Assessing Hydrological Interception

BY

Plantation Forestry

FOR

Application

IN

Water Resources Management

A. J. GREENWOOD

Submitted in fulfillment of the requirements of the degree of Doctor of Philosophy

13 February 2017
ANNO DOMINI NOSTRI IESU CHRISTI

IDIBUS FEBRUARIAS MMDCCCLXX
ANNO URBIS CONDITAE
Abstract

IN the early to mid-1990s, Australia governments adopted significant forestry and water policy agendas. The forest policy stimulated plantation expansion, and articulated benefits to water resource degradation which became a focus of the water agenda. The prospect of changing water availability during a severe drought resulted in the National Water Initiative (NWI), which sought to protect the integrity of water entitlements from plantation expansion.

State government agendas provided additional complexity, notably in South Australia where forest water use became subject to regional regulation in 2004, reminiscent of South African experiences in 1972. Inconsistencies between assessments used to support/contest the sustainability of plantation developments resulted in the amendment of South Australian planning frameworks, to ensure competing policy issues were addressed in a balanced manner.

Mixed progress in implementing the interception clauses of the NWI have been relegated to unfinished business without critically evaluating its capacity to deliver the required policy and scientific outcomes. Here, national and jurisdictional forest water policies are analysed and weaknesses identified in the lack of a cohesive national policy agenda arising from jurisdictional independence under the Australian Constitution. Inefficiencies in implementation are traced to: competing agendas; the complexities of their inter/intra-jurisdictional administration; a lack of regard to relevant international precedents and a tendency for Australian water reform to be initiated as short-term responses to predictable disasters rather than long-term planning.

Reforms under the NWI are found to have had little direct effect in progressing jurisdictional forest water management agendas.

Australian empirical hydrological assessment approaches used to support forest water decision-making are examined in the context of international systems and learnings from South Africa. An approach for evaluating and transparently integrating Australia’s limited forest hydrology datasets with modelled information is developed to improve confidence in decision-making.

A similar group of empirically-based models are subjected to a comprehensive Bayesian evaluation with South African and American approaches to identify an option with the greatest integrity to underpin current Australian forest water management. Systemic limitations in a widely used approach developed by a leading Australian research organisation are confirmed and revealed as being compounded by weak model structure, highlighting the challenges faced by water management agencies in securing research to support defensible decision-making.
Challenges associated with agency capacity limitations and the inevitability of using more complex modelling in supporting future Australian forest water management are addressed by noting South African learnings which identify the importance of growth in plantation water use; and exploring the feasibility of using simpler elements of a sophisticated Bayesian assessment to establish confidence in a plant growth model. Complexity introduced into the 3-PG plant process model is shown to improve the model’s ability to extract information from data providing greater confidence in its potential for future development as a water management tool than more exhaustive, integrated assessments focused on marginal improvements in performance.

Results are discussed in the context of water management and their implications for future research and policy developments.
Declaration

I hereby certify that: this thesis comprises only my original work towards the PhD except where indicated in the Preface; due acknowledgement has been made in the text to all other material used, and, the thesis is fewer than 100,000 words in length, exclusive of tables, maps, bibliographies and appendices.

Ashley Greenwood
August 2016
Preface

IN accordance with university requirements I would like to preface this thesis by acknowledging that Sections 5 and 6 were completed with support and advice provided by co-authors whose contributions are consistent with the work being original and my own as declared in the Contribution to Published Work Forms section below.

Specific material contributions were made by Dr Gerrit Schoups who provided the Matlab version of the generalised likelihood that I translated into R; and Dr Eddy Campbell who provided an example of an adaptive Metropolis algorithm which I modified and incorporated into my own code used in Section 6.

AJBG
August 2016
Contribution to Published Work Forms
Declaration for a thesis with publication

PhD and MPhil students may include a primary research publication in their thesis in lieu of a chapter if:

- The student contributed greater than 50% of the content in the publication and is the “primary author”, i.e., the student was responsible primarily for the planning, execution and preparation of the work for publication
- It has been peer-reviewed and accepted for publication
- The student has approval to include the publication in their thesis from their Advisory Committee
- It is a primary publication that reports on original research conducted by the student during their enrolment
- The initial draft of the work was written by the student and any subsequent editing in response to co-authors and editors reviews was performed by the student
- The publication is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in the thesis

Students must submit this form, along with Co-author authorisation forms completed by each co-author, when the thesis is submitted to the Thesis Examination System: [https://tes.app.unimelb.edu.au/](https://tes.app.unimelb.edu.au/). If you are including multiple publications in your thesis you will need to complete a separate form for each publication. Further information on this policy is available at: gradresearch.unimelb.edu.au/preparing-my-thesis/thesis-with-publication

A. PUBLICATION DETAILS (to be completed by the student)

<table>
<thead>
<tr>
<th>Full title</th>
<th>A method for assessing the hydrological impact of afforestation using regional mean annual data and empirical rainfall–runoff curves.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Authors</td>
<td>Greenwood, A.J.B., Benyon, R.G., Lane, P.N.J.</td>
</tr>
<tr>
<td>Student’s contribution (%)</td>
<td>&gt;95%</td>
</tr>
<tr>
<td>Journal or book name</td>
<td>Journal of Hydrology</td>
</tr>
<tr>
<td>Volume/page numbers</td>
<td>411: 49-65</td>
</tr>
<tr>
<td>Status</td>
<td>[ ] Accepted and In press   [x] Published  Date accepted/ published 2011</td>
</tr>
</tbody>
</table>

B. STUDENT’S DECLARATION

I declare that the publication above meets the requirements to be included in the thesis

<table>
<thead>
<tr>
<th>Student’s name</th>
<th>Student’s signature</th>
<th>Date (dd/mm/yy)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ashley Greenwood</td>
<td></td>
<td>10/05/2016</td>
</tr>
</tbody>
</table>

C. PRINCIPAL SUPERVISOR’S DECLARATION

I declare that:

- the information above is accurate
- The advisory committee has met and agreed to the inclusion of this publication in the student’s thesis
- All of the co-authors of the publication have reviewed the above information and have agreed to its veracity
- ‘Co-Author Authorisation’ forms for each co-author are attached.

<table>
<thead>
<tr>
<th>Supervisor’s name</th>
<th>Supervisor’s signature</th>
<th>Date (dd/mm/yy)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dr Patrick Lane</td>
<td></td>
<td>10/05/2016</td>
</tr>
</tbody>
</table>
Declaration for a thesis with publication

PHD and MPhil students may include a primary research publication in their thesis in lieu of a chapter if:

- The student contributed greater than 50% of the content in the publication and is the “primary author”, i.e. the student was responsible primarily for the planning, execution and preparation of the work for publication.
- It has been peer-reviewed and accepted for publication.
- The student has approval to include the publication in their thesis from their Advisory Committee.
- It is a primary publication that reports on original research conducted by the student during their enrolment.
- The initial draft of the work was written by the student and any subsequent editing in response to co-authors and editors was performed by the student.
- The publication is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in the thesis.

Students must submit this form, along with Co-author authorisation forms completed by each co-author, when the thesis is submitted to the Thesis Examination System: https://tes.app.unimelb.edu.au/. If you are including multiple publications in your thesis you will need to complete a separate form for each publication. Further information on this policy is available at: gradereach.unimelb.edu.au/submitting-your-thesis/thesis-with-publication.

### A. PUBLICATION DETAILS (to be completed by the student)

<table>
<thead>
<tr>
<th>Full title</th>
<th>The first stages of Australian forest water regulation: National reform and regional implementation.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Authors</td>
<td>Greenwood, A.J.B.</td>
</tr>
<tr>
<td>Student’s contribution (%)</td>
<td>100%</td>
</tr>
<tr>
<td>Journal or book name</td>
<td>Environmental Science &amp; Policy</td>
</tr>
<tr>
<td>Volume/page numbers</td>
<td>29: 124-136</td>
</tr>
<tr>
<td>Status</td>
<td>□ Accepted and in press  □ Published  □ Date accepted/ published 2013</td>
</tr>
</tbody>
</table>

### B. STUDENT’S DECLARATION

I declare that the publication above meets the requirements to be included in the thesis.

<table>
<thead>
<tr>
<th>Student’s name</th>
<th>Student’s signature</th>
<th>Date (dd/mm/yy)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ashley Greenwood</td>
<td></td>
<td>10/05/2016</td>
</tr>
</tbody>
</table>

### C. PRINCIPAL SUPERVISOR’S DECLARATION

I declare that:

- the information above is accurate
- The advisory committee has met and agreed to the inclusion of this publication in the student’s thesis
- All of the co-authors of the publication have reviewed the above information and have agreed to its veracity
- ‘Co-Author Authorisation’ forms for each co-author are attached.

<table>
<thead>
<tr>
<th>Supervisor’s name</th>
<th>Supervisor’s signature</th>
<th>Date (dd/mm/yy)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dr Patrick Lane</td>
<td></td>
<td>10/05/2016</td>
</tr>
</tbody>
</table>

The University of Melbourne
CRICOS Provider Number: 00116K

Last Updated 17 August 2015
Declaration for a thesis with publication

PhD and MPhil students may include a primary research publication in their thesis in lieu of a chapter if:

- The student contributed greater than 50% of the content in the publication and is the “primary author”, i.e., the student was responsible primarily for the planning, execution and preparation of the work for publication
- It has been peer-reviewed and accepted for publication
- The student has approval to include the publication in their thesis from their Advisory Committee
- It is a primary publication that reports on original research conducted by the student during their enrolment
- The initial draft of the work was written by the student and any subsequent editing in response to co-authors and editors reviews was performed by the student
- The publication is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in the thesis

Students must submit this form, along with Co-author authorisation forms completed by each co-author, when the thesis is submitted to the Thesis Examination System: https://tes.app.unimelb.edu.au/. If you are including multiple publications in your thesis you will need to complete a separate form for each publication. Further information on this policy is available at: gradresearch.unimelb.edu.au/preparing-my-thesis/thesis-with-publication

A. PUBLICATION DETAILS (to be completed by the student)

| Full title | Bayesian scrutiny of simple rainfall-runoff models used in forest water management. |
| Authors | Greenwood, A.J.B., Schoups, G., Campbell, E.P., Lane, P.N.J. |
| Student’s contribution (%) | >90% |
| Journal or book name | Journal of Hydrology |
| Volume/page numbers | 512: 344-365 |
| Status | Accepted and In press Published Date accepted/ published 2014 |

B. STUDENT’S DECLARATION

I declare that the publication above meets the requirements to be included in the thesis

Student’s name | Student’s signature | Date (dd/mm/yy)
Ashley Greenwood | | 10/05/2016

C. PRINCIPAL SUPERVISOR’S DECLARATION

I declare that:
- the information above is accurate
- The advisory committee has met and agreed to the inclusion of this publication in the student’s thesis
- All of the co-authors of the publication have reviewed the above information and have agreed to its veracity
- ‘Co-Author Authorisation’ forms for each co-author are attached.

Supervisor’s name | Supervisor’s signature | Date (dd/mm/yy)
Dr Patrick Lane | | 10/05/2016
Co-author authorisation form

All co-authors must complete this form. By signing below co-authors agree to the listed publication being included in the student’s thesis and that the student contributed greater than 50% of the content of the publication and is the “primary author” ie. the student was responsible primarily for the planning, execution and preparation of the work for publication.

In cases where all members of a large consortium are listed as authors of a publication, only those that actively collaborated with the student on material contained within the thesis should complete this form. This form is to be used in conjunction with the Declaration for a thesis with publication form.

Students must submit this form, along with the Declaration for thesis with publication form, when the thesis is submitted to the Thesis Examination System: https://tes.app.unimelb.edu.au/

Further information on this policy and the requirements is available at: gradereview.unimelb.edu.au/preparing-my-thesis/thesis-with-publication

### A. PUBLICATION DETAILS (to be completed by the student)

| Full title | A method for assessing the hydrological impact of afforestation using regional mean annual data and empirical rainfall–runoff curves. |
| Authors | Greenwood, A.J.B., Benyon, R.G., Lane, P.N.J. |
| Student’s contribution (%) | >90% |
| Journal or book name | Journal of Hydrology |
| Volume/page numbers | 411: 49-65 |
| Status | □ Accepted and In-press  ☑ Published  Date accepted/published 2011 |

### B. CO-AUTHOR’S DECLARATION (to be completed by the collaborator)

I authorise the inclusion of this publication in the student’s thesis and certify that:

- the declaration made by the student on the Declaration for a thesis with publication form correctly reflects the extent of the student’s contribution to this work;
- the student contributed greater than 50% of the content of the publication and is the “primary author” ie. the student was responsible primarily for the planning, execution and preparation of the work for publication.

<table>
<thead>
<tr>
<th>Co-author’s name</th>
<th>Co-author’s signature</th>
<th>Date (dd/mm/yy)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dr Richard Benyon</td>
<td>[Signature]</td>
<td>11/05/2016</td>
</tr>
<tr>
<td>Dr Patrick Lane</td>
<td>[Signature]</td>
<td>10/05/2016</td>
</tr>
</tbody>
</table>
Co-author authorisation form

All co-authors must complete this form. By signing below co-authors agree to the listed publication being included in the student’s thesis and that the student contributed greater than 50% of the content of the publication and is the “primary author” i.e. the student was responsible primarily for the planning, execution and preparation of the work for publication.

In cases where all members of a large consortium are listed as authors of a publication, only those that actively collaborated with the student on material contained within the thesis should complete this form. This form is to be used in conjunction with the Declaration for a thesis with publication form.

Students must submit this form, along with the Declaration for thesis with publication form, when the thesis is submitted to the Thesis Examination System: https://tes.app.unimelb.edu.au/

Further information on this policy and the requirements is available at: graderesearch.unimelb.edu.au/preparing-my-thesis/thesis-with-publication

A. PUBLICATION DETAILS (to be completed by the student)

<table>
<thead>
<tr>
<th>Full title</th>
<th>Bayesian scrutiny of simple rainfall-runoff models used in forest water management.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Authors</td>
<td>Greenwood, A.J.B., Schoups, G., Campbell, E.P., Lane, P.N.J.</td>
</tr>
<tr>
<td>Student’s contribution (%)</td>
<td>&gt;90%</td>
</tr>
<tr>
<td>Journal or book name</td>
<td>Journal of Hydrology</td>
</tr>
<tr>
<td>Volume/page numbers</td>
<td>512: 344-365</td>
</tr>
<tr>
<td>Status</td>
<td>Accepted and In-press ✔ Published Date accepted/published 2014</td>
</tr>
</tbody>
</table>

B. CO-AUTHOR’S DECLARATION (to be completed by the collaborator)

I authorise the inclusion of this publication in the student’s thesis and certify that:

- the declaration made by the student on the Declaration for a thesis with publication form correctly reflects the extent of the student’s contribution to this work;
- the student contributed greater than 50% of the content of the publication and is the “primary author” i.e. the student was responsible primarily for the planning, execution and preparation of the work for publication.

<table>
<thead>
<tr>
<th>Co-author’s name</th>
<th>Co-author’s signature</th>
<th>Date (dd/mm/yy)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dr Gerrit Schoups</td>
<td></td>
<td>12/05/2016</td>
</tr>
<tr>
<td>Dr Eddy Campbell</td>
<td></td>
<td>18/05/2016</td>
</tr>
<tr>
<td>Dr Patrick Lane</td>
<td></td>
<td>10/05/2016</td>
</tr>
</tbody>
</table>
Acknowledgements

MY sincere thanks and apologies go to Louise and William and Freya and Maxwell. I fear that if I had my time again, I'd have only followed the same path.

Special thanks are owed to people who provided support at important junctures of this work including: Dr Francesco Minunno; Dr Joseph Guillaume; Dr Geritt Schoups; Dr Eddy Campbell; Dr Luke Esprey and my supervisors Dr Craig Beverley, Dr Richard Benyon, Dr Patrick Lane and two anonymous reviewers for their comments and advice.

This project actually commenced in 2007. Colleagues who played a critical part in developing the scientific basis of South Australia’s state-wide forest water policy and helped me start on this path include: Mr David Cresswell, Mr Jason van Laarhoven, Dr David Deane, Dr Theresa Heneker, Mr Darryl Harvey, Mr Steve Barnett and The Very Reverend Glen Scholz.

I feel that special mention should go to the remaining elders whose contribution to my life and our society have resulted in the opportunity to undertake this work and will provide the foundation for those of future generations, specifically: Mrs Elsie Redfern née Parkes, Mr William H Brightman, Mr David B Graham and not least Mrs Marjorie Graham née Greenwood née Redfern.

I would also like to acknowledge the contribution of the McBurneys: Neil, Alice, Tom and particularly Robyn Lea McBurney née Greenwood who had far more to do with this than she knows.

Having done so I feel I ought to also make mention of the late Mr Ernest John Redfern, Mrs Elsie Irene Greenwood née Hunter and even Mr Mervyn Harrison Greenwood, all of whom played their unknowing part in this little adventure.

Finally, I would like to acknowledge the loving memory of Mrs Lilian Brightman née Wilke who might’ve wondered why; and the late Mr Ian Maxwell Greenwood who had no doubt it was all a waste of time. Fair enough.

AJBG
née nay

August 2016,
February 2017
Fullarton, Adelaide
# Contents

1. INTRODUCTION ........................................................................................................ 1-1

## PART I
### FOREST WATER REGULATION

2. AN INTERNATIONAL PERSPECTIVE ........................................................................ 2-1
   2.1 South Africa ............................................................................................................. 2-2
       2.1.1 The Afforestation Permit System ...................................................................... 2-3
       2.1.2 Afforestation as a licensed streamflow reduction activity .............................. 2-6
   2.2 Discussion .................................................................................................................. 2-8

3. AN AUSTRALIAN PERSPECTIVE .............................................................................. 3-1
   3.1 The National Water Initiative .................................................................................. 3-2
       3.1.1 National Water Commission .............................................................................. 3-4
       3.1.2 Unfinished business .......................................................................................... 3-5
   3.2 Other Inter-jurisdictional Initiatives ........................................................................ 3-6
   3.3 Jurisdictional Initiatives ............................................................................................ 3-7
       3.3.1 South Australia .................................................................................................. 3-8
       3.3.2 Victoria ............................................................................................................. 3-10
       3.3.3 Western Australia ............................................................................................... 3-12
       3.3.4 Tasmania ........................................................................................................... 3-14
       3.3.5 New South Wales ............................................................................................... 3-15
       3.3.6 Other Jurisdictions ............................................................................................. 3-16
   3.4 Jurisdictional Capacity to Support Policy Implementation’ .................................... 3-16
       3.4.1 Survey .................................................................................................................. 3-18
       3.4.2 Results ............................................................................................................... 3-20
       3.4.3 Discussion ......................................................................................................... 3-22
   3.5 Discussion .................................................................................................................. 3-22
PART II
FOREST HYDROLOGY AND
WATER RESOURCE ASSESSMENT FOR
FOREST WATER MANAGEMENT

4. HYDROLOGICAL MODELLING FOR FOREST WATER MANAGEMENT
........................................................................................................................................... 4-1
4.1 An Overview of Empirical and Mechanistic Approaches.................. 4-1
  4.1.1 Some historical context................................................................. 4-3
4.2 North America................................................................................. 4-5
4.3 South Africa.................................................................................... 4-6
  4.3.1 Empirical approaches ................................................................. 4-6
  4.3.2 Mechanistic approaches............................................................. 4-12
4.4 Australia......................................................................................... 4-15
  4.4.1 Empirical approaches ................................................................. 4-15
  4.4.2 Mechanistic approaches............................................................. 4-19
  4.4.3 Jurisdictional approaches ........................................................... 4-26
4.5 Discussion....................................................................................... 4-28

5. CONFRONTING EMPIRICAL LIMITATIONS...................................... 5-1

6. A BAYESIAN APPROACH TO MODEL SCRUINITY.......................... 6-1

7. USE OF A PLANT PROCESS MODEL IN WATER MANAGEMENT ... 7-1
  7.1 An Overview of 3-PG .................................................................... 7-2
    7.1.1 Turning light into carbon ......................................................... 7-2
    7.1.2 Canopy processes ................................................................. 7-5
    7.1.3 Soil water balance ................................................................. 7-7
    7.1.4 Site quality and the fertility rating .......................................... 7-7
    7.1.5 Recognised shortcomings..................................................... 7-8
  7.2 3-PG+ in CAT................................................................................. 7-9
    7.2.1 Developments ........................................................................ 7-10
  7.3 Analysis......................................................................................... 7-13
  7.4 Data............................................................................................... 7-16
  7.5 Results.......................................................................................... 7-20
  7.6 Discussion..................................................................................... 7-33

8. DISCUSSION.................................................................................... 8-1

9. CONCLUSIONS.............................................................................. 9-1
10. REFERENCES.................................................................................. 10-2

APPENDICES
Tables

**Table 3.1:** Survey questions (Appendix B)................................................................. 3-19

**Table 4.1:** Plantation impact assessment approaches used in water accounting, management or to inform recent policy development. Queensland and Australian Capital Territory did not consider risks significant. .............................................. 4-27

**Table 7.1:** Sites used in the study, all planted with *E. globulus*.................. 7-16
**Table 7.2:** Data summary. Methods are described in Benyon and Doody (2004); Benyon et al. (2006; 2008)................................................................. 7-17
**Table 7.3:** Site data, soil properties and related model parameters. ............... 7-18
**Table 7.4:** PERFECT soil properties................................................................. 7-18
**Table 7.5:** *E. globulus* species parameters after Feikema et al. (2010); for additional information see their Appendix A and Table A1). **Emboldened** asterisked parameters (*) also shown with prior probability ranges................................................................. 7-19
**Table 7.6:** Run times ................................................................................. 7-20
**Table 7.7:** Model and statistical parameters, ~reasonable expectation, prior ranges, and 95% confidence limits; *lwr* = 2.5%, *upp* = 97.5%.................................................................. 7-31
Figures

Figure 2.1: South African forest research sites, after Scott et al. (2000), map 1. .......2-3

Figure 3.1: Responses to survey Questions 1 to 8. Red histograms repeated through panels (a)-(h) are the responses to Question 2, grey histograms are responses to other questions (see Table 3.1 and text). .................................................................3-21

Figure 3.2: Responses to survey Question 9. Use of tools ..................................3-21

Figure 4.1: USDA-NRCS (2004) curve numbers (Figure 10.2, from Chapter 10 Estimation of direct runoff from storm rainfall). .................................................................4-5

Figure 4.2: (a) Nänni (1970b, Figure 1) curves, from Schulze and George (1987, Figure 4), overlain with Van der Zel (1982) runoff- runoff reduction curves in red (from Figure 1b) for comparison. (b) Van der Zel (1982, Figure 1) curves for 15- (left) and 40- year (right) rotations, taken from Van der Zel (1997, Figure 3). ........................................4-7

Figure 4.3: CSIR/Scott and Smith (1997, Figure 4) age-runoff reduction curves for eucalypts. Curves are differentiated with regard to total annual streamflow and low flows under optimal and sub-optimal growing conditions related to rainfall and temperature. Similar curves were developed for pines (see Scott and Smith 1997, Figure 3). .................................................................4-10

Figure 4.4: Evaporation-rainfall curves for (Holmes and Sinclair 1986, Figure 3). ....4-15

Figure 4.5: (a) Zhang et al. (2001, Figure 9) rainfall-evapotranspiration curves. (b) Zhang rainfall-runoff curves taken from Zhang et al. (2003, Figure 6). .................4-16

Figure 4.6: Figure 6 from Zhang et al. (2003) (see Figure 4.5(b) above), reproduced with the Australian data of Zhang et al. (1999). Data below 1000 mm/year are not segregated by land-cover according to the hypothesis inherent in the curves........4-18

Figure 4.7: Greenwood and Cresswell curves. (a) pre- and post forest rainfall-runoff curves (Greenwood and Cresswell 2007, Figure 6). (b) age-runoff reduction curves (Greenwood and Cresswell 2007, Figure 8). Red and blue curves are from Scott and Smith (1997, Figure 3). .................................................................4-18
Figure 7.1: Conceptual approach used to evaluate the integrity of improvements to 3-PG soil water modelling made under 3-PG+. (a) conceptual plantation with selected species parameters that control processes related to radiation interception ($aC$ quantum canopy efficiency), transpiration ($gC_x$ maximum canopy conductance) and growth ($FR$ fertility rating); (b) stylised depiction of model change from simple soil water store to a multi-layered one; (c) stylised improvement in model parameter identification in the marginal and (d) joint posterior probability distributions associated with the change.

Figure 7.2: Location of sites used in the study. Dark green areas are pine plantations, eucalypt plantations are grey-green and difficult to distinguish among the mosaic of remnant native vegetation. Indications of the wider distribution of plantation types may be found in Benyon et al. (2008, Figure 2); Harvey (2009, Figure 1).

Figure 7.3: Gelman shrink factor for 3-PG model parameters $aC$, $FR$ and $gC_{max}$ for both the original implementation of 3-PG (a-i) and 3-PG+ (j-r) at each site.

Figure 7.4: Gelman shrink factor for statistical parameters of the generalised likelihood for both the original implementation of 3-PG (a-o) and 3-PG+ (p-ad) at each site.

Figure 7.5: DPI – Markov chains 3-PG (a-c) and marginal posterior distributions (d-e); Markov chains 3-PG+ (g-i) and marginal posterior distributions (j-l).

Figure 7.6: Jill – Markov chains 3-PG (a-c) and marginal posterior distributions (d-e); Markov chains 3-PG+ (g-i) and marginal posterior distributions (j-l).

Figure 7.7: Vicki – Markov chains 3-PG (a-c) and marginal posterior distributions (d-e); Markov chains 3-PG+ (g-i) and marginal posterior distributions (j-l).

Figure 7.8: Marginal posterior distributions for 3-PG - DPI (a-e); Jill (f-j); Vicki (k-o) and 3-PG+ statistical parameters - DPI (p-t); Jill (u-y); Vicki (z-ad).

Figure 8.1: South African age-evapotranspiration curves derived from observed data from seven sites from Jarmain et al. (2009) after Dye and Bosch (2000), with mean annual South Australian data from Greenwood and Cressell (2007), where $RR = \text{runoff reduction}$; $MP/R/E/\Delta S = \text{mean annual precipitation/runoff/evapotranspiration/change in storage}$, presented by Greenwood (2012). Maximum impacts manifest at five years in half of the cases (purple line).
Appendices

APPENDIX A: EXTRACTS FROM THE NATIONAL WATER INITIATIVE

APPENDIX B: SURVEY

APPENDIX C: SECTION 3.3.1

APPENDIX D: SECTION 5

APPENDIX E: SECTION 6

APPENDIX F: OTHER PUBLICATIONS

APPENDIX G: PRESENTATIONS GIVEN DURING THE PROJECT
1. INTRODUCTION

“NASA was transformed from a research and development agency to more of a business, with schedules, production pressures, deadlines, and cost efficiency goals elevated to the level of technical innovation and safety goals.”

The top levels of NASA soon began to be occupied by business managers instead of technical engineers. ...

Managers at NASA deliberately took a risk. They believed the risk was quite low or zero, but they had not even done the calculations to know how big the risk was.


AROUND 300 BC the Greek natural philosopher and father of modern botany Theophrastus observed that the environment influenced tree growth (Theophrastus, V.II; V.VIII), some 200 years later Roman authors described forest influences on the environment. Vitruvius (first century BC) noted that trees mitigated evaporation losses by intercepting solar radiation (Vitruvius, VIII, 6); while Pliny the Elder (first century AD) described rising water tables and increases in runoff as a consequence of forest felling, the water being previously expended in the nutriment of the trees (Pliny the Elder, XXXI, 30). However, it was not until the mid-nineteenth century that the influences of forests on the environment received systematic scientific attention, literally as forest influences (Fernow, 1893; Harrington, 1893).

