPS	Luggage Point		
Total mass used in hospitals (g d ⁻¹)	481	340	556	2,640	1,948	2,324
Total mass consumed by the general population (g d ⁻¹)	35,624	5,725	9,542	72,775	72,775	72,775
Total mass in the corresponding STP influent (g d ⁻¹)	36,105	6,065	10,098	79,688		
Overall Contribution	1%	6%	6%	9%		

When looking more specifically at the contributions of individual hospitals to loads of pharmaceuticals in municipal wastewater, for 63 to 84% of the API used, these contributions are likely to be less than 15%. It should be noted that the percentage of APIs for which the contribution of an individual hospital is below 15% decreases with the increasing size of the hospital, in terms of number of beds. For example, the contribution of the smallest hospital QEII (132 beds) was found to be less than 15% for 407 pharmaceuticals, while for RBWH, the largest hospital (822 beds), it was less than 15% for 348 substances. These results suggest that for a large amount of the compounds investigated, hospitals are not a major point source of pharmaceutical residues in municipal wastewater. This implies that for these compounds at least 85% of the loads originate from households and would reach the corresponding STP even if hospital effluents were treated separately. On the contrary, the number of hospital-specific APIs, ie, a contribution between 97 and 100%, increased with the size of the hospital. For instance, 54 hospital-specific APIs were found at QEII and 123 at RBWH, confirming the belief that a higher number of hospital-specific compounds are used at larger hospitals.

In 2009, a sampling campaign carried out by Ort *et al.* (2010) at CAB hospital showed that measured contributions for 75% of the compounds investigated were in good agreement with our predicted contributions. For example, we predicted contributions of 2.5% and 0.5% for metoprolol and atenolol at this hospital. These values compare well with the results obtained experimentally by Ort *et al.* (2010), which fell in the range 2.0 -7.0% for metoprolol and 0.9-3.5% for atenolol. Other compounds investigated by Ort and co-workers at CAB hospital included trimethoprim and roxithromycin. These were the only two substances detected at both the hospital and its corresponding STP which resulted in contribution above 15%, with a maximum contribution of 18% obtained for trimethoprim and 56% for roxithromycin when using a conservative approach to account for experimental uncertainties. However, average contributions experimentally determined by Ort *et al.* (2010) for these two compounds were 10% for trimethoprim and 26% for roxithromycin which are close to the predicted average contributions obtained in our study at CAB hospital (ie, 13% for trimethoprim and 19% for roxithromycin).

Hospital-Specific Compounds

Results of our predictive approach showed that for 54 to 123 compounds, hospitals would be a major point source. However, high contribution from a hospital may not be necessarily associated with high consumption values and excretion in the hospital. For example, the antiviral abacavir returned a 100% contribution at four of the six hospitals. For this compound, consumptions varied from 0.06 g y⁻¹ bed⁻¹ at IPS hospital to 0.3 g y⁻¹ bed⁻¹ at QEII hospital. Based on water consumption at the hospitals, assuming no metabolism, concentrations expected in these hospital effluents would be 0.3 µg L⁻¹ and 1.0 µg L⁻¹. As a comparison, concentrations for abacavir in influent of the STP to which these hospitals discharge their effluent would range from 0.003 to 0.004 µg L⁻¹.

It is also necessary to understand how individual hospital-specific compounds are prescribed and used in hospitals. For instance, in Australia, HIV antiretroviral drugs such as abacavir are subsidised under the “Highly Specialised Drug Program” and are only prescribed by qualified medical practitioners through hospital-based pharmacies (HSDP, 2011). It is therefore not surprising that a 100% contribution is predicted for these types of drugs. It should also be noted that treatments with antiretroviral drugs are not curative but help manage HIV infections, and are long-term treatments (Anderson and Lennox, 2009). Consequently, they are more likely to be excreted at home rather than in hospitals, suggesting that the contributions of hospitals for this type of drugs are extremely overestimated. In fact, a recent report by McArdell *et al.* (2011) experimentally quantifying mass flows of 100 pharmaceuticals in wastewater of a Swiss hospital and municipal wastewater showed that the contribution of that hospital for ritonavir (one of the top 100 of compounds prescribed in that hospital) was only 0.9%.

In order to assess if hospital-specific compounds could have an impact on risks of human exposure to these substances, concentrations in all hospital effluents and municipal wastewater were predicted based on water consumption and compared to effect threshold (ET) concentrations as MOE.

The comparison of concentrations predicted in hospital effluents with the calculated ET for compounds exclusively originating from hospitals showed 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 values. In STP influents, the percentages of compounds for which MOEs would be more than 100-fold lower than ET values increased to values in the range of 90 to 100%. This indicates that only a small percentage of compounds originating from hospitals may be of concern.

The results obtained at QEII hospital are illustrated in Figure 2. In the effluent of this hospital, the concentrations of 15 of the 54 hospital-specific compounds were less than 100-fold lower than the calculated ET values (Figure 2A). MOEs for these compounds varied from one for the local anaesthetics ropivacaine and oxybuprocaine to 70 for the antibiotic meropenem and the anaesthetic agent ketamine. However, when determining MOE values in the influent of the STP to which this hospital discharge, none of these 15 compounds returned a MOE below 100 (Figure 2B). The expected concentrations in STP influent (based on flow rate data and assuming no metabolism) would all be more than 500 times lower than the ET values, making these compounds unlikely to increase risks of human exposure to any of these hospital-specific substances.

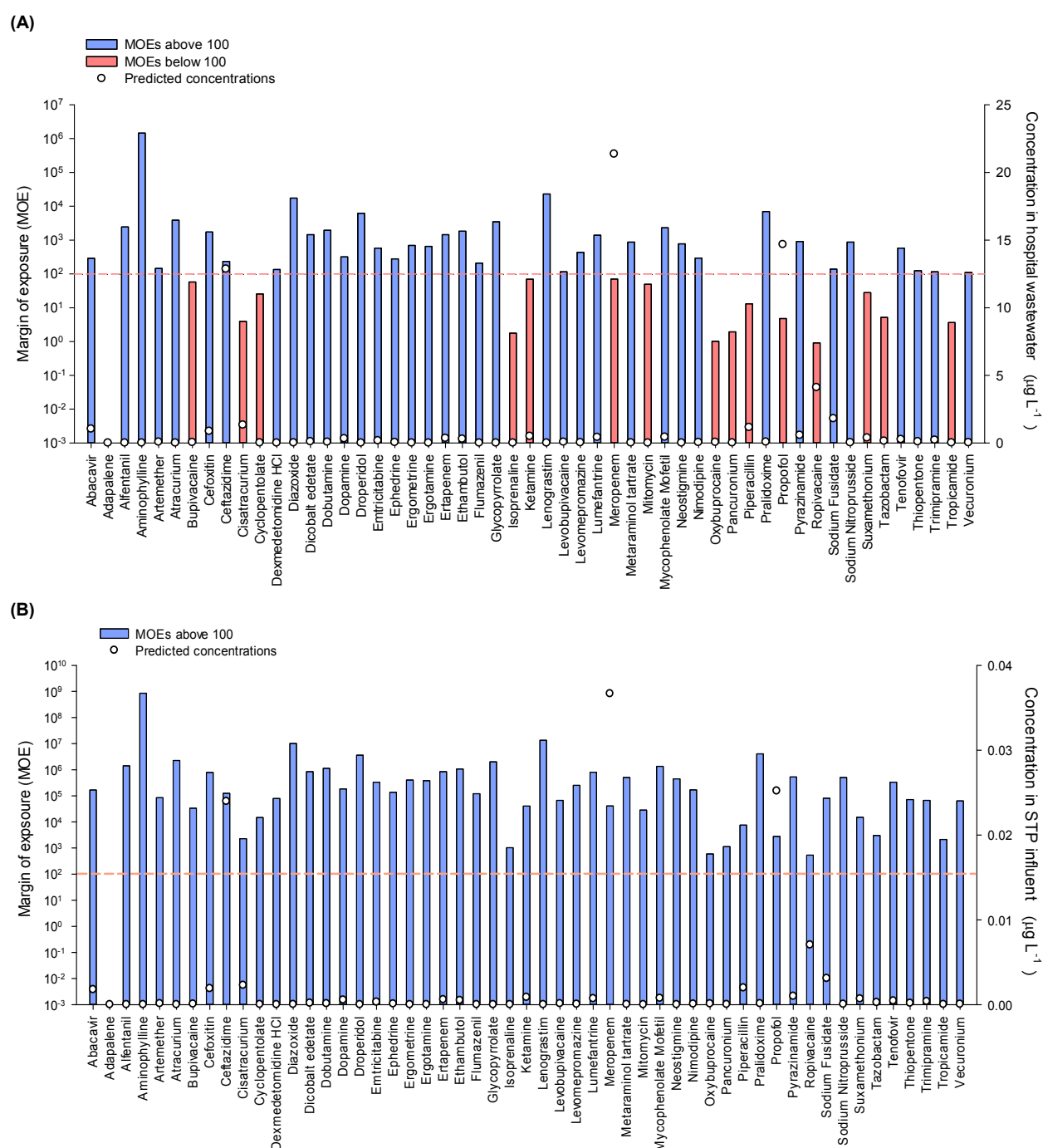


Figure 2. Predicted concentrations and MOE in (A) the effluent of QEII hospital and (B) in the influent of the corresponding STP for hospital-specific compounds (*ie*, 100% contribution). The dashed line corresponds to a MOE of 100.

At the largest hospital, RBWH, MOEs in the hospital effluent were below 100 for 41 out of the 123 hospital-specific compounds. In the influent of the STP to which RBWH hospital discharges its effluent, only nine out of the 123 hospital-specific compounds with MOEs below 100 remained. These included vincristine sulphate (MOE=0.4), tazobactam (MOE=3) and piperacillin (MOE=8).

Table 2 summarises the final list of compounds for which hospital-specific substances would result in MOE values below 100 in influents of the corresponding STPs. As can be seen, only twelve distinct hospital-specific pharmaceuticals remained in concentrations which may be of concern (ie, MOE<100).

Among these, three antibiotics were found: cefazolin, piperacillin and tazobactam. Although risks of direct human exposure to traces of antibiotics in the environment have not been demonstrated yet, major health concerns relate to the possible development of antibiotic-resistance bacteria, hence antibiotic-resistant genes that may transfer to human pathogens (Fick *et al.*, 2009). Reinthaler *et al.* (2003) investigated the resistance of *E. coli* strains isolated in sewage and sludge to a set of 24 antibiotics. These included two of the hospital-specific antibiotic drugs (piperacillin and tazobactam) remaining here with MOEs below 100 in STP influent. No resistance to the combination piperacillin/tazobactam was found by *Escherichia coli* (*E. coli*) strains isolated from the influent of a STP conjointly treating municipal and hospital wastewater. The highest degree of *E. coli* resistance that they observed was for the antibiotic tetracycline. Among the six hospitals investigated in our study, RBWH was found to be a major contributor to the loads of tetracycline in municipal wastewater, but corresponding MOEs in both hospital effluent and municipal wastewater were well above 100,000. This suggests that, if our approach helps screening antibiotics for which hospitals would be major contributors and of potential concern for human health, further investigations on potential human health risks resulting from the spread of antibiotic-resistant bacteria that may originate from hospitals are warranted.

Table 2. List of hospital-specific compounds* with a MOE below 100 in influents of the STPs to which the hospitals investigated discharge their effluents. (values in grey are MOE values above 100).

Hospitals			QEII	CAB	IPS	PC	PA	RBWH
Number of hospital-specific compounds			54	56	74	92	112	123
Corresponding STP			Oxley	Caboolture	Ipswich	Luggage Point		
Number of compounds with a MOE ≤100			0	3	3	8	9	9
Generic name (API)		Therapeutic class	MOE					
1	Bupivacaine	AA	33663	71	47	69	69	69
2	Piperacillin	AB	7599	79	2058	8	8	8
3	Tazobactam	AB	3030	32	820	3	3	3
4	Oxybuprocaine	AA	594	248	126	71	71	71
5	Pancuronium	NB	1122	NU	912	48	48	48
6	Ropivacaine	AA	532	365	892	68	68	68
7	Tropicamide	MY	2121	1415	519	53	53	53
8	Cefazolin	AB	NC	NC	NC	32	NC	NC
9	Infliximab	ARh	NU	NU	NU	NU	81	81
10	Vincristine Sulphate	AN	NU	NU	NU	NU	0.4	0.4
11	Levobupivacaine	AA	67325	447	100	NU	NU	2978
12	Suxamethonium	AA	15213	256	98	371	357	357

AA= Anaesthetic agent; AB= Antibiotic; AN= Antineoplastic; ARh= Antirheumatic agent; MY= Mydriatic; NB= Neuromuscular blocking agent; NU= Not Used at the hospital; NC: not considered (ie, contribution <97%).

*Contributions comprised between 97 and 100 % were taken into account.

Seven antineoplastic agents (anagrelide, capecitabine, procarbazine, carmustine, vincristine, busulfan and mitomycin) presented MOE values below 100 in the hospital effluents at four of the six hospitals investigated. But concentrations in the corresponding STPs dropped significantly, making vincristine the only cytotoxic compound remaining with a MOE below 100 in the catchment of PA and PC hospitals (Table 2), with concentrations below 0.012 µg L⁻¹. Although such a concentration seems low and in accordance with low concentrations typically observed for this category of substances in the environment (Webb, 2004), it would deserve additional investigations. Indeed, anticancer drugs are among the most toxic substances used in medicine and are known to be poorly biodegradable (Aherne *et al.*, 1990, Kümmerer, 2004).

The real impact of hospital effluents on the load of anticancer drugs in municipal wastewater is difficult to assess. The administration of some of these compounds to out-patients as well as the slow excretion of some of these substances (eg, capecitabine, fluorouracil) means that significant fractions of antineoplastic drugs are excreted at home (Johnson *et al.*, 2008). A trend towards home-based administration of anticancer treatments has been recently

confirmed in France by Besse *et al.* (2012). Their analysis of consumption data from a local chemotherapy centre showed that 50% of the antineoplastic agents consumed in that centre were prescribed to out-patients and that only 20% of the drugs prescribed to out-patients were excreted onsite. This trend implies that hospitals may no longer be a major source of chemotherapeutic drugs.

Experimental Approach

In order to validate the outcomes of our predictive study, the contribution of a 296-bed hospital to a STP which serves a population of 75,000 people will be investigated. The concentration of a set of pharmaceuticals previously measured by Ort *et al.* (2010) at CAB hospital will be measured to determine the contribution of this hospital to the loads of the selected compounds in influent of the STP. This will also allow comparing future results with results obtained at CAB hospital. In addition, it is planned to investigate the presence of the 12 hospital-specific substances prioritised using our predictive approach and to also run bioassays on both hospital wastewater and raw municipal wastewater for global toxicity assessments.

In order to collect the hospital wastewater and minimise sampling error, a flow-proportional sampling system has been installed in a sewer collecting effluent from the hospital (Figure 3A and 3B). A similar flow-proportional sampling system has also been installed at Bundamba STP (Figure 3C). Both systems will sample wastewater from both sites over 24 hour cycles during three consecutive days. Samples collected daily will be transported to the University of Queensland, filtered and extracted for pharmaceutical analyses.

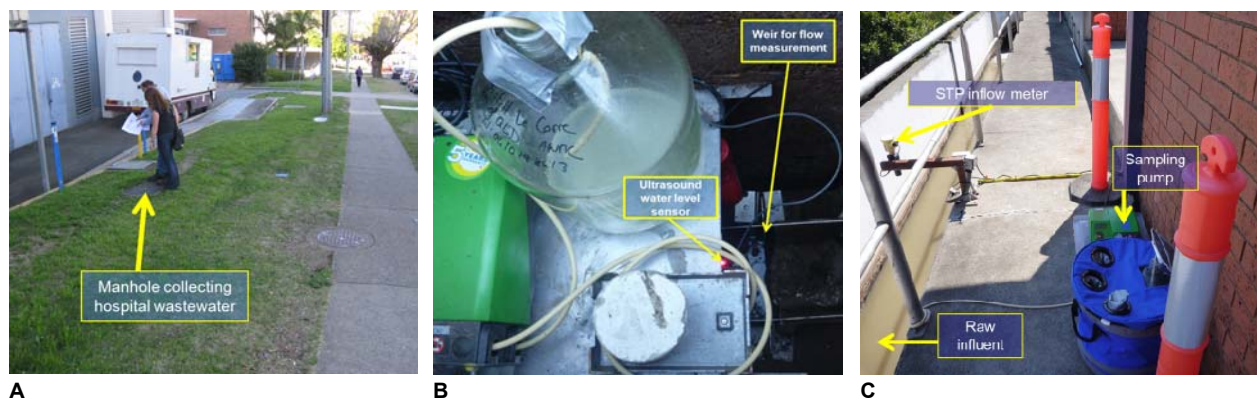


Figure 3. Sampling location. A: sewer location, B: flow-proportional sampling system in the manhole; C: flow-proportional sampling system at the STP.

Conclusions

The consumption-based approach used in this study showed that:

- The impact of SEQ hospitals on pharmaceutical pollution in municipal wastewater is limited.
 - The six hospitals investigated over all contribute less than 6% of the total mass of APIs consumed in a catchment.
 - For up to 84% of the 589 APIs evaluated, the contribution of an individual hospital is likely to be less than 15%.
- Depending on the hospitals investigated, 100% contributions were obtained for 54 at QEII to 123 APIs at RBWH,
 - Among these hospital-specific compounds, the predicted concentrations of only 12 compounds were less than 100 times below a concentration “of no concern” in the influent of STPs. They warrant more detailed investigations including environmental and human toxicity, biodegradation and treatment or source control options.
 - It should be noted that concentrations of pharmaceuticals in raw wastewater are expected to be significantly reduced after conventional wastewater treatment and advanced water treatment. Therefore, the results obtained for hospital-specific compounds indicate that these are unlikely to be present in STP effluents at levels representing a risk to humans.
- The outcomes of this study suggest that decentralised treatment of hospital wastewater to reduce pharmaceutical loads in municipal wastewater would be of limited benefit. However, additional aspects including the impact of hospital wastewater on the propagation of antibiotic resistant bacteria, will require specific attention to fully evaluate whether source treatment of hospital wastewater is relevant or not.

Overall, consumption audit data are a good predictor of a hospital contribution. They allow screening active pharmaceutical ingredients used in hospitals and identifying those of potential concern that may require monitoring. The consumption based approach developed in this study represents an additional step towards prioritisation of pharmaceuticals originating from hospital wastewater that is transferrable to other countries depending on availability and quality of audit data.

Acknowledgements

The authors thank the South East Queensland Urban Water Security Research Alliance for their financial support; Queensland Health and the Medication Services Queensland for providing hospital audit data and their support regarding understanding and processing of the hospital audit data; Greg Jackson from Queensland Health for his helpful comments during this study; Vanna Mabbott and Chris Raymond from the Drug Utilisation Sub-Committee (DUSC) of the Pharmaceutical Benefits Advisory Committee (PBAC) (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, flow data of the STPs investigated and allowing the installation of our sampling systems; Lendlease for the installation of our flow-proportional sampling systems.

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Antibiotic Resistant Bacteria in Hospital Wastewaters and Sewage Treatment Plants

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Summary

We investigated the presence and survival of antibiotic resistant bacteria in untreated hospital wastewaters (UHW) and their transmission to the receiving sewage treatment plant (STP) in South East Queensland (SEQ), Australia. Over eight weeks of sampling, 245 *Escherichia coli* and 167 *Staphylococcus aureus* strains were collected from UHW and its receiving STP inlet (STP-I) and post-treatment outlet (STP-O). These strains were typed using the PhP typing method and random amplified polymorphic DNA-polymerase chain reaction (RAPD-PCR). Among *E. coli* strains collected from UHW, seven common (C) PhP-RAPD types were frequently found in a majority of weekly samples with multi-drug resistant (MDR) C-types also found in STP-I and on three occasions in STP-O. Similarly among *S. aureus* strains, seven C- types were frequently found in a majority of samples with two MDR C-types also found in STP-I and on two occasions in STP-O. Our data suggest that some MDR bacterial strains found in UHW may have the ability to survive transmission to the STP and then through to the final treated effluent before being released into surface waters.

Keywords

Multi-drug resistant bacteria, hospital wastewater, sewage treatment plant.

Introduction

The wide application of antimicrobial agents in clinical settings to treat infectious disease, as well as their use in aquaculture and veterinary medicine, is of great concern to public health as this can lead to the development and evolution of antibiotic resistant bacteria (Islam, 2011; Mazel and Davis, 1999; Wise *et al.*, 1998). This occurs as a result of the high selective pressure that antibiotics place on bacteria, resulting in the proliferation and subsequent dissemination of resistant bacteria. Resistance genes can also be transferred between cells on plasmids or transposons by transductive or conjugative processes (Berger-Bachi, 2002). DNA elements which mediate integration of resistance genes (eg, integrons) may also be involved (Moura *et al.*, 2011) resulting in the further spread of multi-drug resistant (MDR) bacteria.

Hospitals provide an environment conducive to MDR bacteria, making the treatment options limited and expensive (Magiorakos *et al.*, 2011). Furthermore, not enough is known about their release and survival from hospital wastewater, through the sewerage system and finally into treated effluent released by sewage treatment plants (STPs) into the environment. Sewerage systems also carry other waste materials from the community and industry and so MDR bacteria must survive a long, hostile transition route, including final disinfection, before they are released into surface waters.