The study of forest influences loosely encompassed hydroclimatic, physical and socio-economic effects of forests (Pinchot, 1905; Schwappach, 1904) becoming more closely associated with purely environmental influences toward the middle of the twentieth century (Kittredge, 1948; Pavari, 1959; Wicht, 1939; Wilm, 1957; Zon, 1927). The term forest hydrology, the study of the hydrological cycle concerned with forests and forestry was first coined in the late 1940s by Kittredge (1948), but did not emerge as a distinct scientific discipline until the 1960s, marked by the first international conference held in 1965 (Sopper and Lull, 1967).

Early studies in forest hydrology comprised paired catchment experiments which were designed to empirically investigate the impacts of different forest treatments on streamflow compared to an undisturbed or paired control catchment. The earliest of these studies was established in Emmental, Switzerland at the turn of the twentieth century (Engler 1919, in Badoux et al., 2006; Fernow, 1893; McCulloch and Robinson,
INTRODUCTION

1993). Similar investigations commenced shortly thereafter in Colorado by the United States Department of Agriculture (Bates and Henry, 1928) and then in Jonkershoek by the South African government in the mid-1930s (Wicht, 1939).

Swiss and US studies shared similar objectives investigating streamflow increases associated with forest clearing (Badoux et al., 2006; Bates and Henry, 1928) and assessing vegetation controls for flood management and soil conservation (Colman, 1953). In contrast, South African studies were specifically concerned with addressing reductions in water availability associated with afforestation, the intentional establishment of any tree-dominated vegetation on land that was formerly not tree-dominated (FAO, 2005; Van Dijk and Keenan, 2007).

Commercial afforestation had commenced in South Africa in 1876 and by 1915 farmers began to complain about decreases in downstream water availability (Van der Zel, 1995). Paired study sites were first established in 1935 (Wicht, 1939), followed by others set up to examine the influence of different physio-climatic conditions and treatments on tree growth and catchment hydrology (Scott et al., 2000). Research focused on management applications and was eventually used to support the regulation of commercial forestry water use in the early 1970s (Van der Zel, 1982; 1995).

Reflecting on the South African experience, Dye and Versfeld (2007) concluded that regulation was likely to emerge in other countries that encouraged plantation forestry in areas with limited water availability. The sagacity of these views proved germane to Australia which became only the second country in the world to embark on the regulation of plantation forest water use, in accordance with the National Water Initiative (NWI) (COAG, 2004b).

During the late 1990s Australian national forest policy (AG, 1997b) and tax incentives (Leys and Vanclay, 2010) encouraged plantation forest to expand into drier regions (Keenan et al., 2004; Van Dijk et al., 2004). When drought and natural disaster began to threaten the shared water resources of the inter-jurisdictional Murray-Darling Basin (Earth Tech Engineering, 2003; Van Dijk et al., 2006), large-scale plantation forestry became classified as an interception activity under the NWI, an activity with the potential to intercept significant volumes of surface and/or groundwater and impact the integrity of water access entitlements (COAG, 2004b, s 55 - 57).

Interception in the context of Australian water management policy then encompasses any of a range of hydrological processes induced by anthropogenic activities which
result in water being intercepted as it moves from the atmosphere through the landscape to a stream or aquifer. These processes which include infiltration, diversion, storage and evapotranspiration are distinct from the hydrological use of interception, which is used to describe water detained by a canopy or even the plant processes associated with solar radiation incident on it.

At face value the NWI provided a framework to resolve competition between large-scale plantation developments and downstream water users, however national progress has been mixed. South Australia remains the only state of Australia to regulate plantation water use and did so before the NWI was signed (COAG, 2004b; GSA, 2004). In its final (triennial) assessment the National Water Commission (NWC) noted that no state or territory had fully implemented interception arrangements that meet the requirements of the NWI (NWC, 2014a) and significant unfinished business remained (SCEW, 2013). Establishing a clear perspective on the efficacy of the NWI with regard to plantation interception in the different jurisdictions is fundamental to its future progress.

Like the first stages of South African regulation (Scott and Smith, 1997; Van der Zel, 1995), current South Australian forest water management is underpinned by empirically-based hydrological assessments (Greenwood, 2007; Greenwood and Cresswell, 2007; GSA, 2009; Harvey, 2009). The South African experience saw the integrity of empirical relationships challenged because they were not fully documented; only addressed long term average annual streamflow responses; and were not sufficiently flexible to accommodate the needs of land and forest managers such as the effects of different site influences on water use (Bosch, 1982; Bosch and Von Gadow, 1990; Dye and Bosch, 2000). The development of more sophisticated relationships to depict a wider range of conditions and species (Scott and Smith, 1997) did not completely quiet criticism which included persistent uncertainties in extrapolating empirical data to unmonitored sites; impacts on low flows and a sense of forestry being unfairly singled out amongst other dry land activities that used similar volumes of water (Dye, 2013; Dye and Versfeld, 2007; Versfeld, 1996).

Unlike Australia, South Africa progressed to using mechanistic modelling in supporting forest water management (Gush et al., 2002b). While similarly sophisticated modelling was developed by Australian forest hydrologists (Dawes et al., 1997; Hatton et al., 1992; Moore et al., 1986; 1988; O’Loughlin, 1986; 1990; Vertessy et al., 1993; Zhang et al., 1996) and was recognised as superior to empirical approaches (Vertessy et al., 1996), empirical methods have remained the preferred basis of research assessments.
(Bradford et al., 2001; Vertessy and Bessard, 1999; Vertessy et al., 2000; Watson et al., 1999; Zhang et al., 2003b) and management tools (Brown et al., 2006a; 2006b; DPIW, 2008; Greenwood, 2007; Greenwood and Cresswell, 2007; Harvey, 2009; Prosser and Walker, 2009) and continue to be used in significant national initiatives (Post et al., 2012), most recently to calculate volumetric water offsets for use in carbon farming (AG, 2015; Roberts, 2014).

Despite their persistent use, Australian empirical forest water assessment tools have received little critical attention regarding their fitness for supporting water management and defensible decision-making. Similar to the South African experience unresolved issues include: inconsistencies between some established relationships (BRS, 2003) and local data (Greenwood, 2007); the lack of examination against competing alternatives (Clark et al., 2011; Gupta et al., 2012) and whether more can be done to improve the confidence in extrapolating empirical data and relationships more generally.

The trend toward using more complex computer models in natural resources management (Gupta et al., 2008; Hrachowitz et al., 2014; Loucks, 2013; Refsgaard et al., 2005; Sargent, 2013) suggests that more sophisticated modelling approaches will be required to meet the demands of Australian forest water management (Pittock, 2011; Pittock et al., 2013). Lessons from South Africa indicate the need to accommodate more flexible forest management practices; the dynamic role of growth in water use; the influences of different site conditions and the impacts of climate change (Dye, 2013; Dye et al., 2001; 2000; 2008b; Gush et al., 2002b; Le Maitre and Versfeld, 1997; Schulze, 2005; Warburton et al., 2010). After considering a number of alternatives (CPH Water, 2001), South Africa adopted the ACRU agro-hydrological modelling platform (Schulze and George, 1987) to quantify streamflow reductions resulting from commercial afforestation (DWAF, 2005; Gush et al., 2002a).

The plethora of computer models available to contemporary resource management makes choosing the most appropriate model for a specific task difficult (Todini, 2011). South African deliberations were set in the context of developing a decision-support system that incorporated modelling and data requirements; scenario generation; and design and implementation. Forest water modelling requirements were considered as a range of attributes that included conventional hydrological specifications such as the capacity to replicate water operations, applicability to local conditions and, transparency and credibility to support decision-making (CPH Water, 2001).
INTRODUCTION

Australia’s federated Constitution and separate jurisdictional agendas challenge the notion of selecting a uniform approach to support any national management initiative. The development of a national hydrological modelling platform to support the NWI (Farrell, 2012) meets many of the requirements of a national decision-support system (see South African discussion in CPH Water, 2001), however a suite of different approaches continue to be used in assessing the hydrological impacts of afforestation for water management across the country (see Section 4.4.3).

Identifying and promoting a single forest-water modelling approach for adoption across Australian inter-jurisdictional water management frameworks is beyond the scope of a single academic study. The complexities of such work is may be seen in the implementation of the National Water Knowledge and Research Platform (DSEWPC, 2012), while South African experience in refining part of an established forest water assessment platform required the combined efforts of a range of researchers and multiple post-graduate students (Jewitt et al., 2009).

As exhaustive model evaluation demands significant resources (ASCE, 1993; Bennett et al., 2013) which are often in limited supply within water management agencies (COAG, 2004b, s 98-101), this study follows a pragmatic but as curiously untrodden path in Australian forest water management by capitalising on South African experience to select a candidate model and scrutinise it in a way which deliberately augments both Australian and South African work and emphasises water management outcomes rather than technical performance alone.

A number of prospective Australian models have been identified by South African researchers as having potential for forest water management: IHACRES (Dye and Croke, 2003); Macaque (Roelofsen, 2002) and notably 3-PG (Landsberg and Waring, 1997; Sands and Landsberg, 2002). In contrast with IHACRES and Macaque, 3-PG’s potential to support forest water management was recognised in its capacity to mechanistically depict variations in plantation/stand water use according to growth patterns arising from site and climatic conditions (Campion et al., 2005; Dye, 2000a in Dye et al., 2001; Dye, 2001; Dye, 1999 in Dye and Bosch, 2000; Dye et al., 2004; Esprey et al., 2004; Gush, 1999 in Campion et al., 2005).

Australian researchers identified similar potential (Zhang et al., 2003b), particularly when trees were not able to access groundwater (Benyon et al., 2008). Moreover, refinements to the 3-PG soil water module have resulted in improved depictions of water use (Almeida and Sands, 2015; Feikema et al., 2010b). Studies aimed at
supporting forest water management also reported encouraging results but amid a high level of complexity (Feikema et al., 2008; Marcar et al., 2010).

Improved modelled output is not enough to inspire the confidence required by water management (Brugnach et al., 2007; Grayson et al., 1992; Jakeman and Letcher, 2003; Refsgaard and Henriksen, 2004; Sargent, 2013) which needs to balance socio-economic and environmental outcomes (COAG, 2004b, s 80; Crase et al., 2005), a feature of its **wicked** nature (Lach et al., 2005).

Quantifying uncertainty is a recognised way by which confidence can be established in water management and other wicked problems more generally (APSC, 2007; Botterill and Cockfield, 2013; Brugnach et al., 2008a; Head, 2010; Macleod et al., 2008; Maxim and van der Sluijs, 2011; Pahl-Wostl et al., 2011; Van der Sluijs et al., 2005). Many frameworks have been developed to characterise uncertainty (Matott et al., 2009), among them Bayesian approaches explicitly quantify uncertainty; the more information a model is able to extract from the data, the greater the confidence in parameter identification and the modes as a viable hypothesis (Kuczera, 1997; Kuczera and Mroczkowski, 1998; Kuczera and Parent, 1998; Mroczkowski et al., 1997). Consequently parameter identifiability offers an alternative to conventional model evaluation (Schoups et al., 2008) and a way to readily communicate model confidence to stakeholders.

In simpler terms recent improvements to 3-PG (Almeida and Sands, 2015; Feikema et al., 2010b) must unambiguously demonstrate that the right results have been obtained for the right reasons (Beven, 1989; 2006; Clark et al., 2011; Goswami and O'Connor, 2010; Gupta et al., 2012; Gupta and Sorooshian, 1983; Jakeman and Hornberger, 1993; Kirchner, 2006; Klemeš, 1982; Troutman, 1985).

Given the capacity constraints of water management agencies, uncertainty quantification will need to be rigorous but simpler than exhaustive approaches (Kavetski et al., 2006a; 2006b; Kuczera et al., 2006) suggested to support best-practice (Vaze et al., 2012).

If formal Bayesian approaches provide the required rigour (Clark et al., 2012; Stedinger et al., 2008) and the practicality of exhaustive approaches is questionable (McInerney et al., 2016), can a simple Bayesian analysis clarify the vagaries of best practice (Black et al., 2011; COAG, 2010b) and provide sufficient confidence to support water management in terms which decision-makers can readily comprehend?
INTRODUCTION

By equating the probabilistic definition of confidence determined by Bayesian credible intervals (literally the quantification of uncertainty) to the abstract notion of confidence in an outcome (ASCE, 1985), then a narrowing of parameter credible intervals generated under 3-PG+ (Feikema et al., 2010b) compared to a control using the original implementation of 3-PG (Landsberg and Waring, 1997; Sands and Landsberg, 2002) must provide a transparent basis for placing confidence in the integrity of the revised model structure against a null hypothesis of similar or lower confidence reflecting increased equifinality and its subjective manipulation to produce satisfactory results.

This study examines the effectiveness of Australian forest water management with particular attention to scientific assessment methods used to underpin policy.

Part I comprises a comprehensive analysis of a review of international policy initiatives with particular emphasis on South Africa, the first jurisdiction to regulate forest water use following decades of applied scientific research (Section 2). This is followed in Section 2 by a comprehensive analysis of contemporary Australian forest water management policy initiatives, examining the effectiveness of national water reforms and as a case study assesses the capacity of a leading jurisdictional water management agency to implement them (Section 3.4).

Part II examines assessment methods currently used to support forest water regulation, developing tools to support existing methods and providing insights into future needs and research areas from a water management perspective. Section 4 reviews assessment methods; Section 5 develops an approach to support the robustness and integrity of Australian empirical assessment methods and data; Section 6 subjects established tools to rigorous scrutiny incorporating a progressive uncertainty analysis within a Bayesian paradigm and evaluates its value to water management agencies.

Finally, Section 7 explores the future assessment needs of forest water management by determining whether a complex, widely used plant process model can demonstrate potential to support forest water management and decision-making by using a philosophy developed from the findings of Section 6. Results are discussed in Section 8 and final conclusions summarised in Section 9.

A feature of this work is its water management paradigm, which while not rocket science is unusual in Australian hydrology, arguably a neglected driver of hydrological research (Klemeš, 1973).
PART I

FOREST WATER REGULATION
2. AN INTERNATIONAL PERSPECTIVE

We believe that the South African model is indispensable to this country and that the thinking, if not the legislation, should be carried through to all other countries encouraging plantation forestry, where water is in any way a limiting factor. It is very likely that there are many countries where downstream users are suffering the ‘unintended consequences’ of upstream afforestation when there is really no excuse for not recognising and managing this accordingly.

Managing the hydrological impacts of South African plantation forests: An overview, Dye and Versfeld (2007)

DESPITE nearly a century of research the water use of plantation forestry remains a contentious issue in many parts of the globe (Schirmer et al., 2016). The projected growth in the demand for wood products in Asia (Ajani, 2011; NFA, 2012) combined with the potential of commercial afforestation to contribute to \( CO_2 \) abatement initiatives (UNFCCC, 2009), raises the prospect of regional plantation expansion. As water resources become subject to increasing pressure due to growing use (Rockström et al., 2009) and changing availability (IPCC, 2013), the adverse impacts of plantation developments on them will need to be managed to achieve balanced outcomes, especially in drier regions where competition for water is high (Albaugh et al., 2013; Farley et al., 2005; Jackson et al., 2005).

Global research into the hydrological impacts of afforestation has resulted in calls for greater attention to water planning and forest management in South America (Almeida et al., 2007; Buytaert et al., 2007; Ferraz et al., 2013; Huber et al., 2008; ITTO, 2002; Lima et al., 2012; Little et al., 2009; Nosetto et al., 2012) and policy reform in India (Calder et al., 2008a; 2008b), while Chinese research has shown notable progress (He et al., 2012; Li et al., 2013; Liu et al., 2014; Sun et al., 2006; Wei et al., 2005; Yu et al., 2013), on the back of an ambitious program of afforestation (Lei, 2002).

Regulatory responses are varied. Many significant timber-producing jurisdictions adhere to codes of practice (Cashore et al., 2006) or are administered under policy frameworks (FAO, 2010; McDermott et al., 2010), many of which advocate some form of protective regard for the role of native forests in water conservation (Blaser et al., 2011). Examples of regulations which explicitly seek to manage the impacts of afforestation on water availability are less common and are usually associated with tension arising from competing interests.
Concerns in Brazil include a suite of socio-environmental issues associated with the aggressive expansion of eucalypt plantations, including: land rights, social justice, loss of biodiversity, the environmental impacts of pulp mills and the impacts plantations on water resources (May, 2006; Overbeek et al., 2012). In 2001 the legislative assembly of Espirito Santo prohibited the planting of eucalypts for pulp production, a decision challenged by vested interests and overturned in the Supreme Court the following year (ITTO, 2002). Since 2006 plantation forest development proposals greater than 1,000 ha have been subject to environmental impact assessments and required to obtain licences which specify conditions of operation, such as areas to be reserved for native forest and riparian buffers and which may include monitoring biodiversity and water impacts (McDermott et al., 2010).

To date only two jurisdictions have gone further to explicitly regulate plantation forestry water use: South Africa in the early 1970s (De Coning, 2006; Dye, 2013; Dye and Versfeld, 2007; Van der Zel, 1995) and Australia some four decades later (see Section 3 below).

2.1 South Africa

The dearth of natural forests in South Africa meant that establishing self-sufficiency in wood products required the conversion of veldt and grassland to intensive exotic plantations with negative consequences for downstream water users. Commercial afforestation commenced in high rainfall areas of Western Cape province in 1876 and by 1915 farmers had begun to complain about decreases in water availability (Dye and Versfeld, 2007; Dye and Bosch, 2000; Van der Zel, 1995; Wicht, 1967a). Perceptions regarding the socio-economic benefits of forestry and contemporary uncertainty regarding its potential hydrological benefits in promoting rainfall (Fernow, 1893; Harrington, 1893; Schwappach, 1904) initially outweighed concerns about negative downstream impacts. However concerns were exacerbated by drought during the 1920s (Scott et al., 1998), and by the time of the Fourth British Empire Forest Conference held in 1935 at Durban, South Africa (Guillebaud, 1935), the situation was such that the Union of South Africa government requested a committee be formed under the auspices of the conference to report on the effects of tree-planting on water availability (Wicht, 1948; 1959; 1967a). Paired study sites were first established at Jonkershoek near Stellenbosch in the Western Cape province in 1935 (Wicht, 1939), followed by others set up to examine the influence of different physio-climatic conditions and treatments on tree growth and catchment hydrology, most notably: Cathedral Peak (1938), Witklip (early 1940s), Mokobulaan (1956) and Westfalia (early
1970s, although inconclusive private experiments were conducted in the 1930s) (Scott et al., 2000) (see Figure 2.1 below).

![Figure 2.1: South African forest research sites, after Scott et al. (2000), map 1.](image)

South African forest hydrology has been almost entirely concerned with supporting water management and policy development (Le Maitre and Versfeld, 1997). Consequently the role of research in paving the way toward forest water regulation is very well documented and the experience provides valuable insights for water managers (Bosch, 1982; Bosch and Von Gadow, 1990; Dye, 2013; Dye and Versfeld, 2007; Dye, 1996a; Dye and Bosch, 2000; Edwards and Roberts, 2006; Van der Zel, 1995; 1997; Versfeld, 1996; Wicht, 1949; 1959; 1967a).

### 2.1.1 The Afforestation Permit System

At the time of the first South African investigations, scientific consensus held that forests had a positive influence upon the local distribution of rainfall; were beneficial to water supplies in reducing siltation in reservoirs and while of observational data precluded any definite pronouncement on the hydrological impacts of exotic
plantations, they were unlikely to be detrimental to streamflow (Guillebaud, 1935). It is unsurprising then that persistent observations to the contrary generated controversy, challenged the benefits of government afforestation policy and precipitated the South African research program (Wicht, 1935; 1959).

By the 1950s the limited and irregular character of its water supplies led to a growing recognition of the need to manage South African catchments for different water resource priorities. Planning measures for different water management outcomes were proposed which included limiting plantation forestry development to catchments without storages and a minimum requirement for maintaining a stable discharge (Wicht, 1958). Concerns intensified during a drought in the mid-1960s (Van der Zel, 1995) and by the late 1960s a land management and planning philosophy accompanied by specific concerns regarding the impacts of forestry on water resources resulted in a prohibition against establishing plantations in riparian zones under the Soil Conservation Act 1969 (Nänni, 1970b).

By 1970 a high-level government committee recommended the regulation of afforestation (Scott et al., 1998). In 1972 the Forest Act 1968 was amended such that government approval was required to establish a commercial timber plantation on unafforested land (Van der Zel, 1995). The amendments specified the type of land to be considered unafforested and gave the minister power to prohibit the planting of trees within a prescribed area in order to protect water resources (Van der Zel, 1995).

The South African Afforestation Permit System (APS) that the legislation enabled was administered through a centralised committee and regional panels with members drawn from water and forest management agencies at both national and regional levels, along with conservation representatives at the regional level (Van der Zel, 1982; 1995). A feature of the system was the adoption of a three-tiered categorization system which delineated catchment management responses according to identified water resource priorities: different areas of afforestation were permitted in different catchments according to the commitment of their water resources to competing priorities, such as reticulated water supply, and stipulated limits on anticipated reductions in mean annual catchment yield (Van der Zel, 1982; 1995). Permit conditions stipulated the area and genus to be planted and required riparian buffers be maintained between plantations and water bodies (Edwards and Roberts, 2006; Van der Zel, 1995).

The centralised committee initially assessed afforestation permit applications on merit, with advice provided by regional water and forest managers (Van der Zel, 1995).
July 1974, the committee accepted the co-called Van der Zel curves as the scientific basis for assessing reductions in streamflow due to plantation forestry (Van der Zel, 1982; see further discussion in Section 4.3 below). The curves, a pair of cartographic lines depicting reductions in mean annual runoff for 15- and 40-year rotations were developed by government hydrologists using findings from the Cathedral Peak research station in the uplands of KwaZulu-Natal and a range of other investigations before being validated against global data (Van der Zel, 1995).

The adoption of the curves was given careful internal consideration but little apparent external stakeholder consultation (Van der Zel, 1995), nonetheless they can be regarded as improving both administrative efficiency and stakeholder confidence; prior to the adoption of the Van der Zel curves, ad-hoc committee deliberations only issued one-year permits in critical catchments. Following their introduction five-year afforestation permits were able to be approved by delegated administrative staff wherever afforestation was allowed (Van der Zel, 1995).

Over the years the APS was refined to meet stakeholder needs. Different runoff reduction limits were adopted in different regions and afforestation extent could be managed at sub-catchment scales to ensure the permissible plantation area was not concentrated in a single area (Van der Zel, 1995). In the early- to mid-1990s a major overhaul embedded the APS in a computer-based geographic information system (GIS) (Van der Zel and Rabe, 1993). Modifications included the adoption of a country-wide sub-catchment scale planning and the ability to allocate water to any diversion limit in consultation with stakeholders (Van der Zel, 1995). On the eve of legislative reform (Section 2.1.2, below), new information was introduced to improve decision-making in the form of a reference manual, supported by improved empirical assessment models (Le Maitre et al., 1997 in Scott et al., 1998, discussed further in Section 4.3 below).

All in all the APS and its assessment approaches served their purpose quite well (Bosch and Von Gadow, 1990) and facilitated sustainable plantation expansion (Van der Zel, 1995). However it remained controversial causing disquiet in the forestry industry because of its inflexibility; the empirical limitations of the Van der Zel curves; the lack of rigour in documenting their development (see Section 4.3 below); arbitrariness in defining riparian buffers; a need for greater integration with catchment management planning and the perceived unfairness of other dryland crops not being similarly regulated (Bosch and Von Gadow, 1990; Dye and Versfeld, 2007; Dye and Bosch, 2000; Görgens, 2003).
2.1.2 Afforestation as a licensed streamflow reduction activity

Democracy and a new constitution in 1996 set the stage for an energetic phase of legal and policy reform in South Africa which encompassed human rights, distributive justice and water management (De Coning, 2006). In 1997 the South African Council for Scientific and Industrial Research (CSIR) published new runoff reduction models which overcame some of the limitations of the Van der Zel curves (Scott and Smith, 1997; see Section 4.3). In 1998 the National Water Act 1998 took effect which among other things aimed at promoting equitable access to water and redressing the results of past racial and gender discrimination (RSA, 1998). Assessment tools based on Scott and Smith (or CSIR) curves were developed to support forest water management (Scott et al., 1998) and subsequently enshrined in law (DWAF, 1999a).

The South African constitution identifies water management as primarily a national function (Godden, 2005). Water resources are recognised as part of a single, interdependent cycle under the act, and regulated as a national asset by the national government in the role of a public trustee (Godden, 2005). The highest priority in water allocation under the National Water Act 1998 is The Reserve, comprising a basic human needs reserve and an ecological reserve (RSA, 1998, s 16-18). The act also defines Streamflow Reduction Activities (SFRAs as any land-based activity including the cultivation of any vegetation, that is likely to reduce the availability of water in a watercourse to meet the needs of the Reserve, international obligations, or other water users (RSA, 1998, s 36(2)). Currently, the use of land for commercial afforestation is the only declared SFRA (RSA, 1998, s 36(1)(a)).

Before SFRAs are evaluated, the Reserve must be determined for the relevant water resource (RSA, 1998, s 18). The determination of the Reserve entails determining the class of the water resource and its relevant quality objectives, known collectively as resource directed measures (RDMs) (DWAF, 1999b). RDMs determine the quantity and the quality of water required by the Reserve (RSA, 1998, s 13(3)).

Once the Reserve is determined other water use including that of SFRAs, may be allocated in consideration of criteria that reflect water policy principles (DWAF, 1999b). The criteria include, but are by no means restricted to: the likely effect of water use on the resource and on other water users; the RDMs (RSA, 1998, s 27(1)), encompassing the extent to which the effect of proposed SFRAs may be accommodated in water allocation and planning frameworks; the risks they pose to the Reserve and anticipated impacts on streamflow regime, riparian habitats, aquatic biota, and water quality
(DWAF, 1999b). Licensed water allocations and water use charges are based on quantitative afforestation impact information (DWAF, 1999a; 2005).

Clearly, the sophistication of the assessments demanded by this system is far greater than could be met using the Van der Zel curves. The CSIR or Scott and Smith (1997) curves represented a quantum leap forward in this regard accommodating estimates of average reductions in annual and low flows for different species of tree grown in different climates, under different forest management practices at four different research stations (Scott and Smith, 1997; see Section 4.3).

The new curves were integrated into a GIS-based assessment tool which was used to produce a handy reference manual comprising tabulated sub-catchment scale streamflow reductions (Le Maitre et al. 1997 in Scott et al., 1998), that was adopted as the legal basis for assessing volumetric reductions in streamflow arising from SFRAs (DWAF, 1999a). The system largely satisfied legal and administrative needs of being unambiguous, transparent, unbiased, adaptive and easy to use, but major technical shortcomings associated with its underlying empirical basis remained, particularly a lack of representativeness of wider South African climatic variability and the need to take greater account of hydrological variability against an agreed baseline land-cover (DWAF, 1999b; Dye and Bosch, 2000) (see Section 4.3).

The Department for Water Affairs and Forestry (DWAF) and South African Water Research Commission sought to address these issues by developing a range of streamflow reduction tables applicable to all regions of South Africa. The Gush tables were based on the outputs of a mechanistic agro-hydrological model (ACRU) and were adopted as the legal basis for assessing SFRAs in 2005 (DWAF, 2005; Gush et al., 2002a). The approach offered a basis for assessing afforestation situations not covered by the experimental catchments on which the Scott and Smith curves were based, while the data from the catchments afforded an opportunity to verify ACRU, with the objective of providing streamflow reduction estimates that were acceptable to a wide range of stakeholders (Gush et al., 2002b).