We proposed that due to their possible persistence in hospitals, MDR bacterial strains could be frequently found in untreated hospital wastewater (UHW) and, despite the high dilution rate occurring in the sewer systems, these strains may travel to the STP and be detected in both the influent and the treated effluent of the receiving STP. To provide evidence for this hypothesis, we identified bacterial strains found in the untreated wastewater of a hospital in subtropical South East Queensland (SEQ) and traced their movement to the receiving STP to determine if strains survived the sewerage collection and treatment process before treated municipal wastewater is released into the environment. For this study, we focused on two key pathogens: 1) a Gram-negative bacterium *Escherichia coli* which, although a common inhabitant of the human gut flora, can also cause several important nosocomial infections such as urinary tract infection, septicaemia and meningitis (Johnson *et al.*, 2005); and 2) a Gram-positive bacterium *Staphylococcus aureus* which, apart from its well established pathogenicity in hospitalised patients, is also a normal inhabitant of skin of healthy individual and is found in between 25-30% of the interior nares of healthy individuals (Krishna and Miller, 2011).

Methods

Hospital samples were collected from the untreated wastewater outlet pipe of a selected hospital in the subtropical SEQ before it enters the sewer system. The sewer channel taking hospital wastewater to the STP was estimated to be 12.5 km and in view of the high dilution of bacteria in sewage system while travelling to receiving STP, the sampling period was extended for two months to increase the chance of detecting bacterial strains found in UHWW. Using “grab-sampling” technique, water samples were collected for eight consecutive weeks from UHWW and its receiving STP at 10.30am and at 11.00am of the same day respectively. The STP was an activated sludge plant with N and P reduction and services an equivalent population of 130,000 and has a 12-13 day sludge age. Samples were collected from the incoming raw sewage (STP-I) and treated effluent after the activated sludge treatment and chlorination (STP-O). The final effluent is discharged into a nearby waterway. All samples were processed in accordance with the Australian and New Zealand Standards for Water Microbiology and Water Quality Sampling (ANZ standard water microbiology method, 2007). In brief, wastewaters were collected in 500 ml sterile microbiological containers mounted onto a handle of appropriate length. They were transported to the laboratory on ice and processed within 4 hours of collection.

Up to 16 *E. coli* colonies (where possible) were randomly collected from each UHWW sample at each occasion. If samples from the STP outlet were positive for *E. coli* up to 12 colonies (where possible) were isolated for subsequent fingerprinting. In all, 245 *E. coli* isolates were isolated from UHWW (n= 120), STP-I (n=102) and STP-O (n= 23). A similar approach was used for *S. aureus* strains isolated from hospital wastewaters and the STP. In all, 167 *S. aureus* strains were isolated from UHWW (n=85), STP-I (n=74) and STP-O (n=8).

These strains were typed using a high resolution biochemical fingerprinting method (the PhP-RE system for *E. coli* and PhP-FS for *S. aureus*) according to the manufacturer’s instruction and RAPD-PCR method as outlined in Naffa *et al.* (2006). Strains having identical PhP/RAPD pattern were regarded as identical and grouped into common (C) types whilst strains with different PhP and/or RAPD types were regarded as single (S) types. Using the method of Clinical Laboratory Standard Institute (CLSI, 2011), a representative strain of each C-type from UHWW and STP samples was then tested for their antibiotic resistance against nine (for *S. aureus*) and 16 (for *E. coli*) antimicrobial agents. For *S. aureus*, these included tetracycline (30µg), amoxycillin-clavulonic acid (20/10µg), ampicillin (10µg), gentamicin (10µg), ciprofloxacin (5µg), chloramphenicol (30µg), amikacin (30µg), ceftazidime (30µg) and vancomycin (8 µg). For *E. coli* the antimicrobial agents included tazocin (TZP 55µg), cefotetan (CTT 30µg), cefpodoxime (CPD 10µg), ceftazidime (FOX 30µg), imipenem (IMI 10µg), gentamicin (GEN 10µg), nitrofurantoin (NIT 300µg), trimethoprim (TMP 5µg), sulphafurazole (SF 300µg), sulphamethoxazole (RL 100µg), tetracycline (TET 30µg), ciprofloxacin (CIP 5µg), chloramphenicol (C 30µg), nalidixic acid (NAL 30µg), kanamycin (AK 30µg), and norfloxacin (NOR 10µg).

Results

Common and Persistent Strains in UHWW and STP

Among *E. coli* strains collected from UHWW, seven C-PhP-RAPD types were frequently found in the majority of the weekly samples. These strains were all MDR and were also found in STP-I and in three occasions in STP-O (Table 1). Strains belonging to these C-types constituted 43% of the isolates (n=52 out of 120) from UHWW. Sixty-eight *E. coli* isolates from UHWW were found in only one sampling occasion and were regarded as S-types.

Typing of the *S. aureus* isolates also showed the presence of seven C-PhP-RAPD types constituting 65% of the isolates (n=55) in samples collected from UHWW. The remaining 30 isolates were found in only one sampling occasion and were regarded as S-types. Comparison of isolates belonging to C-types with those found in samples collected from the STP (both inlet and outlet) showed the presence of strains with identical PhP-RAPD types (Table 1).

Antibiotic Resistance Patterns of Strains from Hospital Wastewater and STP

The pattern of antibiotic resistance among the strains varied in samples collected from different sites. In all, 92% of *E. coli* strains and 86% of *S. aureus* strains were resistant to more than two antibiotics (MDR strains), with the highest number being resistant to 12 antibiotics for *E. coli* (7%) and nine for *S. aureus* (6%). The mean number of antibiotics to which *E. coli* strains from the UHWW were resistant to (6.72 ± 2.8) was significantly higher than those found among STP isolates (3.1 ± 1.3) ($p < 0.0001$). Similarly, the mean number of antibiotics to which *S. aureus* strains from UHWW were resistant to (5.14 ± 2) was significantly higher than those found in STP isolates (2.9 ± 1.9) ($p < 0.0001$).

Table 1. *E.coli* and *S. aureus* strains belonging to common (C) types found over eight weeks of sampling (W1-W8) from untreated hospital wastewater (UHW) and its receiving sewage treatment plant (STP). STP-I: STP influent, STP-O: STP outlet. Only strains found in both UHW and the STP-I or STP-O have been included. +: indicates the presence of the strains in both type of effluent.

PhP/RAPD C-types	Sites Where Bacteria Were Found	Weeks where Bacteria were Found							
		W1	W2	W3	W4	W5	W6	W7	W8
<i>E. coli</i>									
C1	UHWW and STP-I	+	+	-	-	-	-	+	+
C2	UHWW and STP-I and STP-O	+	+	+	+	+	+	+	+
C3	UHWW and STP-I	-	-	+	+	-	-	+	+
C4	UHWW and STP-I and STP-O	-	+	+	+	+	+	+	-
C5	UHWW and STP-I	+	-	+	-	+	+	+	-
C6	UHWW and STP-I	+	+	-	-	+	+	-	+
C7	UHWW and STP-I and STP-O	+	-	+	+	+	+	-	+
<i>S. aureus</i>									
C1	UHWW	+	+	+	-	+	-	-	-
C2	UHWW and STP-I	+	-	-	-	-	-	-	+
C3	UHWW and STP-I	-	+	+	-	+	+	-	-
C4	UHWW and STP-I	-	-	-	-	-	+	-	-
C5	UHWW and STP-I and STP-O	-	-	-	+	+	+	+	-
C6	UHWW and STP-I and STP-O	-	-	-	-	+	+	+	-
C7	UHWW and STP-I	-	-	-	-	-	+	-	-

Among *E. coli* strains isolated from UHW, the highest resistance was observed against aztreonam, gentamicin, amoxicillin-clavulanic acid, ceftazidime, cefepime ranging from 89% to 79% with the lowest resistance found against ciprofloxacin; norfloxacin, nalidixic acid, nitrofurantoin and chloramphenicol (0% to 12%).

Among the *S. aureus* strains, the highest resistance was observed against ampicillin followed by amoxicillin/clavulonic acid, gentamicin and cefoxitin ranging from 100% to 78% with the lowest resistance to vancomycin (Figure 1).

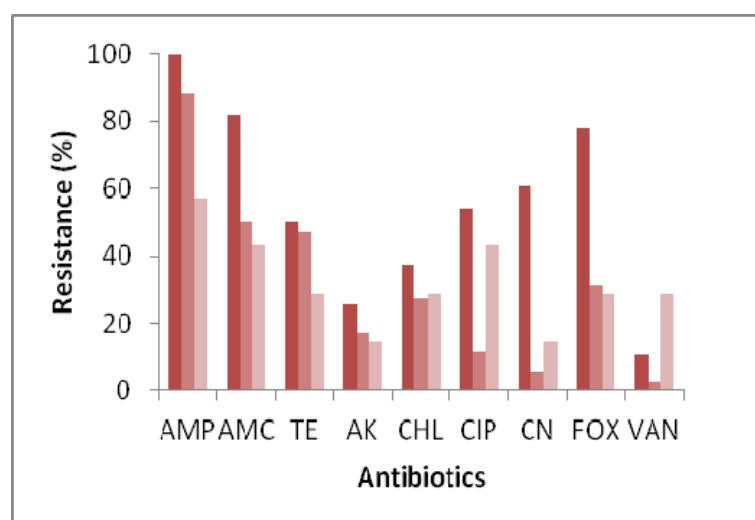


Figure 1. Prevalence of antibiotic resistant *S. aureus* strains isolated from UHW (dark red) and its receiving STP inlet (medium red) and outlet (light red). AMP=ampicillin, AMC= amoxicillin/clavulonic acid, TE=tetracycline, AK=amikacin, CHL=chloramphenicol, CIP=ciprofloxacin, CN=gentamicin, FOX=cefoxitin, VAN=vancomycin.

MDR strains belonging to the same C-types in samples collected from UHW and STP (both inlet and outlet) typically had identical antibiotic resistance patterns although, on some occasions, strains isolated from STP-O showed less resistance to one (*E. coli* C-type 4) or two (*E. coli* C-type 7) antibiotics.

Significance and Impact

In this study we traced the movement and survival of two important bacterial pathogens, *E. coli* and *S. aureus* from the untreated hospital wastewater to a receiving STP and its discharged effluent. Typing of the isolates using a combination of a high resolution PhP typing and RAPD-PCR confirmed that certain PhP-RAPD types of these bacteria were commonly found in untreated hospital wastewaters. In our study, these strains were regarded as common types. Using the sampling protocol in this study we were able to isolate some of the persistent hospital strains from the inlet of the STP and showed that they belonged to the same PhP-RAPD types.

Originally, we were interested in identifying the presence and survival of antibiotic resistant bacterial strains in both the hospital wastewaters and the STP, but in view of the high level of antibiotic resistance found among both *E. coli* and *S. aureus* strains in hospital wastewaters, and in view of the high diversity of the strains that were only found in one sampling occasion from UHWW (68 *E. coli* S-types and 55 *S. aureus* S-types), we made a comparison between the level of antibiotic resistance found among the strains from hospital wastewaters and the STP. The results indicated that the mean number of antibiotics to which bacteria from hospital wastewater were resistant was significantly higher than those found in the STP receiving these wastewaters. This could partly be due to the fact that hospital wastewater contains as much as 100 times higher antibiotic levels than STPs (Baquero *et al.*, 2008; Kummerer 2004).

Interestingly, most of these strains had an identical or very similar antibiotic resistance pattern. Other studies have either identified bacterial strains in hospital wastewater (Ekhaïse and Omawoya, 2008; Nunez and Moreton, 2007), or separately in biosolids from an STP (Burtcher and Wuertz, 2003), but have not investigated the movement or survival of these strains to the extent done in our study.

Hospitals present an environment for a concentrated source of resistant bacteria, which may be released into the sewer system. It is therefore important that any study investigating the prevalence of antibiotic resistant bacteria in hospital wastewater consider factors that impact the level of antibiotic resistant bacteria in such wastewaters. For instance, it is possible that some of these antibiotic resistant bacteria are sourced from community effluents upstream of the hospital since the bulk of antibiotic treatment in the community would occur at home. In this study, we did not test any samples from hospital upstream effluents to rule out this possibility but Galvin *et al.* (2010) have shown that hospital wastewater had a higher proportion of antibiotic resistant *E. coli* than wastewater upstream of the hospital; and that the sewage treatment process was effective at removing the sensitive *E. coli* strains but failed to remove a number of antibiotic resistant strains (Galvin *et al.*, 2010). The original aim of sewage treatment was not for pathogen control, yet we now know that STPs do significantly reduce waterborne pathogen loads in the community.

In our study, we found a high reduction in the antibiotic resistant strains in hospital wastewater by the time they reached the STP. Interestingly, the few surviving strains that appeared to survive hospital discharge and sewage treatment were those that not only were MDR but also belonged to common types and were frequently found in the hospital wastewater at different weeks. We postulate that these common types may constitute the hospital persistent strains that have evolved to persist in the environment, including final chlorination after sewage treatment.

We did not compare the prevalence of antibiotic resistant bacteria found in our samples with those found in a treated hospital wastewater. Several studies have investigated the treatment of hospital wastewater by quantitatively comparing two different treatment methods (ie, conventional activated sludge system and plate membrane bioreactor) of hospital effluent treatment (Pauwels *et al.*, 2006). However, the results indicate that these treatment systems are not always effective against all types of bacteria present in the wastewater from hospitals. For instance, a Brazilian study investigated a hospital sewage treatment plant that used an extended aeration activated sludge process and showed that the treatment of the hospital wastewater may not be totally effective in removing antibiotic resistant bacteria and resistance genes from the hospital wastewater (Chagas *et al.*, 2011). This could partly be due to the ability of MDR bacterial to survive not only the hospital environment but also the municipal wastewater once they are released from hospitals. Such MDR strains, if they gain residency in hospitals, can be constantly released in hospital wastewater as shown in our study. Based on these findings, it can be postulated that on-site treatment of hospital wastewater would have little impact on the presence of MDR bacteria in municipal wastewater. This however, has yet to be fully investigated before any conclusion can be made.

A general conclusion and observation presented here is that we found the presence of MDR *E. coli* and *S. aureus* strains belonging to a few common types in untreated hospital wastewater. These strains were able to survive the journey to the inlet of the STP. However, only strains of certain types were able to survive treatment process of the STP, including chlorination, to be released into the surface waters. These strains in our study were shown to be frequently present in hospital wastewater suggesting their persistence in the hospital environment. The significance of this for public health however, requires further work to fully characterise and quantify the input of MDR bacteria from hospitals compared with those originating in the general community or other wastewater-related sources.

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Faecal Indicators and Zoonotic Pathogens in Household Drinking Water Taps Fed from Rainwater Tanks

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Summary

In this study, the microbiological quality of household tap water samples fed from rainwater tanks was assessed by monitoring the numbers of *Escherichia coli* (*E. coli*) and enterococci from 24 households in South East Queensland (SEQ), Australia. Quantitative PCR (qPCR) was also used for the quantitative detection of zoonotic pathogens in water samples from rainwater tanks and connected household taps. The numbers of zoonotic pathogens were also estimated in faecal samples from possums and various species of birds using qPCR as possums and birds are considered to be the potential sources of faecal contamination in roof-harvested rainwater (RHRW). Among the 24 households, 63% rainwater tank and 58% connected household tap water (CHTW) samples contained *E. coli* and exceeded Australian Drinking Water Guidelines of < 1 colony forming units (CFU) *E. coli* per 100 mL water. Similarly, 92% of rainwater tanks and 83% of CHTW samples also contained enterococci. In all, 21%, 4%, and 13% of rainwater tank samples contained *Campylobacter* spp., *Salmonella* spp., and *Giardia lamblia*, respectively. Similarly, 21% of the samples from rainwater tanks and 13% of the CHTW samples contained *Campylobacter* spp. and *G. lamblia*, respectively. The number of *E. coli* ($P = 0.78$), enterococci ($P = 0.64$), *Campylobacter* spp. ($P = 0.44$), and *G. lamblia* ($P = 0.50$), in rainwater tanks did not differ significantly from numbers observed in the CHTW samples. Among the 40 possum faecal samples tested, the *Campylobacter* spp., *Cryptosporidium parvum*, and *G. lamblia* were detected in 60%, 13%, and 30% of the samples, respectively. Among the 38 bird faecal samples tested, *Campylobacter* spp., *Salmonella* spp., *C. parvum*, and *G. lamblia* were detected in 24%, 11%, 5%, and 13% of the samples, respectively. Household tap waters fed from rainwater tanks tested in the study appear to be highly variable. Regular cleaning of the roof and gutter along with pruning of overhanging tree branches might also prove effective in reducing animal faecal contamination to the rainwater tanks.

Keywords

Roof-harvested rainwater, faecal indicators; zoonotic pathogens, qPCR, health risks.

Introduction

The microbiological quality of RHRW is generally assessed by monitoring faecal indicator bacteria (FIB) such as faecal coliforms, *Escherichia coli* (*E. coli*), and enterococci which are commonly found in the gut of warm-blooded animals including humans (Pinfold *et al.*, 1993; Sazakil *et al.*, 2007). In addition, a number of studies on the microbial quality of roof-harvested rainwater (RHRW) reported the presence of zoonotic bacterial and protozoa pathogens in individual or communal rainwater tanks (Ahmed *et al.*, 2008; Crabtree *et al.*, 1996; Simmons *et al.*, 2001). Most of these studies assessed the quality of the RHRW on the basis of the presence or absence of the specific pathogens, with little information available regarding their numbers or potential sources in RHRW.

Around 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), however, it is usually not recommended to use RHRW for drinking where town water is available. Queensland regulations do not prohibit the plumbing of rainwater tanks to supply drinking water. If a person, however, chooses to use rainwater for drinking or any other purpose, then that person is responsible for ensuring the quality of the water is fit for its intended use. Many householders who drink RHRW use under sink filtration (USF) system in order to reduce the exposure to pathogenic microorganisms, suspended solids and harmful chemicals.

Little is known regarding the prevalence of zoonotic pathogenic microorganism in wild animals such as birds and small mammals which are most likely contaminating RHRW. Mammals can get access to the roof via over hanging trees, electricity cables or climbing onto the roof via walls or other structures attached to the house. Birds can get access to the roof via overhanging trees or mounted structures on the roof such as TV aerials and solar panels. Knowing the source of pathogenic microorganisms is important in order to mitigate management strategies and to reduce public health risks from exposure to pathogenic microorganisms.

The aims of this study were: (i) to investigate the prevalence and numbers of faecal indicators (*E. coli* and enterococci), zoonotic bacteria (*Campylobacter* spp., and *Salmonella* spp.) and protozoa (*Cryptosporidium parvum* and *Giardia lamblia*) pathogens in water samples from rainwater tanks and connected household taps; and (ii) to investigate the prevalence of above mentioned pathogens in faecal samples from possums and various species of wild birds. Conventional culture based methods were used to enumerate *E. coli* and enterococci and quantitative PCR (qPCR) were used to obtain the numbers of zoonotic pathogens in RHRW, CHTW and animal faecal samples.

Results

Rainwater Tank Survey Results

The study area “Currumbin Ecovillage” is known for its sustainable residential developments and is often viewed as a blueprint for future urban developments. Twenty-four households participated in this study. All households use captured RHRW for drinking and other non-potable uses such as car washing, clothes laundering, showering, gardening etc. A sanitary inspection was undertaken to identify factors (ie, the presence of overhanging trees, TV aerials and wildlife faecal contamination on the roof) that may contribute to the faecal contamination of the rainwater tanks. Information on the filtration methods of RHRW prior to drinking was also obtained from the householders. The size of the selected rainwater tanks ranged from 7,200 to 30,000 L and aged between one to five years. Among the 24 households surveyed, six (25%) had overhanging trees ($n = 4$) or TV aerials ($n = 2$) mounted on the roof. Seven (29%) tanks had visible signs of faecal droppings on the roof. Twenty of the tanks (88%) had first a flush diverter installed. Among the 24 households, ten (42%) filtered the water before drinking. Of the ten households, nine (90%) had USF (ie, cartridge type filter, 0.5 μm pore size) and one had both USF and UV installed.

Numbers of Faecal Indicators in Roof-Harvested Rainwater and Connected Household Tap Water Samples

Two water samples were collected from each household (ie, one from the rainwater tank and one from the connected household tap) giving a total of 48 samples. Before sampling, the external taps and connected household cold water taps were wiped with 70% ethanol and allowed to run for 30 to 60 seconds to flush water. Samples were transported to the laboratory and processed within 2-4 hours. The membrane filtration method was used to process water samples for bacterial enumeration. Sample serial dilutions were made and filtered through 0.45 μm pore sized nitrocellulose membranes (Millipore, Tokyo, Japan). The membranes were placed on modified mTEC agar (Difco, Detroit, MI) and mEI agar (Difco) for the isolation of *E. coli* and enterococci, respectively. Among the 24 households, *E. coli* were cultured from 15 (62%) RHRW and 14 (58%) CHTW samples. Similarly, 22 (92%) RHRW and 20 (83%) CHTW samples contained cultured enterococci (Table 1).