Despite technical improvements such as explicit baseline land-covers and a suite of soil properties, limitations in the Gush tables remained including: an inadequate number of verification sites; a dependence on empirical plant growth relationships; a scale of modelling which was too coarse to properly account for physio-climatic variability and uncertainties around low flow estimation (Jewitt et al., 2009). Consequently modelled outputs continue to be contested (Dye and Versfeld, 2007), while research into refining the basis for assessment is ongoing (Jewitt et al., 2009).
2.2 Discussion

Changes in water availability arising from large-scale land-use change have generated stakeholder tensions around the world, particularly in developing countries where water availability is closely associated with other human rights issues.

South African forest water regulation commenced with the recognition of stakeholder concerns regarding water security. Its policy response started with a dedicated, long-term research program focused on providing direct support for policy development, resource management and decision-making. The program retained government support through a series of natural calamities and world war.

Initial regulation under the APS was implemented as a highly centralised system with little consultation. The scientific basis of assessing afforestation impacts was completely empirical and unrepresentative of most forestry areas; it lacked transparency and was contested by stakeholders. Mechanistic modelling brought greater flexibility to the system; provided more defensible decision-making; effected management at smaller scales; improved community consultation and ultimately facilitated significant plantation development and water resources protection for 26 years.

National reforms brought more sophisticated water legislation with a higher demand for scientific information to support policy. Stakeholder consultation was given primacy as South Africa sought to promote regional decision-making and redress past inequities. The demands for sophisticated information under the current legislation resulted in the adoption of mechanistic modelling as the basis for assessing hydrological afforestation impacts. Valid concerns regarding the system’s limitations persist but improvement is ongoing, according to a progressive philosophy of stakeholder consultation and capacity building (Jewitt et al., 2009).

South African water management is underpinned by the national government’s responsibilities which are enshrined in its constitutional and legislative frameworks (Godden, 2005). While the country faces a swathe of social, cultural and economic reform agendas which compete with water reforms for limited financial resources, it is arguable that it has produced a more broadly based approach to ensuring sustainable environmental and social outcomes than countries with comparable resource management challenges, like Australia (Godden, 2005).
3. AN AUSTRALIAN PERSPECTIVE

Normally the government drifts along at about 8,000 metres...

Every few years, there’s a drought. When it gets really bad it suddenly rains politicians, experts
and media.

They form pools of expertise and funding to cope with the drought cycle ...

As soon as the good years return, they evaporate back to 8,000 metres.


In Australia, State and Territory governments are responsible for managing water
resources under a provision of the Australian Constitution which prohibits the
Commonwealth from enacting laws that limit their right to do so:

100. Nor abridge right to use water

The Commonwealth shall not, by any law or regulation of trade
or commerce, abridge the right of a State or of the residents
therein to the reasonable use of the waters of rivers for
conservation or irrigation (CA, 2010, Chapter IV, Finance and Trade).

Despite obvious administrative inefficiencies, water issues with inter-jurisdictional
impacts, like drought or cross-border resource management initiatives are addressed
according to the legal and policy frameworks of each of the jurisdictional governments
concerned. Consequently implementing Australian national water reforms has
required negotiations be conducted between the federal government and each of its
State and Territory partners in an institutional environment that fragments and
complicates water resources management (Fisher, 2000), more than would be the
case under a unitary constitution such as South Africa’s (Godden, 2005; Kildea and
Williams, 2010).

The most common approaches toward national accord have been inter-governmental
agreements (IGAs) (Kildea and Williams, 2010). The last 20-years have seen
Australia in the midst of a period of unprecedented national water reforms
implemented under two Commonwealth-State IGAs. The intention of the first was to
address issues that included widespread natural resource degradation (COAG, 1994),
the second known as the National Water Initiative (NWI) was initiated to refresh the
1994 reforms at a time of severe drought, and included arrangements to adapt to future changes in water availability (COAG, 2004b).

Among other things, the NWI recognised the need for secure, nationally-compatible water access entitlements. It further recognised that the integrity of entitlements needed protection from the expansion of certain land-use change or interception activities (including large-scale plantation forestry) which where necessary, should be subject to planning and/or regulatory controls (COAG, 2004b, s 55 – s 57).

The hydrological impacts of plantation forestry are at their most significant at smaller, sub-catchment and property scales (CSIRO, 2008c; Polglase and Benyon, 2009; SKM et al., 2010), as reflected in regional policy issues that emerged pre-NWI in South Australia, Victoria and Western Australia (discussed in ensuing sections). The NWI targeted large-scale interception activities initially associated with risks to the shared water resources of the Murray-Darling Basin (Earth Tech Engineering, 2003). Its association with large-scale risks arguably magnified the extent of plantation impacts to policy makers, ensuring the NWI would become the guiding instrument for implementing forest water management in all Australian jurisdictions.

### 3.1 The National Water Initiative

The NWI comprises 108 sections arranged under nine headings. Sections most germane to this study are shown emboldened below and presented in Appendix A.

<table>
<thead>
<tr>
<th>Section</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.</td>
<td>Preamble (s 1 – s 7);</td>
</tr>
<tr>
<td>2.</td>
<td>Implementation (s 8 – s 14);</td>
</tr>
<tr>
<td>3.</td>
<td>Commencement (s 15);</td>
</tr>
<tr>
<td>4.</td>
<td>Interpretation (s 16 – s 17);</td>
</tr>
<tr>
<td>5.</td>
<td>Roles and responsibilities (s 18 – s 22);</td>
</tr>
<tr>
<td>6.</td>
<td>Objectives (s 23);</td>
</tr>
<tr>
<td>7.</td>
<td>Key Elements (s 24 – s 101);</td>
</tr>
<tr>
<td>i)</td>
<td>Water Access Entitlements and Planning Framework (s 25 – s 57);</td>
</tr>
<tr>
<td>ii)</td>
<td>Water Markets and Trading (s 58 – s 63);</td>
</tr>
<tr>
<td>iii)</td>
<td>Best Practice Water Pricing (s 64 – s 77);</td>
</tr>
<tr>
<td>iv)</td>
<td>Integrated Management of Environmental Water (s 78 – s 79);</td>
</tr>
<tr>
<td>v)</td>
<td>Water Resource Accounting (s 80 – s 89);</td>
</tr>
<tr>
<td>vi)</td>
<td>Urban Water Reform (s 90 – s 92);</td>
</tr>
<tr>
<td>vii)</td>
<td>Knowledge and Capacity Building (s 98 – s 101); and</td>
</tr>
<tr>
<td>viii)</td>
<td>Community Partnerships and Adjustment (s 93 – s 97).</td>
</tr>
</tbody>
</table>
8. Variation (s 102 – s 103);

The essence of the NWI is contained within ten objectives (s 23) and eight key elements (s 24 – s 101), which place the security of water entitlements as their first priority in sections 23(i) and 24(i) (see Appendix A). Specific reference is made to protecting the integrity of water entitlements from unregulated growth in interception through land-use change (s 25(xi)). Land-use change activities are addressed in sections 55 to 57 under the primary key element Water Access Entitlements and Planning Framework and are defined as activities that have the potential to intercept significant volumes of surface and/or ground water and include farm dams and bores; intercepting and storing of overland flows and large-scale plantation forestry (s 55).

The intention of sections 55 to 57 is to ensure the significance of interception activities is assessed in a water management context and where necessary, implement appropriate planning, management and/or regulatory measures to protect the integrity of water access entitlements and the achievement of environmental objectives (s 56).

Where interception activities are assessed as being significant and located in heavily allocated systems, they should be systematically recorded and a threshold for the activities set. Any increase in significant interception activities above the threshold should be subject to a water access entitlement and a monitoring regime implemented (s 57(i)). In less heavily allocated systems significant interception activities should be identified and the amount of water likely to be intercepted estimated. As for heavily allocated systems, a threshold for the activities should be set and progress toward the threshold monitored (s 57(ii)).

The provisions described above provide a robust framework for any jurisdiction acting in good faith to protect the integrity of water access against the impact of interception activities, but only once measures are found to be necessary (s 56). This qualification enables jurisdictions considerable discretion in addressing interception issues. For instance, small-scale issues may be negated in favour of large-scale plantation forestry developments by simply increasing the scale of assessment, where they appear less conspicuous (CSIRO, 2008c; Polglase and Benyon, 2009; SKM et al., 2010).

More satisfactory outcomes may be possible if the NWI’s somewhat disparate sections were better integrated. Water Resource Accounting (s 80 – s 89) is a Key Element (ch 7 pt v) of the NWI, which seeks to ensure that adequate measurement, monitoring and
reporting systems are in place to support public and investor confidence in the amount of water being traded, extracted and managed (s 80, see Appendix A). Actions include developing and implementing consolidated water accounts to protect the integrity of the access entitlement systems (s 82), aligning well with s 56 described above. Moreover water resource accounts are to include information to enable the production of a national water balance, including consideration of land use change and other externalities among its elements (s 82(c)(iii)). Without the recognition that robust water accounting is required to protect the integrity of access entitlement systems against interception activities (s 82), the effectiveness of their management will remain mixed across jurisdictions.

### 3.1.1 National Water Commission

Along with the NWI, COAG (2004a) committed to establishing the National Water Commission (NWC) to assess the progress of implementing the NWI. The legislation under which the NWC was established was scheduled to expire on 30 June 2014, subject to an independent review of its ongoing role. The review concluded that a number of core functions would be required to continue to progress NWI reforms (Rosalky, 2011). The National Water Commission Act 2004 was subsequently amended to extend the NWC’s in a role which focused on monitoring, auditing and assessment to be reviewed before the end of 2017 (AG, 2014).

The activities of NWC extended to facilitating NWI implementation by funding a range of projects over all elements of the NWI according to accredited jurisdictional implementation plans (ANAO, 2010). The work programme included developing national water planning guidelines (COAG, 2010b), conducting assessments of interception activities (Duggan et al., 2008; SKM et al., 2010) and developing tools to assess the risks posed by plantation forestry (Gilfedder et al., 2010).

In 2012 the (then) COAG Standing Council on Environment and Water (SCEW, now National Environment Protection Council see [www.swew.gov.au](http://www.swew.gov.au), accessed 15 October 2014) developed an enhanced water reform agenda in response to reviews of the NWI (COAG, 2008c; 2010a) and the NWC itself (Rosalky, 2011), and published a report describing the next stages of Australian water reform (SCEW, 2013). The report identified significant unfinished business from the NWI and proposed a five-year work plan to address it, the three elements of which were:

- the further development and implementation of water plans, including their accommodation of significant interception activities (s 23 – s 24);
• the better recognition of Indigenous needs in relation to water access and management (s 25); and,
• fully implementing the NWI in managing surface-groundwater connectivity (s 25).

In September 2013 a new Australian government announced the NWC’s closure by 31 December 2014 (DE, 2014) and the disbanding of SCEW (NWC, 2014a). The functions carried out by the NWC were transferred to other government agencies (CA, 2014).

The final NWC (triennial) assessment released in October 2014 (NWC, 2014a) contained little new information concerning interception by plantation forestry. Progress in South Australia was noted; work reviewing the methods used to assess interception activities was reported (Barma and Holz, 2013) and the importance of unfinished business reiterated. Significantly, the report noted that the lack of a standing council with responsibility for considering national water policy could result in a loss of momentum in pursuing national water reforms and that several agreed actions had already been discontinued, including fully implementing NWI interception commitments (NWC, 2014a, Appendix B).

3.1.2 Unfinished business

The unfinished business left by the termination of the NWC to fully implement the interception commitments of the NWI included:

• a review of progress in addressing interception, encompassing currently available tools and information (Item 2, Table 1); and,
• develop enhancements to the NWI Policy Guidelines for Water Planning and Management (COAG, 2010b) (Item 5, Table 1) (SCEW, 2013).

While Item 5 (Table 1) prioritised climate change and Indigenous engagement, the full text referred to under Schedule A (SCEW, 2013, p19) provided information germane to the scope of the review in Item 2, committing to the development of additional tools, methodologies, modules, or case studies to address a range of water management issues, including the management of interception by plantations.

The required actions relevant to interception comprised ongoing biennial reporting on the implementation of interception management arrangements (COAG, 2010b) (SCEW, 2013, Deliverable R1, Table 2). The arrangements were detailed under a set of principles which emphasised the preference for bringing interception activities under
a water access entitlement system, underpinned by assessments and reporting and requiring entitlements to offset plantation water use (s 3.6.5, COAG, 2010b).

Offset entitlements should be of high-reliability to address the variability of the hydrological impact of the plantation during the life of a rotation and periods of changing water availability, and their volume estimated (as always) using the best available science. The last but certainly not least consideration is that offset entitlements should be tradeable, with implications for the integrity of the method of estimation.

3.2 Other Inter-jurisdictional Initiatives

Inter-governmental agreements have underpinned other significant Australian water reforms related to plantation forestry, the most salient of which are those made to manage the shared resources of the inter-jurisdictional Murray-Darling Basin (the Basin) under the Commonwealth Water Act 2007 and Murray-Darling Basin Plan (see COAG, 2008a; 2008b; MDBA, 2013 and related agreements therein).

At the beginning of the Millennium Drought (SEACI, 2011; Van Dijk et al., 2013) the then Murray-Darling Basin Commission investigated a number of factors that had the potential to impact the capacity of the Murray-Basin resources to meet the needs of the environment and other users. The factors included: climate change, reafforestation, groundwater extraction, changes to return flows from irrigation areas, farm dam development, vegetation regrowth after bushfires, industry change and water trade (Earth Tech Engineering, 2003).

The supporting forest hydrology information was provided by the Australian CSIRO, as part of the Cooperative Research Centre for Catchment Hydrology (Zhang et al., 2003a). The information was presented in the context of the prospect of doubling and trebling plantation area in the Basin by 2020. Australian empirical assessment methods were promoted over physically-based models which were described as problematic due to limited data availability and parameter estimation. A single South African study was cited to demonstrate the hydrological impacts of different species on different flow regimes at different ages (Scott and Smith, 1997), however the learnings 30 years of plantation forest water regulation were given no further attention.

While water trade and industry change were considered unlikely to impact on water resources (Earth Tech Engineering, 2003), while the impacts of bushfire, groundwater over-allocation, climate change, growth in farm dams, expansion of plantation forestry
and re-vegetation and reduced return flows from increased water use efficiency in irrigation were recognised by high-level decision-makers as significant risks (MDBMC, 2004, meeting 35, 26 March).

In June 2004, the NWI identified a number of land-use change activities that without some form of planning and regulation, presented a risk to the integrity of water access entitlements: farm dams and bores, intercepting and storing of overland flows and large-scale plantation forestry (COAG, 2004a, s 55 and 56, as described in Section 3.1 above). By 2006, the hydrological impacts of afforestation were entrenched in the Australian national water policy reform narrative as one of The Six Risks to the Basin’s shared water resources (Dyson, 2005; Van Dijk et al., 2006).

The effects of the Millennium Drought and associated natural disasters (bushfires), brought home the prospect of changing water availability due to climate change (Arnell et al., 2001) and became the driving force in progressing reforms under both the NWI and the Murray-Darling Basin Plan (COAG, 2008a; 2008b; 2008c) and changing policy directions regarding the hydrological impacts of plantation forestry.

Before the worst effects of the drought were felt in the early 2000s (BOM, 2002; Lu and Hedley, 2004), conversion of pasture to plantation forestry was regarded as having clear economic and environmental benefits, including carbon sequestration and managing dryland salinity (Zhang et al., 2003a). From a water resources management perspective, salinity was the problem and trees were the answer (Passioura, 2005).

The benefits of afforestation were promoted under the National Forest Policy (AG, 1997b; CA, 1992), an agenda released around the same time as the initial COAG water reforms (COAG, 1994), which informed the analysis of inflated plantation expansion scenarios described by Zhang et al. (2003a) (above). Driven by tax incentives awarded under managed investment schemes (Brown and Beadle, 2008), the policy of plantation expansion would have the unforeseen consequence of establishing competing agendas as forestry developments extended into drier regions (Keenan et al., 2004; Van Dijk et al., 2004), where competition for limited water resources was already high and subject to jurisdictional water management initiatives.

### 3.3 Jurisdictional Initiatives

A number of significant national-scale studies have reviewed jurisdictional responses to managing plantation interception: three under the NWI which encompassed all interception activities (Barma and Holz, 2013; Duggan et al., 2008; SKM et al., 2010);
one, focusing on the risks posed by afforestation to Murray-Darling water resources through both traditional forestry and carbon sequestration (SMEC, 2010); another reviewed jurisdictional forest water policy responses to inform the Northern Territory on implementing the NWI (Hutley et al., 2012) and the last, conducted for a forest industry group in the context of the NWI, was distinct in pursuing a sub-jurisdictional (regional) perspective (Polglase and Benyon, 2009).

All studies were written with regard to the NWI; all examined assessment methods and all sought to identify further scientific work. However, the quality of jurisdictional policy analyses, largely undertaken by scientists was mixed. With the notable exception of Polglase and Benyon (2009) policy analyses were either broad-brush; limited solely to NWI objectives (assumed to be the driving force behind reform); heavily reliant on parsimonious jurisdictional NWI implementation plans. Hutley et al. (2012) omitted Victorian work which featured a well-developed forest water policy. Few bothered to distinguish between investigations targeted at supporting forest water management and more general research.

The approach adopted in the ensuing sections is not dissimilar, however it has a significant point of distinction in that it attempts to step outside the NWI paradigm to provide a more balanced account of policy initiatives in the context of State forest water management agendas. In doing so it provides a more critical evaluation of the NWI’s contribution to progressing jurisdictional agendas and a clearer indication of the value of NWC-funded research, particularly in the light of the capacity of State agencies to support NWI implementation.

### 3.3.1 South Australia

South Australia is the first and currently the only Australian jurisdiction to regulate the water use of plantation forestry (Victorian, legislation is not yet passed and Western Australian legislation is currently in development, see relevant sections below). National reforms were key motivators in developing a state-wide forest water policy framework (GSA, 2009) and legislation (GSA, 2011; 2012), but the first stages of regulation were enacted regionally in the Lower South East, pre-dating the NWI by a number of days (COAG, 2004b signed 25 June; GSA, 2004 published 3 June).

Indeed regional water resources have a long history of inter-jurisdictional management across the South Australian and Victorian border under the Border Groundwaters Agreement (GSA, 1985). However, while the agreement aims to protect groundwater resources by promoting their cooperative management, it does not take water
extracted for domestic and stock use or the impacts of plantation forests into account (SAVBG, 2013).

Nonetheless principles of resource protection and equitable sharing established in the 1980s and extended through ensuing groundwater prescription processes, motivated stakeholder concerns regarding the impacts of plantation forestry expansion on the security of water entitlements in the mid-1990s (PSA-SCWA, 1999). Consequently South Australia’s response to the NWI (GSA, 2005) may be seen as only part of a longer and more complex story which illustrates how competing administrative and scientific agendas can interfere with the efficient implementation of water reforms and even destabilise regional planning frameworks.


Recent developments

South Australia is arguably the jurisdiction closest to meeting the NWI’s intentions with regard to plantation forestry. Its state-wide policy recognises that large-scale plantation forestry is an issue for sustainable water resources management in higher rainfall areas (GSA, 2009) and its legislation was amended to address the issue in the context of water allocation planning (GSA, 2011).

In areas where water allocation planning overlap with the Border Groundwaters Agreement, the latter takes precedence. Licensees are able to transfer allocations between irrigation activities or to a forest water licensee, provided that the volume of water permitted to be extracted from licensed wells under the agreement is not exceeded (GSA, 2013). However change may be in wind. The quantity of plantation forestry water use in the border zone is recognised by inter-jurisdictional partners as being a significant component of the regional water balance, and attaining long-term sustainability will mean developing a new management approach which may require reductions in plantation forestry development (SAVBG, 2013).

On June 25th, 2014 exactly a decade after South Australia signed the NWI, the Lower Limestone Coast Prescribed Wells Area was declared a forestry area under a water allocation plan (SENRMB, 2012), enabling the introduction of volumetric water access entitlements for commercial forestry water use in the Lower South East. The trigger for moving to volumetric licences was consistent with the NWI (s 55-57), satisfying the
relevant jurisdictional authority that commercial forestry was having or was reasonably likely to have a *significant* hydrological impact on the prescribed water resources (GSA, 2014). The interception sections of the NWI (s 55-57) are punctuated with the need for determining the *significance* of impacts (see discussion in Section 3.1). There is currently no consistent, national or jurisdictional framework for transparently determining how the significance of the hydrological impacts of an interception activity should be assessed. Building the case to convince the South Australian authority (the Minister) took many years, many studies and much public engagement (see discussion in Avey and Harvey, 2014; notably Prosser and Walker, 2009). It seems inevitable that risk management approaches will become more *significant* in future Australian water management decision-making (Avey and Harvey, 2014).

The South Australian experience is remarkable for its parallels with South Africa: regulation emerged in response to concerns regarding plantation impacts on water availability; it was initiated as a permitting system; initially underpinned by empirical relationships and coloured by stakeholder dissatisfaction proportional to the relative weaknesses of government consultation processes. Distinctive contrasts include the greater legal and administrative complexities arising from Australia’s federated constitution and the progression of South Africa to mechanistic computer modelling as a basis of assessment.

### 3.3.2 Victoria

The Victorian NWI implementation plan included a three stage response to managing interception by large-scale plantation forestry to be completed by 2008, comprising:

- recording current interception activities by characterising catchment stress across the state in terms of the hydrologic stress and the capacity for further plantation development;
- determining thresholds by developing a methodology for assessing environmental impacts and socio-economic considerations; and,
- implementing a compliance and monitoring regime by developing policies and management tools to account for and manage the hydrological impacts of afforestation (VG, 2006).

Like South Australia the framework that supported this approach pre-dated the NWI. In April 2003, the Victorian Government responded to severe drought and water resource degradation by issuing a ministerial statement setting out a broad agenda of water policy reform (Thwaites, 2003). A green paper was released in August of the
same year, followed by a white paper on 23 June 2004 (two days before the NWI was signed), containing strategies to support long-term Victorian growth in the face of changing water availability (DSE, 2004).

The white paper recognised that the quantitative impacts of afforestation on water resources were not addressed by contemporary Victorian legislative frameworks, but needed to be considered in planning frameworks to protect the future integrity of water entitlements and achieve environmental objectives (DSE, 2004). Specific actions included undertaking a statewide assessment to identify high, medium or low hydrologic impact zones for new plantation developments and developing appropriate tools to account for the impacts of new plantations on water resources, an approach which was to form the basis of the Victorian NWI implementation plan (VG, 2006). Existing planning arrangements were to be applied in the interim (DSE, 2004).

By September 2011 Victoria still had no specific state-wide policy for addressing interception by plantation forestry (NWC, 2011, Appendix B). However, by November Victoria had not only released a state-wide policy (DSE, 2011b), but also released comprehensive strategic regional assessments containing further detail of the policy and its implementation (DSE, 2011a; 2011c). The failure of managed investment schemes and an increased competition for land abated new forestry development in Gippsland, such that the Victorian Government did not consider it necessary to declare intensive management areas in the region (DSE, 2011a). However in western Victoria, blue gum plantation expansion was recognised as contributing to pressures on regional water resources, and had resulted in groundwater licence applications being rejected in some areas (DSE, 2011c).

The response set out in the Western Region Sustainable Water Strategy (DSE, 2011c) is based on three main elements articulated in the Victorian state-wide policy: obtaining the best estimates of water use; legislative reform (in accordance with the white paper); and controlling plantation expansion in declared intensive management areas over stipulated thresholds. Water use estimates are to be made using the best available computer modelling, empirical measured streamflow, groundwater and water balance information, while the suitability of remote sensing techniques will be reviewed and ways to combine the different approaches considered (DSE, 2011c).

The required legislative changes are contained in the Water Bill 2014 (PV, 2014b, currently in its second reading, moved 26 June), which among other things, seeks to repeal the Water Act 1989 and includes mechanisms to establish targeted controls on new forestry plantations and implement actions in the Gippsland and Western Region
Sustainable Water Strategies (PV, 2014a). The relevant catchment management authorities (CMAs) may identify areas of intensive management for further consideration as declared forest plantation areas under the new legislation. In anticipation of this process, the Glenelg Hopkins CMA Draft Western Region Sustainable Water Strategy identified areas likely to be a higher management priority within the Glenelg River catchment (DSE, 2011d). Priority areas were delineated through a consultative process using the best information available and relied heavily on modelling conducted during the Water and Land-use Change (WatLUC) study (SKM, 2005; 2008a; 2008b).

To provide certainty to proponents, the Minister for Water will issue guidelines to inform landholders and to ensure applications for new forestry developments in a declared area can be assessed rapidly. The Department of Sustainability and Environment will develop these guidelines in consultation with water corporations, catchment management authorities, the forestry industry and other stakeholders. In developing the guidelines, the Department will identify the technical support required by the industry when seeking approval for new developments in intensive management areas (DSE, 2011c, section 5.4).

Once a forest plantation is declared, intensive management prescriptions become possible under the new legislation. Existing water use will be recognised and new forestry developments will be restricted if they cover more than 20 ha or 10 per cent of the property, whichever is greater (DSE, 2011c). Afforestation of pasture and cropping land exceeding this area will require a take and use licence authorising the interception of water by a forest plantation unless an offset area approval has been obtained (PV, 2014b, s 67(1)). The resource assessment required to support both licensing and offsetting purposes must estimate the annual average amount of water intercepted by the forest plantation that would otherwise flow directly or indirectly into a water source (PV, 2014b, s 67(1)(d)). The average annual amount of water will be determined in accordance with the regulations (PV, 2014b, s 67(1)(e)), which have not yet been developed (Steendam and Schreiber, 2014, pers. comm.).

3.3.3 Western Australia

After some years of planning and consultation the Government of Western Australia began actively reforming its water legislation (GWA, 2009b; 2013). In 2012 the Water Services Act 2012 was enacted to update legislation related to water, sewerage, irrigation and drainage services (GWA, 2015). The reform was the first step in
consolidating six different acts that regulate the take and use of water, protect waterways, manage drainage and protect public drinking water sources and supply into two modernised water resources management acts (GWA, 2016).

Drought and climate change have been a long-time driver of Western Australian water reform. In 2002, the Government of Western Australia held a symposium to brainstorm water planning responses to drought and changing water availability (Gallop, 2002). It recognised the impacts of climate change and the need to improve water management planning (WALA, 2002). Outputs from the symposium and associated regional water fora led directly to a strategy designed to underpin WA’s future water security in the face of intense drought and long-term climate change (GWA, 2003).

The strategy acknowledged the impacts of permanent climate change and the role of plantation forestry in exacerbating the stress on some water resources and recommended a whole-of-government review of irrigation activities. The review (IRSC, 2005), identified the need to consider a water allocation approach that recognised the impacts of different land uses on water resources to ensure bottom-line requirements such as environmental flows were met and described principles to guide water reform in the context of the National Water Initiative, which was signed by WA in April the following year.

The NWI provided momentum to the WA reforms, marking the beginning of a period of intensive policy development. In October 2006 a draft State Water Plan was released in followed by a Blueprint for Water Reform in December (WRIC, 2006). A government response to the Blueprint was released in February 2007 (GWA, 2007a) followed by the NWI implementation plan in April (GWA, 2007c) and the final State Water Plan 2007 in May (GWA, 2007b).

The NWI implementation plan committed WA to accounting for interception in water planning processes, setting thresholds and developing local rules where the impacts were substantial. These specific goals were supported by a broad plan of policy development and legislative reform (GWA, 2007c, p 39).

Current WA water management legislation, the Rights in Water and Irrigation Act 1914, does not recognise plantation forestry as a water taking activity subject to licensing (GWA, 1914, s 2 and 5) or provide for statutory water plans as required under the NWI (GWA, 2007c), however the scene has been set for reform for some years. The impact of plantations on other water users; the methods by which forestry water entitlements should be quantified; and whether a plantation is impacting
groundwater, surface water or both, have been defined in guidelines (GWA, 2009a). The circumstances which would require plantations to be brought under a water entitlement system have been well-developed for some time (GWA, 2009b), and a legislative and policy framework was developed in September 2013 (GWA, 2013). Salient features of the framework build on the NWI and inter-state experiences and seek to give water users and the wider community a clear understanding of the proposed reforms and garner stakeholder feedback (GWA, 2013).