Table 1. Numbers of faecal indicators in roof-harvested rainwater (RHRW) and connected household tap water (CHTW) samples.

Household ID	Numbers (mean) of Faecal Indicators per 100 mL of Water			
	<i>E. coli</i>		Enterococci	
	RHRW	CHTW	RHRW	CHTW
H1	1.5×10^1	2×10^1	2.1×10^1	1.3×10^1
H2	3×10^0	4×10^0	1.2×10^1	1.3×10^1
H3	1×10^0	4×10^0	9.1×10^1	1×10^2
H4	2×10^0	ND	3×10^0	6×10^0
H5	2×10^0	3×10^0	3×10^0	1×10^1
H6	2.3×10^2	6.7×10^1	2.7×10^1	ND
H7	1×10^0	2×10^0	3.7×10^1	8.2×10^1
H8	8.9×10^1	6×10^0	4.0×10^1	6.1×10^1
H9	ND	ND	4×10^0	1×10^0
H10	2×10^0	ND	1.7×10^1	2.5×10^1
H11	5×10^0	9×10^0	2.8×10^1	3.6×10^1
H12	1.2×10^1	6×10^0	ND	ND
H13	ND	ND	3×10^0	ND
H14	5×10^0	2×10^0	5.4×10^1	6.1×10^1
H15	1.2×10^1	3×10^2	7.5×10^1	1.1×10^2
H16	ND	ND	2.3×10^1	1.8×10^1
H17	ND	3×10^0	2×10^0	2×10^0
H18	ND	ND	4.9×10^1	4.1×10^1
H19	1×10^0	1×10^0	1.5×10^1	4×10^1
H20	ND	ND	ND	2.4×10^1
H23	ND	1.5×10^1	1.1×10^2	2.5×10^1
H25	ND	ND	5×10^0	3×10^0
H29	1×10^0	ND	2×10^0	ND
H35	ND	ND	4×10^0	3×10^0

ND: Not detected.

The numbers of *E. coli* in these samples ranged from 1×10^0 to 2.3×10^2 per 100 mL (for RHRW) and 1×10^0 to 3.0×10^2 colony forming units (CFU) per 100 mL (for CHTW) of water, respectively. For enterococci, these numbers were 2×10^0 to 1.1×10^2 CFU per 100 mL (for RHRW) and 1×10^0 to 1.1×10^2 (for CHTW) CFU per 100 mL, respectively. Enterococci were more frequently detected in both RHRW (22 of 24 samples contained enterococci) and CHTW (20 out of 24) than *E. coli* (15 out of 24; RHRW and 14 out of 24; CHTW). Among the 24 samples tested from RHRW tanks, 96% samples contained at least one faecal indicator and 58% samples contained both indicators. Similarly, among the 24 samples tested from the connected household taps, 92% samples contained at least one faecal indicator and 50% were positive for both indicators.

Numbers of Zoonotic Pathogens in Roof-Harvested Rainwater and Connected Household Tap Water Samples

Approximately 20 L of water from each rainwater tank and household tap was concentrated by hollow-fiber ultrafiltration system, using Hemoflow HF80S dialysis filters (Fresenius Medical Care, Lexington, MA, USA). The samples were concentrated to approximately 100 mL. Each 100 mL sample was further centrifuged at 3,000 *g* for 30 minutes at 4°C. The supernatant was discarded, and the pellet was resuspended in 5 mL of sterile distilled water. For qPCR analysis of pathogens, DNA was extracted from the pellet obtained from 1.5 mL of concentrated samples (ie, 48 samples) using a DNeasy Blood and Tissue Kit (Qiagen, Valencia, CA) and stored at -80°C until use. Standards for qPCR of the *Campylobacter* spp. 16S rRNA, *Salmonella invA*, *C. parvum* COWP and *G. lamblia* β -giardin genes were prepared from the genomic DNA. qPCR assays were performed according to previously published methods (Ahmed *et al.*, 2012).

Among the 24 households, five (21%), one (4%), and three (13%) RHRW samples contained *Campylobacter* spp. 16S rRNA, *Salmonella invA*, and *G. lamblia* β -giardin genes, respectively (Table 2). Similarly, five (21%) and three (13%) of the CHTW samples contained *Campylobacter* spp. 16S rRNA, and *G. lamblia* β -giardin genes, respectively. *Salmonella invA* gene could not be detected in CHTW samples. For the estimation of pathogen numbers, genomic copies (determined by qPCR) of each pathogen was converted to bacterial cells or protozoa cysts.

After conversion of genomic copies to number of cells, the number of *Campylobacter* spp. in RHRW and household tap water samples ranged from 5×10^0 to 1×10^2 (in RHRW) and 1×10^1 to 1.9×10^1 (in CHTW) cells per L of water. Similarly the estimated number of *Salmonella* spp. was 7.3×10^3 (in RHRW) cells per L of water. The numbers of *G. lamblia* cysts ranged from 1.2×10^2 to 5.8×10^2 (in RHRW) and 1.1×10^2 to 1.4×10^2 (in CHTW) per L of water.

Table 2. Numbers of zoonotic pathogens in roof-harvested rainwater and connected household tap water samples.

Household ID	Numbers (mean) of Bacterial Cells and Protozoa Cysts per L of Water					
	<i>Campylobacter</i> spp.		<i>Salmonella</i> spp.		<i>G. lamblia</i>	
	RHRW	CHTW	RHRW	CHTW	RHRW	CHTW
H1	ND	ND	ND	ND	1.2×10^2	1.4×10^2
H2	ND	ND	ND	ND	ND	ND
H3	ND	ND	ND	ND	ND	ND
H4	ND	ND	7.3×10^3	ND	ND	ND
H5	ND	ND	ND	ND	ND	ND
H6	1.1×10^2	+ ^a	ND	ND	ND	ND
H7	ND	ND	ND	ND	1.6×10^2	1.4×10^2
H8	ND	ND	ND	ND	ND	ND
H9	ND	ND	ND	ND	ND	ND
H10	4.7×10^1	ND	ND	ND	ND	ND
H11	ND	1.4×10^1	ND	ND	ND	ND
H12	+ ^a	1.1×10^1	ND	ND	ND	ND
H13	ND	ND	ND	ND	ND	ND
H14	5×10^0	1.2×10^1	ND	ND	5.8×10^2	1.1×10^2
H15	3×10^1	1.9×10^1	ND	ND	ND	ND
H16	ND	ND	ND	ND	ND	ND
H17	ND	ND	ND	ND	ND	ND
H18	ND	ND	ND	ND	ND	ND
H19	ND	ND	ND	ND	ND	ND
H20	ND	ND	ND	ND	ND	ND
H23	ND	ND	ND	ND	ND	ND
H25	ND	ND	ND	ND	ND	ND
H29	ND	ND	ND	ND	ND	ND
H35	ND	ND	ND	ND	ND	ND

ND: Not detected; a: Not quantifiable.

Correlation between Zoonotic Pathogens and Indicators in Roof-Harvested Rainwater and Connected Household Tap Water Samples

The number of faecal indicators and pathogens were pooled for all RHRW and CHTW samples to determine whether there was any correlation between the numbers of RHRW and CHTW samples. The number of *E. coli* ($P = 0.78$), enterococci ($P = 0.64$), *Campylobacter* spp. ($P = 0.44$), and *G. lamblia* ($P = 0.50$), in RHRW did not significantly differ from those numbers in CHTW samples as determined by Wilcoxon's signed-rank test. The numbers of *E. coli* and enterococci were analysed to determine whether the numbers within the RHRW and CHTW correlated with each other. Significant correlations were observed between *E. coli* and enterococci in water samples from RHRW ($r_p = 0.33$; $P = 0.005$) and CHTW ($r_p = 0.28$; $P = 0.01$) as determined by Pearson's multiple correlation.

Numbers of Zoonotic Pathogens in Animal Faecal Samples

Brush tail possum faecal samples ($n=40$) were obtained from a possum removal service in Brisbane (<http://www.possumman.com.au>). Bird faecal samples ($n=38$) were collected from the botanical gardens, a bird sanctuary and a veterinary hospital. The bird species include plover, wood duck, noisy minor, pacific black duck, blue faced honey eater, magpie, crow, ibis, seagull, topknot pigeon, crested tern, juvenile black swan, pacific baza, fantail cuckoo, rainbow lorikeet, and tawny frogmouth. Up to three samples were collected from each species of bird. All samples were transported to the laboratory, stored at 4°C and processed within 24 h. DNA was extracted from fresh faeces (ie, 80-220 mg) from each individual animal by using a QIAmp Stool DNA kit (Qiagen).

Among the 40 possum faecal samples tested, the *Campylobacter* spp. 16S rRNA, *C. parvum* COWP, and *G. lamblia* β -giardin genes were detected in 60%, 13% and 30% of the samples, respectively. After conversion of genomic copies to number of cells, the number of *Campylobacter* spp. in possum faecal samples ranged from 2×10^5 to 2×10^7 . *G. lamblia* was detected in 12 samples, however, only seven were quantifiable. The numbers of *G. lamblia* in possum faecal samples ranged from 2.1×10^1 to 1.6×10^3 cysts 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.

Among the 38 bird faecal samples tested, the *Campylobacter* spp. 16S rRNA, *Salmonella invA*, *C. parvum* COWP, and *G. lamblia* β -giardin genes were detected in 24%, 11%, 5%, and 13% of the samples, respectively. The numbers of *Campylobacter* spp. *Salmonella* spp. and *G. lamblia* organisms in bird faecal samples ranged from 6.6×10^4 to 6.6×10^6 , 6.3×10^2 to 1.8×10^3 , and 1.3×10^0 to 1.0×10^2 cysts per gm of faeces, respectively. *C. parvum* COWP gene was not quantifiable.

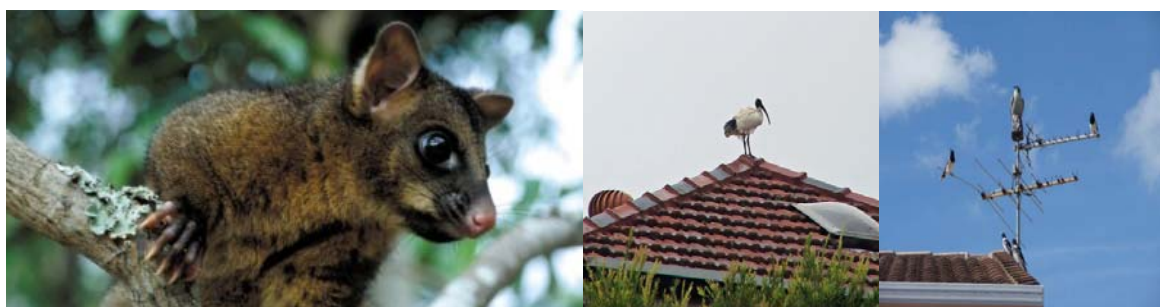


Figure 1. Dominant sources of faecal contamination in rainwater tanks.

Discussion

In this study, 62% of the RHRW and 58% of the CHTW samples fed from the RHRW tanks exceeded Australian Drinking Water Guidelines (ADWG, 2004) of < 1 CFU *E. coli* per 100 mL water. The pooled numbers of *E. coli* and enterococci in the CHTW samples did not differ significantly from the numbers found in the RHRW samples. It has to be noted that 58% of households in this study did not use any filtration methods, therefore, the presence of faecal indicators in the CHTW samples was not unexpected. Ten (42%) households had USF installed, however, these systems do not appear to be effective in removing faecal indicators. For example, households H3, H8, H11, H12, H15, H18 and H35 had USF, however, the numbers of faecal indicators in CHTW samples did not differ significantly from rainwater samples.

Five (21%) of the 24 RHRW tanks tested in this study contained *Campylobacter* spp. H6, H10, H12 and H15 were four of the five households where there were overhanging trees (H10 and H12) or evidence of wild life faecal droppings (H6 and H15) on the roofs. Two households (H12 and H15) had USF installed, however, *Campylobacter* spp. was detected in the CHTW samples suggesting the poor efficacy of USF systems. For *Campylobacter* spp., most human infections (ie, 95%) are caused by *C. jejuni* and *C. coli* (Butzler, 2004) and therefore, all *Campylobacter* spp.

PCR positive samples were further tested for the presence of *C. jejuni* and *C. coli*. Three RHRW tank and two CHTW samples contained *C. coli*. None of the RHRW tank and CHTW samples, however, contained *C. jejuni* (data not shown). *G. lamblia* was detected in three (13%) of the RHRW tanks tested in this study. H1 and H7 were two of the three households where there was evidence of wildlife faecal droppings. All three CHTW samples contained *G. lamblia*. It has to be noted that these households did not apply any filtration methods for rainwater purification prior to drinking. The high numbers of *G. lamblia* in both RHRW and CHTW samples from households H1, H7 and H14 may pose serious health risks to the consumers because of the low infectious dose of *Giardia*.

To obtain an insight on the magnitude of health risks, genomic copies of *G. lamblia* were converted to cysts numbers. The *G. lamblia* β -giardin gene is expressed as a single-copy gene within the nucleus of each trophozoite. 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 resulting in 16 copies of the β -giardin gene per *Giardia* cyst. The number of *G. lamblia* appeared to be one order of magnitude higher in rainwater samples in this study compared to our previous study (Ahmed *et al.*, 2010). It has to be noted that in the current study, 20 L water samples were tested whereas in the previous study, smaller sample volumes (ie, 2-2.5 L) were tested. The concentration of large volumes of water samples may have increased the detection sensitivity. *C. parvum* could not be detected in any of the samples tested, however, the presence of *Cryptosporidium* spp. in RHRW samples has been reported in US Virgin Islands and Denmark (Albrechtsen, 2002; Crabtree *et al.*, 1996). *Salmonella* spp. was detected in only one rainwater tank and none of the CHTW samples were positive for *Salmonella* spp.

Wild animals such as birds, mammals and reptiles are the most likely sources of faecal contamination in RHRW as these animals have access to the roof surface. In all, 60% possum and 24% bird faecal samples contained *Campylobacter* spp. All bird faecal samples contained *C. jejuni*. None of the possum faecal samples contained *C. jejuni* (data not shown). Possum and bird faecal samples also contained *G. lamblia* and the numbers of cysts ranged from 2.1×10^1 to 1.6×10^3 (for possums) and 1.3×10^0 to 1.2×10^2 (for birds) per gm of faeces. Previous research studies also reported the presence of *G. lamblia* in possum and bird faeces in North Island, New Zealand (Chilvers *et al.*, 1998). In this study, five possum and two bird faecal samples were also positive for *C. parvum*. The prevalence of *C. parvum* in possum and bird faecal samples was lower than *G. lamblia*. Chilvers *et al.*, (1998) reported similar findings and suggested that this could be because the duration of *Cryptosporidium* infection is much shorter than *Giardia* infection. 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 rainwater 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 rainwater tanks.

To date, several disease outbreaks and clinical cases associated with rainwater consumption have been reported (Ashbolt and Kirk, 2006; Brodribb *et al.*, 1995; Merritt *et al.*, 1999). In contrast, an epidemiological study of young children in South Australia reported that the consumption of RHRW did not increase the risk of gastroenteritis as opposed to mains water (Heyworth *et al.*, 2006). The results of the current study indicate that certain householders may have been exposed to potential pathogenic bacteria and protozoa, however, no increase in reported cases of illnesses is evident. 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.*, 2006). Before the disease can be reported in the Notifiable Diseases Surveillance System, 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 were viable and infective.

The faecal contamination of RHRW appears to be limited to systems that are not well maintained. It has been suggested that all RHRW systems should be appropriately maintained, including ensuring the cleanliness of the systems before rainfall events, especially roofs and gutters, which should be cleaned frequently, while the receiving tanks should be cleaned at least two times per year to improve the quality of water (Cunliffe, 1998). The roof should be kept clear of overhanging trees which may provide access to wild animals. Indeed, the high numbers of bacterial and protozoa pathogens in possum and bird faecal samples indicates the need for good maintenance of the roof and gutter and elimination of overhanging tree branches to minimise faecal contamination of RHRW.

It is evident that further information relating to the occurrence of pathogens throughout the year and the viability of pathogens in rainwater tanks is needed. In addition, more information is required on the survival of bacterial and protozoa pathogens in rainwater tanks. In a previous study, after estimating health risks associated with the rainwater use, it was suggested to disinfect rainwater before using it as potable water, especially for drinking (Ahmed *et al.*, 2010). The householders were asked to provide information on the types of filters installed in their USF systems and as well as information on the maintenance regimes. Certain householders did not follow the manufacturer's guidelines, and therefore, the presence of faecal indicators and pathogenic microorganisms in CHTW samples was not unexpected. The quality of the RHRW and CHTW can be improved by implementing effective point-of-use treatment procedures such as filtration followed by disinfection with UV treatment, ozone disinfection or ultra membrane filtration.

Conclusions

In conclusion, household tap waters fed from rainwater tanks in Currumbin Ecovillage 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 that RHRW may not be suitable for drinking. Although 42% householders filtered RHRW prior to drinking, the poor microbiological quality suggests the inefficacy of filtration methods when not used in accordance with manufacturer's guidelines. In view of this, it is recommended that RHRW should be disinfected using effective treatment procedures prior drinking.

The high prevalence of bacterial and protozoa pathogens in possum and bird faecal samples indicates these animal species are most likely the sources of faecal contamination in rainwater tanks. Therefore, maintenance of good roof and gutter hygiene and elimination of overhanging tree branches and other structures where possible to prevent the flocking of possums and birds should be considered.

Acknowledgements

This research was undertaken and funded as part of the 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.

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Climate and Water in South East Queensland: Past and Future

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Summary

The decade long “Millennium Drought” in South East Queensland (SEQ) was broken in 2009 with above average rainfall in 2010 and devastating floods and extreme rainfall in January 2011, raising the questions as to whether climate change plays a role and what the future hydroclimatic conditions will be like over the region. The *Climate and Water* project within the Urban Water Security Research Alliance has been focusing on these issues, including: (i) what causes drought and floods in the SEQ region; (ii) how the properties of drought (intensity, duration and frequency) may change in a warming climate, in addition to forcing by multidecadal variability; and (iii) how best to translate global projections of climate change (rainfall and other hydrological fields) into future water availability information. Here we provide a brief synthesis, showing that in terms of rainfall, the fluctuations that we experienced can be accounted for by multidecadal variability in the Pacific Ocean and its modulation on the regional rainfall teleconnection with El Niño-Southern Oscillation. Further, we show that while the direction of future rainfall change remains uncertain, the impact of rising temperature will increase the risk of longer-lasting and deeper droughts in the future, with significant implication for inflows into dams and water resource security. This is manifested through an increase in the proportion of SEQ experiencing exceptionally hot conditions, and a similar increase in the incidence of potential evaporation extremes and in the proportional area experiencing severe water balance deficit. The implications for extreme floods are yet to be examined.

Keywords

Droughts, floods, interdecadal climate variability, future climate.

Introduction

Since the 1980s, multidecadal-mean rainfall over South East Queensland (SEQ) has been far smaller than that in the previous decades. The impact of the rainfall decline, particularly since 1998, culminated in 2007 when the combined water storage level in the drinking water dams supplying SEQ dropped below 20%, or less than one year of water supply. Is this an indication of what is to come in a warming climate? Why was the impact of the recent drought so severe? Is there any evidence supporting an impact from climate change? The regional summer rainfall is affected by the El Niño-Southern Oscillation (ENSO) cycle, but the ENSO teleconnection is modulated by the Pacific Decadal Oscillation (PDO), which is alternatively referred as the Interdecadal Pacific Oscillation (IPO) (Cai *et al.*, 2010). Does the evolution of the PDO-IPO contribute to the drought?

We explore the potential role of climate change on the recent drought using climate model experiments under observed climate change forcing factors. These climate models are useful in terms of gauging the impact of climate change. Each model has its own variability, which is independent from one model to another, so that when averaged across many models, it is reasonable to assume the variability components will cancel out, leaving largely the climate change component. We show that climate change may not play a significant role in driving the latest rainfall deficiency. We then explore natural variability processes whereby the PDO-IPO influences rainfall variability over the region. When the PDO-IPO is in a positive phase, there is a breakdown of the impact that La Niña has on SEQ rainfall.

Is the regional rainfall linkage with the PDO-IPO able to explain the devastating SEQ flood and the associated extreme rainfall in January 2011? And what does the flood tell us about the status of the PDO-IPO? Although a PDO-IPO transition to a negative phase was suggested to take place in the early 2000s (McPhaden *et al.*, 2004), the exact PDO-IPO status was unclear at the time of the flood. Using several lines of supporting evidence, we propose that the SEQ 2011 austral summer rain constitutes a confirmation of a transition to a negative phase of the PDO-IPO, which will have implications for rainfall in the next decades (Cai and van Rensch 2012).

Although the rainfall changes may not be linked to global warming, a rising evaporation rate due to global warming can change occurrences and areal extent of droughts. Using outputs from 12 climate downscaling experiments, we show that even without changes in rainfall, the impact of increasing evaporation going into the future could be substantial. Because the *Climate and Water* project primarily focuses on droughts and the associated impact on water resources, our analysis examines whether the properties of droughts will change in a warming world rather than on potential future extreme rainfall events. A component of the *Climate and Water* project also uses the outputs from the climate models to force the Brisbane Integrated Water Quantity and Quality Model (IQQM) and examine impacts on water storage, yielding highly policy-relevant results, which will be discussed in a separate paper.

Results

A Lack of Consensus from Climate Models on SEQ Rainfall Trends

We first address whether the observed rainfall trend is attributable to climate change. We examine the 20th century experiments submitted to the IPCC AR4, comprising a total of 71 simulations made available through the Third Climate Model Intercomparison Project (CMIP3). Model names and details are listed in Table 1 of Cai and Cowan (2007) with references to further documentation. A total of 48 experiments include stratospheric ozone forcing components, while the rest do not, allowing stratification into groups with and without ozone depletion to separate the impact of depleting stratospheric ozone. Ozone forcing may not be the only difference between these two groups and all models include an anthropogenic aerosol forcing in one form or another. We have also analysed a series of targeted multi-member ensemble experiments. We used the CSIRO Mk3A model (Rostayn *et al.*, 2008) for this analysis, as it was designed to isolate the impacts of individual forcing factors. The impact of ozone depletion (four ensemble members) and greenhouse gas-only (four ensemble members) in these targeted experiments is realised by forcing the model with a time-evolving forcing alone. However, the impact of anthropogenic aerosol forcing is obtained by comparing two sets of experiments (eight ensemble members each) with and without increasing aerosols, both in the presence of all other forcing factors. For all experiments, the ensemble-mean trend over the 1950-2000 period is calculated and the uncertainty range of the trend is estimated from the standard error.

Several important results emerge from these model results (Figure 1). Firstly, only four models (red crosses, GFDL-CM2.1, MIROC3.2(hires), INM-CM3.0, IPSL-CM4) produce a statistically significant rainfall declining trend over SEQ comparable to the observed. Secondly, the observed summer rainfall reduction is not generated in an all-model ensemble mean (orange cross), which, as an entity, contains all climate change forcing factors, such as greenhouse gases, aerosols, and stratospheric ozone depletion. Thirdly, results from targeted experiments forced by ozone depletion only (green circle), or a comparison between the CMIP3 model groups with and without ozone depletion (orange circles), show that ozone depletion has little impact. Finally, increasing aerosols, if anything, tend to increase SEQ summer rainfall, although the increase is very small (green crosses). In summary, the observed rainfall decrease is not produced by the majority of models, nor is it attributable to any individual climate change forcing factor.

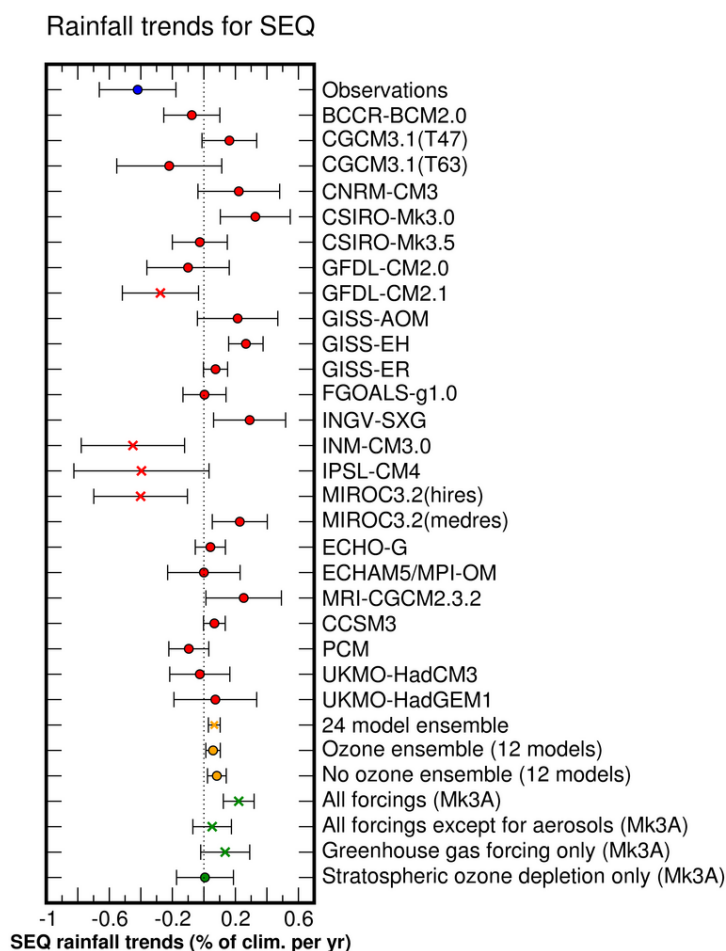


Figure 1. Rainfall trends in terms of percentage of climatology in the observed (1950-2008, blue), and in various climate models (1951-2000) with different forcing scenarios. Only four models produce a trend that is statistically significant and in the same direction as in the observed.

Impact of ENSO and PDO-IPO on Droughts and Floods

PDO-IPO and the Recent Drought

We then show that there is an asymmetry in the regional rainfall teleconnection with ENSO. This is depicted in Figure 2, showing that ENSO is a rainfall-generating mechanism for the region for the pre-1979 period. The La Niña-rainfall relationship is statistically significant; rainfall increases with La Niña amplitude. By contrast, the El Niño-induced rainfall reductions do not have a statistically significant relationship with El Niño amplitude. Since 1980, this asymmetry no longer operates, and La Niña events no longer induced a rainfall increase, leading to the observed SEQ rainfall reduction.

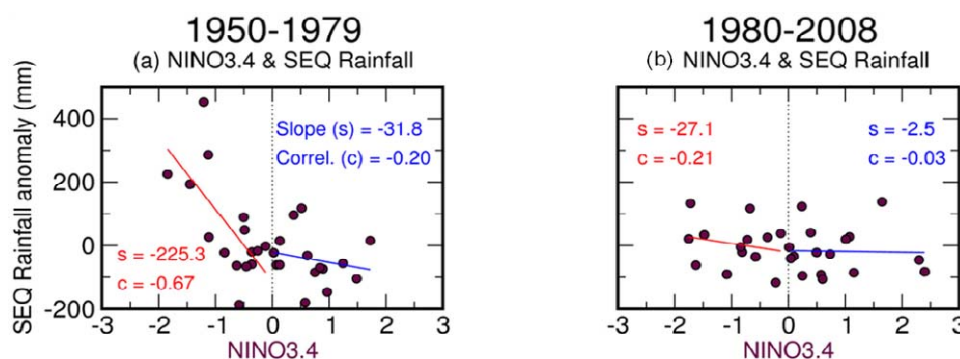


Figure 2. Scatterplot of detrended SEQ DJF rainfall anomalies vs. detrended Niño3.4, with linear fits using samples with positive and negative Niño3.4 values.

This breakdown is caused by an eastward shift in the Walker circulation and the convection centre near Australia's east coast, in association with a post-1980 positive phase of the Interdecadal Pacific Oscillation (IPO). Because the strong relationship of the region's rainfall with La Niña during the pre-1980 period, there is an overall statistically significant correlation between the region's rainfall and ENSO, which is *not* present in the post-1980 period. The evolution of the time-varying correlations is seen between rainfall and the Southern Oscillation Index using a 13-year sliding window (Figure 3a). Such a breakdown in the ENSO-rainfall teleconnection occurred before 1950, confirming the importance of an influence from multi-decadal variability. An aggregation of outputs from climate models discussed above to distil the impact of climate change suggests that the asymmetry and the breakdown may not be generated by climate change.

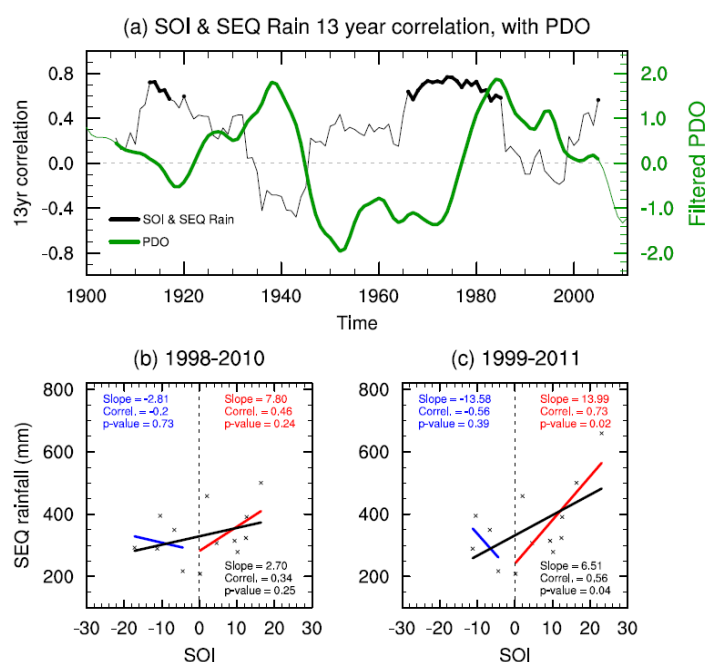


Figure 3. (a) Time series of the SEQ rainfall-ENSO teleconnection calculated as correlations (black curve), using a 13-yr sliding window of SEQ DJF rainfall with the SOI over the period 1900-2011 (statistically significant correlation at the 95% confidence level is 0.55, indicated by a solid grey line). A PDO index is overlaid in green (Mantua *et al.*, 1997) (an uncertain status is shown as a thinner curve). The correlation between the rainfall-ENSO teleconnection and the PDO is -0.41. (b) and (c) Scatter plot for the 13 years ending in 2010 and 2011 respectively, with slopes, correlations and p-values of the linear fits conducted separately using samples corresponding to all values (black line), positive (red line, La Niña) and negative (blue line, El Niño) SOI values.

PDO-IPO and the Recent Flood

The 13-year correlation ending 2011 is statistically significant at the 95% confidence level (Figures 3a and 3c), but the correlation ending 2010 is not (Figure 3b). Thus, the 2011 SOI-rainfall pair is such an extreme outlier that it changes the statistical properties of the other 12 common samples (years 1999-2010). The 1999-2011 ENSO-SEQ rainfall correlation (Figure 3c) has a strong asymmetry, as in Figure 2a. The PDO-IPO (green) shows a sign of transitioning to a negative phase. There have been only two such transitions since 1900, each is accompanied by either a concurrent, or a delayed, strengthening of the ENSO-SEQ rainfall teleconnection toward the statistical significance at the 95% confidence level, as in the 1913-1922, and the 1965-1977 periods, respectively. There is no precedence in which such an occurrence is not accompanied by a PDO-IPO transitioning to, or already in, a negative phase. This is a line of evidence supporting that the PDO-IPO has entered into a negative phase.

Figure 4 provides a historical perspective in the context of the well-known SOI-SEQ summer rainfall relationship. The summer 2011 year stands out with an unprecedented SOI value, contributing to the asymmetry of the relationship. The well-known ENSO-rainfall teleconnection for the region relies on events similar to that in DJF 2010/2011. Figure 4 also identifies summers since 1900 when both the rainfall anomalies and SOI values were greater than a one-standard deviation value (large circles). These include the 1956, 1971, 1974, 1976, 2008, and 2011 wet summers. Excluding these samples (linear fit black line), the La Niña-rainfall teleconnection over SEQ disappears (with a statistically insignificant correlation of 0.03), highlighting the importance of these “pinnacle” extreme rainfall-SOI pairs and their high nonlinearity in contributing to the well-known ENSO-SEQ summer rainfall relationship. The wet summers before 1980 were associated with a negative PDO-IPO. The PDO-IPO phase for 2008 and 2011 is uncertain. However, from the 111-year record, there are only three SEQ summers (including that of 2011) when the regional rainfall anomaly surpasses 300 mm. There is no precedence of this occurring during a positive PDO-IPO phase. This also constitutes a line of evidence supporting a switch in the PDO-IPO to a negative phase.

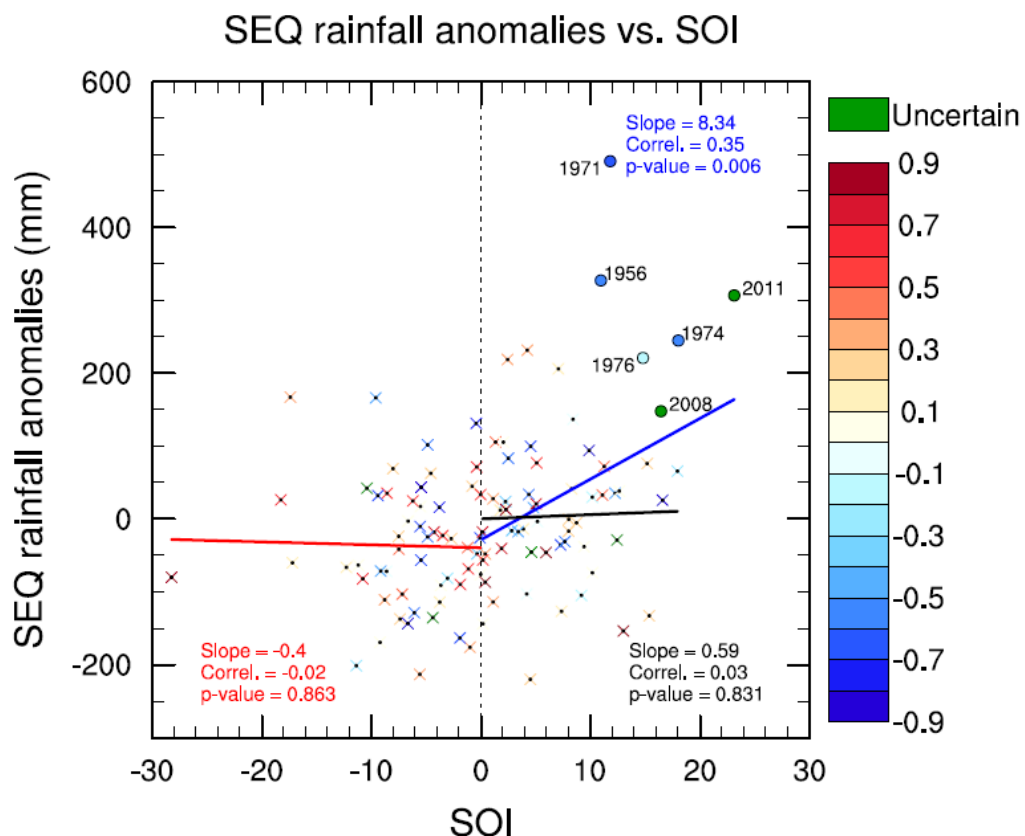


Figure 4. Scatter-plot of SEQ DJF rainfall anomalies and SOI values over the period 1900-2011, with slope, Pearson's correlation coefficients (square value to obtain r^2 for goodness of fit), and p-values for statistical significance of the linear fits, conducted separately using samples corresponding to positive (blue line, La Niña) and negative (red line, El Niño) SOI values. Samples marked with circles indicate summers with both the SOI and the positive rainfall anomalies greater than one standard deviation. When these “extreme” summers are excluded, the linear fit (black line) is not statistically significant. The colour bar indicates the phase of the PDO (representing PDO-IPO, green curve of Figure 3a). The phase of 2008 and 2011 is marked as uncertain “green,” as it requires data up to 2014 and 2017 respectively, to know the actual value.

Future Climate Extremes

Because the *Climate and Water* project primarily focuses on droughts and the associated impact, our analysis examines whether the properties of droughts will change in a warming world. Twelve dynamical downscaling simulations have been performed using the CSIRO Conformal-cubic Atmospheric Model (CCAM) at a resolution of 20km. Each simulation was forced by the sea surface temperature (SST) fields produced by one of the General Circulation Models (GCM) submitted to the World Climate Research Programme's CMIP3 multi-model dataset. Most of these models are forced by the A2 emission scenario. In each case, the SST fields were bias corrected before they were used to force CCAM, see McGregor and Nguyen (2009) for more on this process. The downscaling process saves output for a comprehensive range of variables on a sub-daily time-scale and at a resolution of around 20km over much of eastern Australia. Rainfall, temperature and pan evaporation fields for a region bounded by 20–29°S and 145–155°E are extracted from this dataset and analysed.

The twelve downscaling experiments project a hotter, more drought-prone SEQ. These changes appear to be driven primarily by a local warming and associated increase in evaporative potential and not by any clear signal in rainfall. Using climatological rainfall in combination with an increase in evaporative potential produced similar results. A method similar to that of Hennessy *et al.* (2008) is used to assess changes in simulated incidence of exceptional events. Using this method, any value that falls outside the 5th percentile of the reference period (1971–2000) is considered to be exceptional. For example, a value lower than the 5th percentile rainfall for a location is an exceptionally dry value for that location and a value greater than the 95th percentile is considered exceptionally wet.

Figure 5 shows 30-year running mean values of the percent area experiencing exceptional events for each of rainfall, temperature, potential evaporation and the Palmer Drought Severity Index (PDSI, Palmer (1965) modified to use simulated evaporation potential as an input), shows all hotter and drier extremes.

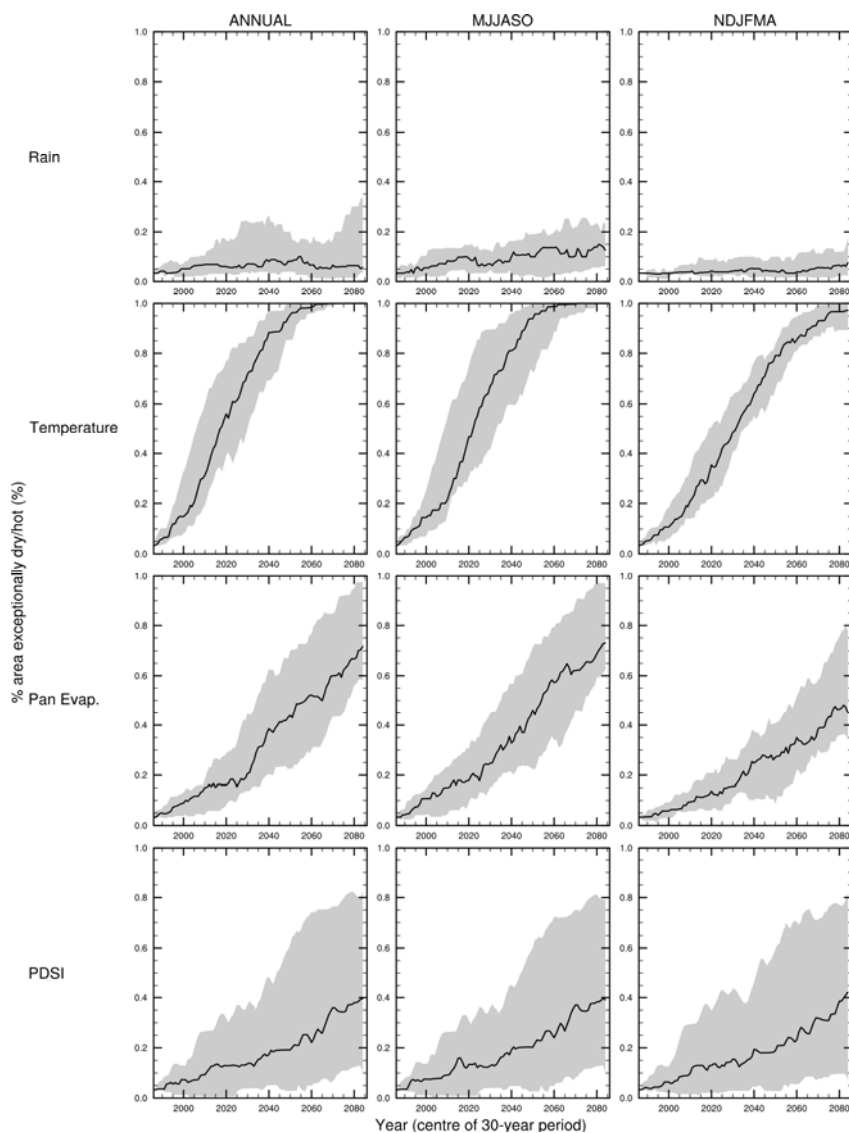


Figure 5. 30-year running mean time series of percent area of SEQ experiencing exceptionally dry (top row, in terms of rainfall), hot (second row), high evaporation potential (third row) and severe drought (bottom row) years (left column), dry half-years (centre column) and wet half-years (right column).

The proportion of SEQ experiencing exceptionally low rainfall (top row), high temperatures (second row), high pan evaporation (third row) and severely negative PDSI (bottom row) is shown in Figure 5. The shaded area represents the 10th to 90th percentile range of the 12 downscaling simulations with the bold line showing the median. The left column shows the full year, the middle column the dry-half year (May - October) and the right shows the wet half-year (November - April). The simulations exhibit strong increases in the proportion of SEQ experiencing exceptionally hot conditions, with more rapid changes over the dry half-year than the wet. There is a similar increase in the incidence of potential evaporation extremes and in the proportional area experiencing severe water balance deficit. However, there is little to no trend in the proportional area simulated to experience exceptionally dry conditions, particularly over the wet-season.

At the other end of the spectrum, the proportion of SEQ experiencing exceptional rainfall events shows little change over the twenty-first century, suggesting that the frequency of periods of successive wet years (and half-years) are unlikely to change. All of the simulations show an absence of cold periods and suggest that by 2050 the occurrence of periods of low evaporation will reduce (Figures not shown).