3.3.4 Tasmania

Tasmania is yet to develop a formal policy framework to address interception by plantation forestry. Its NWI implementation plan recognised the importance of the issue and aimed to ensure that plantation expansion would maximise the environmental and other benefits associated with forestry activities while minimising any negative impacts on water resources. To that end, Tasmania committed to scoping and developing a comprehensive catchment planning tool, test it in the Ringarooma catchment in the state’s north-east and by December 2006, incorporate the results into the Draft Ringarooma Water Management Plan (DPIW, 2006, see Implementation Timetable).

Concerns raised regarding the impacts of plantation forestry interception on water resources were raised during the public consultation phase of the planning process, which was halted while the issue was investigated further. The planning tool was indeed developed and trialed in the Ringarooma catchment, but the results indicated that plantation conversion had minimal impacts on water availability at a catchment scale. Challenges facing the wider implementation of the tool, mostly in addressing uncertainties associated with input data, were identified as needing further consideration in a policy context (DPIW, 2008).

Consequently, further forest water policy development has not occurred. A recent government overview of Tasmanian water management arrangements is silent on forest water management (DPIPWE, 2010), and despite comments by NWC (2011) to the contrary, the Draft Ringarooma Water Management Plan (eventually released in October 2012) contains no measures to address the issue (DPIPWE, 2012).

Given the political sensitivities around Tasmanian forestry, it is unlikely there was ever any real intent to do so. Tasmanian commitments in their NWI implementation plan were couched in the environmental benefits of plantation forestry and were not
supported by a program of legislative reform to regulate plantation forestry as a *water taking* activity under the *Water Management Act 1999*.

### 3.3.5 New South Wales

New South Wales’ approach to addressing interception by large-scale plantation forestry under the NWI was restricted to assessing its significance in cooperation with CSIRO and the (then) Murray-Darling Basin Commission. Within that context, NSW committed to developing and applying a methodology to identify changes in plantations over time, assessing their impacts on a *catchment scale* and depending on the scale of the impacts, bringing plantations within the water entitlement framework. Details regarding the substance of these commitments such as timelines, were not documented (GNSW, 2006).

In a survey of jurisdictional programs, Duggan et al. (2008) reported that NSW considered large-scale plantation forestry a significant interception activity in some areas, and intended to determine what (if any) action was necessary based on the *outcomes of the MDBC Risks to Shared Water Resources Programme*, specifically CSIRO’s *Murray-Darling Basin Sustainable Yields* assessments (CSIRO, 2008c).

NSW noted that the sustainable yields assessments found that the greatest threat to water related objectives across the Murray-Darling Basin was climate change. From a Basin-wide and valley scale (NSW Murray-Darling Basin valleys comprise multiple sub-catchments and extents typically greater than 100 km, see Ribbons, 2009), the impacts of projected increases over 25 years in farm dam, plantations forest, and groundwater development did not present a compelling case to change management or policy (OWNSW, 2010).

Accordingly, NSW has not developed any specific policy to address plantation interception and its management strategy has entailed guidance by *state-wide policies* with links to relevant water sharing plans (NWC, 2014c). However the most recent NWC report card indicates that NSW water sharing plans only address NWI interception activities *to some extent*. Despite the presence of significant plantation forestry and its potential for expansion in some catchments, specific provisions for compliance with interception provisions of the NWI are simply not evident in the relevant plans (NWC, 2014c).
3.3.6 Other Jurisdictions

In areas where its impacts were assessed, plantation forestry was not identified as a significant interception activity in Queensland (GQ, 2006). A specific forest-water policy was not considered necessary as Queensland’s existing regulatory and planning framework could be used to address the issue. In particular, statutory water plans prepared under the Water Act 2000 could be amended if plantations proved to be a risk to the future integrity of water access entitlements and the achievement of environmental objectives for the water system (GQ, 2006).

Land tenure in the Australian Capital Territory is such that any significant land use change is subject to an environmental impact assessment, which includes impacts on water resources. Its commitment to environmental protection, the strength of its water management and planning framework (see for example ACTG, 2014a; 2014b) and the low likelihood of plantation expansion in the ACT, rendered any specific forest water policy unnecessary (ACTG, 2006).

In the Northern Territory, significant forest plantation proposals are assessed for possible water resource impacts under the Environmental Assessment Act. Developments are required to operate under accredited plans which include management measures to ensure that interception does not reach unacceptable levels (NTG, 2006).

While the potential for plantation expansion was identified as low, the NT recognised its potential to alter water balances at regional scales and its NWI implementation plan committed to considering the possibility of further regulation subject to assessments conducted in the Daly River area and evaluating the possibility of extending the findings to the Darwin rural area and the Tiwi Islands (NTG, 2006).

Since then, NT plantation development has expanded, albeit at modest levels (from 16,000 to 44,000 ha between 2005 and 2013, Gavran, 2014). Recent assessments have found that plantations use similar amounts of water to tropical savannah and pose a low risk to water resources, rendering further regulation unnecessary at this time (Hutley et al., 2012 and references therein).

3.4 Jurisdictional Capacity to Support Policy Implementation’

Of the jurisdictional interception reviews cited in Section 3.3 (Barma and Holz, 2013; Duggan et al., 2008; Hutley et al., 2012; Polglase and Benyon, 2009; SKM et al., 2010), four included surveys of agency staff and specialists to determine the
significance of NWI interception activities and the capacity of jurisdictions to respond to them (COAG, 2004b, s 98, Key Element 7, Knowledge and Capacity Building).

Duggan et al. (2008) reviewed the potential impact of different types of intercepting activities, their impact and significance, geographic variability, jurisdictional responses and methods available for integrating interception into NWI-consistent water planning processes. They targeted different questions to agency staff, water experts, industry groups, staff involved in regional NRM case studies.

Polglase and Benyon (2009) interviewed representatives from all state and territories to identify progress on policy development, regionally-specific hydrological and other issues considered important in relation to plantations and water interception, to identify where research could inform and advance science-based policy implementation.

SMEC (2010) aimed to gain a clear understanding of scientific knowledge and gaps on the impacts of afforestation for carbon sequestration benefits and on catchment water balances; and identify current legislation and policies in each jurisdiction relating to afforestation for carbon sequestration, with a focus on the water use impacts of these policies. They interviewed specialists from a range of disciplines relevant to carbon sequestration and forest hydrology covering research organisations, environmental management agencies, catchment management groups, and the forestry industry.

Following on from the assessments of SKM et al. (2010), Barma and Holz (2013) synthesised the technical methods used to assessed the impacts of interception activities. They surveyed national and international methods drawing on available documentation and discussions with key staff from state and territory agencies.

Each of these surveys reported insights into forest water science-policy agenda via opinions provided by a range of specialists and agency representatives. The approach assumed that survey participants had sufficient insight into science-policy issues to adequately represent the issue at hand. This assumption is likely to be invalid for a number of reasons, including:

- many of the agencies canvassed belong to the federal government, research organisations or universities, none of which have any direct role in the day-to-day business of jurisdictional water management (see participants of the studies cited above and ‘key players’ identified by Chartres, 2006);
- unfounded confidence generated by the perceived professional authority of the participants whose opinions are taken on faith, despite not having current
knowledge or relevant experience in the field, but whose reputation means they are less likely to be questioned (ASCE, 1985);

- jurisdictional water management functions are often fragmented between agencies, making the identification of key players difficult (Chartres, 2006);
- researchers investigating the issue invariably stand on the science side of the science-policy interface rather than the policy or management side, underpinning the bias evident in the preceding bullet points.

The consequence of this bias is a series of assessments of limited value to those working on the management side of the interface, developing and implementing policy frameworks. It is indicative of the problem that the most cohesive review of international plantation forest water research in Australian water management literature was not compiled until nine years after signing the NWI (Barma and Holz, 2013), a mere 10 months before the closure of the NWC was announced (15 May 2014, NWC, 2014b) and 12 months too late to dispel Australia’s leading forest water management jurisdiction (CCWPL, 2012; Jasper, 2015) from the misapprehension that it was the first in the world to enact relevant legislation (WIA, 2012a).

The surveys cited above reflect a paradigm seated on the science side of the science-policy interface. They were designed to explore known unknowns. In contrast, the perspective of this thesis is from the management side of the interface. The lack of satisfactory outcomes from the conventional scientific perspective, prompts testing a less conventional one, exploring blind spots in water management agencies or unknown unknowns (Rumsfeld, 2002), a concept capable of providing valuable scientific insights before gaining infamy in 2002 (Logan, 2009).

### 3.4.1 Survey

A survey was designed to identify scientific knowledge gaps that may be unknown to water management experts. It was given to applied scientists and policy professionals with experience supporting or developing forest water policy within the South Australian agency responsible for managing forest water, an organisation on the management side of the science-policy interface.

The survey questions are shown in Table 3.1 (below) and the complete document is presented in Appendix B. Question 1 attempted to clarify the experience of the respondent. It was used to screen whether the survey would be included in further analyses. The level of experience was also used to validate the simple self-evaluation of knowledge on a scale of one to 10 in Question 2 (Table 3.1). Validation was
undertaken after the survey by reflecting on the responses to Question 1 according to experience/involvement criteria shown at the bottom of Table 3.1. The degree of accord between self-adjudged knowledge (Question 2) and independently adjudged experience was used to provide an indication of the reliability of the cohort’s responses.

Questions 3 to 8 were designed to unpick the claims of Questions 2 from a forest science perspective. Some accord would be expected between responses to Question 2 and Questions 3 to 8 from a well-trained staff in a leading water management agency already leading the nation in supporting forest water policy development, unless gaps in the knowledge identified in Questions 3 to 8 were evident.

Question 9 is one of general knowledge of tools and Question 10 aims to explore how successful the NWI has been in reaching the applied realms of water management.

Table 3.1: Survey questions (Appendix B).

<p>| | | | | | | | | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>1. What experience do you have in impact assessment or supporting initiatives to address the ecohydrological impacts of afforestation by plantation forestry?</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2. Please rate your knowledge of issues associated with assessing/ managing the ecohydrological impacts of afforestation.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>low knowledge</td>
<td>general knowledge</td>
<td>detailed knowledge</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
<td>6</td>
<td>7</td>
<td>8</td>
<td>9</td>
</tr>
<tr>
<td>3. Radiation; radiation interception and canopy dynamics</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>4. Canopy processes and conductance</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5. Photosynthetic production and canopy light efficiency</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>6. Stomatal conductance and processes</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>7. Plant–root–soil–water interactions and processes:</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>8. Soil–water (balance) dynamics:</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>9(a) What tools/approaches have you used to assess impacts?</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(b) Can you name other tools/approaches?</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>10. With specific regard to assessing the impacts of interception by plantation forestry:</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(a) Do you know of any agency initiatives targeted at identifying the key knowledge and capacity building priorities needed to support the NWI? If so what?</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(b) Can you identify any agency initiatives targeted at more effectively coordinating the national water knowledge effort?</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
3.4.2 Results

A total of 19 surveys were distributed to staff within the Science, Monitoring and Knowledge Group of the Department of Environment Water and Resources, South Australia during the first half of 2015. Of the 19 surveys, 18 were completed and returned, two of which were excluded on the basis of insufficient experience in forest water management leaving a total of 16 surveys for analysis. The professional background of the cohort comprised hydrologists (7; 1 insufficient experience), hydrogeologists (3; 1 failed to return), ecologists (5; 1 insufficient experience) and an economist/policy expert. Fifteen of the respondents held post-graduate degrees, five of whom held PhDs.
Figure 3.1: Responses to survey Questions 1 to 8. Red histograms repeated through panels (a)-(h) are the responses to Question 2, grey histograms are responses to other questions (see Table 3.1 and text).

Figure 3.1 shows the survey results. Significance was evaluated using the non-parametric Mann-Whitney test `wilcox.test()` in R (R Core Team, 2015).

Responses to Question 2 are repeated through each panel in Figure 3.1(a)-(h) as red histograms. Figure 3.1(a) shows responses to Question 2 (red) and the verified responses to Question 1 (grey). The difference between the two data is not significant ($p > 0.05$) indicating that the self-evaluated level of knowledge indicated by responses to Question 2 is generally verified by experience. In contrast, the responses to Questions 3 to 6 (Figures 3.1b-e) are significantly different to the responses to Question 2 ($p < 0.05$), indicating a lack of knowledge regarding plant processes in forest water policy development.

![Graph showing survey responses]

**Figure 3.2:** Responses to survey Question 9. Use of tools

The strength of expertise in the cohort lies in soil-water (Figure 3.1g) and to a lesser extent plant-root processes (Figures 3.1f), reflecting the strength of hydrological and hydrogeological expertise in the cohort. The lack of knowledge of growth processes suggests an explanation for South Australia’s simple empirical approaches to accounting for forest water use, a point reflected in the salience of empirical methods and expert opinion among other tools used in Figure 3.2.
Mechanistic models and field measurements were more widely used than plant process models and only a limited number of tools were actually used compared to a wider but still limited knowledge of other approaches (Figure 3.2, ‘used’ versus ‘known’).

Finally, responses to Question 10 indicated that agency initiatives targeted at identifying key knowledge and capacity building priorities to support the NWI were limited and staff awareness of them was very low, indicating either a lack of activity in the area or a lack of staff engagement.

### 3.4.3 Discussion

The survey indicates a well-educated cohort of professionals engaged in forest water policy development and management, PhDs comprised 30% of the sample indicating an applied scientific background rather than research.

South Australia is regarded as a leader in forest water management (CCWPL, 2012; Jasper, 2015; WIA, 2012a). It appears to have attained this position using relatively unsophisticated empirical methods and without regard to a significant number of climatically-driven processes that drive plantation water use, a consideration that will need to be taken further into account as climate change begins to exert greater impacts.

Needless to say, expertise in soil-water processes will not be sufficient to account for the impacts of plantation water use in a changing climate, however the cohort was largely blind to the role of climatically driven processes in determining plant water use.

Moreover a programme of internal capacity building as per the NWI (s 98) is not in place to address these needs. Growth in scientific capacity is sought in terms of establishing cooperative research partnerships with external agencies (DEWNR, 2016). This study highlights how care is needed to deliver the required management outcomes using this approach, which inevitably retains a perspective biased toward the research side of the science-policy interface. Safely situated on the research side of the science-policy interface the approach can only do much to narrow it, without seeking to cooperatively grow internal agency knowledge and capacity.

### 3.5 Discussion

The jurisdictional responsibilities for water management under Australia’s federated constitution make for a fragmented institutional environment that complicates the
management of water resources (Fisher, 2000; Godden, 2005; Kildea and Williams, 2010). National water reforms are typically progressed through IGAs, formed from political responses to natural disaster which have delivered uneven outcomes due largely to the considerable jurisdictional discretion they preserve.

In terms of managing the impacts of large-scale afforestation activities, the NWI has had limited impact in so far as jurisdictions had begun developing or had enacted land-use change policies and legislation in regions where the effects were felt before the NWI was signed. In 2004, South Australia was the only jurisdiction to enact such legislation and remains the only one to have done so following ten years of national reform later. Victoria and Western Australia are currently in the process of reforming forest water law as they were before the NWI commenced.

Fundamental to these initiatives was prolonged drought and concerns regarding water availability under a changing climate. The same issues prompted national reforms in accordance with Murray-Darling Basin agendas, which proved ineffective and even problematic in progressing on-ground, regional management outcomes (Greenwood, 2013, Section 3.3.1).

Meanwhile the qualified terms of the NWI permitted other jurisdictions with recognised afforestation issues (Tasmania and NSW, Duggan et al., 2008) to shelve further action, arguably inconsistent with the intent of the NWI in protecting the integrity of water access entitlements from unregulated growth in interception (COAG, 2004b, s 25(xii)).

The wider context of managing interception under the NWI includes knowledge and capacity building (Section 3.1). One hundred and seventy eight activities were undertaken to this end under the NWC Raising Water Standards programme (NWC, 2013). At its conclusion the performance of the programme was assessed through a mixture of reviews, surveys and consultation with decision-makers, experts and NWC executives and staff. The assessment found that knowledge, information and tools delivered through the program had supported improved decision-making and many project outputs had a tangible and wide-reaching impact, pointing to improved water management and implementation of water reforms (NWC, 2013).

While the evaluation acknowledged that gaps remained in the overall picture of national water reform, the findings of this work suggest that at least from the perspective of contributing to the management of interception by large-scale plantation forestry, the NWI's contribution is not particularly tangible or wide-reaching.
PART II

FOREST HYDROLOGY

AND

WATER RESOURCE ASSESSMENT

FOR

FOREST WATER MANAGEMENT
4. HYDROLOGICAL MODELLING FOR FOREST WATER MANAGEMENT

It now seems clear that the computer has done a disservice to many branches of science ... As Fiering (1976) puts it, “Fascination with automatic computation has encouraged a new set of mathematical formalisms simply because they now can be computed; we have not often enough asked ourselves whether they ought to be computed or whether they make any difference ....”


MANAGEMENT frameworks developed in South Africa and Australia to support the regulation of plantation forest water use are underpinned by hydrological modelling. The initial phases of forest water regulation in both countries were underpinned by simple empirical models (DWAF, 1999a; Greenwood and Cresswell, 2007; GSA, 2009; Scott and Smith, 1997; Van der Zel, 1995). Complex mechanistic models have been adopted in South Africa (DWAF, 2005; Gush et al., 2002a) where alternatives, a number of which were developed in Australia (Dye et al., 2002; Dye and Croke, 2003; Dye et al., 2008b), continue to be evaluated for their management potential.

The similarities between the Australian and South African experience are striking, featuring expansion of plantation developments into areas with limited water availability and concerns of established users regarding the security of their access to water (Dye and Versfeld, 2007; Dye and Bosch, 2000). While the need for more sophisticated assessments has been recognised in Australia (Prosser and Walker, 2009), a comprehensive review of the South African experience has never been compiled in the context of informing Australian forest water management.

4.1 An Overview of Empirical and Mechanistic Approaches

Different systems may be modelled using different approaches to ensure that parameterisation is consistent with the assumptions made to describe the processes of interest and that parameters can be identified with reasonable confidence (Clarke, 1973; Dooge, 1968). The hydrological impacts of plantation forestry are determined by the interplay of conceptual climatic and environmental (site) mechanisms (Dye, 1996a; Landsberg and Sands, 2011), which lend themselves to mechanistic or deterministic modelling that is, phenomena can be described by mathematical relationships that provide an output that is invariably determined by a given input (Clarke, 1973; Dunin, 1975). Mechanistic models may be further categorised as either lumped or distributed,
according to the degree to which they represent catchment processes at smaller scales (Chow et al., 1988).

Simple, lumped mechanistic models which concentrate on the replication of observations with little regard to underlying processes have also been described as empirical. Clarke (1973) used the empirical to describe simple models of this type with parameters that reflected physical processes. The use of the term empirical in forest hydrology is more literal, and may include cartographic lines (Holmes and Sinclair, 1986; Van der Zel, 1982) and simple functions without any particular physical basis (see those described in Body, 1975).

Their simplicity, transparency and often graphic character has made empirical models useful in disseminating complex information to stakeholders (Bots et al., 2011; Brugnach et al., 2007; Van Daalen et al., 2002) and supporting management frameworks (Passioura, 1996). Their capacity to represent acceptable compromises as the best available information in a field characterised by complex natural systems and a dearth of data has been fundamental to their role forest water management (Greenwood, 2013, Section 3.3.1 this work; Scott and Smith, 1997; Van der Zel, 1995).

Their direct reliance on site-specific observations means that an empirical model’s integrity should not be judged by its accord with local data alone, but its capacity to represent conditions beyond those in which they were developed or their underlying data collected. The degree to which empirical models can achieve this is termed their robustness (Passioura, 1996).

Distributed, mechanistic models are necessarily more complex than their lumped counterparts, seeking to describe physical processes (Clarke, 1973). The distinction between the simplest mechanistic models and those described as empirical above is arbitrary, however some level of theory around conceptual processes should be evident (corresponding to the deterministic-empirical models of Clarke, 1973). Consequently the term mechanistic may encompass conceptual, physically-based, process-based (Beven, 1993) or even bottom-up models (Sivapalan et al., 2003a).

Models of this type have the potential to characterise important processes ignored by simple models. They may improve prediction according to existing, changed or new physical conditions and enable parameter estimation through field measurements (Abbott et al., 1986a in Beven, 1989; Dye, 2001). Their superior flexibility in replicating site conditions and plant responses compared to empirical models resulted in their adoption for forest water management in South Africa (DWAF, 2005; Gush et al.,
2002a). However the success of complex models has been challenged due to a range of issues that contribute to uncertainty including tenuous links between model functions and reality, multiple optimal parameter sets and difficulties in validating outputs (Beven, 1989; 1993; Franks et al., 1997; Grayson et al., 1992).

Uncertainties associated with modelling used to support decision-making have begun to attract increased scrutiny from planners and regulators (Beven, 2000). Recognised limitations in South African mechanistic modelling led to the investigation of the viability of using plant process-modelling in water management (Dye, 2001; Dye et al., 2008b). Plant process models are a class of mechanistic models developed to predict plant growth as a response to complex environmental and physiological processes (Battaglia and Sands, 1998; Ford and Bassow, 1989; Landsberg, 2003; Reed and Waring, 1974). Plant process models offer the potential to predict plantation water use arising from growth rates determined by site characteristics, a key requirement of water managers who face the consequences of competing water use at small-scales. However it is unclear whether uncertainties in such complex models can be overcome to provide the confidence required in water management decision-making.

4.1.1 Some historical context

Forest hydrology is based on a rich legacy of empirical observation. Its persistent use of empirical methods reflects the role of measurement in production forestry and the traditions of catchment studies (Ice and Stednick, 2004; McCulloch and Robinson, 1993; O'Loughlin and Bren, 1982). Empirical observations of periodic tree growth increments have been systematically recorded since the late 18th century (Schwappach, 1904). The use of empirical site-growth curve relationships became widespread during the 19th century (Roth, 1916) and integrated into timber yield modelling frameworks in the early 20th century (Bates, 1918; Frothingham, 1921; Watson, 1917; Woodward, 1917), an approach still used in contemporary forest management (Carmean et al., 1989; Cieszewski et al., 2007; Monserud, 1984; Rutishauser et al., 2013).

Unfortunately when empirical evidence is scarce, weakly founded and at times misguided belief systems have a tendency to fill knowledge gaps. A paucity of available hydrological data at the end of the 19th century resulted in a general consensus of scientific opinion that forests promoted rainfall and that afforestation with exotic species should yield similar benefits (Wicht, 1959), a belief that extended well into the 20th century (McCulloch and Robinson, 1993).
Complex mechanistic modelling commenced in the early 1960s in both hydrology (Crawford and Burges, 2004; Crawford and Linsley, 1966; Dawdy and O’Donnell, 1965; Donigian and Imhoff, 2006) and plant biology (De Wit, 1965; Duncan et al., 1967; Loomis and Williams, 1963; Sinclair and Seligman, 1996; Thornley, 1972). In the 1970s a veritable flood of mechanistic models followed, in both hydrology (Beven and Kirkby, 1979; Chapman and Dunin, 1975; Dooge, 1975; Stephenson and Freeze, 1974; Swank, 1981); agro-hydrological plant modelling, the latter encompassing plant processes, sophisticated soil moisture structures (Holtan and Lopez, 1971; Holtan et al., 1975; Ritchie, 1972; see also Van Keulen et al., 1981 and references therein) and specialised tree growth models created to explore physiological responses to environmental variables (Reed and Waring, 1974; Running et al., 1975).

Forest hydrology was slow to embrace early computer models because of their limitations in representing responses from non-static land-cover (Swank, 1981). Efforts to address these limitations included integrating empirical growth relationships into established mechanistic hydrological models in South Africa (Pitman, 1973; 1978) and the development of more sophisticated models, representing more complex soil-vegetation-atmospheric interactions in the USA (Federer and Lash, 1978; Knisel, 1980; Rutter et al., 1975; Goldstein & Mankin, 1972 in Swift et al., 1975). By 1980, mechanistic models had emerged with the capacity to explicitly assess the hydrological responses of changing land-use (Aston and Dunin, 1980).

Throughout the 1980s, hydrological (Beven and Wood, 1983; O’Loughlin, 1986; O’Loughlin, 1990), agro-hydrological (see Williams et al. 1983; Ritchie 1985; Littleboy et al. 1989 in De Jong and Bootsma, 1996; Schulze, 1984a) and plant process modelling (McMurtrie and Wolf, 1983; Running, 1984; Running and Coughlan, 1988) attained higher levels of sophisticated determinism, albeit with varying degrees of empiricism, particularly in describing site-specific growth relationships (Affholder et al., 2012; Battaglia and Sands, 1998; Korzukhin et al., 1996; Mäkelä et al., 2000). Accordingly, forest hydrology modelling approaches became more sophisticated, exploding in a number and variety too wide to be described here (Beckers et al., 2009).

Modelling systems have been developed to support forest water management in the two jurisdictions where the hydrological impacts of afforestation are controlled (Section 2 and 3), however the most sophisticated empirical system for assessing land-use change was developed in the United States and is still used in the hydrological
modules of widely used sophisticated mechanistic models (Arnold et al., 2012; Knisel, 1980; Littleboy et al., 1999).

### 4.2 North America

While the water used by afforestation is not regulated in United States, land-use change research has a long history with significant implications for water management, mostly with regard to increasing its availability through clearing (Ponce and Meiman, 1983).

![figure1](image)

**Figure 4.1:** USDA-NRCS (2004) curve numbers (Figure 10.2, from Chapter 10 Estimation of direct runoff from storm rainfall).

Protracted study programs at dedicated research stations facilitated the development of empirical rainfall-runoff relationships (Bates and Henry, 1928; Hoover, 1944; Rallison, 1980), culminating in USDA curve numbers (USDA-NRCS, 2004) originally developed in the 1950s to predict event-based runoff from different land-cover complexes for flood and catchment management (Ponce and Hawkins, 1996; Rallison, 1980; Williams et al., 2012) (Figure 4.1).

The system went on to become the most widely used rainfall-runoff assessment method in the world (Boughton, 2005), and despite well-articulated concerns around its empirical limitations (Ponce and Hawkins, 1996), has found application in a range of hydrological, land-use change, erosion and water quality models (Bingner et al., 2009;
Knisel, 1980; Littleboy et al., 1992; Neitsch et al., 2005; PRF, 2004; Schulze, 1984a; USDA-NRCS, 2002; 2009; Williams, 1995; Young et al., 1989).

4.3 South Africa

South African forest hydrology research has been targeted at providing information to support policy development and legislation to regulate the impacts of afforestation on water resources (Le Maitre and Versfeld, 1997). Two generations of empirical tools were used before mechanistic agro-hydrological modelling was finally adopted. While the developments have been generally acknowledged as improving management concerns regarding the system remain (Dye and Versfeld, 2007).

4.3.1 Empirical approaches

In 1935 the South African government established the Jonkershoek Forest Influences Research Station near Stellenbosch, in the winter rainfall region of the south western Cape. A second research station was established to collect information in the summer rainfall areas at Cathedral Peak in the Drakensberg Range of KwaZulu-Natal in 1938.

The research catchments were equipped with high-quality hydrometric recording equipment and their climatic and environmental characteristics scrupulously recorded including the composition and rate of growth of native, pre-treatment vegetation communities. Treated catchments at Jonkershoek were mostly planted with Pinus radiata and P. patula at Cathedral Peak (Nänni, 1956; Wicht, 1948; see also Wicht, 1949).