Conclusions

The devastating SEQ extreme rainfall in January 2011 and the preceding decade-long drought over SEQ were climate extremes that can be generated in the model without climate change. Indeed, climate models forced by observed climate change forcing factors suggest that the drought could occur without climate change. We show that rainfall fluctuations that we experienced can be accounted for by the PDO-IPO. Further, we show that while the direction of future rainfall change remains uncertain, the impact of rising temperature will increase the risk of longer-lasting and deeper droughts going into future, with significant implication for water resources. This is manifested through an increase in the proportion of SEQ experiencing exceptionally hot conditions, and a similar increase in the incidence of potential evaporation extremes. This will result an increase in the proportional area experiencing severe water balance deficit and dry catchment conditions. This is projected to decrease the amount of runoff from rainfall and the consequent inflows to the water storage dams, impacting on future water security in SEQ.

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A Research Agenda for Water Smart Tropical and Sub-Tropical Cities and Towns

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Summary

This paper describes the process and outcomes relating to the development of a research agenda for Water Smart Tropical and Sub-tropical Cities. The aim of the work is to advance the current understanding of the role and importance of water in achieving sustainable urban development. The area of “Water Smart Cities” was identified as a strategic research priority by the Urban Water Security Research Alliance in 2010. A research project was accordingly initiated with the purpose of delivering a research program as a “legacy”, for subsequent implementation. A dialogue-driven approach was used to design the research agenda to achieve a convergence between the requirements of key policy and decision-making stakeholders in the region, and identified opportunities for advancing the science from a literature review. An iterative process was employed with option development and refinement occurring through increasingly focused stakeholder feedback supported by an increasingly targeted review of scientific and industry and practice literature.

There are a number of related initiatives and approaches currently under way, including Water Sensitive Cities, Cities of the Future and Integrated Urban Water Management. While these efforts cover significant ground, what is clear from the literature, and from stakeholders, is that gaps remain. For example, there are further research opportunities relating to a guiding conceptual framework, the application in tropical and sub-tropical urban contexts, a set of integrated water indicators, and evidence for participative planning and implementation. A scan of the Queensland policy and planning environment identified the regional planning and the associated state of the region reporting and monitoring, as the most appropriate planning instrument to consider influencing. The paper concludes with the suggestion of a Planning Support System for Water Smart Tropical and Sub-Tropical Cities and Towns, as a potential high-impact delivery platform for the research outputs.

Keywords

Water Smart Cities, sustainable water management, conceptual framework, sustainability assessment, integrate water indicators, urban metabolism.

Purpose

The issue of Water Smart Cities was identified by stakeholders of the Urban Water Security Research Alliance as a strategic research priority late in 2010. Further direction from the stakeholders at that stage was that the proposed research should be more applied, situated at the science-policy interface, intentionally directed at particular Queensland-specific policy products for more direct impact. In addition, the stakeholders set the challenge to differentiate the research geographically by requiring that it be appropriate for the range of tropical and sub-tropical biophysical and socio-economic contexts of Queensland. As a consequence, a project was initiated with the intention of delivering a research program; a “legacy”, for implementation at a later stage. This paper describes the process and outcomes relating to the development of a research agenda to advance the current understanding of the role and importance of water in achieving sustainable urban development, specifically in the context of tropical and sub-tropical climates and at the science-policy interface.

A dialogue-driven approach was used to design the research agenda to achieve a convergence between the requirements of key policy and decision-making stakeholders in the region, and identified opportunities for advancing the science from a literature review. An iterative process was employed with option development and refinement occurring through increasingly focused stakeholder feedback supported by an increasingly targeted review of scientific and industry and practice literature.

Results

Stakeholder Needs Assessment

During 2010, a series of workshops with key stakeholders identified the development of a conceptual framework as a research priority. Such a framework is necessary to guide a range of policy and planning informing instruments to better integrate and mainstream water thinking into urban planning processes. A conceptual framework should:

- clearly define urban development objectives, based on an integrated socio-economic and ecological system approach, incorporating the concepts of resilience and risk;
- be framed in the broader context of planning for Cities of the Future with the perspective being the role of water and its interaction with other agents and flows within the urban development process;
- be multi-scale with a clear definition of system boundaries for and linkages between scales;
- differentiate between cities and towns with a range of local biophysical and socio-economic contexts;
- differentiate between qualities of water for different uses and for achieving different outcomes (fit-for-purpose);
- guide the formulation of associated outcomes-oriented metrics for applying principles in practice and assessing the sustainability of options and for monitoring progress on a comparable basis; and
- focus on the factors with the biggest impact and, in the shorter term, on those for which information is available to enable measurement.

In 2012, a workshop was convened to specifically explore needs and opportunities for advancing to more water smart cities and towns in tropical and sub-tropical Queensland. Areas for change were prioritised, in decreasing order of importance, as:

- people - bringing the community along so as to bridge the apparent disconnect between vision and reality, bringing about the practical application of readily available knowledge already present in existing guidelines and policy;
- governance changes to more adaptable and flexible political, planning control and legislative systems;
- integration of scales and sectors (planning), more specifically the linkages between energy-water-carbon, transport, water and energy infrastructure and cities within their regional context;
- adaptive/flexible infrastructure planning and delivery, including natural and built environment assets and focused on issues of centralised vs decentralised infrastructure, different contexts, full cost recovery; and
- processes of planning and decision-making including improved levels of state and local planning integration, reducing inconsistencies between departments and greater levels of community, city planners and designers work together to design urban areas.

Most subsequent actions identified by the participants, related to the first priority of better engaging “People” and communities in translating the vision into practice and are mostly to do with better information for participatory, evidence-based decision-making and for motivating behavioural change. The set of easier to achieve, but effective actions identified, relate primarily to high level community engagement, education, advocacy and awareness-raising to effect behavioural change. Effective actions, identified as more difficult to achieve, relate to the provision of evidence-based, targeted information to create a deeper level of awareness and understanding of issues to not only effect a change in behaviour, but to enable community participation in infrastructure planning and visioning ie, participatory decision-making where information such as costs and benefits are made clear and transparent so that informed decision-making can take place on the basis of a shared understanding of the implications of decisions.

Literature and Industry Practice and Policy Review

A number of major initiatives have recently been launched in Australia and internationally to help progress the management of aspects of urban water sustainability. Launched by the International Water Association, the Cities of the Future program focuses on water security and the design of cities to minimise the use of scarce natural resources and increase the coverage of water and sanitation in lower and middle income countries (<http://www.iwahq.org/3p>, accessed 17/5/2012).

In Australia, the recently-announced Cooperative Research Centre for Water Sensitive Cities (CRCWSC) will address four key themes (Society, Future Technologies, Water Sensitive Urbanism, and Adoption Pathways) identified as critical to encourage community acceptance of, and participation in, the development of water sensitive cities (CRCWSC Bid Document). The CRC is hosted in the Water Sensitive Cities Centre at Monash University, Melbourne. The current activities of the Centre relate to enabling the wider uptake of WSUD with a strong focus on the sustainable integration of stormwater into the planning of cities, and the development and use of cities as water supply catchments (<http://www.watersensitivecities.org.au/about-us/cwsc/background-and-aims>, accessed 17/5/2012). In striving towards more sustainable development, the water industry and research community have produced a range of frameworks, designs, demonstrations, case studies, principles and guidelines using the concepts of Integrated Urban Water Management (IUWM) (Mitchell, 2006; Maheepala *et al.*, 2010) and Water Sensitive Urban Design (WSUD) (Wong and Brown, 2009). In practice, WSUD has been implemented at a range of building, precinct and neighbourhood scales. More recently, the importance of integrating water sensitive design within the wider urban planning context at not only the local development scale but at the scale of cities has been recognized (Binney *et al.*, 2010; Burn *et al.*, 2011).

A scan of the Queensland policy and planning environment has identified the regional plan and the associated state of the region reporting and monitoring, as the most appropriate planning instrument to consider influencing. Under the Sustainable Planning Act (SPA) (2009), Regional Plans are a statutory state planning instrument, requiring for designated regions, to provide: an integrated planning policy for a designated region through the identification of desired regional outcomes and the policies and actions for achieving them; the desired future spatial structure of the region including land use and infrastructure; and key regional environmental, economic and cultural resources to be preserved, maintained or developed (Queensland Government, 2009a). Regional Plans are currently in various stages of development throughout Queensland and are all quite different in terms of content detail and also associated regulatory controls. This has implications for the degree to which innovative water smart solutions can be tried and tested in practice and an opportunity exists to investigate the role of regional planning statutory strictures in limiting or promoting the implementation of water smart solutions across Queensland.

The stated purpose of the most recent SEQ Regional Plan 2009-2031 is to manage regional growth and change in the most sustainable way to protect and enhance quality of life in the region (Queensland Government, 2009b). As part of the SEQ State of the Region Report prepared every five years, sustainability indicators based on the desired regional outcomes are captured to monitor changes in economic, environmental and social factors and to report on the progress of the region towards sustainability (Queensland Government, 2008). The sections relating to the water and integrated sustainability desired regional outcomes provide the most relevant opportunity for achieving impact for the intended research.

The initial review of scientific and industry and practice literature, concluded that significant opportunities for further research to enhance urban sustainability from a water perspective, lie in the area of resource efficiency, in particular through the integration of: energy, water and nutrient flows; resource efficiency and the human (socio-economic) dimensions of urban development; and resource efficiency, resilience and risk (Priestley, 2011). A comparative assessment of a number of alternate approaches to understanding resource efficiency in urban contexts has indicated that the urban metabolism approach, first articulated by Wolman (1965), is a solid basis from which to begin (Priestley *et al.*, 2011). This is because it has been demonstrated to, amongst others:

- build on a solid foundation of material and energy balances which underpin the resource flows throughout the city rather than simply “accounting” for them as inflows and outflows;
- incorporate social and physical sciences (eg, bridging some of the typically divided science elements);
- quantify the performance of cities (eg, the hydrological performance);
- drive accuracy and structure into water (and other urban) accounting systems;
- allow for the captures of flows beyond the urban boundary;
- allow for capture of virtual water flows and nutrients;
- be extendable, with a number of attempts having been made, in theory and practice, to extend the urban metabolism to account for broader human and ecological processes;
- be divisible, possible to apply only selected parts of the urban metabolism model to understand particular aspects. Techniques such as mass and energy flow analysis are a simplified subset of urban metabolism and provide a basic example of what the approach can offer; and
- can be applied using pragmatic methods to acquire data including downscaling techniques, in recognition of the complexity and data-hungriness in applying the urban metabolism concept.

Recommendations for the application of an urban metabolism approach based on the initial literature review included: developing simplified urban metabolism models to focus on “big gains” factors and drivers; investigating the integration of urban metabolism models dealing primarily with resource use efficiency with other tools used for the assessment of system resilience, risk and cost; and developing and evaluating approaches to the clear communication of the outputs of urban metabolism models to urban planners, politicians and the general public in relation to resource efficiency indicators and their changes over time (Priestley *et al.*, 2011).

A subsequent, more targeted literature review, has revealed a number of alternative approaches and theoretical advances to urban metabolism (Newman, 1999; Alberti, 2003; Alberti, 2005; PMSEIC, 2010; Resilience Alliance, 2007 and 2010; Minx *et al.*, 2011). These may provide a useful input to the formulation of an appropriate conceptual framework to guide the water smart cities and towns research agenda in SEQ. It is clear from this literature that the conceptual framework needs to view urban areas as integrated, complex, adaptive, social-ecological systems with emergent properties. Until recently, the human and ecological processes occurring in urban areas have generally been studied as separate phenomena. This approach precludes gaining an understanding of how human-dominated ecosystems emerge from interactions between socio-economic and ecological processes, both locally and over larger spatial scales.

An extended metabolism model has been proposed and applied in assessing the sustainability of a number of cities by Newman (1999). The inclusion of livability factors, in addition to resource efficiency factors, is advocated to enable the identification of how a simultaneous increase in livability benefits alongside a reduction of resource inputs and waste outputs, as the goal of sustainability in cities, might be achieved (Newman, 1999). The extended

metabolism model has been used as a basis for deriving a set of principles to guide IUWM (Burn *et al.*, 2011). The Resilience Alliance (2007; 2010) has developed a resilience assessment framework for assessing the resilience in social-ecological systems based on the four interconnected themes of metabolic flows, social dynamics, governance networks, and the built environment.

In an attempt to more fully capture the interactions and feedbacks in human-dominated ecosystems, Alberti *et al.* (2003), have elegantly postulated and applied (Alberti, 2005) an integrated model of human and ecological processes to understand the forces driving urban development patterns, resulting patterns of natural and developed land, impacts on social and biophysical processes, resulting environmental and social changes and effects and feedback on human and biophysical drivers. The Prime Minister's Science, Engineering and Innovation Council (PMSEIC, 2010) has recognised that energy, water, carbon and other natural cycles, such as nutrient cycles, are significant interconnectors between the natural environment and human society. Accordingly, an integrative, resilience-based approach has been advocated as an appropriate framework for addressing the combined and intersecting energy, water and carbon challenges of Australian cities and towns.

In a recent report to the European Environmental Agency, Minx *et al.* (2011) present an extended concept of urban metabolism as part of developing a pragmatic approach to assessing urban metabolism in Europe. Three major extensions are recommended in going beyond a purely metabolic assessment: 1) linking the urban metabolism to environmental pressures and aspects of environmental quality at multiple spatial scales from local to global (with implications for concepts of carrying capacity and resilience); 2) linking urban metabolism to urban drivers, patterns (physical structure of the city) and lifestyles to enable an assessment of causes; and 3) linking urban metabolism to aspects of quality of life to understand consequences. Important contributions relating to the application of the model and the derivation of indicators in accordance with the model have been made: 1) the necessity of being pragmatic in relation to data availability; 2) the importance of analysis at a finer grained spatial scale; 3) ways in which downscaling techniques can be used to disaggregate higher level input-output stocks and flow data to smaller scale geographic units; and 4) the necessity of incorporating links and relationships from and to the specific geographic area of interest and between different scales so as to be able to explore questions related to the localisation and globalisation of resource flows (Alberti, 2003; Minx *et al.*, 2011).

Emerging Research Agenda

Themes

Responding to stakeholder inputs and the literature review, emerging priority research themes are:

- guiding, conceptual framework based broadly on the paradigm of cities as complex adaptive socio-ecological systems and incorporating extended urban metabolism approaches with the water “thread” and intersections of water with other elements and flows, being clearly identified and mapped;
- understanding the inherent characteristics of tropical and sub-tropical Queensland biophysical and socio-economic contexts and the implications for sustainable city policy and practice;
- a set of scale-related integrated water indicators (to amongst others, inform State of the Region reporting);
- spatial methods and information of areas of water smart need, potential and suitability to inform amongst others, the land use and infrastructure part of the regional plans;
- case study-based comparative assessment of Regional Plans and their associated regulatory controls to understand the influence of regulatory and governance impacts on regional innovation – eg, Townsville vs SEQ; and
- an underpinning information portal and pragmatic database to provide comparative and updated information to enable and support addressing all the other above-mentioned themes (Figure 1).

Impact

In accordance with the directive to situate the research agenda at the science-policy interface and in response to the strong stakeholder requirement for evidence-based, targeted, clear and transparent information to enable participatory decision-making, it is envisaged that the proposed research agenda could be designed around the delivery of a Planning Support Systems (PSS) product. Based on the well-established concept and knowledge base relating to the development and practical implementation of PSS are geo-information-based instruments, providing an integrated environment of multiple technologies with a common interface, incorporating a suite of theories, data, information, knowledge, methods and tools to support the planning tasks of problem diagnosis, data collection, spatial and trend analysis, geodata modeling, spatial scenario building, visualisation and display, plan formulation, prediction and forecasting, plan evaluation, monitoring, enhanced participation and collaborative decision-making (Geertman, 2006). PSS function as “information frameworks” that integrate the full range of information technologies useful for supporting the specific planning context for which they are designed (Klosterman, 1997; Geertman and Stillwell, 2003). It is envisaged that the various components of the research agenda (Figure 1) could be incorporated into a science-policy, water-urban planning “boundary object” (Guston, 2001): a Planning Support System for Water Smart

Tropical and Sub-Tropical Cities and Towns, providing an integrated, flexible, user-friendly, accessible, repository or “intelligent digital toolbox” (Klosterman, 1994), of data, information, tools and models from which users (planners and communities) can select to support the particular planning task at hand.

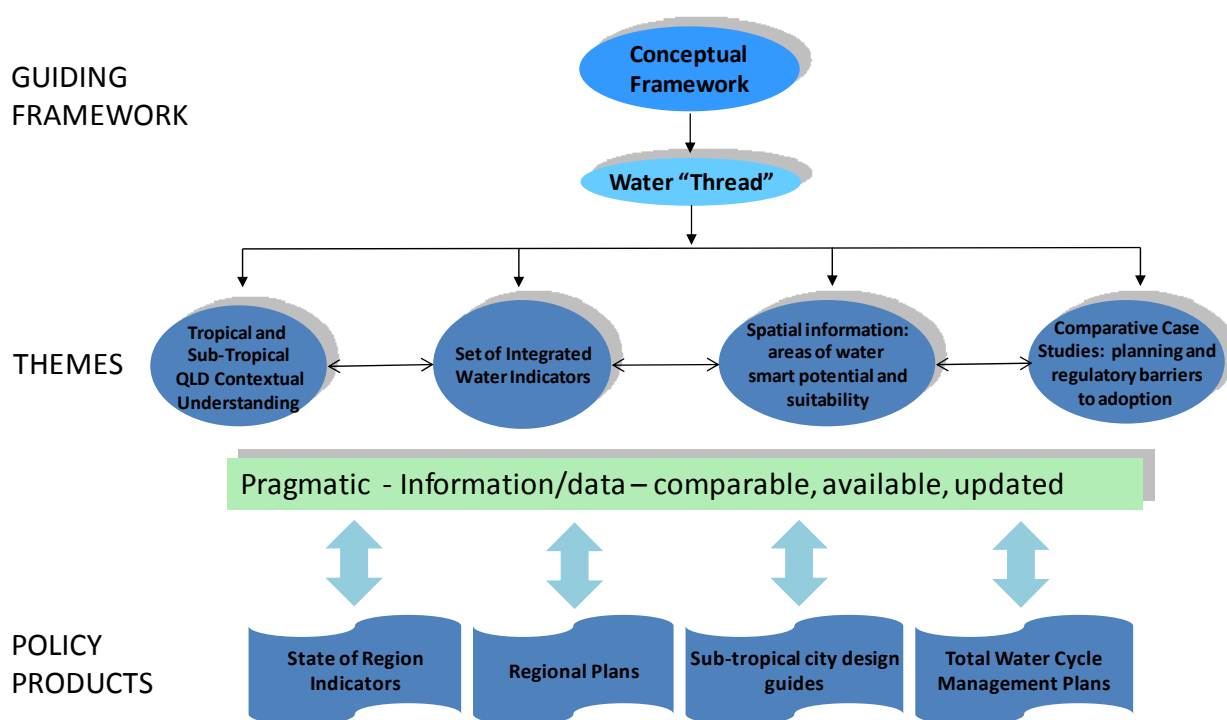


Figure 1. Emerging themes of a “legacy” Research Agenda for Water Smart Tropical and Sub-Tropical Cities and Towns.

Conclusion

This paper has described the process and outcomes relating to the development of a research agenda for Water Smart Tropical and Sub-tropical Cities with the aim of advancing the current understanding of the role and importance of water in achieving sustainable urban development. In response to the identifications of the area of “Water Smart Cities” as a strategic research priority by key stakeholders of the Urban Water Security Research Alliance in 2010, a research project was initiated with the purpose of delivering a research program as an Alliance “legacy”, for subsequent implementation.

An iterative process was employed with option development and refinement occurring through increasingly focused stakeholder feedback supported by an increasingly targeted review of scientific and industry and practice literature. There are a number of related initiatives and approaches currently underway, including Water Sensitive Cities, Cities of the Future and Integrated Urban Water Management. While these efforts cover significant ground, what is clear from the literature, and from stakeholders, is that gaps remain. Further research needs and opportunities identified include: a guiding conceptual framework; an understanding of the range of tropical and sub-tropical Queensland biophysical and socio-economic context and how water smart cities approaches may differ from other contexts; enhanced evidence for participative planning and implementation including a set of integrated water indicators, spatial information of areas of water smart potential and suitability; and comparative case studies investigating planning and regulatory and barriers to adoption.

Within the Queensland policy and planning environment, regional planning instruments together with the associated state of the region reporting and monitoring, have emerged as the most appropriate planning instrument to consider influencing. It is proposed that a Planning Support System for Water Smart Tropical and Sub-Tropical Cities and Towns is potentially an appropriate, high-impact delivery platform for the research outputs.

It is intended that this emerging set of research activities will be reviewed and refined through a final set of interactions with stakeholders and researchers to ultimately reach agreement on a full program of activities with scheduling, resourcing and implementation details.

This process of iterative dialogue represents a significant opportunity in generating a genuinely trans-disciplinary, multi-scale basis for better integrating water within the urban development process. The state of Queensland, with its rapidly expanding urban population and economic growth, and tropical and sub-tropical climate, provides a unique context for making a step-change in progressing towards the next generation of urban sustainability.