_Midgley and Pitman, Nänni and Van der Zel curves_

In the late 1960s South Africa completed the first in what was to become a series of national (WR90) water resources assessments undertaken for broad planning purposes (Midgley and Pitman, 1969). The study developed an approach to estimating regional mean annual runoff using a rainfall-runoff power function. Optimal parameters were estimated using observed data, which were then regionalised according to physio-climatic catchment characteristics, and used to generate mean runoff estimates at quaternary sub-catchment scales (a quaternary sub-catchment represents the fourth level of catchment subdivision, where the primary catchment represents an entire river system, Edwards and Roberts, 2006). Of all the catchment characteristics associated with runoff, land-cover was found to exert the strongest influence (Midgley and Pitman, 1969).
Figure 4.2: (a) Nänni (1970b, Figure 1) curves, from Schulze and George (1987, Figure 4), overlain with Van der Zel (1982) runoff-runoff reduction curves in red (from Figure 1b) for comparison. (b) Van der Zel (1982, Figure 1) curves for 15- (left) and 40-year (right) rotations, taken from Van der Zel (1997, Figure 3).

Nänni (1970a; 1970b) published a Midgley and Pitman (1969) rainfall-runoff curve modified in its upper parts to account for data collected from paired catchment studies at Cathedral Peak. The curve was accompanied by average, post-treatment runoff reduction curves for maximum impacts observed during 15- and 40-year plantation rotations (Figure 4.2a).

In 1972 forest water use was regulated under the Afforestation Permit System (APS) and by 1974 the South African government had developed a system to support the definition of maximum permissible plantation areas according to their estimated impacts on water resources at catchment and sub-catchment scales (Van der Zel, 1982) (Figure 4.2b).

The system was broadly based on Nänni curves, modified to reflect ideas from South African and international assessments, specifically: the global review of Hibbert (1967); the deforestation studies of Hewlett and Hibbert (1961) at Coweeta in the USA; and unpublished government reports not sighted in this study (McDonald, 1973; Aspeling, Roberts and Language, 1973, both cited in Afrikaans; and Wand, 1974) (Van der Zel, 1995).
The curves so derived were compared to a global dataset in another unsighted reference (Van der Zel, 1981) and considered a safe over-estimate of experimentally found results, before being accepted as the basis of calculation by a central government committee in July 1974 (Van der Zel, 1995). The adjustments resulted in greater reductions in lower rainfall areas than those described by the Nänni curves, and lesser reductions in higher rainfall areas (see superimposed curves in Figure 4.2a).

The basis of assessment was not limited to runoff reduction curves. A planning system was established which limited the levels of water resource development according to established levels of development at a catchment-scale, and prescribed buffers around water bodies. Catchments where competing priorities for water were so great that no further afforestation could be allowed were termed Category I. Category II catchments were already heavily committed but were considered able to support additional afforestation to the extent that it would not reduce mean annual yield by more than 5%. The remaining catchments were designated Category III and were assigned an arbitrary use limit of 10% of pre 1972 mean annual runoff (Van der Zel, 1995). Buffers were initially set at 10 metres for perennial streams, and were gradually increased up to 30 metres and 50 metres for wetlands (Edwards and Roberts, 2006; Van der Zel, 1995).

Despite its success in facilitating sustainable plantation expansion (Van der Zel, 1995), the empirical limitations of the basis of assessment posed problems to industry, researchers and water managers alike, including:

- observations made in high rainfall areas were not representative of where plantation forestry expansion was occurring;
- there was no account of the plant growth processes that drive tree evapotranspiration;
- relationships were based on pine trees replacing a montane grassland under an assumed and fixed management regime;
- impacts did not reflect the changing water use over the life of the rotation or in response to management practices;
- the mean annual time-step did not accommodate considerations of inter- or sub-annual variability or impacts on different parts of the flow regime, particularly low flows;
- a lack of documented rigour in their derivation;
• buffer widths were not supported by science (Bosch, 1982; Bosch and Von Gadow, 1990; Dye and Bosch, 2000).

Persistent dissatisfaction and the potential demonstrated by more sophisticated approaches (Boden, 1991; Le Maitre and Versfeld, 1997; Schulze and George, 1987), led to the development of more sophisticated empirical runoff reduction models, which were able to address limited differences between site characteristics, flow regimes and tree species.

In what was to prove a giant step toward future water resource planning the early 1990s saw the afforestation permit system delivered within a GIS framework, enabling the integration of a range of spatial land-use data available at different scales (Van der Zel and Rabe, 1993). The combination of new models into GIS set the scene for a new era in forest water management (Scott et al., 1998).

**Scott and Smith or CSIR curves**

In a review of results from South African paired catchment studies, Bosch (1982) plotted observations of changes in annual evapotranspiration at four sites with hand-fitted curves. Differences in response over time were attributed to differences in climate, stand structure, growth rates and physiology. Bosch and Von Gadow (1990) updated the data with an additional two sites in their critique of the Afforestation Permit System (APS), and fitted them with the sigmoidal Chapman-Richards function, the basis of many empirical tree growth models (Pienaar and Turnbull, 1973):

\[ E_t = A (1 - e^{-kt})^m \]  

where \( E_t \) is evapotranspiration, \( t \) is time in years and \( A, k \) and \( m \) are parameters.

By the early 1990s, impacts on low flows and the different impacts of different species in South Africa were established (Smith and Scott, 1992). The need to take account of these influences when considering permit applications led the South African government to commission the Council for Scientific and Industrial Research (CSIR) to develop models to better depict total and low-flow reductions due to afforestation by either eucalypts or pines, growing under optimal or sub-optimal conditions (Smith and Scott, 1992 in Scott and Lesch, 1995).
Figure 4.3: CSIR/Scott and Smith (1997, Figure 4) age-runoff reduction curves for eucalypts. Curves are differentiated with regard to total annual streamflow and low flows under optimal and sub-optimal growing conditions related to rainfall and temperature. Similar curves were developed for pines (see Scott and Smith 1997, Figure 3).

The work was later published by Scott and Smith (1997), who adopted an approach where streamflow reductions were expressed as a percentage of the expected flow, removing year-by-year variation caused by climatic fluctuations and facilitated fitting curves to reductions as a function of plantation age. Pine data came from *Pinus patula* plantations at Cathedral Peak and Mokobulaan B and *P. radiata* plantations at Jonkershoek. Eucalypt data came from *Eucalyptus grandis* plantations at Westfalia and Mokobulaan A. The result was a series of four curves each for pines and eucalypts, depicting proportional reductions for low and total-flows, at optimal and sub-optimal sites (Figure 4.3).

Cathedral Peak and Jonkershoek data were found to show close accord and were pooled into a single dataset. Mokobulaan B reductions were distinctly different from those of Cathedral Peak and Jonkershoek which was attributed to site qualities, giving rise to a classification of *optimal* to Mokobulaan and Westfalia, known for their productivity due to high rainfall, deep soils and temperate climate. As both eucalypt sites were optimal, a set of *sub-optimal* eucalypt curves were synthesised based on the differences noted between optimal and sub-optimal pine sites.
The authors’ recognised that the use of proportional reductions may unintentionally imply the same relative impacts of afforestation in all rainfall zones and stressed that extrapolation into drier areas (mean annual precipitation <900 mm/year) was uncertain. A compounding issue not addressed was the likely influence of the diverse pre-forest land-covers, which ranged from montane grassland, to native scrub to a partially (57%) afforested catchment. However the authors’ took the view that in fitting pooled data across wide climatic and geographical ranges for two different (pine) species, the models demonstrated convincing robustness (Scott and Smith, 1997), and were incorporated into a spatial system to support South African forest water regulation (DWAF, 1999a; Le Maitre and Scott, 1995; Scott et al., 1998).

The system represented a marked increase in the sophistication of forest water impact assessment and accounting. Impacts due to afforestation were adjusted to account for planning constraints around riparian buffers, while impacts at dry sites were considered optimal in accordance with the observation that the driest catchments were quickest to reach the maximum proportional reductions (Scott et al., 1998).

Quaternary sub-catchments were used as the minimum hydrological planning unit. Sub-quaternary mean annual rainfall for plantation sites was estimated with each using aerially weighted WR90 national water resource assessment isohyets (Midgley et al., 1994 in Scott et al., 1998). Mean annual low flows were estimated from a frequency analysis of WR90 monthly flow data and assumed to be uniform across each quaternary sub-catchment.

The reference yield against which proportional streamflow reductions are quantified is of fundamental importance, particularly when reductions are used for legal purposes with the potential to limit economic activity (DWAF, 1999a). Mean annual yield data from native scrub was estimated using the WR90 empirical rainfall-runoff relationships for different climate-related zones across South Africa (Scott et al., 1998). Because the yield from pre-afforestation native scrub was generally not known, it was agreed that natural vegetation would be represented by the land-covers of Acocks (1988), which provide an indication of natural vegetation had probably been, across the country (Edwards and Roberts, 2006).

Despite their greater sophistication in incorporating different tree genera, species and a wider range of site characteristics, the applicability of the CSIR curves remained limited because the range of conditions at the research sites still did not reflect the diversity of most prospective plantation sites or contain the flexibility to depict the influence of different forest management practices (Dye and Bosch, 2000).
4.3.2 Mechanistic approaches

Concerns regarding the CSIR curves eventually combined with a need for more realistic depictions of hydrological responses to plant growth and silvicultural practices to drive the evaluation of a mechanistic agro-hydrological model for use in forest water management.

In 1979 the South African Water Research Commission funded a five year program to assess flood events through hydrological modelling. The project aims were threefold, comprising: the further development of the USDA curve number method for South African conditions; the development of a hydrological soil classification for use in Southern African modelling; and the development and testing of other hydrological models for eventual general application to small catchments in Southern Africa. The latter was met through the development of ACRU, which became the basis of assessment for streamflow reduction activities in 2005 (DWAF, 2005). There is a fair bit of hydrological distance between flood prediction and land-use modelling, so why choose ACRU?

ACRU

ACRU was developed in the Agricultural Catchment Research Unit (ACRU) of the Agricultural Engineering Department at the University of Natal. Its origins in an agricultural rather than a civil engineering paradigm give an indication of its intended application in depicting conventional surface and sub-surface hydrology as well as conceptualising vegetation water use from physically-based soil moisture stores (Schulze, 1984b). This physical-conceptual approach reflected its intended application in exploring scenarios within the context of its physical limitations and measurable or derivable catchment characteristics, rather than providing calibrated outputs (Jewitt and Schulze, 1999; Schulze, 1984a; Schulze and George, 1987).

The model uses a daily time step and empirical relationships to depict plant growth-water use from multiple soil horizons derived from local and international studies (Schulze, 1984a; 1984b). Soon after its initial development Schulze and George (1987), reviewed the limitations of the empirical curves used to underpin the APS and, recognising the needs of forest managers to dynamically model changes in tree water use with growth or management treatments, began addressing the problem within ACRU.
Contemporary physically-based forest-water models were considered (Federer and Lash, 1978; Rogerson 1982; Simpson and Fritschen 1982 in Schulze and George, 1987), but were rejected as not having a dynamic structure and being too difficult to adapt to local conditions. ACRU modifications included incorporating canopy interception and evaporation rates as empirical functions of rainfall, leaf area index (LAI) and potential evapotranspiration. Changes in water use in response to growth were accommodated in a dynamic file containing monthly changes in relevant variables and parameter values such as LAI, the actual to potential transpiration ratio, stress thresholds, rooting depth or proportion of roots in the topsoil. The estimated rates of change were based on values published in South African and international literature, and expert advice. Early trials were considered successful but improvement was envisaged with the inclusion of additional hydrological parameters and more realistic growth relationships (Schulze and George, 1987).

Subsequent improvements included distributed catchment representation (Tarboton and Schulze, 1991) and the capacity to model different tree genera, the impacts of leaf litter and site preparation into a decision-support system (Jewitt and Schulze, 1991). In what appears as anticipation of its use under the new National Water Act 1998, ACRU was verified on three research catchments by its developers and deemed viable for supporting decision-making when used in conjunction with the forest decision support system, despite poor results at two of the sites (Jewitt and Schulze, 1999).

Apparently encouraged by this work a joint venture was formed between CSIR and ACRU developers to further verify the model and produce tabulated streamflow reductions acceptable to a wide group of stakeholders (Gush et al., 2002b). In 2005 ACRU supplanted the CSIR curves in underpinning forest water regulation (DWAF, 2005; Gush et al., 2002a).

**Other work**

The formal prescription of modelling approaches under South African law have not precluded the evaluation of other approaches (see for example Dye and Croke, 2003). While South African forest hydrology research focused on supporting water regulation, forestry research pursued a parallel agenda of improving productivity by manipulating forest management practices without giving adequate attention to water management.

Le Maitre and Versfeld (1997) attempted to quantify plantation water use in terms of stand growth to support decision-making under the APS. Outputs from empirical site index growth equations regressed against observed catchment water balance data
from research sites gave viable predictions of mean annual evapotranspiration compared to Van der Zel and CSIR curves, while also providing additional flexibility to incorporate some forest management regimes and insights into plant-water dynamics.

Greater attention has been given to evaluating process models, notably those from Australia, which allowed the separate influences of weather, stand characteristics and site conditions on carbon allocation patterns and transpiration rates to be assessed (Dye et al., 2001). The most promising models 3-PG (Landsberg and Waring, 1997; Sands and Landsberg, 2002) and WAVES (Zhang et al., 1996) are capable of simulating growth and water use over a wide range of conditions. In the earliest evaluations (Gush, 1999 in Campion et al., 2005; Dye, 2000a in Dye et al., 2001; Dye, 1999 in Dye and Bosch, 2000), 3-PG which showed great promise in improving the understanding of links between growth and water use, but more work was required to place predictions on a firmer scientific footing, such as species parameterisation and the collection of a broader range of interpretive data (Dye, 2001).

An evaluation of a spatial version of 3-PG and the use of remote sensed leaf data confirmed the model’s potential and identified the need for detailed soil data and highlighted several limitations concerned with soil water interactions (Dye et al., 2002; 2004). The importance of realistic, good quality, local data remained a key issue in the performance of both 3-PG (Campion et al., 2005; Esprey et al., 2004) and WAVES (Dye et al., 2008a; Dye et al., 2008b), but 3-PG parameterisation could be achieved over a relatively short period for a range of species, using relatively simple field equipment and techniques (Dye, 2001; Dye et al., 2004). Recognition that different models have the capacity to suit different forest types and the growing local database for some genera has seen 3-PG continue to be employed in South African plantation water use studies (Dye et al., 2008a; Wise et al., 2011).

While the South African work has been encouraging some 3-PG parameters, specifically maximum available soil-water and soil fertility, remained difficult to measure in the field and required indirect estimation (Campion et al., 2005; Dye et al., 2004). Parameterisation was found to be a progressive process, with improvements gained through experience (Dye, 2001), suggesting a potential for encapsulating expert opinion rather than an objective interpretation of mechanistic causality (see also Landsberg and Sands, 2011, Chapter 9).
4.4 Australia

Unlike South Africa, Australian research into land-use change and catchment yield were not driven by a national agenda. The scale of the issue and disparate jurisdictional research priorities has meant that Australian forest hydrology has lacked the consistency of purpose to support one (Bren and McGuire, 2007; Costin and Slatyer, 1967; O’Loughlin, 1988). Until quite recently, the risks of afforestation to water availability were considered subordinate to its benefits to water quality and land management (Section 3.3.1). It is unsurprising then, that the paucity of hydrological afforestation data in Australia has been noted in a number of studies (Brown et al., 2005; Lane et al., 2005; O’Loughlin, 1988; Vertessy, 1999) with particular implications for the development of empirical models.

4.4.1 Empirical approaches

The first Australian afforestation models did not appear until the 1980s (Holmes and Sinclair, 1986; Kuczera, 1987), trailing mechanistic hydrological land-use change modelling by some years (Aston and Dunin, 1980). Holmes and Sinclair curves were developed using information from 19 sites in Victoria and South Australia (Figure 4.4, below). Based on data from relatively high rainfall areas, the authors considered the curves to be unsuitable for use in catchments drier than 900 mm/year (p 218, Holmes and Sinclair, 1986).

Holmes and Sinclair “forests” included mixed eucalypt native vegetation and pine plantations, while the Kuczera curve was based on native Eucalyptus regnans re-growth forest in the water supply catchments of Melbourne. Later work extended the application of both of these relationships into assessment tools (Holmes and Sinclair curves in MAYA, Vertessy et al., 2000; Kuczera curve in Macaque, Watson et al., 1997; later modified by Watson et al., 1999).

As a consequence of its limited database, Australian empirical afforestation modelling approaches have tended to be

Figure 4.4: Evaporation-rainfall curves for (Holmes and Sinclair 1986, Figure 3).
used outside their range of applicability. Holmes and Sinclair curves were used in the cleared sections of mid-to upper Murrumbidgee catchment in NSW (Vertessy and Bessard, 1999), where the majority of which has an average annual rainfall of between 500 and 800 mm/year (Green et al., 2011). The quality of the results was mixed. However, the tendency of mean annual data to dampen inter-annual variability and selectively reported performance criteria enabled the results to be described in positive terms to encourage further application at other sites.

A lack of rigorous scrutiny also led to the widespread misuse of another Australian empirical curve system with consequences for the initial stages of forest water regulation in South Australia and ongoing consequences for national water and carbon policy development (Section 3.3.1).

**Zhang curves**

At around the same time, the most influential empirical afforestation models to emerge from Australia were developed (Zhang et al., 1999; 2001). Like Holmes and Sinclair curves, Zhang curves represent conceptual relationships between mean annual rainfall and evapotranspiration for grassland and forest (Figure 4.5).

![Figure 4.5: (a) Zhang et al. (2001, Figure 9) rainfall-evapotranspiration curves. (b) Zhang rainfall-runoff curves taken from Zhang et al. (2003, Figure 6).](image)

Their derivation used the functional form of Milly (1994) and the boundary conditions of Budyko (1974). A plant available water coefficient (\(\omega\)) was used to represent the ability of plants to access water within the root zone, similar to Milly’s soil water capacity-precipitation ratio. Zhang et al. (1999; 2001) assumed \(\omega\) took a value of 0.5 for “grassland” and 2.0 for “forests” and estimated values for mean annual
evapotranspiration relative to each land-cover using non-linear regression and data from over 250 sites around the world. By assuming a steady-state in the catchment water balance where the long-term change in storage could be ignored, the curves were translated into mean annual rainfall-runoff curves for conceptual grassland and forests land-covers (Figure 4.5b). While no mention was made of their range of climatic applicability, uncertainties were recognised as greater in areas of low rainfall (Zhang et al., 1999, p 22).

Zhang curves have been endorsed by influential Australian scientists (Brown et al., 2005; Vertessy, 2001; Vertessy et al., 2003) and leading government research organisations (BRS, 2003; Zhang et al., 2007). They have used as the basis for runoff reductions in a number of forest hydrology modelling products (Brown et al., 2006a; 2006b; Dawes et al., 2004; HTC, 2007) and have found wide application in a range of studies both in Australia and internationally (Bren et al., 2006; Brown et al., 2007; Herron et al., 2002; Sun et al., 2005; 2006; Wang et al., 2008; Zhang et al., 2003b; 2010; Zhang et al., 2008 among others).

Decision-making and policy development

A feature of the Zhang curves is the status they have attained as established relationships in Australian forest hydrology (BRS, 2003). As the backdrop to the national capital, the Murray-Darling Basin plays an important role in shaping Australian water policy development (Section 3.3.1) and Zhang curves have played a prominent role in a series of significant Murray-Darling assessment, none more so than in supporting the development of the Murray-Darling Plan through the Murray-Darling Sustainable Yields Project (CSIRO, 2008c; Post et al., 2012) where they were used in the flow duration curve adjustment approach of (Brown et al., 2006b). They were key in assessing the benefits of afforestation during the salinity crisis that pre-dated the National Water Initiative (Bradford et al., 2001; Dowling et al., 2004; Van Dijk et al., 2004; Zhang et al., 2003b); extended their influence in water security assessments after the National Water Initiative (Polglase and Benyon, 2009; SKM et al., 2010; Zhang et al., 2010); found application in economic assessments (Hafi et al., 2010; Polglase et al., 2008) and the impacts of climate change (Austin et al., 2010).

Smaller-scale assessments have been conducted to evaluate their capability to support water management modelling (Brown et al., 2007; Herron et al., 2002), while in South Australia, they were used to support plantation forestry expansion in a water supply catchment (Bren 2004, in Greenwood, 2007) and formed the initial basis for
assessing the impacts of afforestation on significant wetlands in district council planning frameworks (DWLBC-PIRSA, 2007).

However, variance between Zhang curves and Australian data (Figure 4.6, see also Greenwood, 2007, and comments by Herron et al., 2002, p 375; Podger, 2005, p 32; Brown et al., 2006b, p 13), prompted South Australian water managers to develop new empirical relationships based on local data, simple rainfall-runoff functions and South African CSIR runoff reduction curves (Greenwood and Cresswell, 2007) (Figure 4.7).

**Figure 4.6:** Figure 6 from Zhang et al. (2003) (see Figure 4.5(b) above), reproduced with the Australian data of Zhang et al. (1999). Data below 1000 mm/year are not segregated by land-cover according to the hypothesis inherent in the curves.

**Figure 4.7:** Greenwood and Cresswell curves. (a) pre- and post forest rainfall-runoff curves (Greenwood and Cresswell 2007, Figure 6). (b) age-runoff reduction curves (Greenwood and Cresswell 2007, Figure 8). Red and blue curves are from Scott and Smith (1997, Figure 3).
Inconsistencies arising from the competing assessment systems eventually destabilised planning frameworks, requiring legal intervention to clarify decision-making processes (GSA, 2008). The situation was not fully resolved until an independent review was conducted to clarify the matter (Prosser and Walker, 2009). In the end, runoff reductions quantified using Greenwood and Creswell curves were incorporated into a state-wide forest water policy framework, augmenting relationships developed in the State’s South East to manage groundwater impacts (GSA, 2009) (Section 3.3.1).

Despite their questionable robustness under Australian conditions the presence of Zhang curves in national-scale assessments continues, most recently in integrated water-carbon assessments related to CO₂ mitigation (Polglase et al., 2013; SMEC, 2010) where they have entered Australian law as the basis of calculating water offsets required in the establishment of plantations for carbon sequestration (CA, 2011; Roberts, 2014).

4.4.2 Mechanistic approaches

Compared to mainstream hydrology, forest hydrology was slow to embrace complex computer modelling (Swank, 1981). Australia was no exception, despite a conspicuous presence in the vanguard of early hydrological and agro-hydrological computer modelling (Aston and Dunin, 1980; Boughton, 1966; Chapman, 1970; Fitzpatrick and Nix, 1969; Mount, 1972 in Kuczera, 1988; Porter and McMahon, 1971). In the first Australian national symposium on forest hydrology, only two of the 22 papers presented were directly concerned with computer simulation (O’Loughlin et al., 1982; Ronan et al., 1982). While ensuing modelling studies did not form part of a cohesive, nationally supported research program (O’Loughlin, 1988), the sophisticated mechanistic approaches they produced were eventually progressed under national agendas, but without exploring their potential in water management.

In the 1980s, O’Loughlin (1981; 1986) developed a saturated partial area concept of runoff based on a geomorphic approach to hydrological modelling proposed by Beven and Kirkby (1979) and Beven and Wood (1983). A number of studies used the concept linked to a mechanistic rainfall-runoff model which utilised empirical functions and pan data to simulate evapotranspiration, with applicability in modelling the hydrological impacts of land-use change under steady state conditions (Grayson et al., 1988; Moore et al., 1986; 1988) or dynamically (Moore and Grayson, 1991). At around the same time, and using similar geomorphic hydrological principles O’Loughlin et al.
(1989) and O'Loughlin (1990) developed a similar model TOPOG, which Hatton et al. (1992) combined with plant process modelling by Wu et al. (1994), that incorporated physiological relationships (Running and Coughlan, 1988; Running and Gower, 1991), becoming the first Australian model to integrate sophisticated hydrological concepts with growth-dependent, process-based evapotranspiration.

In the early 1990s Australian forest hydrology attained a greater consistency of purpose (Costin and Slatyer, 1967) with the establishment of a quasi-national-scale research agenda under the Cooperative Research Centre for Hydrology (CRCCH). The agenda was dominated by eastern state institutional interests (eWater, 2011). Independent lines of forest water research included simple hydrological approaches (Post et al., 1996) and coupled energy-water balance modelling (Silberstein and Sivapalan, 1993; 1995; Silberstein et al., 1999). Zhang et al. (1996) developed WAVES, a soil-vegetation-atmosphere transfer (SVAT) model based on the plant process modelling of Wu et al., (1994) which was capable of representing relationships within more complex vegetation assemblages, but without the dynamic soil-moisture capacity of TOPOG (Zhang and Dawes, 1998).

In its early days, CRCCH researchers advocated mechanistic over empirical modelling approaches due to their potential superiority in management applications (Vertessy et al., 1996). Work included further development of TOPOG (Hatton et al., 1995; Vertessy et al., 1998; 1996; 1993) and integrating empirical leaf conductance relationships into mechanistic hydrological models (Peel et al., 2000; 2003; Watson et al., 1998). However subsequent work placed heavier emphases on empirical approaches (Bradford et al., 2001; Vertessy, 1999; 2001; Vertessy and Bessard, 1999; Vertessy et al., 2000), doubtless related to the challenges in running mechanistic models (Vertessy et al., 2003) and their onerous data requirements (Dawes et al., 1997).

Simpler approaches were also pursued outside leading research organisations. The need to evaluate hydrological responses to land-cover change in the context of salinity management led to the development of SoilFlux in Victoria (Daamen et al., 2003) and LUCICAT in Western Australia (Bari and Smettem, 2006). Like ACRU, these systems are unable to simulate growth processes per se, requiring leaf area index (LAI) and rooting information as model inputs, nonetheless results are acceptable where sufficient information is available (Bari et al., 2010; Benyon et al., 2008). LUCICAT has the capacity to represent hill-slope processes and unsaturated soil dynamics particularly germane to streamflow production in the south west of WA (Bari and
Smettem, 2006), while Soilflux implements a one-dimensional Richards equation but has also been coupled with MODFLOW to accommodate lateral flow (Daamen and Hoxley, 2003).

While LUCICAT was developed to assess land-use change impacts on salinity (Bari and Smettem, 2006), recent work has been included changes to water availability (Bari et al., 2011; CSIRO, 2010). SoilFlux was used in conjunction with an empirically based streamflow-land-use change model ForestImpact in the Water and Land Use Change (WatLUC) study (see Daamen and Hoxley, 2003). WatLUC investigated land-use change issues in south-western Victoria with a view to supporting water management, and contributed to the initial identification of areas considered to be prospective as prescribed *intensive management areas* under plantation water legislation currently before parliament (DSE, 2011d).

Notwithstanding their sophisticated approaches to modelling soil-water relationships their reliance on empirical growth input data will always limit their flexibility in depicting the impacts of different climatic influences and forest management practices at different sites.

**Plant process models and water impacts**

The routine use of increasingly sophisticated modelling approaches to support decision-making and the need to account for forest growth and management practices in water management have arguably overwhelmed concerns of a decade ago regarding data availability and computational challenges (Dye, 2013; Pittock et al., 2013; Polglase et al., 2013). The integration plant process models into hydrological and agro-hydrological platforms is now well-established, offering the potential to realistically predict the impacts of conditions on plant growth and consequently water use (Miehle et al., 2009; Whitehead and Beadle, 2004).

Plant process models were first developed in the 1970s to explore physiological responses to environmental variables (Reed and Waring, 1974; Running et al., 1975). Their potential to support *forest management* was realised in the 1990s (Battaglia and Sands, 1998; Vanclay and Skovsgaard, 1997), prompting their further development for practical application using readily available data, fewer parameters and generating practical outputs for managers (Battaglia and Sands, 1997; Battaglia et al., 2004; Huth et al., 2001; Landsberg and Waring, 1997).
A number of studies have evaluated the relative strengths of plant process models in predicting growth and support forest management (Cameron et al., 2013; Miehle et al., 2009; Minunno et al.; Van Oijen et al., 2013; Van Oijen and Thomson, 2010), but less work has been specifically targeted at improving the capacity to predict plantation water use (Almeida and Sands, 2015; Feikema et al., 2010b). Examples of work conducted in context of water management are even less common (Feikema et al., 2008; Marcar et al., 2010).

Benyon et al. (2008) evaluated 3-PG (Landsberg and Waring, 1997; Sands and Landsberg, 2002), CABALA (Battaglia et al., 2004) and SoilFlux (Daamen et al., 2003). All were able to produce unbiased predictions of annual water use by plantations with closed canopies that didn’t access groundwater. SoilFlux and 3-PG models underestimated groundwater uptake, while CABALA over-estimated it. CABALA explained the greatest amount of total inter-site variation in mean annual evapotranspiration and provided the best estimates of monthly evapotranspiration and its seasonal variability. SoilFlux and to a lesser extent 3-PG, under-estimated evapotranspiration in winter and over-estimated it in late spring and early summer (Benyon et al., 2008).

SoilFlux model was the best of the three for simulating annual evapotranspiration from plantations without access to groundwater. The high quality LAI data and inferred rooting depth information from the Benyon et al. (2008) study sites gave SoilFlux a considerable advantage over CABALA and 3-PG, which are known to have difficulties in simulating both leaf and root growth prior to canopy closure (Sands and Landsberg, 2002). Further development was considered necessary if process models were to be used to accurately simulate growth and water use of plantations with access to groundwater (Benyon et al., 2008).

Similar conclusions were reached by Feikema et al. (2009), who compared the efficacy of Macaque (Watson et al., 1998) and 3-PG+ on assessing the hydrological impacts of native scrub in Melbourne’s water supply catchments. The absence of a module that simulates growth and water use in response to changing environmental conditions meant that Macaque was unable to reflect leaf growth dynamics across the catchment, limiting its capacity to model evapotranspiration for hydrological purposes. Being capable of modelling leaf growth, 3-PG was superior in this regard, but was better suited to single-species plantations than native forests.

With the possible exception of the work by Benyon et al. (2008), 3-PG has never been formally subjected to a selection process which would identify it as a leading candidate.
for further use among a range of competing alternatives. Nonetheless it has found the
widest application in further exploring the potential of process models in Australian
water management.

Improving the representation of plantation water use in 3-PG

Australian efforts to improve the capability of 3-PG in modelling plant water use
commenced at around the same time as South African evaluations of its potential for
water management (Section 4.3.2) (Morris, 1999; Morris and Collopy, 2001; Claridge
et al. 2001 in Williams et al., 2009).

Morris and Collopy (2001) noted that despite adequate performance of 3-PG in
predicting overall growth, compensating errors in model structure and parameter
values had the potential to compromise the integrity of intermediary results, like water
use. Consequently the model was modified to introduce the capacity to validate the
model against water use variables in an effort to identify structural modifications that
may be required before comprehensive multivariate parameterisation was attempted.
While growth predictions were robust, summer water use was over-estimated, possibly
related to problems representing canopy conductance at times of high vapour pressure
deficit. Notwithstanding these shortcomings, 3-PG appeared to be capable of
predicting annual and probably monthly stand water use with acceptable accuracy
(Morris and Collopy, 2001). Modifications to 3-PG (Morris, 1999; Morris and Baker,
2003; Morris and Collopy, 2001) were described by Morris (2003) in a re-worked
version called 3-PG+, which he concluded could be further improved by refining plant
processes, but a more appropriate approach may lie in developing the model as part of
an integrated catchment-scale framework (Morris et al., 2004) (see further description
in Section 7).

Integrated catchment modelling

A number of studies have integrated catchment modelling using both 3-PG and Zhang
curves, some with particular reference to water management. Claridge et al. 2001 in
Williams et al. (2009), conducted a pilot study in southeast Queensland aimed at
integrating transpiration outputs from 3-PG (3PG-SPATIAL Tickle et al., 2001), a crop
model (APSIM) and a pasture model (GRASP) into a terrain-based subsurface
hydrology model (PHROG), to explore land use options to ameliorate the accumulation
of shallow groundwater (see also RIRDC, 2002). PHROG was developed expressly
to improve the accuracy of water balance modelling for use with 3-PG, and while it was
found effective in the wider study, its limitations in responding to changes in soil
moisture induced by 3-PG modelled water use and conceptual problems associated shallow groundwater dynamics meant that further development was required to achieve satisfactory integration (Williams et al., 2009).

Herron et al. (2002) used a spatial version of 3-PG to develop tree planting scenarios for an assessment of the impacts of afforestation on streamflow in the Macquarie River, catchment in NSW under different global warming forecasts. Zhang curves were then used to adjust yield from afforested areas modelled using IQQM (Simons et al., 1996 664), the NSW hydrological modelling platform used in water allocation and planning tool, to assess impacts on the security of reservoir inflows, downstream users and the environment. While 3-PG played no role in quantifying impacts it is germane to the preceding section and discussion to follow that Zhang curves for pre-afforestation conditions were found to over-estimate runoff compared to calibrated IQQM results; became less reliable with decreasing annual rainfall and reductions had to be applied proportionally, rather than as absolute quantities to maintain the integrity of the IQQM calibration (Herron et al., 2002 p 375).

A similar exercise was undertaken by Zhang et al. (2003b) who modified the GIS implementation of Zhang curves by Bradford et al. (2001), using gridded impacts in the Goulburn catchment of Victoria. Pre-afforestation yield was provided by gauged data in wetter areas and Zhang curves elsewhere. Differing levels of evapotranspiration were determined using Zhang curves, apportioned according to 3-PG modelled leaf area index (LAI) modelling. Sites with higher productivity were assigned higher levels of evapotranspiration estimated from confidence intervals calculated around the Zhang evapotranspiration curves using the Zhang et al. (1999) global data set. The timing of the maximum impacts as indicated by the attainment of maximum LAI was estimated to range between eight and 16 years according to the site.

A flow duration curve analysis identified heavier impacts on low flows. When the impacts of afforestation were incorporated into the Goulburn Simulation Model, an implementation of the Victorian water allocation and management model REALM used to manage system entitlements (Perera et al., 2005), the security of the regulated seasonal allocations showed a minor (<5%) decline. Unregulated flows past Goulburn Weir decreased by up to 27% with implications for downstream ecosystems.

Short-comings in the performance of the 3-PG water balance model (Dye et al., 2004; Morris and Collopy, 2001; Williams et al., 2009), prompted Feikema et al. (2007) to embed 3PG+ (Morris, 2003) into an agro-hydrological modelling platform (PERFECT, Littleboy et al., 1999; 1992) to exploit its more sophisticated (one-dimensional) multi-
layered soil water and daily time-step. PERFECT itself has been incorporated into the Victorian catchment analysis tool (CAT), a platform which combines a number of mature models to implement of sophisticated distributed catchment modelling (Beverly et al., 2005; Feikema et al., 2008). While the PERFECT water balance was calculated on a daily time-step (rainfall, transpiration, soil evaporation, infiltration, drainage), 3PG+ growth processes such as canopy dynamics, continued to operate on a monthly time-step. The approach improved evapotranspiration estimates (Feikema et al., 2007; 2009), but sub-daily processes lead to an inability to represent daily evapotranspiration variability, while the water dynamics of deeper soil profiles remained a challenge (Feikema et al., 2010b).

When 3PG+ was run in the spatially explicit CAT incorporating surface- and subsurface hydrology in five catchments in Victoria and Tasmania, Feikema et al. (2008) found that smaller catchments dominated by plantations delivered good calibrations, while larger one suffered from a lack of knowledge in representing evapotranspiration and growth of native forests. Nonetheless, the integration of 3PG+ into CAT was considered a valuable tool with potential for further development.

The most recent efforts in integrating 3-PG into a hydrological assessment framework closely resembled those of Feikema et al. (2008). Funded as part of a National Water Commission project to develop Methods to Accurately Assess Water Allocation Impacts of Plantations (Gilfedder et al., 2010), the work distinguished between a relatively simple (Zhang curve-based) approach for estimating whole-of-catchment impacts (Zhang et al., 2010) and the need to use a more complex approach to reflect within-catchment variability. The complex approach combined plantation growth and groundwater modelling to estimate likely impacts, using a one-dimensional daily time-step version of 3-PG (described as 3-PG+_FLUSH without further citation) and pasture/crop models in PERFECT within or in conjunction with a surface- and subsurface hydrological catchment model, 2CSalt (Marcar et al., 2010).

Their use of 3-PG in a catchment modelling framework enabled the authors to conclude that the impacts of afforestation on water resources would vary according to the location of the conversion due to differences in climate, soil properties, slope and the amount of area converted to plantation (Marcar et al., 2010, third bullet point p 53). The final report concluded that the approach could be suitable for medium risk situations that justified the collection of additional data to reduce uncertainty, but was not as suitable for high risk situations, which would justify more complex approaches.
such as TOPOG (Dawes et al., 1997), which had not formed part of the study (Gilfedder et al., 2010).

4.4.3 Jurisdictional approaches

The range of assessment approaches used across Australia is shown in Table 4.1, below. While far from comprehensive Table 4.1 provides a useful jurisdictional perspective on the approaches most germane to forest water policy development and management.

Queensland and Australian Capital Territory are not mentioned as they did not consider that large-scale plantation forestry presented significant risks to water resources under the National Water Initiative and have not embarked on any significant assessment initiatives beyond the Murray-Basin Sustainable Yields project (CSIRO, 2008c), used by New South Wales to preclude the need for further policy development (Section 3.3.5).

At the time of writing South Australia remains the only jurisdiction to regulate large-scale plantation forest water use and does so using different empirical relationships in different regions according to different hydrological and industry demands (Section 3.3.1).

Victoria has also adopted a regional approach, using a one-dimensional soil water model reliant on empirical relationships, both as internal model inputs governing plant growth and evapotranspiration, or as the basis of supporting models, to quantify potential streamflow reductions and identify areas warranting further investigation.

Tasmania has developed and trialed a tool based on a number of empirical relationships proposed by CSIRO (Brown et al., 2006a). The timing and magnitude of streamflow reductions induced by afforestation was derived by re-calibrating a form of the exponential CSIR Scott and Smith curves adjusted for rainfall, using five selected South African sites (Lambrechtbos A and B, Biesievlei, and Cathedral Peak I and II), a New Zealand site (Glendhu) and one from mainland Australia (Red Hill). Ungauged, pre-afforestation reference runoff was estimated using Zhang curves in the original tool (Brown et al., 2006a), while modelled daily data were used in the trial (DPIW, 2008).

A mechanistic model was developed in Western Australia capable of depicting the hydrological impacts of land-use change using empirical leaf growth data to quantify evapotranspiration. While its performance was considered satisfactory by jurisdictional researchers, it was not used in the final Western Australian Sustainable Yields
assessment which employed Zhang curve-based yield reductions (Bari et al., 2010; CSIRO, 2009), an approach adopted in some water planning processes (DW, 2011).

**Table 4.1:** Plantation impact assessment approaches used in water accounting, management or to inform recent policy development. Queensland and Australian Capital Territory did not consider risks significant.

<table>
<thead>
<tr>
<th>Region</th>
<th>Description</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>South Australia</strong> – commercial forestry water use managed regionally according to a state-wide policy (GSA, 2009) and regulated in declared areas.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lower South East</td>
<td>Deemed annualised recharge reductions; Annualised direct groundwater extractions averaged over the rotation based on empirical observations.</td>
<td>Harvey (2009); Benyon and Doody (2004).</td>
</tr>
<tr>
<td><strong>Victoria</strong> – commercial forestry water use managed regionally according to a state-wide policy (DSE, 2011b), regulation in declared management areas before parliament</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Gippsland Region</td>
<td>Risks not considered significant due to lack of recent expansion.</td>
<td>DSE (2011a).</td>
</tr>
<tr>
<td>Western Region</td>
<td>Priority areas identified in a regional review. Water balances prepared areas using SoilFlux in the Water and Land Use Change (WatLUC) study.</td>
<td>DSE (2011c); DSE (2011d); SoilFlux, Daamen et al. (2003); WatLUC, SKM (2005; 2008a; 2008b).</td>
</tr>
<tr>
<td><strong>Western Australia</strong> – commercial forestry water use is not regulated, but guidelines for its management are available (GWA, 2009a), and policy development and legal reforms are ongoing. Water allocation plans consider plantation water use where significant.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Greater south west</td>
<td>CSIRO sustainable yields, using Zhang curve-based FCFC as well as LUCICAT. Groundwater modelling using coupled WAVES and MODFLOW, see below.</td>
<td>CSIRO (2008a); Zhang et al. (2001); FCFC, Brown et al. (2006b); LUCICAT, Bari and Smettem (2006).</td>
</tr>
<tr>
<td>Perth regional</td>
<td>Perth Regional Aquifer, including Gnangara mound, using WAVES coupled to MODFLOW through a Vertical Flux Model (VFM).</td>
<td>VFM, DW (2009a); coupled MODFLOW, DW (2009b); WAVES, Zhang and Dawes (1998).</td>
</tr>
<tr>
<td>Great Southern</td>
<td>Salinity assessments LUCICAT, MAGIC.</td>
<td>Kent River, De Silva et al. (2006).</td>
</tr>
<tr>
<td><strong>Tasmania</strong> – commercial forestry water use is not regulated and following a catchment-scale modelling assessment, not considered a significant risk.</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>New South Wales</strong> – commercial forestry water use is not regulated and its management is guided by state-wide policies.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Murray-Darling Basin</td>
<td>Response to NWI determined by outcomes of CSIRO sustainable yields assessments, using Zhang curve-based FCFC.</td>
<td>CSIRO (2008c); Zhang et al. (2001); FCFC, Brown et al. (2006b).</td>
</tr>
<tr>
<td><strong>Northern Territory</strong> – commercial (mahogany) forestry water use is not regulated and following local-scale assessments, not considered a significant risk.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Daly River</td>
<td>Assessed using WAVES, and 3-PG supported by intensive data collection. Zhang curves discussed as boundary conditions in 3-PG work.</td>
<td>WAVES, Cresswell et al. (2011) in Hutley et al. (2012); 3-PG, Hutley et al. (2012).</td>
</tr>
</tbody>
</table>
Complex plant process modelling studies conducted in the Northern Territory found that mahogany plantations had similar levels of water use as native savannah and posed no significant risk to water resources.

4.5 Discussion

Empirical relationships have underpinned a century of forest management. The initial stages of both South African and Australian forest water regulation were underpinned by empirical relationships. The experience of both jurisdictions highlighted the vulnerability of empirical systems to challenge in the context of competing demands on water resources on the grounds of their robustness and limited applicability beyond the conditions of their derivation.

Where Australian forest water management remains underpinned by empirical relationships South Africa moved to mechanistic modelling. While mechanistic approaches address some of the limitations of empirical methods including greater flexibility in responding to forest management practices and the variable influences of site conditions and environment, they remain limited by uncertainty.

The persistent use of empirical methods in Australia suggest that effort should be put into overcoming their limitations, while a practical perspective is needed on the quantification of uncertainty for use in water management. Moreover the models used to underpin Australian forest water management have never been subjected to objective evaluation to ensure that decision-making is defensible. The competing alternatives should include well-established systems from the United States and those used in South African forest water management.

Australian researchers continue to use mechanistic and process models to assessing the hydrological impacts of afforestation. The increasing reliance of contemporary water management on complex models and the need to evaluate impacts under different site and climatic conditions makes the likelihood of their application in underpinning Australian forest water management appear inevitable. If a consistency of purpose is needed in progressing a national forest water management agenda, the field of competing priorities will need to be formally evaluated and issues of uncertainty confronted.

Experience in both South Africa and Australia suggests a number of alternatives, including ACRU, WAVES, TOPOG and 3-PG. South African modelling demonstrates the need to represent the dynamic relationship between water use and tree growth and
model pre-forest reference yield against which to estimate impacts.; Australian research has identified the need for soil water modelling consistent with site hydrology and the challenges of managing water in both countries under changing climatic drivers.

Large-scale, integrated catchment assessments are of limited value to water management as their complexity compounds uncertainty. While stand-scale assessments suffer from simplistic treatments of streamflow processes, they offer the potential to reflect site-specific issues more flexibly than empirical approaches and without the compounding uncertainties of large-scale, integrated models.

The implementation of 3-PG+ in PERFECT meets many of the criteria described above, it has the capacity to model a range of pre-afforestation land-covers, offers 3-PG’s pragmatic approach to modelling plant growth and incorporates a multi-layer soil water module on a daily time-step. However the 3-PG+ improvements have increased the complexity of the model and the studies to verify their integrity have not been conclusive. It remains to be established whether they can demonstrate additional confidence for use in water management.
5. **CONFRONTING EMPIRICAL LIMITATIONS**

*Is this long-term experimentation based on upon a philosophy of despair? .... Are we going to put all our energy in just measuring what happens, or shall we put a little more effort in research to try and find out why things happen?*

H. L. Penman, in discussion of "The validity of conclusions from South African multiple watershed experiments," Wicht (1967b)

AUSTRALIAN water reforms, regulation and policy frameworks underpin public and investor confidence (COAG, 2004b, s 31(vii), 80, 93). Confidence in water information is based in science which in Australian forest water policy development has relied heavily on empirical information. However the robustness of some information may be questionable (Section 3.3.1) and the capacity of water management agencies to undertake the analyses required to address them is a recognised limitation to water policy reform (COAG, 2004b, s 80 – s 101). Consequently the need for greater scrutiny and transparency of empirical information is a high priority for forest hydrology and water management.

6. A BAYESIAN APPROACH TO MODEL SCRUTINY

For a good mathematical model it is not enough to work well. It must work well for the right reasons.

Dilettantism in hydrology: transition or destiny? Klemeš (1986)

THE evaluation of hydrological models is the subject of a considerable body of research (Bai et al., 2009; Beven, 2006; Clark et al., 2008; Gupta et al., 2008; Jakeman et al., 2006; Refsgaard and Henriksen, 2004; 2005; see citations in Hrachowitz et al., 2013; Wagener and Gupta, 2005), but contention remains regarding how it should be undertaken (Clark et al., 2011; Gupta et al., 2012).

Model scrutiny is centered around quantifying uncertainty (Kuczera et al., 2010; Refsgaard, 2007; Renard et al., 2011; Sivapalan et al., 2003b) but uncertainties associated with water management modelling have not been closely analysed (Etchells and Malano, 2005; Lowe et al., 2009) and frameworks intended to guide water managers provide little practical guidance (Black et al., 2011; Blackmore et al., 2009; Vaze et al., 2012).

Their basis in probability makes Bayesian methods particularly appealing as a starting point, but their application in assessing the hydrological impacts of afforestation is uncommon and effectively unknown in forest water management. Work is needed to establish whether these sophisticated techniques can reasonably be considered suitable for use in water management agencies, recognised as having significant capacity constraints (COAG, 2004b, s 80 – s 101).

7. USE OF A PLANT PROCESS MODEL IN WATER MANAGEMENT

*Given the frame of mind that a person has when creating a model to provide himself with a framework, a week’s non-modelling thought would probably lead him to the same conclusions that a year’s modelling would.*

*Sense and nonsense in crop simulation, Passioura (1973)*

FROM their emergence in the 1970s as biological research tools, complex plant-process models have become a feature of integrated hydrological assessments (Section 4.4.2). Their potential to support water management has been recognised in South Africa (Section 4.3.2) and they have been used to inform decision-making in Australia (Section 4.4.3, for example DW, 2009a; Hutley et al., 2012).

The structure and parameterisation of complex models can give rise to *equifinality*, where the same results may be achieved by a number of equally likely, physically reasonable parameter sets (Beven, 2006; Clark et al., 2011; Franks and Beven, 1997; Franks et al., 1997; Grayson et al., 1992; Jakeman et al., 1994; Schoups et al., 2008), undermining confidence in model predictions and rendering decision-making vulnerable to challenge (Beven, 2000; Sivapalan et al., 2012). The issue is particularly germane to water management where the use of complex modelling is now routine (Gupta et al., 2008; Hrachowitz et al., 2013; Liu et al., 2008; Loucks, 2013; Montanari et al., 2013) and the confidence of policy-makers in models, modelling and modellers may be low (Brugnach et al., 2007).

Confidence in science can be difficult to come to terms with (ASCE, 1985) and water management is wicked (Lach et al., 2005). Moreover, verifying computer models of natural systems has been described as impossible (Oreskes et al., 1994) while evaluating alternative model structures by fitting their outputs to observed data favours complex models due to their greater flexibility (Schoups et al., 2008).

This study uses a Bayesian approach to evaluate the potential of improvements to the soil-water modelling of a widely used plant-process model (3-PG) to support decision-making. It does not focus on the capacity of more complex implementations of 3-PG to generate acceptable results (already demonstrated by Almeida and Sands, 2015; Feikema et al., 2010b), but examines whether the additional complexity of the improved model (3-PG+, Feikema et al., 2010b) can increase confidence in decision-making by extracting more information from data during parameter identification.
The study proceeds in Section 7.1 with an overview of 3-PG and the modifications implemented in 3-PG+ (Section 7.2). The analysis and data are described in Section 7.3 and Section 7.4 respectively. Results are presented in Section 7.5 and discussed in Section 7.6.

7.1 An Overview of 3-PG

Physiological Principles for Predicting Growth or 3-PG, is a canopy carbon balance model developed to bridge the gap between conventional empirical and process-based growth models. The key elements of 3-PG as it pertains to water use are outlined below, while comprehensive descriptions are available elsewhere (Landsberg and Sands, 2011). The version of 3-PG used in both studies described above was 3-PG+ which contains a number of developments designed to improve the model's ability to depict plant water use (see Section 4.4.2 above and 7.2 below).

7.1.1 Turning light into carbon

Ultimately, plant process models seek to depict photosynthesis as summarised in Equation 7.1, below. Respiration then reverses as the plant uses carbohydrates and oxygen to generate energy to support tissue maintenance and grow wood, roots and leaves. Limiting any factor on the left hand side of Equation 7.1 will limit production on the right, regardless of the availability of the others.

\[ \text{CO}_2 + \text{H}_2\text{O} + \text{energy} \rightarrow \text{carbohydrates} + \text{O}_2 \]  
\[ \text{Equation 7.1} \]

The 3-PG model follows the same basic principles at a stand scale: radiation energy is intercepted by the canopy and converted into carbohydrate assimilates, which are then allocated to foliage, stems and roots, with losses to respiration, litterfall and root turnover.

The first step in the process is calculating gross primary carbon production \( P_g \) as a function of radiation intercepted by the canopy. This process is simplified by using a light-use efficiency approach as depicted in Equation 7.2:

\[ P_g \propto \varepsilon_g \phi \]  
\[ \text{Equation 7.2} \]

where \( \phi \) is the radiation energy intercepted by the canopy and \( \varepsilon_g \) is light-use efficiency (\( g \text{ MJ}^{-1} \)).
Green-leaved plants only absorb the visible/near-visible wavelengths of the short-wave radiation incident on the earth’s surface \((\phi_i, \lambda \sim 0.4 - 4.0 \text{ m}^{-6})\), termed photosynthetically active radiation \((\phi_p)\), estimated in 3-PG by assuming a simple linear relationship with incident radiation: \(\phi_p = \phi_i/2\) (Landsberg and Waring, 1997). The photosynthetically active radiation absorbed by the stand \((\phi_a)\), is given by Beer’s Law, which treats the canopy as a single big leaf:

\[
\phi_a = \frac{\phi_i \zeta}{2} \left(1 - e^{-k \text{LAI}/\zeta}\right)
\]

where \(\zeta\) is the proportion of ground covered by the canopy, a quantity assumed to increase uniformly over time until it reaches unity at a given stand age (Landsberg and Sands, 2011; Sands, 2004); \(k\) is a canopy light extinction coefficient and \text{LAI} is leaf area index; the projection of total leaf surface area per unit ground area.

Gross primary production can then be written:

\[
P_g \propto \varepsilon_g \frac{\phi_i \zeta}{2} \left(1 - e^{-k \text{LAI}/\zeta}\right).
\]

Light-use efficiency \((\varepsilon_g)\) is determined from canopy (quantum) light-use efficiency \((\alpha_c)\) to facilitate direct comparison with leaf- or canopy-level photosynthetic carbon production per absorbed radiation:

\[
\varepsilon_g = b \alpha_c
\]

where \(b\) is a constant that adjusts units of \(\alpha_c \text{ (mol C mol}^{-1}\text{ photons)}\) to those of \(\varepsilon_g \text{ (g MJ}^{-1}\text{)}\) (Landsberg and Sands, 2011).

Part of the gross primary production is required for respiration processes associated with plant maintenance, leaving the net primary production \((P_n)\) to be allocated to new growth \((P_n = P_g - \text{respiration})\). In 3-PG, \(P_n\) is estimated as a constant proportion of gross primary production: \(P_n = Y P_g\), where \(Y = 0.47\) (Sands and Landsberg, 2002), which may be used with the equations above to write an expression for net primary production:

\[
P_n = \alpha_c \phi_i \zeta \left(1 - e^{-k \text{LAI}/\zeta}\right)B
\]

where \(B = b Y/2\) combines the constants from Equations 7.3 and 7.5.

The efficiency of a canopy in converting light into carbon is determined by environmental and physiological constraints. Constraints include: soil water
availability, atmospheric water availability (vapour pressure deficit) and stand age, each of which also influence canopy conductance (discussed below) (Landsberg and Waring, 1997); mean air temperature and site nutrition (Sands and Landsberg, 2002). In 3-PG, each of these constraints is expressed as a dimensionless modifier in the range [0,1], and used in a growth modifier approach (Landsberg and Sands, 2011, p137) which limits a stand’s ability to convert light into carbon, termed canopy quantum efficiency:

\[ \alpha_C = \alpha_{Cx} \min(f_\theta f_D f_A f_F f_T f_N f_{Ca}) \]

7.7

where \( \alpha_{Cx} \) is the maximum possible canopy quantum efficiency, a species-specific parameter and:

- \( f_\theta \) is the average monthly soil moisture ratio,
- \( f_D \) the average monthly vapour pressure deficit,
- \( f_A \) stand age and the associated decrease in stem hydraulic conductivity,
- \( f_F \) the number of frost days (\( f_{\theta,D,A,F} \) Landsberg and Waring, 1997) ; and
- \( f_T \) monthly mean air temperature, and,
- \( f_N \) site fertility (\( f_{T,N} \) Sands and Landsberg, 2002); and,
- \( f_{Ca} \) atmospheric CO\(_2\) concentration modifier introduced by Almeida et al. (2009).

Modification for atmospheric CO\(_2\) also included a separate modifier for canopy conductance (\( f_{Cg} \)) discussed further under Section 7.1.2, below.

While \( \alpha_{Cx} \) is conceptualised as a constant species-specific parameter, it is known to vary according to the seasonal break-down in chlorophyll (Landsberg and Waring, 1997) and has been varied across different sites to produce optimal results (Sands and Landsberg, 2002). Typical values include 0.03 \( \text{mol \ C mol}^{-1} \text{photons} \) for mixed deciduous forests (Landsberg and Waring, 1997) and 0.05 – 0.07 for \textit{Eucalyptus globulus} (Sands and Landsberg, 2002).