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Genetic Markers for the Detection of Sewage Pollution in Environmental Waters in Brisbane

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Summary

Environmental water samples ($n = 13$) were collected during the dry and wet weather conditions and tested for the presence of the sewage-associated markers, namely, *Methanobrevibacter smithii* *nifH* gene marker, *Enterococcus* surface protein (*esp*) found in *Enterococcus faecium*, *Bacteroides* HF183, adenoviruses (AVs) and polyomaviruses (PVs). The host-specificity of the genetic markers to differentiate between human and animal faeces was > 0.95 (maximum value of 1), while the overall sensitivity of these markers in human sourced faeces and wastewater ranged between 0.78 to 1.0 (maximum value of 1). Among the 13 environmental water samples tested, two (15%), three (23%), ten (77%), five (38%), and four (31%) were positive for the *nifH*, *esp*, HF183, AVs and PVs, respectively. The prevalence of the *nifH* marker in environmental water samples was low compared to other markers, suggesting that the use of this marker alone may not be sensitive enough to detect faecal pollution in environmental waters. Based on our results, it is recommended that a combination of sewage-associated markers should be used for the accurate and sensitive detection of faecal pollution in South East Queensland's waterways.

Keywords

Faecal pollution, microbial source tracking, faecal indicator bacteria, genetic markers, PCR.

Introduction

The source identification of faecal pollution in environmental waters is vitally important to minimise public health risks associated with the exposure to enteric pathogens. Faecal pollution tracking in environmental waters is challenging, however, due to the diffuse nature of different sources of pollution. In recent years, library-independent microbial source tracking (MST) methods have been developed to detect, and in some cases, quantify sewage and animal faeces associated markers in environmental waters (Bernhard and Field, 2000; Fong *et al.*, 2005; McQuaig *et al.*, 2009). The commonly used markers include anaerobic bacterial gene markers (Bernhard and Field, 2000) bacterial toxin gene markers (Scott *et al.*, 2005) and viral markers (Fong *et al.*, 2005; McQuaig *et al.*, 2009).

Ideally, a MST marker should have certain characteristics such as: (i) it should be specific to only the target host group (known as host-specificity); (ii) it should be present in all members within a host group (known as host-sensitivity); (iii) it should exhibit temporal and geographical stability; and (iv) the decay rate should be similar to those of pathogens (Field and Samadpour, 2007; Stoeckel and Harwood, 2007). Among these characteristics, host-specificity and -sensitivity are considered as important traits because these could influence the false positive and negative detection of faecal pollution in environmental waters. It has been reported that certain markers are highly host-specific while other markers have been reported to exhibit low host-specificity. The host-specificity of a particular marker, especially anaerobic bacterial gene or toxin gene markers may vary across different geographical locations (Field and Samadpour, 2007; Stoeckel and Harwood, 2007). Because of this, validation of the MST markers against reference faecal samples has been recommended in a new geographical area before application (Stoeckel and Harwood, 2007). None of the sewage-associated markers reported in the research literature has all the desired characteristics. It is, therefore, recommended that a combination of markers (ie, multiple markers) should be used in environmental studies to obtain confirmatory results.

The primary aim of this study was to evaluate the host-specificity and -sensitivity of sewage-associated markers in faecal samples from 11 host groups in South East Queensland (SEQ), Australia. In addition to the testing of specific host groups, environmental water samples were also tested for the presence of the *Methanobrevibacter smithii* *nifH* gene marker (Ufnar *et al.*, 2006), *Enterococcus* surface protein (*esp*) found in *Enterococcus faecium* (Scott *et al.*, 2005), *Bacteroides* HF183 (Seurinck *et al.*, 2005), adenoviruses (AVs) (Fong *et al.*, 2005) and polyomaviruses (PVs) (McQuaig *et al.*, 2009). The host-specificity and -sensitivity of the sewage-associated markers along with the polymerase chain reaction (PCR) results were then used to validate the presence of sewage pollution in SEQ environmental waters.

Methods

Host-Specificity and -Sensitivity of the Genetic Markers

To determine the host-specificity and -sensitivity of the genetic markers, up to 500 faecal samples were collected from 11 host groups using aseptic techniques. The host groups include human, birds, cattle, horses, pigs, chickens, ducks, dogs, sheep, kangaroos and possums. These host-groups were chosen because they were considered as the

potential sources of faecal pollution in catchment water. The sensitivity and specificity of the genetic markers were determined as: sensitivity = $a/(a + c)$ and specificity = $d/(b + d)$, where 'a' is true positive (samples were positive for the marker of its own species), 'b' is false positive (samples positive for the PCR marker of another species), 'c' is false negative (samples were negative for the marker of its own species), and 'd' is true negative (samples were negative for the PCR marker of another species) (Gawler *et al.*, 2007).

PCR Inhibitors

An experiment was conducted to determine the effects of PCR inhibitors on the detection of sewage-associated markers in environmental water samples. Ten-fold serial dilutions were made and all DNA samples (undiluted, 10-fold and 100-fold dilutions) were spiked with 10^3 gene copies of the *Campylobacter jejuni mapA* gene. The threshold cycle (C_T) values obtained for the DNA samples from the spiked environmental water samples were compared to those of the DNA samples from distilled water.

Environmental Water Sampling

Environmental water samples ($n=13$) were collected between March 2010 and July 2011 from the Fitzgibbon (FG) stormwater drain ($n=4$), the Brisbane River (BR) ($n=3$), Cabbage Tree Creek (CT) ($n=3$), and Oxley Creek (OC) ($n=3$). Among the 13 samples, six were collected during dry weather conditions and the remaining seven were collected following wet weather events. The sampling site FG is located in a storm water drain that receives runoff from a surrounding 290 ha residential catchment. The suspected source of faecal pollution in this site includes sewage network pipes and small numbers of horses and cattle (Table 1). The site BR is located in the Brisbane River and is tidally influenced. This site receives urban run-off through stormwater drains. Sampling site CT is located in the Cabbage Tree Creek Catchment and contains residential and industrial developments and is serviced by a sewage treatment plant (STP). Sampling site OC is a tributary of the Brisbane River. This site is also tidally influenced and is highly populated, and characterised by industrial areas, as well as a STP. Water samples were collected in sterile plastic containers at 30 cm below the water surface of all sites, and transported to the laboratory and processed within four to eight hours.

Table 1. Sampling sites description and location around Brisbane.

Sampling Sites	Site Description	GPS Coordinate
FG	Receive runoff from low density urban areas, some animal input such as cattle and horses	27°20'08.7"S; 153°01'14.5"E
BR	Stormwater drain outlets from urban area, dilution effect due to tidal influence	27°28'50.05"S; 152°59'53.84"E
CT	Medium density residential and industrial developments and serviced by a wastewater treatment plant	27°20'59.7"S; 153°02'06.6"E
OC	Tributary of the Brisbane River, industrial area, medium density population, and serviced by a wastewater treatment plant	27°32'07.8"S; 152°59'31.4"E

Microbiological Analysis

The membrane filtration method was used to process environmental water samples for faecal indicator bacteria (FIB) isolation and enumeration. Sample serial dilutions were made and filtered through 0.45 μ m 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 *Enterococcus* spp., respectively. Variable volumes (ranged from nine L to 19 L) of water samples from each site were concentrated by hollow-fiber ultrafiltration system (HFUS), using Hemoflow HF80S dialysis filters (Fresenius Medical Care, Lexington, MA, USA) as previously described (Hill *et al.*, 2005). For real-time PCR analysis of the *nifH*, HF183, AVs and PVs, DNA was extracted from the pellet (obtained from 1.5 mL of concentrated samples) using the PowerSoil DNA Kit (MOBIO Laboratories, CA, USA) following the manufacturer's instructions. Extracted DNA samples were stored at -20°C until processed. Real-time PCR assays were performed using previously published primers, probes and cycling parameters (Fong *et al.*, 2005; McQuaig *et al.*, 2009; Scott *et al.*, 2005; Seurinck *et al.*, 2005; Ufnar *et al.*, 2006). For each PCR experiment, positive controls (eg, corresponding plasmid DNA or genomic DNA) and negative control (eg, sterile water) were included. The PCR was performed using the Bio-Rad iQ5 thermal cycler (Bio-Rad Laboratories).

Results and Discussion

Host-Specificity and -Sensitivity of the Genetic Markers

The *esp*, HF183 and PVs showed absolute host-specificity value of 1.0 (maximum value of 1.0) (Table 2). The *nifH* and AVs also exhibited high host-specificity (> 0.95) suggesting these markers can be used to differentiate faecal pollution from sewage to animals. The HF183 and AVs showed high-sensitivity value of 1.0 (maximum value of 1.0) suggesting high prevalence of these markers in human sewage. The *nifH* (ie, 0.81), *esp* (ie, 0.90) and PVs (ie, 0.78) were less prevalent in human sewage than HF183 and AVs. Host-specificity and -sensitivity are two important characteristics of molecular markers because markers with low specificity and sensitivity may result in false positive and false negative detection of sewage pollution in environmental waters (Noble *et al.*, 2003). It is desirable that a marker should be highly host-specific (value of 1), however, the specificity of a particular marker may vary from study to study (Ahmed *et al.*, 2008; McQuaig *et al.*, 2009). US EPA recommended that a marker with specificity < 0.8 may not be useful for MST studies (US EPA, 2005). Several published studies reported the specificity of marker(s) > 0.9 and therefore, marker(s) showing a value above 0.9 could be considered as suitable for MST studies (Ahmed *et al.*, 2008a, Ahmed *et al.*, 2008b; McQuaig *et al.*, 2006; Shanks *et al.*, 2009). It has been recommended that the specificity and sensitivity of MST markers need to be tested prior to their application for environmental studies, especially for geographical locations where the specificity has never been tested (Field and Samadpour, 2007; Stoeckel and Harwood, 2007). In this study, the specificity of the genetic markers was rigorously evaluated by screening around 500 faecal samples from 11 animal species. Care was taken to prevent PCR false positive results. DNA extracted from animal faeces and wastewater was serially diluted, and tested with the PCR to confirm that inhibitors did not mask the amplification. To prevent PCR cross contamination, animal faecal and wastewater samples were processed before sewage samples.

The genetic markers were occasionally detected in non-target faecal samples such as from dog faeces. The presence of the HF183 marker in animals has been explained by the fact that horizontal transfer of faecal bacteria may occur among animal species in close contact such as humans and their pets (ie, cats and dogs) (Carson *et al.*, 2005). None of the sewage-associated bacterial markers reported in the research literature was shown to be absolutely host-specific. For example, the *esp* and the HF183 markers were detected in non-target faecal samples (Ahmed *et al.*, 2008a, Carson *et al.*, 2005; Whitman *et al.*, 2007).

Table 2. Host-specificity and -sensitivity of the genetic markers.

Genetic Markers	Host-Specificity	Host-Sensitivity
<i>nifH</i>	1.0	1.0
<i>esp</i>	1.0	0.90
HF183	>0.95	0.81
AVs	>0.95	1.0
PVs	1.0	0.78

PCR Inhibitors

For the spiked distilled water, the mean C_T value for the *mapA* gene was 32.4 ± 0.6 . For environmental water samples, the mean C_T values were 32.4 ± 0.8 (for undiluted DNA), 31.8 ± 1.0 (for 10-fold dilutions) and 31.3 ± 0.7 (for 100-fold dilutions). One-way ANOVA was performed to determine the differences between the C_T values obtained for distilled water and those obtained for surface water samples. No significant difference ($P > 0.05$) was observed among the C_T values for spiked distilled water, undiluted, 10-fold and 100-fold diluted DNA indicating the environmental samples were potentially free of PCR inhibitors.

Number of Faecal Indicators and the Prevalence of Sewage-Associated Markers in Environmental Water Samples

The numbers of FIB in water samples collected during dry conditions ranged from 7.7×10^1 to 5.3×10^2 (for *E. coli*) and from 1.3×10^2 to 5.1×10^2 (for *Enterococcus* spp.) (Table 3). For wet conditions, these figures were 4.7×10^2 to 8.4×10^3 (*E. coli*) and 1.8×10^3 to 2.5×10^4 (*Enterococcus* spp.). The numbers of both *E. coli* and *Enterococcus* spp. were generally one to two orders of magnitude higher during the wet conditions compared to dry conditions. The numbers of both *E. coli* and *Enterococcus* spp. in the dry conditions differed significantly ($P < 0.001$, *E. coli*; $P < 0.001$, *Enterococcus* spp.) from the wet conditions. Among the 13 samples tested during the dry and wet conditions, ten (77%) and 13 (100%) samples exceeded the Australian and New Zealand Environment and Conservation Council (ANZECC) water quality guidelines of 150 faecal coliforms and 35 *Enterococcus* spp. per 100 mL of water for primary contact.

Among the 13 samples tested, two (15%), three (23%), ten (77%), five (38%), and four (31%) were positive for the *nifH*, *esp*, HF183, AVs and PVs markers, respectively (Table 3). Among the 13 samples tested, 13 (100%) were positive for at least one marker, five (38%) were positive for at least two markers, one (8%), from BR taken after a runoff event, was positive for all five markers tested in this study. The presence of sewage pollution in environmental waters could not be detected in 11 (85%) samples, if the *nifH* marker was used alone in this study. Similarly, sewage pollution could not be detected in ten (76%), three (23%), eight (62%), and nine (70%) of samples, if the *esp*, HF183, AVs and PVs markers, respectively, were used alone.

The *E. coli* and enterococci numbers in environmental water samples collected during the wet conditions were significantly ($P < 0.001$, *E. coli*; $P < 0.001$, enterococci) higher than those in dry conditions. This is in agreement with previous studies and not unexpected because, after rainfall, waterways receive faecal pollution from various point and non-point sources (O'Shea and Field, 1992). The *nifH* marker was detected in two samples from the sites FG and BR indicating potential sewage pollution. Both these sites receive stormwater through urban runoff. The presence of the *nifH* marker in stormwater runoff has been reported in recent studies (Sercu *et al.*, 2011; Parker *et al.*, 2010). The *esp* marker was detected in three samples from the sites BR (two samples) and OC (one sample). Overall, the HF183 was more frequently detected in water samples than other markers. Three of four samples from the site FG were PCR positive for the HF183, suggesting high prevalence of this marker in stormwater runoff. Viral markers AVs and/or PVs were also detected in several samples suggesting the presence of sewage pollution in the tested environmental water samples. Only one sample from site BR was positive for all five markers. It has to be noted that this site receives urban run-off through stormwater drains and possibly exposed to more human faecal matters compared to other sites.

Table 3. The number of *Escherichia coli* and Enterococcus spp. and PCR positive/negative results of sewage associated markers in water samples collected from the FitzGibbon (FG), Brisbane River (BR), Cabbage Tree Creek (CT) and Oxley Creek (OC) South East Queensland, Australia.

Sampling Site	Event	Sampling Condition (rainfall)	Faecal Indicators CFU/100 mL		Sewage-Associated PCR Marker Results				
			<i>E. coli</i>	Enterococcus spp.	<i>nifH</i>	<i>esp</i>	HF183	AVs	PVs
FG	1	Dry (0.4 mm)	3.6×10^2	5.1×10^2	ND ¹	ND	+	ND	ND
	2	Wet (49.8 mm)	4.7×10^3	1.8×10^3	+	ND	+	ND	ND
	3	Dry (2.6 mm)	1.3×10^2	1.3×10^2	ND	ND	+	ND	ND
	2	Wet (14.8 mm)	3.4×10^3	2.3×10^4	ND	ND	ND	+	ND
BR	1	Dry (0 mm)	7.7×10^1	3.3×10^2	ND	ND	+	ND	+
	2	Wet (10.2 mm)	4.7×10^2	3.4×10^3	ND	+	+	ND	ND
	3	Wet (19.4 mm)	6.0×10^3	8.1×10^3	+	+	+	+	+
CT	1	Dry (0 mm)	4.8×10^2	2.0×10^2	ND	ND	+	ND	ND
	2	Dry (2.6 mm)	5.3×10^2	4.6×10^2	ND	ND	+	ND	+
	3	Wet (19.4 mm)	8.4×10^3	2.5×10^4	ND	ND	+	+	ND
OC	1	Wet (15 mm)	1.6×10^3	1.1×10^4	ND	ND	ND	+	ND
	2	Dry (0 mm)	9.0×10^1	4.0×10^2	ND	+	+	ND	ND
	3	Wet (7 mm)	3.5×10^3	9.9×10^3	ND	ND	ND	+	+

1 ND = Not Detected.

The prevalence of the *nifH* marker was low in environmental samples compared to the *esp*, HF183, AVs and PVs. Sewage pollution could not be detected in 11 (85%) samples when the *nifH* marker was used alone in this study. The presence of PCR inhibitors in environmental water samples masking the *nifH* detection can be ruled out as all water samples were spiked with low gene copies (ie, 10^3) of *C. jejuni mapA* gene. The amplification of the *mapA* gene indicated the samples were free from PCR inhibitors. Larger volumes (9-19 L depending on the turbidity) of water samples were tested in this study for the sensitive detection of the sewage-associated markers in environmental waters. A recent study reported the application of HFUS combined with the PCR detection of the MST markers in fresh and estuarine water in the USA (Lesiken *et al.*, 2010) and determined that the HFUS method with the PCR detection was more sensitive compared to the membrane filtration method with the PCR detection.

The low prevalence of the *nifH* marker in environmental samples could be due to the fact that these markers either have different decay rates in environmental waters compared to other markers or because of their low prevalence in sewage. A recent study reported the low prevalence of the *nifH* marker in two urban watersheds in California, USA compared to the HF183 markers (Sercu *et al.*, 2011). The authors modified the *nifH* protocol into a two round PCR for increased sensitivity. Despite that, the HF183 marker was more frequently detected in environmental samples compared to the *nifH* marker (Sercu *et al.*, 2011b). The absence of the *nifH* marker, however, in an environmental

water sample does not rule out the presence of sewage pollution. For the accurate and sensitive identification of human faecal pollution, it is recommended that multiple markers should be used to obtain confirmatory results (McQuaig *et al.*, 2006; Ahmed *et al.*, 2010a). The findings of the present study also suggest that multiple markers should be used in environmental studies to reduce uncertainties associated with a particular marker that fails to detect faecal pollution in environmental waters.

Conclusions

The genetic markers tested in this study appear to be highly sewage specific and can be used to distinguish human faecal pollution from animal faecal pollution. The numbers of both *E. coli* and *Enterococcus* spp. were generally one to two orders of magnitude higher during the wet conditions compared to dry conditions. Among the 13 samples tested during the dry and wet conditions, ten (77%) and 13 (100%) samples exceeded the ANZECC water quality guidelines suggesting the water may not be suitable for primary contact. Sewage pollution was detected in a number of water samples. HF183 was the most and the nifH was least prevalent among the markers tested. The findings of the present study also suggest that multiple markers should be used in environmental studies to reduce uncertainties associated with a particular marker that fails to detect faecal pollution in environmental waters. Further research is required to investigate the correlation between these markers and pathogens in environmental waters.

Acknowledgements

This research was undertaken and funded as part of the SEQ Urban Water Security Research Alliance, a scientific collaboration between the Queensland government, CSIRO, The University of Queensland and Griffith University.

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Poster - Abstracts

Quantification of Evaporation from a Small Water Body Using the Scintillometry, Eddy Covariance and Mass Transfer Techniques

McGloin, R.¹, McJannet, D.², McGowan, H.¹, Cook, F.² and Burn, S.³

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³ CSIRO Land and Water, Highett, Victoria

Accurate quantification of evaporation from small water storages is essential for water management and planning, particularly in water-scarce regions. In order to ascertain suitable methods for direct measurement of evaporation from small water bodies, this research presents a comparison of three evaporation evaluation methods of varying technological complexity and cost (eddy covariance, scintillometry and the mass transfer model) at a small reservoir in south east Queensland, Australia. Theoretical concerns relating to the application of these techniques to the unique environment of a small water body were analysed, including a detailed analysis of the spatial dimensions of the measurement footprints for both techniques. Overall, excellent correlation was shown between the latent and sensible heat flux measurements of the eddy covariance and scintillometry techniques across a variety of seasonal weather conditions. Excellent agreement was also observed between eddy covariance measurements and estimates from the much less expensive and time consuming mass transfer method.

Mechanisms of Micropollutants Removal by BAC Filtration

Rattier, M.¹, Reungoat, J.¹, Gernjak, W.¹, Joss, A.² and Keller, J.¹

¹ The University of Queensland, Advanced Water Management Centre (AWMC), St Lucia, Queensland

² Eawag, Swiss Federal Institute of Aquatic Science and Technology, Duebendorf, Switzerland

Organic micropollutants are discharged into surface water via wastewater treatment plant effluent and subsequently are suspected to impact aquatic wildlife and contaminate drinking water sources. Biological Activated Carbon (BAC) filters have been shown to be effective in removing micropollutants from water via adsorption and/or biodegradation removal processes. It is not known which mechanism is involved and to what extent. This has implications for the design of efficient filters. Filtration involving multi-component mixtures and competitive adsorption present complexity with regards to evaluating adsorption and biodegradation. The purpose of this experimental study is to quantify the adsorption and/or biodegradation of micropollutants in BAC filters. A research approach was designed to follow the long-term performance of a set of lab scale columns filtering treated effluent. The inhibition of the microbial community and use of non-sorptive carbon media were selected to discriminate each mechanism from the BAC performance. The results demonstrate adsorption of micropollutants was prevalent during the initial stage of the BAC columns and was decreasing due to loading with organic matter. Interestingly, the non-sorptive media provided efficient removal for several compounds even after having filtered 8,000 bed volumes. The use of inhibitors provided evidence that the active biomass covering the media was responsible for varying proportions of the removal of the micropollutants present in the treated effluent.