3-PG+ (Section 7.2, below) scales canopy efficiency to convert incident radiation \( \text{MJ m}^{-2} \) to production in \( \text{g cm}^{-2} \) which is directly acted on by the modifiers above, resulting in a species estimate of \( \alpha_{Cx} \) simply termed \( \alpha_C \) by Feikema et al. (2010b). Typical values of \( \alpha_C \) are 1.85 \( \text{mol C mol}^{-1} \text{photons} \) for \textit{Pinus radiata} and 2.30 for \textit{E. globulus} (Feikema et al., 2010a; 2010b).
Once the pool of carbon is quantified by modifying the conversion of incident radiation through environmental modifiers, it is allocated to foliage \((W_F, t \text{ ha}^{-1})\), stem \((W_S, t \text{ ha}^{-1})\) and root stores \((W_R, t \text{ ha}^{-1})\) according to partitioning rates determined by site conditions, growth and tree size, derived from relationships developed from empirical observations. The approach adopted in 3-PG follows that of McMurtrie and Wolf (1983), and interested readers are directed to detailed discussions in Landsberg and Sands (2011) and citations therein.

### 7.1.2 Canopy processes

Leaves play a fundamental role in plant growth and regulating water use. 3-PG uses a big leaf approach to modelling canopy processes which negates complexities arising from real-world three-dimensional architecture and assumes that foliage is uniformly distributed across the stand, a significant limitation in accurately portraying canopy processes for widely-spaced young trees (Landsberg and Waring, 1997). Moreover, 3-PG infers LAI from the foliage biomass allocation \((W_F, t \text{ ha}^{-1})\) of the carbon pool generated by the light intercepted by the canopy according to Equation 7.8:

\[
LAI = \frac{\sigma_{W_F}}{10} \tag{7.8}
\]

where \(\sigma\) is the specific leaf area \((m^2 \text{ kg}^{-1})\) and the denominator a conversion factor. The sensitivity of LAI to carbon assimilation during the early stages of canopy growth prompted Sands and Landsberg (2002) to develop an aged-dependent empirical function to account for observed variations in \(\sigma\) over the first years of a rotation:

\[
\sigma(t) = \sigma_1 + (\sigma_o - \sigma_1) e^{-(\ln2)(t/t_o)^2} \tag{7.9}
\]

where \(\sigma_o = 11 \text{ m}^2 \text{ kg}^{-1}\), \(\sigma_1 = 4 \text{ m}^2 \text{ kg}^{-1}\) and \(t_o = 2.5 \text{ years}\).

### Canopy conductance

Canopy conductance \((g_c)\) represents the interface between the atmospheric water balance and the stand canopy, increasing with LAI up to an unconstrained conductance \(g_x\). It is depicted in 3-PG as a bulked vegetation and surface conductance, estimated from stomatal conductance and LAI after Kelliher et al. (1995) and modified for physiological limitations due to vapour pressure deficit (Landsberg and Waring, 1997); soil moisture stress and age (Sands and Landsberg, 2002); and atmospheric \(CO_2\):
\[ g_c = g_x \min(f_\theta, f_D) f_{AfCB} \]

where the modifiers \( f_{A,\theta,D} \) are defined in Equation 7.7 above and \( f_{CB} \) is a modifier taking into account the effects of atmospheric CO\(_2\) on stomatal conductance (Almeida et al., 2009). Unconstrained canopy conductance \( g_x \) is a function of LAI and the species-specific maximum canopy conductance \( g_{Cx} \) (Landsberg and Sands, 2011).

Values of \( g_{Cx} \) for a range of woody vegetation are around 0.018 m s\(^{-1}\) (Kelliher et al., 1995), while 0.02 m s\(^{-1}\) is the recommended default for \( E. \) globulus by Sands and Landsberg (2002) and 0.018 m s\(^{-1}\) (as \( g_{Cmax} \)) by Feikema et al. (2010b).

**Evapotranspiration**

Evapotranspiration is calculated on a monthly time-step. The original implementation of 3-PG depicted rainfall interception by the stand canopy as a fixed proportion of rainfall (Landsberg and Waring, 1997), which was modified in later implementations as a function of LAI and subject to direct evaporation given by the Penman equation for wet surfaces (Landsberg and Sands, 2011). Evaporation from wet leaf surfaces \( (E_{mm \ month^{-1}}) \) is calculated at the potential rate using the Penman-Monteith equation for wet surfaces (Monteith, 1965, see Equation 2.32 in Landsberg and Sands, 2011).

Evapotranspiration \( (E_t \ mm \ month^{-1}) \) is calculated using the Penman-Monteith equation (Monteith, 1965, see Equation 2.48 in Landsberg and Sands, 2011):

\[
\lambda E_t = \frac{s\phi_n + \theta \phi_n c_{pa} D}{s + \gamma (1 + \theta / \gamma)}
\]

where,

- \( \lambda \) is the latent heat of vaporization of water \((\approx 2.44 MJ kg^{-1})\);
- \( s \) is the rate of change of saturated specific humidity with temperature \((2.2 \text{ at } 20^\circ C, \text{ dimensionless})\);
- \( \phi_n \) net radiation retained by the canopy surface: \( \phi_n = 0.8 \phi_i - 90 \), input as irradiance \( (W m^{-2}) \) and integrated over daylight hours;
- \( \rho_a \) air density \((\approx 1.2 kgm^{-3} \text{ at } 20^\circ C)\);
- \( c_{pa} \) specific heat of dry air \((1004 J kg^{-1}K^{-1})\);
- \( D \) vapour pressure deficit of the surrounding air \((kPa)\);
- \( \gamma \) psychometric constant \((66.1 Pa K^{-1})\);
combined leaf and canopy boundary conductance, calculated as a function of wind-speed (m s\(^{-1}\)); and,

g_c \quad \text{canopy conductance (m s}^{-1})\), physiological control on \(E_T\); if the canopy is wet \(g_c \to \infty\).

7.1.3 Soil water balance

The soil water balance is calculated as the difference between the monthly evapotranspiration (calculated as above) and monthly precipitation using a single-layered storage (Landsberg and Waring, 1997). A maximum available soil water content (\(\theta \text{ mm}\)) is estimated from site soil properties and the rooting depth of the species. A moisture ratio is estimated for the stand (\(r_\theta\)), calculated as:

\[
r_\theta = \frac{\text{current soil water content} + \text{water balance}}{\text{available water}}
\]

If the numerator of Equation 7.12 is greater than \(\theta\), it is set to \(\theta\), and \(r_\theta = 1\), with the excess assumed to have left the system by surface yield or drainage; if the numerator is negative, \(r_\theta = 0\).

The soil water modifier \(f_\theta\), is calculated using:

\[
f_\theta = \frac{1}{1 + [(1-r_\theta)/c_\theta]^{n_\theta}}
\]

where \(c_\theta\) and \(n_\theta\) take different values for different soil types (Denmead and Shaw, 1962).

The over simplification represented by the monthly time-step was recognised as inherently unsatisfactory, resulting in the distortion of recharge and depletion patterns. However, addressing the matter entailed higher data demands and increased complexity, and was arguably inconsistent with the extant crudeness of the soil water modifier approach to controlling water availability (Landsberg and Waring, 1997).

7.1.4 Site quality and the fertility rating

Predictions by 3-PG are highly sensitive to estimates of site fertility (Esprey et al., 2004; Miehle et al., 2009). Of all the modifiers in Equation 7.7 (above) only site fertility \((f_N)\) is not directly dependent on environmental characteristics or plant response. Instead, it is dependent on an abstract notion of fertility, the fertility rating \((F_F)\) (Miehle et al., 2009):
where \( f_{N0} \) is the value the value of \( f_N \) when \( F_R = 0 \) (Sands and Landsberg, 2002). \( F_R \) ultimately controls the productivity or quality of the site through \( f_N \), and plays a direct role in modifying allocation to the root-zone in:

\[
\eta_R = \frac{\eta_{Rx} \eta_{Rn}}{\eta_{Rn} + (\eta_{Rx} - \eta_{Rn}) m f_{\text{Amin}}(f_D)}
\]

where:

- \( \eta_R \) is the fraction of net primary production allocated to the roots,
- \( \eta_{Rx}, \eta_{Rn} \) are species parameters which stipulate the maximum and minimum fractions of production partitioned to the roots, and,
- \( m \) is the root biomass allocation modifier: \( m = m_0 + (1 - m_0) F_R \), where \( m_0 \) is the value the value of \( m \) when \( F_R = 0 \) (Landsberg and Sands, 2011; Sands and Landsberg, 2002).

Where fertilisation data are available, empirical relationships can be developed to assign \( F_R \) to a particular site (Almeida et al., 2010; Miehle et al., 2009; Stape et al., 2004). Where nutrient availability can be assumed to be non-limiting, \( F_R \) can be set to unity (Sands and Landsberg, 2002). Where detailed site information are unavailable, \( F_R \) can be used to calibrate predicted growth volumes against observations (Dye et al., 2004; Landsberg et al., 2003; Sands and Landsberg, 2002), however \( F_R \) can be a surrogate for various factors that influence site productivity (Almeida et al., 2004), and may not directly represent fertility.

### 7.1.5 Recognised shortcomings

The hydrological water balance in the initial versions of 3-PG was extremely simple, comprising a single-layer soil-water model run on a monthly-step. It was recognised by its developers as inherently unsatisfactory, however daily modelling was regarded as representing an unjustified level of increased complexity with high data demands (Landsberg and Waring, 1997). Nonetheless, its limited representation of soil-water interactions has been linked to shortcomings in 3-PG’s ability to replicate monthly evapotranspiration (Benyon et al., 2008; Dye et al., 2004; Morris and Collopy, 2001; Tickle et al., 2001).

Soil-water dynamics are critical in controlling tree growth, particularly in dry areas (Beadle and Inions, 1990; Coile, 1952; Harper et al., 2009; Honeysett et al., 1992;
Jackson and Gifford, 1974; Ralston, 1964; Ryan et al., 2002; 2008). In areas with less than 1000mm/year rainfall, plantations tend to use all the available rainfall, higher leaf areas and growth are possible where stress can access alternative water sources, such as stores which may have accumulated in the deep regolith during wet years (White et al., 2000). When water is not limiting, productivity bears a direct relationship to evapotranspiration (Beadle and Inions, 1990; Honeysett et al., 1992), increasing with increasing soil water availability to a maximum threshold level associated with leaf area (White et al., 2000).

Trees can access water from greater than five metres below the surface (Benyon et al., 2008; Carbon et al., 1980; Dye, 1996b; Greenwood et al., 1985; Knight, 1999; 2002), beyond the limit of most widely available soil information (McKenzie et al., 2003). Deep soils have been linked to productivity (Harper et al., 1999; McGrath et al., 1991; Ryan et al., 2002; Turvey et al., 1986), tree survival in areas with extended dry seasons (Harper et al., 1999; 2001; McGrath et al., 1991) and extreme hydrological impacts (Dye and Poulter, 1991; Scott and Lesch, 1997).

Their significance to tree growth and the recognised limitations of 3-PG made of soil-water dynamics the focus of further developments in 3-PG+.

### 7.2 3-PG+ in CAT

The version of 3-PG used in this study 3-PG+, emerged from a University of Melbourne project to develop an accepted planning tool capable of predicting the hydrological impacts of plantations in relation to their location and management (Feikema et al., 2008). It represents the culmination of a number of developments introduced by Morris and Collopy (2001), Morris (2003) and Morris and Baker (2003) and was implemented as a one-dimensional module within the Victorian Government’s Catchment Analysis Tool farming system model CAT1D or CAT hereafter (Beverly et al., 2005; Beverly, 2009), which has found application in water quantity, quality (Beverly and Hocking, 2009; 2011) and catchment studies (Feikema et al., 2008).

3-PG+ was developed by Dr Jim Morris (University of Melbourne, deceased) and Dr Craig Beverly (Government of Victoria) in Fortran 95. The code includes an implementation of the 3-PG (Feikema et al., 2010a, Appendix C) which was stringently verified against output from the original model (Beverly pers comm).

The main benefits of integrating 3-PG+ in CAT were identified as:
• the ability to access the daily time-step, multi-layered soil moisture module of the agro-hydrological framework PERFECT (Littleboy et al., 1999; 1992) also implemented in CAT;
• the ability to represent soil evaporation under canopy closure and the use of the USDA curve number method (also implemented in PERFECT) to model direct soil runoff;
• providing CAT with a growth-based approach to modelling evapotranspiration which was capable of reflecting dynamic environmental and forest management influences, rather than the simple proportional monthly vegetation cover approach of Allen et al. (1998) (Feikema et al., 2010a).

The first two points feed into the third which if convincingly demonstrated, would literally revolutionise plantation forest water management.

### 7.2.1 Developments

Feikema et al. (2010a) described the enhancements of 3PG+ in CAT as:

1. Limiting growth in stands prior to canopy closure by limiting potential radiation absorption.
2. The formulation of the temperature, soil water, and fertility growth modifiers.
3. Varying specific leaf area with age.
4. Varying branch fraction with age and stand density.
5. Varying litterfall rate with drought.
6. Providing for tree mortality in young stands, prior to the effect of the self-thinning rule.
7. Including canopy interception of rain.
8. Including runoff of rain and evaporation from surface soil.
9. Calculation of transpiration, as affected by the formulation of canopy conductance.
10. Providing for soil layering with depth, and root zone water movement.

Those of greatest interest to this study are described further below, while more detailed information is available in Appendix 1 of Feikema et al. (2010b).

**Radiation absorption before canopy closure (point 1)**

The assumption that leaf area is evenly spread across the canopy at low values of $LAI$ prior to canopy closure tends to result in the overestimation of absorbed radiation and
therefore growth. Consequently 3PG+ was incorporates two species parameters (early\textsubscript{co} and early\textsubscript{exp}) that limit incident photosynthetically active radiation (\(\phi_p\)) absorbed in stands with unclosed canopies to \(\phi_{early}\):

\[
\phi_{early} = \frac{\phi_p}{1 + early\textsubscript{co} e^{-early\textsubscript{exp} LAI}}
\]  

7.16

**Soil water growth modifier (point 2)**

Point 2 encompasses changes to the temperature, soil water, and fertility growth modifiers. Of particular note to this study are the soil water and fertility growth modifiers.

Like 3-PG, the 3-PG+ soil water modifier \(f_\theta\) is calculated using two empirical parameters used to reflect differences in the relationship between transpiration and soil water content for different soil textures (after Denmead and Shaw, 1962). However the soil parameters \((n_\theta, c_\theta)\) in 3PG+ are specified for each soil layer and single representative values over the rooting zone derived by weighting the layer-specific parameters according to the plant available water-holding capacity of each layer. Like later versions of 3-PG, 3-PG+ applies the fertility modifier \(f_N\) to estimate primary production and biomass allocation to foliage (Feikema et al., 2010a; Sands and Landsberg, 2002).

**Canopy rainfall interception (point 7)**

Interception has been modelled using a number of different approaches of the different versions of 3-PG (Sands and Landsberg, 2002), the most recent resembling the above, accommodating throughfall and evaporation from the moisture in the canopy following a storm event (Landsberg and Sands, 2011). Rainfall interception is modelled in 3-PG+ \((I_R, mm)\) after the approach used in the plant process model CABALA (Battaglia et al., 2004):

\[
I_R = \min[I^*R, I_L LAI - R_C + E_C]
\]  

7.17

where \(I^*\) is the maximum fraction of incident rainfall \(R\) that can be intercepted by the canopy (the remainder becoming effective precipitation or throughfall); \(I_L\) \((mm LAI^{-1})\) is an empirical term that links maximum canopy rain storage to \(LAI\) and \(E_C\) \((mm)\) is direct evaporation from the canopy rain store.
Runoff and soil evaporation (point 8)

Runoff from the soil is generated in 3-PG when the volumetric soil water content exceeds the soil moisture capacity (Landsberg and Sands, 2011), however in 3-PG+ runoff is modelled using the USDA curve number approach in PERFECT (Littleboy et al., 1999; 1992) which uses a retention parameter to account for antecedent soil moisture and forest floor characteristics inferred from model outputs that modify runoff (Feikema et al., 2010b). Further information on the curve number approach can be found in Section 5 (Greenwood et al. 2011) and the references therein.

Evaporation from surficial soil layers is calculated using Ritchie’s (1972) two-stage evaporation algorithm in PERFECT (Littleboy et al., 1999). In stage 1, drying occurs at a potential rate from bare soil, modified for the effects of soil cover such as modelled litterfall. Once a specified limit related to soil properties is reached, stage 2 evaporation proceeds at a rate reflecting diffusion processes that are assumed proportional to the square root of time:

$$E_o = [E(1 - (c_m + c_c)) e^{(-0.22W)}]$$

where $E_o$ is potential evapotranspiration, $E$ is effectively pan evaporation calculated using Penman–Monteith method, required due to legacies in the development of PERFECT (Littleboy et al., 1999, p16); $c_m, c_c$ are the fraction of litter and canopy cover respectively and $W_i$ is the absolute mass of litter, all parameters calculated using 3-PG+ outputs.

Transpiration affected by canopy conductance (point 9)

Transpiration is calculated in both 3PG and 3PG+ using Penman–Monteith evapotranspiration ($E_t$) with the appropriate canopy conductance values (see Equation 7.11, Section 7.1.2). However canopy conductance $g_c$ is calculated in 3-PG+ by incorporating the temperature modifier $f_T$ into Equation 7.10 (Section 7.1.2) and

$$g_c = g_x \min(f_B, f_D) f_4 f_T \min(1, \tanh(LAI/3))$$

where the modifiers $f_{A,B,D,T}$ are defined in Equation 7.7 above and the hyperbolic tangent function was introduced to ensure the threshold value of $LAI = 3$, is approached smoothly (Feikema et al., 2010b).
Soil layering and root zone water movement (point 10)

Short-comings in the original 3-PG soil water model (Landsberg and Waring, 1997) prompted the use of multi-layered models to examine moisture variations within the soil profile (Almeida and Sands, 2015; Almeida et al., 2007; Feikema et al., 2010b) and in more recent times have resulted in driving the single-layered soil water dynamics by partitioning monthly rainfall into periods of equal duration according to the number of rainy days (Landsberg and Sands, 2011).

In 3-PG, all plant available water (PAW) in the soil is immediately available for transpiration from the beginning of a simulation. The soil water balance at the end of the month is calculated as the previous month’s PAW plus current month’s rainfall less the current month’s transpiration. If the balance is less than zero, the PAW content is set to zero. In some circumstances the water balance permits more transpiration more the calculated PAW in a given month. Moreover, the single layer can not portray the heterogeneity typical of most soil profiles or the plant-water dynamics produced by roots progressively occupying soil horizons with different moisture content (Feikema et al., 2010b).

The PERFECT soil water model in CAT comprises of a series of layers specified by thickness, texture, bulk density, proportion of coarse fragments and soil water retention characteristics. These data are used to calculate PAW capacity for each layer in to a specified maximum rooting depth. Rainfall excess is cascaded through the layers after accounting for interception, runoff and soil evaporation. PAW is partitioned into two zones in each layer according to root depth and occupancy simulated by 3-PG+. The first zone represents the area from which roots can access water, the second zone represents areas in the soil horizon where water is not accessible to roots. The 3-PG+ soil water balance calculation is analogous to 3-PG but on a daily time-step (see Equations 7.12, 7.13, Section 7.1.3).

More information is available in Feikema et al. (2010a; 2010b) and Littleboy et al. (1999).

7.3 Analysis

The analysis follows that developed in Section 6 (Greenwood et al., 2014). The approach is consistent with frameworks that critically evaluate gains in performance against model complexity and structure. It emphasises evaluating the relative merit of competing models by assessing their ability to extract information from data in
parameter identification rather than attempting to distinguish between similar results that may have more to do with weakly structured models and equifinality (see Sadegh and Vrugt, 2013, and references therein) than significant differences in model integrity, particularly in high dimensional parameter space with unquantified uncertainties and the vagaries of expert opinion (Clark et al., 2011; Gupta et al., 2012). Its basis in probability theory makes it conducive to uncertainty analysis (Kuczera et al., 2010 among many), a recognised challenge to establishing the confidence in modelling outputs for water resources management and decision-making (Brugnach et al., 2008a; 2012; 2008b).

Figure 7.1 summarises the approach. A plantation uses water in response to climate, metabolic growth processes and site conditions (Figure 7.1a). Water use and growth are modelled using both simple (Figure 7.1b(i)) and complex soil water balance modules (Figure 7.1b(ii)). Data from a number of sites are used in a Bayesian calibration to identify parameters chosen to represent key plantation processes. Since results of acceptable veracity may be generated by different parameterisations (eg Franks et al., 1997, and references therein), improvements to model structure can’t be convincingly demonstrated by commonly used performance criteria alone (Krause et al., 2005; Greenwood et al., 2014, and references therein). If model is improved by the additional complexity, parameter identification will improve by a narrowing of confidence limits in marginal posterior distributions (Figure 7.1c), or improved structure in multi-dimensional parameter space (Figure 7.1d).

**Figure 7.1:** Conceptual approach used to evaluate the integrity of improvements to 3-PG soil water modelling made under 3-PG+. (a) conceptual plantation with selected species parameters
that control processes related to radiation interception ($a_C$, quantum canopy efficiency),
transpiration ($g_{cx}$, maximum canopy conductance) and growth ($F_R$, fertility rating); (b) stylised
depiction of model change from simple soil water store to a multi-layered one; (c) stylised
improvement in model parameter identification in the marginal and (d) joint posterior probability
distributions associated with the change.

Similar approaches have used in analysing a number of process models to examine
the impact of different model developments on predictions of growth (Minunno et al.,
2013; Van Oijen et al., 2011; 2013; 2005), but have not focused on plant water-
relations, a long-recognised limitation of 3-PG (Almeida and Sands, 2015; Benyon et
al., 2008; Landsberg and Waring, 1997; Morris and Collopy, 2001).

Of approximately 40 parameters in 3-PG+ (see Feikema et al., 2010b, Table A1), three
were chosen for their role in controlling water use and sensitivity to environmental
interactions (Almeida and Sands, 2015; Esprey et al., 2004; Song et al., 2013):
radiation interception via quantum canopy efficiency ($a_C$); transpiration via maximum
canopy conductance ($g_{cx}$) and growth via the fertility rating ($F_R$).

Markov chain Monte Carlo (MCMC) concepts are explained in more detail in
Greenwood et al. (2014) (Section 6). In this analysis MCMC is implemented using the
differential evolution adaptive Metropolis (DREAM) algorithm of Vrugt et al. (2008;
2009a; 2009b) to cope with the high-dimensional complexity of 3-PG. Based on the
shuffled complex evolution Metropolis of Vrugt et al., (2003), the scheme maintains
detailed balance and ergodicity while facilitating efficient convergence by
simultaneously employing multiple adaptive Markov chains to tune the proposal
distribution in highly nonlinear and multimodal target distributions. Proposals are
generated from the $N$ simultaneous Markov Chains as an $N \times d$ matrix, where $d$ is the
number of parameters. Jumps in each chain are generated by adding a multiple of the
difference of the states of randomly chosen pairs of chains. After burn-in,
convergence can be monitored with the Gelman and Rubin (1992) shrink factor $\hat{R}$.

DREAM was implemented using the dream() function of Guillaume and Andrews
(2012) in the R (R Core Team, 2015) package hydromad (Andrews et al., 2011). The
3-PG+ code was compiled as a Fortran subroutine into a shared object along with
scripts to pass parameters between R and Fortran. The shared object was called in
an R wrapper function with the generalised likelihood of Schoups and Vrugt (2010),
translated from Matlab into R by Greenwood et al. (2014) (Section 6).
The wrapper function was then simply called by the `dream()` function in R, and run using the settings specified in the `dream()` control argument, which were set to default values: bound handling of proposal values was set to `reflect` and ten, $5 \times 10^4$-iteration chains were thinned by a factor of 10, to give a posterior sample of $5 \times 10^4$ parameter sets before removing burn-in.

The length of simulations was decided following test runs on different platforms including the University of Melbourne’s high performance computer “Edward”. The analyses were finally run on a 64-bit Intel® Core™ i5-520M, 2.40 GHz with a solid state disk in an Ubuntu 14.04 Linux environment.

7.4 Data

Data used in the MCMC calibration were stand volume, transpiration and soil moisture to capture dynamics of growth and plant-water dynamics. The data were provided by the courtesy of CSIRO and have been used in a number of hydrological water balance studies (Benyon and Doody, 2004; 2015; Benyon et al., 2006) and modelling investigations (Almeida and Sands, 2015; Benyon et al., 2008) under different site names as listed in Table 7.1 (below). Benyon et al. (2008) found that 3-PG+ performed best when trees did not have access to groundwater. Consequently blue gum (*E. globulus*) sites with a median depth to groundwater exceeding eight metres were chosen for the economic significance of the species and greatest likelihood of obtaining good modelled results. Site locations are shown in Figure 7.2, and climatic and site-related data used in the study are summarised in Table 7.2 (both below).

**Table 7.1:** Sites used in the study, all planted with *E. globulus*.

<table>
<thead>
<tr>
<th>Site name</th>
<th>E (MGA)</th>
<th>N (MGA)</th>
<th>GW depth (m)</th>
<th>Alternate site name</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jill</td>
<td>451,826</td>
<td>5,863,636</td>
<td>10.5</td>
<td>McCourt 4</td>
<td>Benyon et al. (2008), site 15</td>
</tr>
<tr>
<td></td>
<td>140.45 lon</td>
<td>-37.35 lat</td>
<td></td>
<td>McCourt 4 site 6</td>
<td>Benyon and Doody (2015), site 17</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Jill</td>
<td>Benyon et al. (2006)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Almeida and Sands (2015)</td>
</tr>
<tr>
<td>Vicki</td>
<td>600,303</td>
<td>5,782,255</td>
<td>8.3</td>
<td>Macarthur</td>
<td>Benyon et al. (2008), site 1</td>
</tr>
<tr>
<td></td>
<td>142.15 lon</td>
<td>-38.10 lat</td>
<td></td>
<td>Macarthur site 9</td>
<td>Benyon and Doody (2014), site 1</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Vikki</td>
<td>Benyon et al. (2006)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Almeida and Sands (2015)</td>
</tr>
<tr>
<td>DPI</td>
<td>594,533</td>
<td>5,810,554</td>
<td>10.0</td>
<td>DPI</td>
<td>Benyon et al. (2008), site 2</td>
</tr>
<tr>
<td></td>
<td>142.05 lon</td>
<td>-37.85 lat</td>
<td></td>
<td>DPI</td>
<td>Benyon and Doody (2015), site 2</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Not included in Benyon et al. (2006)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>DPI</td>
<td>Almeida and Sands (2015)</td>
</tr>
</tbody>
</table>
**Figure 7.2:** Location of sites used in the study. Dark green areas are pine plantations, eucalypt plantations are grey-green and difficult to distinguish among the mosaic of remnant native vegetation. Indications of the wider distribution of plantation types may be found in Benyon et al. (2008, Figure 2); and Harvey (2009, Figure 1).

**Table 7.2:** Data summary. Methods are described in Benyon and Doody (2004); Benyon et al. (2006; 2008).

<table>
<thead>
<tr>
<th>Data</th>
<th>Diameter at breast height (dbh)</th>
<th>Transpiration</th>
<th>Soil moisture</th>
<th>Stand volume</th>
</tr>
</thead>
<tbody>
<tr>
<td>source:</td>
<td>dendrometer, mean from 6 trees – all sites</td>
<td>mean daily sap velocity heat pulse</td>
<td>calibrated neutron moisture meter</td>
<td>calculated fromdbh</td>
</tr>
<tr>
<td>units:</td>
<td>cm</td>
<td>daily mm</td>
<td>mm/m</td>
<td>m³/ha</td>
</tr>
<tr>
<td>time step:</td>
<td>approx. monthly</td>
<td>monthly total</td>
<td>approx. monthly</td>
<td>approx. monthly</td>
</tr>
</tbody>
</table>

**period of record of above:**

|---------------------------|----------------------------------------------------------|------------------------------------------|------------------------------------------|------------------------------------------|------------------------------------------|------------------------------------------|

**forest data**

<table>
<thead>
<tr>
<th>established</th>
<th>stocking³ (stems/ha)</th>
<th>thinning</th>
<th>% logs</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jill</td>
<td>Jul 1996</td>
<td>1200</td>
<td>none</td>
</tr>
<tr>
<td>Vicki</td>
<td>Aug 1998</td>
<td>925</td>
<td>*</td>
</tr>
<tr>
<td>DPI</td>
<td>Jul 1996</td>
<td>950</td>
<td>*</td>
</tr>
</tbody>
</table>

³ initial and final
Table 7.3: Site data, soil properties and related model parameters.