Water Governance Under Pressure: Demand Management Policy Influence on Water Management Practices in South East Queensland

Lewis, B.R.

The University of Queensland

The millennium drought brought about very real water scarcity concerns in South East Queensland (SEQ) and posed serious challenges for policy makers. A key feature of the State Government's response was the development and implementation of a comprehensive demand management strategy targeting both industry and residents. The strategy comprised a mixed suite of *regulatory, incentives based and suasive policy instruments* deployed to encourage greater water use efficiency.

In this poster presentation the author presents preliminary findings from his current PhD research reviewing the demand management strategy implemented in SEQ. A goal of the research is to tell the story of demand management policy in SEQ, and to explore its role in shaping the changing water management practices of industry and the community for the period 2005 – 2010. The research will inform our understandings of how well the mix of policy instruments worked in practice, and what learnings can be drawn from policy challenges that occurred and were overcome. The study also seeks to explore whether or not other factors have been significant in influencing, enabling and/or constraining adoption of water efficiency practices by SEQ industry and residents. The findings draw from qualitative research undertaken with policy makers, industry representatives and water professionals.

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

Makki, A.A.^{1,2}, 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 indoor 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 water meters in South East Queensland. ANOVA, as well as multiple regression analysis statistical techniques, were used to reveal the determinants (eg, demographic, socio-demographic, stock efficiency, and appliance physical characteristics) of household indoor end use consumptions (eg, shower, tap, dishwasher, clothes washer and toilet). The study revealed a list of the most significant determinants for each of end use consumption at the household level. The study discusses the significant end use determinants and how this statistical approach will be followed to predict overall household residential indoor consumption. Implications of the research on conservation strategies and policy design are discussed, along with future research directions.

Intelligent Sensors to Determine Water End-Use

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³ Centre for Infrastructure Engineering and Management, Griffith University, Southport, Queensland

As part of their efforts to secure our water supplies into the future, Australian water utilities have been investigating the effectiveness of various water savings measures on individual end uses at particular tap outlets and appliances. At present this is done using software analysis of flow measurements at the water meter, which is collated with the results of householder diaries. This is a labour-intensive exercise, and only gives approximate results as many water end usage patterns cannot be reliably distinguished (eg, laundry trough from hand basin or kitchen sink). CSIRO are currently developing a proof of concept intelligent sensor system to automatically discriminate individual end use. Results from a 2 week field trial in one house from the current SEQREUS project are presented and validated against TraceWizard® water end use data.

Crop Mapping and Water Balance Modeling to Extrapolate Applied Irrigation in the Lockyer Valley, Queensland

Ellis T.W.¹, Hartcher, M.¹, Grimm M.², Kodur S.³ and Wolf L.¹

¹ CSIRO Land and Water, CSIRO Land and Water, Ecosciences Precinct, Dutton Park, Queensland

² Department of Civil Engineering, Geo and Environmental Sciences, Institute of Applied Geosciences, Karlsruhe Institute of Technology, Germany

³ Department of Environment and Resource Management, Toowoomba, Queensland

There is a proposal to make available purified recycled wastewater (PRW) to augment groundwater irrigation in the Lockyer Valley, west of Brisbane. A groundwater flow model of the local aquifer will be used to evaluate supply scenarios including direct supply of PRW to farm gate or aquifer injection. Landsat satellite imagery was used to map crops in the Lockyer Valley to inform a crop water balance model, Howleakey. Irrigation demand and deep drainage outputs from Howleakey were used to inform the groundwater flow model. This paper describes the crop mapping and Howleakey methodology, its evaluation against measured groundwater extractions from the Central Lockyer Valley, and its extrapolation to the remaining irrigated areas in the Valley.

N₂O and Ozone Layer Depletion – a New Consideration for Urban Water Planners?

Lane, J.^{1,2} and Lant, P.²

¹ Advanced Water Management Centre, The University of Queensland, St Lucia, Queensland

² School of Chemical Engineering, The University of Queensland, St Lucia, Queensland

In the course of using Life Cycle Assessment (LCA) to inform the Total Water Cycle Planning process in South East Queensland, a review has been conducted on the relevance of state-of-the-art LCA impact assessment models for use in an urban water systems context. Metrics for Ozone Depletion Potential are commonly used in international LCA studies, although LCA researchers typically adopt the broader community view that the Montreal Protocol has largely ‘fixed’ the ozone layer problem through the regulation of CFCs and HCFCs.

While not recognised by either the broader public or default LCA approaches, it is well accepted in the atmospheric science community that N₂O depletes the ozone layer. Recent science has demonstrated that anthropogenic N₂O emissions are now the biggest cause of ozone layer damage. We use a number of case studies to demonstrate the potential significance of this to ascertaining the environmental burden of wastewater treatment plants, and urban water systems more broadly.

There remains some uncertainty about how best to account for the effect of N₂O on the ozone layer in multi-criteria environmental analysis, and the broader significance of the remaining ozone layer ‘problem’. Nonetheless, our results suggest that ozone layer depletion could become a significant management issue for urban water utilities should the ozone threat from N₂O find its way into prominent policy debate. Continued effort into understanding the sources of N₂O emissions from the urban water system, and the potential for mitigation, would therefore be beneficial.

Human Factors in Water Safety Systems

Cloete, S., Horberry, T. and Head, B.

The University of Queensland, St Lucia, Queensland

Many industries now recognise the need to integrate human factors research and principles into their overall risk management schemes. Two broad sets of circumstances have presented us with new opportunities to examine the interactions of human operators and technology in the SEQ water grid: firstly, organisational changes, both recent and forthcoming, to the ownership and operation of water grid assets; and secondly, the recent occurrence of widely publicised incidents in which human error was deemed a contributory factor. This research targeted operations in control rooms for both water treatment and distribution, with a particular focus on the operators’ experiences with alarm systems, human machine interfaces and automation. Results collected to date indicate several opportunities for improving aspects of control room design and work practices, which would bring the SEQ water grid into line with best practice in other process-control industries. Implementing these changes would ultimately reduce the likelihood of human error.

Posters - Presented

Quantification of Evaporation from a Small Water Body using the Scintillometry, Eddy Covariance and Mass Transfer Techniques

McGloin, R.¹, McJannet, D.², McGowan, H.¹, Cook, F.² and Bum, S.²

¹ Climate Research Group, School of Geography, Planning and Environmental Management, The University of Queensland, ² CSIRO Land and Water

Introduction

Accurate quantification of evaporation from small water storages is essential for water management and planning, particularly in water-scarce regions. In order to ascertain suitable methods for direct measurement of evaporation from small water bodies, this study presents a comparison of eddy covariance (EC) and scintillometry measurements from a reservoir in South East Queensland, Australia. The potential for evaluating evaporation from a small reservoir using the much less expensive and time consuming theoretical mass transfer method was also assessed.

Methodology

The primary instrumentation included a pontoon-mounted EC unit incorporating an open-path infrared gas analyser and 3D sonic anemometer (Figure 1a) and a scintillometer consisting of an emitter that transmits an open-path infrared beam to a receiver on the opposite bank of the reservoir (Figure 1b). Evaporation is calculated by the EC unit using the covariance of vertical wind speed and water vapour density, while the scintillometer uses scintillation of the infrared beam in conjunction with standard meteorological measurements to derive evaporation. The mass transfer method is based on a simple equation where evaporation is determined as a function of wind speed and vapour pressure gradient.



Figure 1. (a) Eddy Covariance apparatus and (b) scintillometer receiver.

Results and Discussion

Overall, excellent agreement was observed between the evaporation measurements of the eddy covariance, scintillometry and mass transfer techniques during the majority of seasonal weather conditions found at the study site (Figure 2).

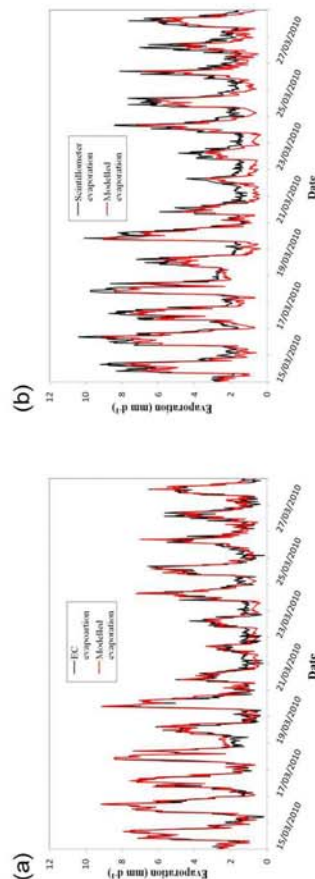


Figure 2. Time series of evaporation modelled using a mass transfer equation and (a) EC evaporation, and (b) scintillometer evaporation during the March 2010.

Theoretical concerns such as the dimensions of the measurement footprint (defines the spatial context of measurements) for EC and scintillometry were also analysed (Figure 3). It was found that the majority of the footprint contributions for both EC and scintillometry originated from the water surface.

Conclusions

Results suggest that the three evaporation techniques all show excellent agreement with one another and are capable of obtaining datasets of reasonable size and quality in the majority of conditions found at Logan's Dam. The simple nature of the mass transfer model and its excellent correlation with evaporation measurements in this study make it an attractive option for evaporation quantification at water bodies of all types.

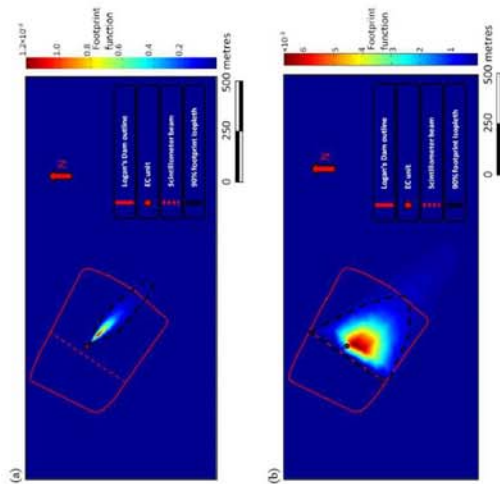


Figure 3. Footprint diagrams at Logan's Dam for (a) the EC system and (b) the scintillometer.



Water Governance Under Pressure: Demand Management Policy Influence on Water Management Practices in SEQ

Brad Lewis

Institute for Social Science Research - The University of Queensland

Summary

This study examines the role of demand management policy in changing industry and community water management practice in South East Queensland (SEQ) from 2005 to 2010. It has two key research components - a review of:

- Policy effectiveness in shaping residential water use behaviours, primarily based on reviewing previous analyses; and
- Policy effectiveness in shaping corporate water use behaviours, based on document analysis and interviews with water and industry leaders.

Results and Discussion

The research suggests that implementation of the Government's demand management policy played a major role in bringing about declines in the average water use by individual SEQ households and corporations.

- **Residential** water use was especially influenced by **regulatory** approaches targeting outdoor water use, ie, a regime of water restrictions ranging from levels 1-6, together with **suasive** or awareness raising campaigns, ie, the Target 140 campaign.
- **Corporate** water use was especially influenced by **regulatory** approaches requiring compulsory implementation of a Water Efficiency Management Plan (WEMP) requiring a 25% reduction in water use or demonstrated adoption of industry best practice.
- Households and corporate water users were also influenced by: **Incentive** approaches, ie, pricing reform and rebate schemes, eg, Business Water Efficiency Program and the Home Waterwise Rebate Scheme; and **suasive** approaches including EcoBiz.



Key Findings

Residential water use:

- Water restrictions had already reduced outdoor household water use (eg, gardens, car washing, pools). Yet 'demand hardening' meant further water savings were unlikely / impossible.
- Policy makers had limited options as monitoring and regulating indoor household water use was problematic. Policy makers under pressure needed the suasive Target 140 campaign to work, and it did!

Corporate water use:

- Urban based industries are highly diverse and there has been a limited history of engaging urban industries in water reform.
- WEMPs required businesses to change their management approach – this was easier for some and harder for others.
- Saving water can also lead to saving energy and money.



Preliminary Conclusions / Impacts

- The effectiveness of demand management policy highlights: a) the effective choice and implementation of policy tools; b) apprehension by stakeholders regarding mandatory powers of the state; c) a degree of trust in public officials; and d) capacities for changed behaviour.
- The influence of social norms on both residential and corporate water use is under-researched. There exists potential for policy makers to investigate potential linkages with policy reform options.



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 water demand forecasting model using aligned household makeup, stock efficiency, socio-demographic and natural science data sets collected from 200 households fitted with water smart meters in South East Queensland. Dummy Coding, ANOVA, as well as multiple regression analysis statistical techniques, were utilised to reveal the determinants (eg, household makeup, stock efficiency, income, etc) of household end use consumptions (eg, shower, toilet, dishwasher and clothes washer).



Figure 1. Significant determinants of daily per household end uses consumption

Results to Date

The study revealed a list of the most significant determinants for each end use consumption at the household level, as illustrated in Figure 1.

End use forecasting models were developed following a block-wise entry multiple regression approach, illustrated in Figure 2, using the revealed statistically significant determinants. Most significant combinations of determinants (shown in bold font in Figure 1) were selected as predictors for each end use based on their ability to explain variation in consumption volumes.



Figure 2. General modelling approach

Key Findings From Developed End Use Models

- Teenagers, females and showerhead efficiency ratings are the most significant determinants of daily household shower consumption.
- Frequency of usage and appliance physical characteristics (eg, efficiency, type, capacity, ECO mode) are the most significant determinants of dish washer, clothes washer and toilet daily household consumption.
- The developed forecasting models for shower, clothes washer, dishwasher and toilet are explaining 90.2%, 89.1%, 85.8% and 76.2% of variation in consumption, respectively.

Future Direction

- Revealing significant determinants and building forecasting models for other water end uses (eg, tap, bathtub).
- Development of a forecasting tool that provides an evidence-based forecast of domestic household indoor demand.

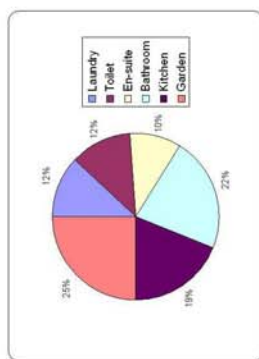
Intelligent Sensors to Determine Water End Use

Roger O'Halloran, Michael Best and Nigel Goodman
CSIRO

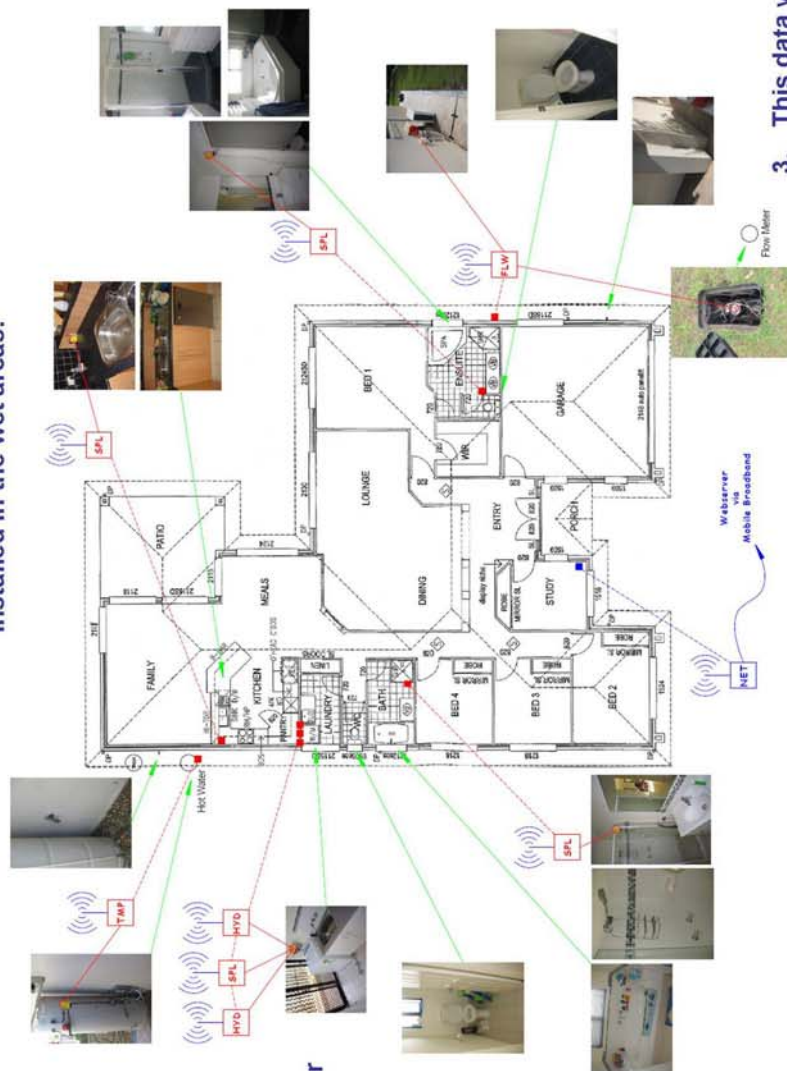
Understanding water usage is of great importance for predicting our water needs into the future. Consequently, we have developed a novel acoustic system to monitor domestic water end use. The results were very promising, and we are planning further trials in conjunction with a commercial partner.

1. A wireless network of sound pressure level sensors (SPL) were installed in the wet areas.

4. The results allowed water use at each outlet to be identified.



2. Hydrophones (HYD) were installed in the laundry plumbing, along with a temperature sensor (TMP) on the hot water service outlet.



3. This data was evaluated in conjunction with flow data (FLW) from a high resolution water meter.

Urban Water Security Research Alliance - Science Forum, June 2012



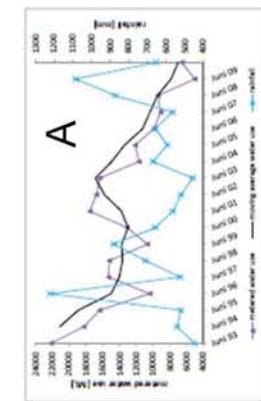
Crop Mapping and Water Balance Modelling to Extrapolate Applied Irrigation in the Lockyer Valley, Queensland

Ellis T. W.¹, Hartcher, M.¹, Grimm M.², Kodur S.³, Henderson, C.⁴, Robinson, B.³, Foley, J.³, Hodgen, M.¹ and Wolf, L.¹

¹ CSIRO Land and Water, Ecosystems Precinct, Dutton Park, Queensland; ² Department of Civil Engineering, Geo and Environmental Sciences, Karlsruhe Institute of Technology, Germany; ³ Department of Environment and Resource Management, Toowoomba, Queensland; ⁴ Department of Employment, Economic Development and Innovation, Gatton, Queensland

Introduction / Summary

The Lockyer Valley is a highly productive irrigation area west of Brisbane. A protracted drought between 2001 and 2007, significantly depleted groundwater reserves (Figure A). Recycled urban wastewater has been considered for augmentation of groundwater for future droughts.

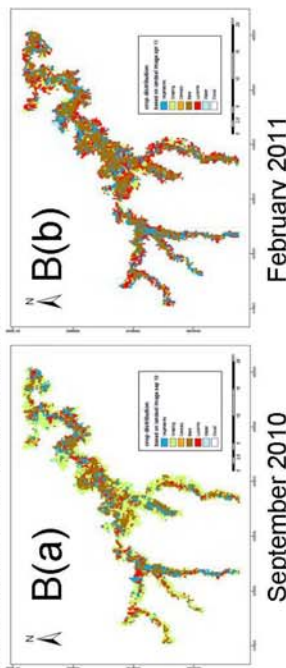


To investigate water supply scenarios, a groundwater flow model was developed. Satellite imagery (Landsat) was used to map crops. Water balance was simulated (using the model HowLeaky?) to estimate potential irrigation demand and deep drainage fluxes. Applied irrigation was estimated from groundwater use data.

Results and Discussion

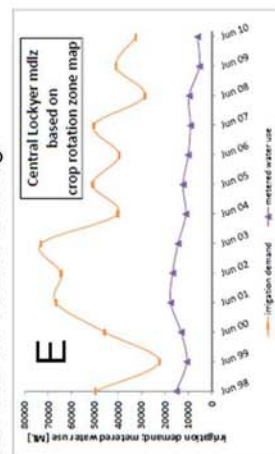
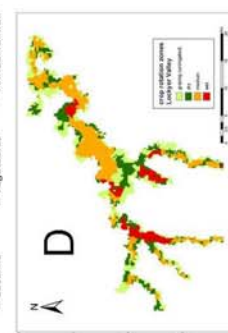
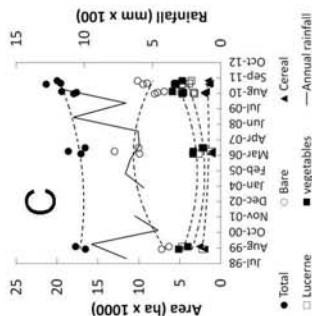
Landsat5 imagery was classified according to ground-truthing data from September 2010 and July 2011. Patterns of bare soil and crop type reflected seasonal variation and damage from the January 2011 flood [Figure B (a) and (b)]. Extrapolation of this method to "before" and "during" the 2001 to 2007 drought indicated greater bare area but similar rotation types (Figure C). Zones of "rotation type" were identified (Figure D):

- Dry = mainly cereals and lucerne - green
- Medium = mainly cereals, vegetables - orange
- Wet = continuous vegetables - red



Key Findings

The HowLeaky? simulations upscaled with rotation patterns suggested much higher irrigation "demand" compared to measured groundwater use (Figure E). This indicates farmer irrigation management rules may differ from those in the modelling and/or our allocation of crop rotation types (Figure D) was unsuitable. Preliminary error analyses suggested these errors were larger than those associated with the remote sensing.



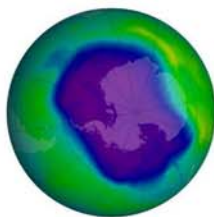
Conclusions / Impacts

- 1) Crop mapping from satellite imagery was successful as a "proof of concept";
- 2) Delineation of "rotation type" zones was possibly simplistic and did not adequately reflect drought effects;
- 3) Future modelling should represent a wider range of cropping and long-term climatic conditions.

N₂O and Ozone Layer Depletion

A New Consideration for Urban Water Planners...?

Joe Lane and Paul Lant
The University of Queensland

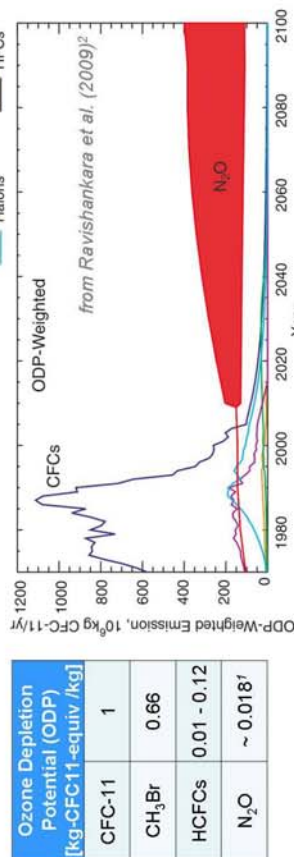


Summary

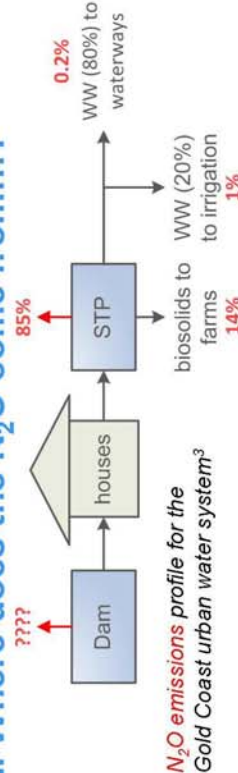
Recent scientific publications have highlighted the role of N₂O in ozone layer depletion. Case study analysis shows that, should the ozone threat from N₂O become a focus of policy action, ozone layer depletion might become a management concern for water utilities.

1. Ozone Layer Science

Anthropogenic N₂O is now the biggest cause of ozone layer depletion. However, climate change will return ozone concentrations to "acceptable" levels before the end of the century.

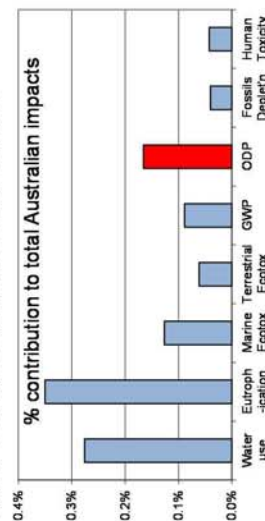


2. Where does the N₂O come from...?

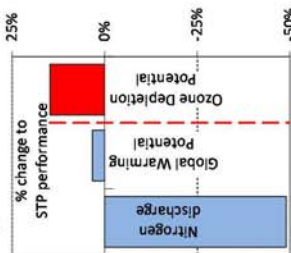


3. Significance for Water Systems Planning

Water/WW contribution to Australia's ozone layer effects is comparable to other notable environmental issues³



Including ODP could change tradeoffs analysis¹



4. What Should Industry do...?

- International policy response is not clear – ozone layer protection could confront urban water utilities in the future.
- Avoid strategies that increase intensity of N₂O emissions.
- GHG accounting will not ensure that this happens.
- Improved understanding of N₂O emissions and mitigation opportunities.

Further Reading:
1 Lane, J. and Lant, P. (2012) Including N₂O in ozone depletion models for LCA, LILCA, 17 (2):252-257
2 Ravishankara, A. et al (2009) Science 326 (5949):123-125
3 Lane, J. et al (2011) Life cycle assessment of the Gold Coast urban water system. UWSRA, Brisbane

Human Factors in Urban Water System Safety

Steven Cloete, Tim Horberry and Brian Head

The University of Queensland

Introduction

The role of the human element in the safe and efficient operation of complex sociotechnical systems has been well understood in many industries, including aviation, nuclear power and healthcare. Surprisingly, bulk water storage, treatment and distribution has received little empirical attention. This project began in response to stakeholder concerns over recent, well publicised incidents in the South East Queensland (SEQ) Water Grid, with an agreed focus on interactions between operators and technology.

Results and Discussion

Interviews and observations of operators were conducted at the control rooms of major water grid participants. Operator's experiences and opinions with current SCADA technologies were measured with two standardised questionnaire/interview instruments derived from EEMUA and Australian Standard guidelines.

Methods

Observations and interview results revealed that much of the operators' time in the control room was consumed by activities which disrupt system monitoring and supervision. These tasks are excellent candidates for automation. Outstanding examples were:

- 1) Logging of site access by maintenance and cleaning personnel (ingress and egress) is performed by telephone;
- 2) Event logging is performed manually using an Excel spreadsheet;
- 3) Routine activities (eg, reservoir flushes) require significant interactions with control software, which introduce opportunities for human error.



Figure 1. LinkWater Control Room

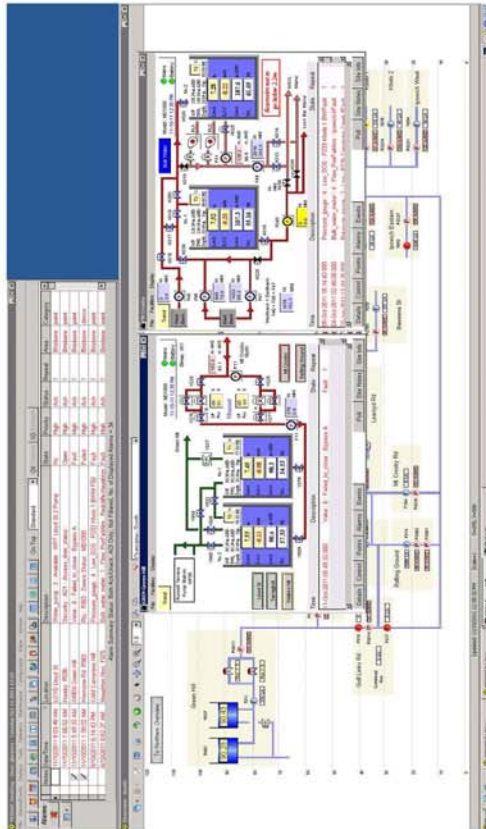


Figure 2. LinkWater Supervisory Control Interface.

Operators have developed creative strategies and workarounds to cope with inadequacies in infrastructure and work practices. They are acutely aware of shortcomings and are articulate advocates for change. They are therefore well positioned to consult in a 'participatory ergonomics' capacity, and should be included in all future infrastructure procurement and upgrade decisions.

Conclusions / Impacts

- The SEQ Water Grid has undergone sweeping changes in recent years, including the introduction of new technologies and changes in the ownership of large assets.
- Changes in technological infrastructure to support operator activities are catching up, eg, LinkWater control room upgrade.
- Improvement of technologies must proceed with integration and information sharing in mind – within and between entities – and with the input of operators.

Awards

Best Presentation

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| Kelly Fielding | – | Water End Use Feedback Produces Long-Term Reductions in Residential Water Demand |
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Highly Commended Presentations:

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| Fran Sheldon | – | Urban Creeks in SEQ - can bugs live anywhere? Role of habitat and hydrology |
| Andrea Walton | – | Community Acceptance of Policy Options for Managing the Maintenance of Rainwater Tanks |
| Fred Leusch | – | Optimising Micropollutants Extraction for Analysis of Water Samples: Comparison of Different Solid Phase Materials and Liquid-Liquid Extraction |
| Rodney Stewart | – | Implications of Resource-Efficient Technology on Peak Water Demand and Water-Related Energy Demand |

Best Paper

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| Maria José Farré | – | Case Study - Occurrence of Non-Regulated Disinfection By-Products from the Capalaba Region's Distribution System |
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Best Poster

- | | | |
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| Maxime Rattier | – | Mechanisms of Micropollutants Removal by BAC Filtration |
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Recipients were presented with a bottle of wine.

Delegates

Official Opening

Mark McArdle, MP, Minister for Energy and Water Supply, QLD

Keynote Speaker

James Cameron, National Water Commission, ACT

Guest Speakers

Helena Amaro, Sydney Water, NSW
Ian Law, IBL Solutions and AWRCE, NSW
Kathryn Linge, Curtin University, WA
Mark O'Donohue, Australian Water Recycling Centre of Excellence, Qld
Michael Kane, ULDA, Qld
Neil Palmer, National Centre of Excellence in Desalination Australia, WA
Ray Beaton, Yarra Valley Water, Melbourne, Vic
Rolando Fabris, SA Water, SA
Rosemary Leonard, Goyder Institute / CSIRO, WA
Tony Weber, BMT WBM Pty Ltd, Qld

UWSRA Management Board

Chris Davis, Chair, NSW
Alan Gregory, CSIRO, NSW
Larry Little, Griffith University, Qld
Scott Keyworth, CSIRO, ACT

UWSRA Research Advisory Committee

Paul Greenfield, Chair, UWSRA Research Advisory Committee, QLD
Mark Pascoe, International Water Centre, QLD
Tad Bagdon, Queensland Water Commission, QLD
Ted Gardner, Department of Environment and Resource Management, QLD

Attendees

Abel Immaraj, Queensland Water Commission, Qld
Alan Hoban, Bligh Tanner Pty Ltd, Qld
Alexander Mofidi, AECOM, Qld
Allan Lush, AECOM, Qld
Amanda Macdonald, Cardno, Qld
Amber Perry, Department of Environment and Heritage Protection, Qld
Amy Piekala-Fletcher, Seqwater, Qld
Anas Makki, Griffith University, Qld
Andrea Walton, CSIRO, Qld
Andres Rojo, International Water Centre, Qld
Andrew Cook, Department of Energy and Water Supply, QLD
Andrew Moir, LinkWater, Qld
Andrew O'Neill, Healthy Waterways, QLD
Andrew Palmer, CSIRO, Qld
Andrew Sloan, Unitywater, Qld
Ann Gooding, South East Water, Melbourne, VIC
Annalie Roux, Seqwater, Qld
Anne Cleary, Healthy Waterways, QLD
Anne Gardiner, Office of the Water Supply Regulator, QLD
Anne Roiko, University of the Sunshine Coast, Qld
Anne Simi, Brisbane City Council, Qld
Anthony Coates, Local Government Infrastructure Services, Qld
Anthony van Herwaarden, CSIRO, Qld
Arseto Bagastyo, The University of Queensland, QLD
Ashok Sharma, CSIRO, VIC
Audrey Van Beusichem, Department of Natural Resources and Mines, Qld
Barry Dennien, SEQ Water Grid Manager, Qld
Barry Holcroft, Unitywater, Qld
Barry Spencer, Seqwater, Qld

Beate Escher, The University of Queensland, Qld
 Belinda Young, Department of Science, Information Technology, Innovation and the Arts, Qld
 Brad Dines, Department of Environment and Heritage Protection, Qld
 Brad Lewis, The University of Queensland, Qld
 Bradley Rhodes, Veoliawater, Qld
 Brian Head, The University of Queensland, Qld
 Brian McIntosh, International Water Centre/CSIRO, QLD
 Bruce Gaydon, Toowoomba City Council, QLD
 Cameron Mathison, Ningi Distributions P/L, Qld
 Cara Beal, Griffith University, QLD
 Carol Farbotko, CSIRO, Qld
 Cedric Robillot, Seqwater, QLD
 Charles Lemckert, Griffith University, QLD
 Chris Owens, SEQ Water Grid Manager, Qld
 Chris Pipe-Martin, GHD, Qld
 Claire Downey, Policy Advisor to Minister Andrew Cripps, QLD
 Clinton Parker, Local Government Infrastructure Services, Qld
 Colleen Hanahan, Office of the Water Supply Regulator, QLD
 Craig Ma, Queensland Urban Utilities, Qld
 Craig McMahon, Toowoomba City Council, QLD
 Craig Reading, Tankworks Australia, Qld
 Craig Walton, Department of Natural Resources and Mines, QLD
 Dan Spiller, SEQ Water Grid Manager, Qld
 Dana Hallett, Queensland Urban Utilities, Qld
 Daniel Field, Qld Health, Qld
 Danielle Butcher, Queensland Water Commission, QLD
 Darren Thompson, Department of Energy and Water Supply, QLD
 David Hamlyn-Harris, Bligh Tanner Pty Ltd, Qld
 David King, Bureau of Meteorology, Vic
 David Merheb, Queensland Urban Utilities, Qld
 David Robinson, Department of Science, Information Technology, Innovation and the Arts, Qld
 David Solley, GHD, Qld
 Declan Page, CSIRO, SA
 Diana Dawson, Council of Mayors (SEQ), Qld
 Dimity Lynas, Queensland Water Commission, Qld
 Donald Mackenzie, Logan City Council, Qld
 Edward Morgan, Griffith University, Qld
 Elizabeth Kellet, Queensland Water Commission, QLD
 Eloise Larsen, LinkWater, QLD
 Emma Anastasi, University of the Sunshine Coast, QLD
 Emma Brewis, Queensland Water Commission, Qld
 Emma Hunter, Rous Water, Lismore, NSW
 Emma Newland, International Water Centre, Qld
 Emmanuelle Filloux, The University of Queensland, Qld
 Esther Coultas, CSIRO, VIC
 Evelyn Rodrigues, Water Services Association Australia, NSW
 Ferry Firmansyah, Urban Utilities, QLD
 Fiona McFarlane, CSIRO, Qld
 Fran Sheldon, Griffith University, Qld
 Fred Leusch, Griffith University, Qld
 Gemma Hamilton, Office of the Water Supply Regulator, Qld
 Grace Tjandraatmadja, CSIRO, VIC
 Greg Jackson, Qld Health, QLD
 Greg Laves, Griffith University, Qld
 Greg Tucker, Brisbane City Council, Qld
 Heather Uwins, Office of the Water Supply Regulator, QLD
 Ian Hanks, Queensland Water Commission, Qld
 Ian White, Department of Energy and Water Supply, QLD
 Imraan Yousuf, Allconnex, Qld
 Jack Mullaly, Healthy Waterways, Qld
 Jackie Hammond, Tankworks Australia, Qld
 Jane Bullock, Local Government Infrastructure Services, Qld
 Jane-Louise Lampard, Griffith University, Qld
 Janet Cumming, Qld Health, Qld
 Jatinder Sidhu, CSIRO, Qld
 Jessica Geranio, Allconnex, Qld
 Jim Mahon, Skills Tech, Qld
 Jinaraj Rajakaruna, Department of Energy and Water Supply, Qld
 Jo Csik, Allconnex, Qld
 Joanne Macdonald, University of the Sunshine Coast, Qld
 Jodieann Dawe, Water Quality Research Australia, SA

Joe Lane, The University of Queensland, Qld
 John Gardner, CSIRO, Qld
 Julia Playford, Department of Science, Information Technology, Innovation and the Arts, Qld
 Julien Reungoat, The University of Queensland, Qld
 Karen Pither, Office of the Water Supply Regulator, Qld
 Karl King, Allconnex, Qld
 Katrin Doederer, The University of Queensland, QLD
 Kelly Fielding, The University of Queensland, Qld
 Kristell Le Corre, The University of Queensland, Qld
 Laura Roff, GHD, Qld
 Lee Williamson, Allconnex, Qld
 Leif Wolf, CSIRO, Qld
 Lucy Peljo, BMT WBM Pty Ltd, Qld
 Macpherson Nthara, International Water Centre, Qld
 Magnus Moglia, CSIRO, Qld
 Maria José Farré, The University of Queensland, Qld
 Mariel Forrester, Department of Energy and Water, Qld
 Mark Askins, Queensland Water Commission, Qld
 Mark Christl, Brisbane City Council, Qld
 Mark Quayle, Irrigation Australia Ltd, Qld
 Mark Wilson, Gold Coast Water, Qld
 Matthew Bower, Urban Utilities, Qld
 Matthew Ingerman, Allconnex, Qld
 Matthew Walsh, Cardno, Qld
 Max Yui, GHD, Qld
 Maxime Rattier, The University of Queensland, Qld
 Melissa Green, CSIRO, QLD
 Meng Chong, CSIRO, Qld
 Michael Bartkow, Seqwater, Qld
 Michael Hopewell, Allconnex, Qld
 Michael Stevens, Aquiba Pty Ltd, Qld
 Michele Akeroyd, Goyder Institute, SA
 Mick Hartcher, CSIRO, Qld
 Mike Taylor, Seqwater, QLD
 Mohammad Katouli, University of the Sunshine Coast, Qld
 Murray Hall, CSIRO, Qld
 Nancy Esler, Department of Natural Resources and Mines, Qld
 Naomi Soustal, Healthy Waterways, Qld
 Neil Tindale, University of the Sunshine Coast, Qld
 Neil Tripodi, Department of Science, Information Technology, Innovation and the Arts, QLD
 Neranjala Fernando, Queensland Water Commission, Qld
 Nicholas Stuckey, University of Southern Queensland, QLD
 Nicole Davis, GHD, Qld
 Nicole Knight, Griffith University, Qld
 Nicole Masters, University of the Sunshine Coast, Qld
 Nigel Garson, Office of the Water Supply Regulator, Qld
 Pam Pittaway, University of Southern Queensland, Qld
 Pat Bourke, Brisbane City Council, Qld
 Paul Burrell, SEQ Water Grid Manager, Qld
 Paul Sherman, Queensland Urban Utilities, QLD
 Peta Neale, The University of Queensland, Qld
 Peter Dillon, CSIRO, Qld
 Peter Kuras, Brisbane City Council, Qld
 Peter Nardi, Queensland Water Commission, Qld
 Pushpa Onta, Department of Natural Resources and Mines, Qld
 Raed Ahmed Mahmood, The University of Southern Queensland, Qld
 Ram Gurung, CSIRO, QLD
 Raquel Esteban, Office of the Water Supply Regulator, Qld
 Regina Souter, International Water Centre, Qld
 Rodney Stewart, Griffith University, Qld
 Roger O'Halloran, CSIRO, Vic
 Rolf Rose, Queensland Water Commission, QLD
 Rupak Aryal, The University of Queensland, Qld
 Ryan McGloin, The University of Queensland, Qld
 Sanja Oldridge, Sprout Water Management, QLD
 Santosh Aryal, CSIRO, ACT
 Sarah Haydon, Australian Water Recycling Centre of Excellence, Qld
 Sarah Smith, Office of the Water Supply Regulator, Qld
 Saskya Hunter, GHD, Qld
 Scott Eade, Allconnex, Qld
 Shelley Luxton, Department of Natural Resources and Mines, QLD

Shiroma Maheepala, CSIRO, Vic
Shivanita Umapathi, CSIRO, Qld
Simon Catzikiris, Office of the Water Supply Regulator, Qld
Simon Cowland-Cooper, Irrigation Australia Ltd, QLD
Simon Taylor, GHD, Qld
Simon Toze, CSIRO, Qld
Sriya Fernando, Queensland Urban Utilities, Qld
Stephanie Ashbolt, CSIRO, Vic
Stephen Fisher, Department of Environment and Heritage Protection, Qld
Steve Spencer, CSIRO, Qld
Steven Cloete, The University of Queensland, Qld
Stewart Burn, CSIRO, Vic
Sue Larsen, Department of Energy and Water Supply, Qld
Tad Bagdon, Queensland Water Commission, Qld
Tamara Stevenson, Allconnex, Qld
Tan Trieu, Urban Utilities, Qld
Tasleem Hasan, Office of the Water Supply Regulator, Qld
Tim Bechenham, Office of the Water Supply Regulator, Qld
Tonia Scholes, Healthy Waterways, Qld
Tony Hall, Griffith University, Qld
Tony Purdon, Banana Shire Council, Qld
Trevor Lloyd, Lloyd Consulting, Qld
Vanessa Hutton, Office of the Water Supply Regulator, Qld
Vlad Matveev, CSIRO, Qld
Vourn Lutton, Queensland Water Commission, Qld
Wade Manuel, LinkWater, Qld
Wai-Tong Wong, Queensland Water Commission, Qld
Wenju Cai, CSIRO, Vic
Wolfgang Gernjak, The University of Queensland, Qld
Yvonne Xu, International Water Centre, 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

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