<table>
<thead>
<tr>
<th>Site</th>
<th>SILO climate site¹</th>
<th>water table (m)</th>
<th>Site properties soil profile</th>
<th>profile depth (m)</th>
<th>GW obs (mm)</th>
<th>ntheta/ctheta²</th>
<th>theta-max³ (mm)</th>
<th>soil layers (mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jill</td>
<td>142.05 E -37.85 N</td>
<td>10.5</td>
<td>0-5.0m loamy sand</td>
<td>5</td>
<td>mean of 5 wells 0-75 75-150 150-300 &amp; 300 spaced to end, all sites</td>
<td>0.70/9.0</td>
<td>451</td>
<td>0-300</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>300-600</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>600-1200</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1200-2400</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2400-3600</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>3600-4800</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>4800-5100</td>
</tr>
<tr>
<td>Vicki</td>
<td>142.15 E -38.10 N</td>
<td>8.3</td>
<td>0-0.3m clay loam</td>
<td>6</td>
<td>as above</td>
<td>0.25/1.0</td>
<td>925</td>
<td>as above</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.3-5.7m heavy clay</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>4800-5700</td>
</tr>
<tr>
<td>DPI</td>
<td>142.05 E -37.85 N</td>
<td>10.0</td>
<td>0-0.2m clay loam</td>
<td>6</td>
<td>as above</td>
<td>0.30/1.0</td>
<td>950</td>
<td>as above</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.2-5.8m clay</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>4800-5700</td>
</tr>
</tbody>
</table>

¹Data drill http://www.nrm.qld.gov.au/silo
²Soil water parameters
³Maximum soil water capacity

A nuance of the developments implemented under 3-PG+ is that while the monthly time-step, single-layered soil-water module of 3-PG was replaced by the daily time-step, multi-layered module of PERFECT, the calculation of transpiration by 3-PG+ remains on a monthly step. Consequently daily transpiration data could only be meaningfully calibrated to 3-PG+ output as monthly totals. Implications of this limitation are discussed further in Section 7.6 (below).

Modelled soil layers were chosen to align with soil profile and water measurements (Table 7.3). PERFECT soil properties were based on site data, prior studies and regional data (see Table 7.4).

Table 7.4: PERFECT soil properties.

<table>
<thead>
<tr>
<th>Site</th>
<th>Soil type</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vicki</td>
<td>Dr 3.13</td>
<td>Massive sandy red duplex soil with structured loamy or sandy duplex soils (Northcote, 1979); CSIRO data used to support Benyon et al. (2008).</td>
</tr>
</tbody>
</table>
Table 7.5: *E. globulus* species parameters after Feikema et al. (2010); for additional information see their Appendix A and Table A1. **Emboldened** asterisked parameters (*) also shown with prior probability ranges.

<table>
<thead>
<tr>
<th>Description</th>
<th>Symbol</th>
<th>Units</th>
<th>Value</th>
<th>Description</th>
<th>Symbol</th>
<th>Units</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Beer’s law constant</td>
<td>$k_p$</td>
<td>-</td>
<td>0.55</td>
<td>Threshold value of $f_{soil}$ for drought litterfall</td>
<td>$l_{thresh}$</td>
<td>-</td>
<td>0.1</td>
</tr>
<tr>
<td>VPD modifier for canopy conductance</td>
<td>$k_g$</td>
<td>-</td>
<td>0.45</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Maximum canopy conductance*</td>
<td>$g_{cmax}$</td>
<td>$m s^{-1}$</td>
<td>0.018</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>[0.001,0.100]</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mortality function factor (parabolic phase)</td>
<td>$\beta_{para}$</td>
<td>-</td>
<td>-0.45</td>
<td>Allometric foliage coefficient</td>
<td>$a_f$</td>
<td>-</td>
<td>0.05</td>
</tr>
<tr>
<td>Mortality function factor (power phase)</td>
<td>$\beta_{power}$</td>
<td>-</td>
<td>78</td>
<td>Allometric stem coefficient</td>
<td>$a_s$</td>
<td>-</td>
<td>0.05</td>
</tr>
<tr>
<td>Fraction of mean stem biomass lost when tree dies</td>
<td>$mort_{fact}$</td>
<td>-</td>
<td>0.2037</td>
<td>Factor to convert C mass to biomass</td>
<td>$C_{biomass}$</td>
<td>$kg kg^{-1}$</td>
<td>2.2</td>
</tr>
<tr>
<td>Constant for age effect on growth</td>
<td>$max_{age}$</td>
<td>years</td>
<td>80</td>
<td>Mean stemwood basic density</td>
<td>$\rho$</td>
<td>$kg m^{-3}$</td>
<td>525</td>
</tr>
<tr>
<td>Power for age effect on growth</td>
<td>$n_{age}$</td>
<td>-</td>
<td>2</td>
<td>Maximum temperature for growth</td>
<td>$T_{max}$</td>
<td>°C</td>
<td>32</td>
</tr>
<tr>
<td>Canopy quantum efficiency*</td>
<td>$\alpha_c$</td>
<td>-</td>
<td>2.3</td>
<td>Minimum temperature for growth</td>
<td>$T_{min}$</td>
<td>°C</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td></td>
<td>[0.5]</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ratio of net to gross primary production</td>
<td>$c_{pp}$</td>
<td>-</td>
<td>0.47</td>
<td>Optimum temperature for growth</td>
<td>$T_{opt}$</td>
<td>°C</td>
<td>19</td>
</tr>
<tr>
<td>Initial mass (above &amp; below-ground) of a single tree</td>
<td>$w_{init}$</td>
<td>kg</td>
<td>0.04</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fraction of initial tree mass that is foliage</td>
<td>$fol_{frac}$</td>
<td>-</td>
<td>0.5</td>
<td>Branch fraction final value</td>
<td>$BF_{finat}$</td>
<td>-</td>
<td>0.25</td>
</tr>
<tr>
<td>Fraction of initial tree mass that is stem</td>
<td>$stem_{frac}$</td>
<td>-</td>
<td>0.4</td>
<td>Branch fraction exponential decline factor</td>
<td>$BF_{dec}$</td>
<td>-</td>
<td>-0.6</td>
</tr>
<tr>
<td>Fraction of initial tree mass that is root</td>
<td>$root_{frac}$</td>
<td>-</td>
<td>0.1</td>
<td>Coefficient to limit early growth</td>
<td>$early_{co}$</td>
<td>-</td>
<td>1.5</td>
</tr>
<tr>
<td>Root turnover fraction per month</td>
<td>$\gamma_r$</td>
<td>-</td>
<td>0.1</td>
<td>Coefficient to limit early growth</td>
<td>$early_{exp}$</td>
<td>-</td>
<td>1.2</td>
</tr>
<tr>
<td>Root allocation</td>
<td>$r_a$</td>
<td>-</td>
<td>0.6</td>
<td>Specific leaf area for younger trees</td>
<td>$SLA_{init}$</td>
<td>$m^2 kg^{-1}$</td>
<td>16</td>
</tr>
<tr>
<td>Root allocation</td>
<td>$r_b$</td>
<td>-</td>
<td>8</td>
<td>Specific leaf area for older trees</td>
<td>$SLA_{finat}$</td>
<td>$m^2 kg^{-1}$</td>
<td>4</td>
</tr>
<tr>
<td>Maximum litterfall fraction per month</td>
<td>$\gamma_{fmax}$</td>
<td>-</td>
<td>0.05</td>
<td>Specific leaf area exponential decline factor</td>
<td>$SLA_{dec}$</td>
<td>-</td>
<td>-0.8</td>
</tr>
<tr>
<td>Constant in litterfall function</td>
<td>$c_f$</td>
<td>-</td>
<td>6</td>
<td>Maximum root depth</td>
<td>$d_{wemax}$</td>
<td>mm</td>
<td>4000</td>
</tr>
<tr>
<td>Constant in litterfall function</td>
<td>$k_f$</td>
<td>-</td>
<td>0.5</td>
<td>Term linking crown storage of rainfall to LAI</td>
<td>$l_L$</td>
<td>$mm L^{-1}$</td>
<td>0.45</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Maximum fraction of rainfall intercepted per day</td>
<td>$l^*$</td>
<td>-</td>
<td>0.75</td>
</tr>
</tbody>
</table>

Monthly mean wind speed data were taken from BOM data collected at Mount Gambier ([http://www.bom.gov.au/climate/averages/tables/cw_026021.shtml](http://www.bom.gov.au/climate/averages/tables/cw_026021.shtml)) as the average between 9am and 3pm mean wind speed (km/h) for years 1941 to 2010. 3-
PG+ species parameters used for *E. globulus* after Feikema et al. (2010), are shown in Table 7.5.

Uniform distributions were assigned over prior ranges centered on the values in Table 7.5 above ($g_{c,m,a,x}$ [0.001,0.100]; $\alpha_c$ [0.5,5.0]), while fertility rating $F_R$ was varied over its full range of [0,1].

### 7.5 Results

The analysis was characterised by extended run times. Each iteration of 3-PG+ took approximately two seconds to complete, limited by the time taken to execute the Fortran object. Implementing parallel programming was beyond the capacity of the project which meant that access to a supercomputer provided little increase in efficiency. Simulation times extended to 24 hours for 3-PG monthly runs and nearly 30 for daily 3-PG+ runs (Table 7.6, below).

<table>
<thead>
<tr>
<th>Site</th>
<th>Duration 3-PG</th>
<th>Duration 3-PG+</th>
</tr>
</thead>
<tbody>
<tr>
<td>DPI</td>
<td>23:35:38 ; 1.7</td>
<td>28:46:49 ; 2.1</td>
</tr>
<tr>
<td>Jill</td>
<td>23:32:48 ; 1.7</td>
<td>28:22:31 ; 2.0</td>
</tr>
<tr>
<td>Vicki</td>
<td>24:36:45 ; 1.8</td>
<td>27:38:22 ; 2.0</td>
</tr>
</tbody>
</table>

Results of the $\hat{R}$ statistic (Gelman and Rubin, 1992) indicates acceptable mixing ($\hat{R} < 1.2$) (see Vrugt et al., 2009a) after 1,000 thinned iterations for 3-PG and after 2,000 for 3-PG+ for the selected model parameters $\alpha_c$, $F_R$ and $g_{c,x}$ (Figure 7.3, below), the delayed convergence of the 3-PG+ simulations is consistent with its greater complexity.

Patterns are similar among the statistical parameters (Figure 7.4) however convergence varied notably between sites. Monthly data from the Vicki site showed unconvincing convergence among the $std_\alpha$, $\beta$ and $\xi$ parameters (Figures 7.4g,i and j). The $\hat{R}$ statistic for the $\beta$ parameter is particularly chaotic (Figure 7.4i).
Figure 7.3: Gelman shrink factor for 3-PG model parameters $\alpha_c$, $F_R$ and $g_{c_{\text{max}}}$ for both the original implementation of 3-PG (a-i) and 3-PG+ (j-r) at each site.
Figure 7.4: Gelman shrink factor for statistical parameters of the generalised likelihood for both the original implementation of 3-PG (a-o) and 3-PG+ (p-ad) at each site
Figure 7.5: DPI – Markov chains 3-PG (a-c) and marginal posterior distributions (d-e); Markov chains 3-PG+ (g-i) and marginal posterior distributions (j-l).
Figure 7.6: Jill – Markov chains 3-PG (a-c) and marginal posterior distributions (d-e); Markov chains 3-PG+ (g-i) and marginal posterior distributions (j-l).
Figure 7.7: Vicki – Markov chains 3-PG (a-c) and marginal posterior distributions (d-e); Markov chains 3-PG+ (g-i) and marginal posterior distributions (j-l)
Figure 7.8: Marginal posterior distributions for 3-PG - DPI (a-e); Jill (f-j); Vicki (k-o) and 3-PG+ statistical parameters - DPI (p-t); Jill (u-y); Vicki (z-ad).
Based on these results, a further 2,000 iterations of each chain were discarded and the remaining 3,000 pooled into the final 30,000 sample posterior distribution from each site for inference. Markov chain traces and marginal posterior probability distributions for selected model parameters from 3-PG and 3-PG+ are shown for each site in Figures 7.5 to 7.7. Marginal posterior probability distributions for the likelihood statistical parameters for each site and both models are shown in Figure 7.8.

**DPI**

Markov chains from the 3-PG analysis at DPI showed good mixing/convergence (Figure 7.5a-c) consistent with the low 쌨 factors in Figure 7.3(a)-(c), but model parameter identification (a_c; F_R; g_{cmax}) was generally poor (Figure 15e-f) and peaks of maximum likelihood were not consistent with known theory as reflected in *E. globulus* parameterisation (see Table 7.5 and ‘est’ data in Table 7.7). The addition of a daily time-step and a multilayered soil water balance in 3-PG+ led to tighter confidence limits and a greater ability to delineate complexity in the posterior distribution (Figure 7.5j-l). Minor peaks at: a_c ~ 2.5; F_R ~ 0.8 - 0.9; g_{Cx} ~ 0.035 are either close to expected values or at least in the right order of magnitude (g_{Cx}).

**Table 7.7:** Model and statistical parameters, ~reasonable expectation, prior ranges, and 95% confidence limits; lwr = 2.5%, upp = 97.5%.

<table>
<thead>
<tr>
<th>pars: est</th>
<th>a_c</th>
<th>F_R</th>
<th>g_{Cx}</th>
<th>std_o</th>
<th>std_t</th>
<th>b</th>
<th>l</th>
<th>φ_1</th>
</tr>
</thead>
<tbody>
<tr>
<td>prior</td>
<td>0.50</td>
<td>0.10</td>
<td>0.93</td>
<td>6.6e^3</td>
<td>1.0e^3</td>
<td>143</td>
<td>200</td>
<td>2.3e^2</td>
</tr>
<tr>
<td></td>
<td>2.50</td>
<td>0.70</td>
<td>1.00</td>
<td>3.2e^2</td>
<td>9.7e^2</td>
<td>141</td>
<td>190</td>
<td>-9.5e^2</td>
</tr>
<tr>
<td></td>
<td>1.15</td>
<td>0.18</td>
<td>1.00</td>
<td>1.2e^2</td>
<td>1.0e^1</td>
<td>27.1</td>
<td>35.1</td>
<td>0.56</td>
</tr>
<tr>
<td></td>
<td>3.85</td>
<td>4.58</td>
<td>0.75</td>
<td>9.1e^2</td>
<td>1.0e^1</td>
<td>18.3</td>
<td>26.4</td>
<td>0.19</td>
</tr>
<tr>
<td></td>
<td>0.51</td>
<td>0.19</td>
<td>0.98</td>
<td>4.9e^3</td>
<td>9.8e^2</td>
<td>300</td>
<td>-30</td>
<td>-1.00</td>
</tr>
<tr>
<td></td>
<td>2.23</td>
<td>5.00</td>
<td>0.38</td>
<td>2.2e^2</td>
<td>6.1e^2</td>
<td>158</td>
<td>207</td>
<td>-2.6e^1</td>
</tr>
</tbody>
</table>

1 model parameters from Table 7.5, F_R, reasonable expectation; statistical parameters for a normal distribution – prior ranges are shown in [ ]; 
2 3-PG results; 3 3-PG+ results.

Mixing in the 3-PG+ simulation was not quite as convincing as the monthly 3-PG simulation. R hovered around the Vrugt et al. (2009a) limit of 1.2 (Figure 7.3g-i) and a large amount of probability mass fell near the upper limit of the prior distribution (Figure 7.5j-l), indicating a weakness in model structure resulting in the posterior distribution being unduly influenced by the choice of the prior range.

Confidence intervals in DPI statistical parameter estimates (std_o, std_t, b, l, φ_1) were similar in both 3-PG and 3-PG+ simulations (compare Figures 7.8a-e and 7.8p-t), 3-PG+ producing greater symmetry and less ambiguous inference in some cases (Figure 7.8r). Confidence in statistical parameter estimates would likely improve with the
running of longer simulations, however the additional investment in time would not substantially change the conclusions drawn in this study.

**Jill**
Monthly 3-PG results from Jill showed comparably good mixing (Figure 7.6a-c) and somewhat better parameter identifiability than at DPI (Figure 7.6d-f). Model structure showed improvement under 3-PG+ with tighter confidence limits around model parameter estimates but $\alpha_c \sim 4.2$ was inconsistent with theory while: $g_{Cx} \sim 0.1$ was determined by the upper limit of the prior range suggesting an incompatibility between 3-PG+ model structure and/or the information content of the data. Like DPI, the statistical parameters were well identified by both and showed improved symmetry in some parameters in the 3-PG+ simulation (Figures 7.8f-j) and 7.8u-y).

**Vicki**
Vicki showed improved parameter identification between the 3-PG and 3-PG+ simulations. Like DPI and Jill sites, mixing was acceptable in the 3-PG analysis but unlike DPI and Jill, Vicki retained better mixing under 3-PG+ than was evident at the other sites (Figures 7.7a-c and Figure 7.7g-i). Parameter estimates from the monthly 3-PG run had similar probability across the prior range (Figures 7.7d-f). A peak at $\alpha_c \sim 1$ among a background of high probability is the only indication of the model's discriminatory power.

$\hat{R}$ of the model parameter chains indicated reasonable convergence for both the 3-PG and 3-PG+ runs (Figure 7.3g-i and Figure 7.3p-r), however $\hat{R}$ of some statistical parameter chains in the 3-PG run were high (Figure 7.4k,n) and at times chaotic (Figure 7.4m).

Output from 3-PG+ indicated more convincing mixing (Figure 7.4z-ad), consistent with the relatively good mixing evident in Figure 7.3(p)-(r). Marginal statistical posterior distributions generated by the monthly 3-PG simulation show a mass of low-level probability (Figure 7.4k-o), suggesting the need for longer Markov chains not evident in the tighter densities generated by the daily 3-PG+ simulation with the same number of iterations (Figure 7.4z-ad).

Maximum likelihood model parameters identified by 3-PG+ ($\alpha_c \sim 2.6; F_R \sim 0.9; g_{Cx} \sim 0.03$) showed close accord with expected values in Table 7.7 ($\alpha_c = 2.3; F_R = 0.8 \pm 0.9; g_{Cx} = 0.018$).
7.6 Discussion

While veracity is necessary to both research and management, it is not sufficient to completely satisfy the needs of either. Research requires originality (Melbourne Research, 2010) and as originality is to research, confidence is to management (Bennett et al., 2013; Sargent, 2013).

Where related work has demonstrated that increased complexity in 3-PG’s soil-water module improved the veracity of modelling results (Almeida and Sands, 2015; Feikema et al., 2010b), this work demonstrates that it enables the model to extract more information from the data, improving confidence by producing tighter credible intervals around the parameter estimates; in other words producing the right results for the right reasons (Beven, 1989; 2006; Clark et al., 2011; Goswami and O’Connor, 2010; Gupta et al., 2012; Gupta and Sorooshian, 1983; Jakeman and Hornberger, 1993; Kirchner, 2006; Klemeš, 1982; Troutman, 1985).

The finding that 3-PG+ benefitted from the introduction of additional complexity is notable. Recent directions in model development favour simplicity (Clark et al., 2011; Gupta et al., 2012; Sivapalan et al., 2003a) due to the higher levels of uncertainty associated with greater complexity (Jakeman and Hornberger, 1993; Schoups et al., 2008). However the simplicity of the single layer monthly water balance was conspicuous against the sophistication of the rest of the model and adding complexity is not necessarily anathema to prudent model development, particularly when balanced by expert-knowledge to constraining (Hrachowitz et al., 2014; Schoups et al., 2008).

Almeida and Sands (2015) used insight based on the expert-knowledge of developers (Sands and Landsberg, 2002) to produce a two-zone soil moisture model and an event-driven pseudo time-step commensurate with the pragmatic philosophy of 3-PG while Feikema et al. (2010b) used the generic seven-layer PERFECT soil module for 3-PG+. Only two layers were required to run the 3-PG+ analyses (five being redundant), and outputs from the daily time-step were aggregated into a monthly time-step for use in the 3-PG+ transpiration-growth calculations. Despite the immediate success of incorporating the capability of PERFECT into 3-PG+, its lack of capacity in modelling sub-monthly transpiration is likely to remain a limitation to further improvement.

A benefit of 3-PG+ within PERFECT is the access it brings to a suite of pre-afforestation land-cover. Establishing yield from standard pre-conversion land-cover
was recognised as an essential part of a forest management framework in South Africa (Edwards and Roberts, 2006; Gush et al., 2002a). South Australian regional runoff reductions are estimated at 85% of pre-plantation yield (GSA, 2009), the reduction is based on observed yield from a small, two-thirds cleared pine plantation on the slopes of a reservoir with shallow soils (Greenwood and Cresswell, 2007). Impacts on yield in deeper soils are known to be greater in South Africa (see Dye and Poulter, 1991; Scott and Lesch, 1997).

The limitations of using South Australian runoff reductions have been identified by independent reviewers (Prosser and Walker, 2009) but their implications have yet to be appreciated by decision-makers. As it stands the approach is likely to underestimate impacts in some areas and lack the flexibility of mechanistic modelling to be convincingly adjusted for different site and climatic conditions.

The importance of site conditions and data quality were apparent in this analysis. The 3-PG monthly analysis of Vicki data showed problems in convergence of the $\beta$ kurtosis statistical parameter (Figure 7.4m). Model parameter identification improved in the 3-PG+ analysis (Figure 7.7) but less convincingly than other sites. Posterior distributions of the statistical parameters from the monthly run show a broad background of low probability indicating greater difficulty in parameter identification and a need for a longer chain and more burn-in (Figure 7.8k-o). While the daily 3-PG+ run showed excellent improvements in statistical parameter identification (Figure 7.8z-ad) the mixed convergence and need for longer run times indicates a problem with the Vicki data quality, at least at a monthly time-step. No immediate explanation is apparent except that problems with Vicki data have been noted elsewhere (site Vikki in Almeida and Sands, 2015).

The generalised likelihood of Schoups and Vrugt (2010) is not currently a part of the hydromad R package (Andrews et al., 2011). It was used to overcome violating the assumptions inherent in parameter estimation in the presence of non-Gaussian errors; a particular concern when pooling disparate information (transpiration, soil water and growth) into a likelihood without regard for compatibility (Kuczera, 1983), a practice used in forest growth studies that been found to reduce uncertainty (Minunno et al., 2013; Van Oijen et al., 2011; Van Oijen et al., 2013; Van Oijen et al., 2005).

Process models used in research are often calibrated manually using expert judgement (Almeida and Sands, 2015; Landsberg and Sands, 2011). In contrast, hydrological models used to support decision-making are mostly calibrated using
automated optimisation algorithms (Efstratiadis and Koutsoyiannis, 2010). A strength in using automated calibration is the reduced reliance on expert opinion which can be perceived as weakening the defensibility of decision-making (Krueger et al., 2012; Mizrahi, 2013; Sutherland and Burgman, 2015; Sutherland et al., 2013). The effectiveness with which 3-PG+ was able to identify parameters using an automated calibration algorithm speaks for its potential to be developed toward supporting forest water management.

Unfortunately the method used to achieve calibration is not particularly well-suited to water management. While implementing MCMC in R is described as straightforward (Robert and Casella, 2010), it requires well-developed knowledge of statistics and probability theory much more widely available among researchers than water managers (Section 3.4). The benefits of conducting Monte Carlo uncertainty analyses are many, but they are offset by heavy computational demands and extended run-times, which are not conducive to fast turn-around sometimes required in supporting the provision of policy advice. Routine employment of these techniques in Australian water management agencies will require overcoming significant limitations in scientific capacity (Chartres, 2006).

If 3-PG+ does indeed have potential to support water management, it needs to be demonstrated more convincingly than was done by (Marcar et al., 2010) and (Feikema et al., 2008) and with greater attention to notions beyond performance as was done for 3-PG+ by (Feikema et al., 2010b) or by (Almeida and Sands, 2015) in their latest invocation of 3-PG. Water management is highly pragmatic in its use of science (Klemeš, 1973; Passioura, 1996), can elements of a sophisticated Bayesian analysis be brought across the science-policy interface and by unambiguously demonstrating the benefits of the 3-PG+ modifications (Feikema et al., 2010b), provide a convincing argument for further development for application in forest water management?
8. DISCUSSION

... we are not bound to accept a chain of reasoning because it is self-consistent and keeps to the line laid down, if it starts from false premises. [Moreover, the more logical the system is, the more the premises are proved to be false by the obvious falseness of the conclusions.]

(Cicero, de Finibus Bonorum et Malorum, IV.xix, 53)

WHILE the hydrological impacts of afforestation continue to be recognised as a risk to water resources around the world (Cao and Zhang, 2015; Döll et al., 2015; Veldman et al., 2015; Zheng et al., 2016) particularly in dry regions (Bodirsky and Popp, 2015; Dye and Versfeld, 2007), only South Africa and Australia have regulated plantation water use. Four decades of continuous empirical observation provided the scientific basis to develop relationships that underpin South African forest water regulation, which was introduced in 1972 (Van der Zel, 1982). Australian catchment studies commenced in the 1950s but were not driven by the consistency of purpose of a dedicated research organisation or a cohesive national water management agenda (Costin and Slatyer, 1967) and suffered from short-term funding cycles of the various jurisdictions (Bren and McGuire, 2007).

In the early 1990s disparate hydrology research agendas began to align under cooperative research organisations (eWater, 2011) and widespread resource degradation prompted national water reform (COAG, 1994). The hydrological impacts of afforestation were considered a positive means of managing dryland salinity and driving plantation expansion by federal tax incentive schemes seemed like an efficacious way to address the salient water management issue of the time (Leys and Vanclay, 2010; Passioura, 2005).

The Millennium Drought brought the prospect of changing water availability to the Federal Government in Canberra (Van Dijk et al., 2013) in the centre of the nation’s iconic Murray-Darling Basin, prompting the invigoration of its 1994 reform agenda under the National Water Initiative (NWI) which recognised large-scale plantation forestry as an activity with the potential to threaten the security of water access entitlements (COAG, 2004a).

**Australian forest water regulation and the National Water Initiative**

The NWI raised awareness of the risks large-scale afforestation posed to water resources and provided a national framework in which to manage them. However it,
delivered little additional value where jurisdictions were already addressing forest water issues at regional scales.

At the time of signing the NWI in 2004, South Australia had already introduced forest water legislation in the State’s Lower South East; Victoria and Western Australia had commenced relevant water reforms that encompassed managing the water use of plantation forestry; Tasmania had recognised the issue as key and developed an assessment tool (Brown et al., 2006a) funded through a pre-NWI water quality initiative (AG, 1997a), while New South Wales had recognised it as significant (Duggan et al., 2008).

After 10-years of NWI implementation, South Australia remains the only jurisdiction with legislation to regulate forest water use and pre-NWI reforms in Victoria and Western Australia are yet to be enacted. The mixed progress was a consequence of declining plantation expansion due to taxation changes related to managed investment schemes and industry contraction during the global financial crisis (Treasury, 2011) and the constitutional independence of jurisdictional water management responsibilities.

The wording of the relevant NWI sections (s 55 - 57) enables jurisdictions to mark their own scorecards on water reform (NWC, 2014a). A jurisdiction is only obliged to enact the relevant provisions of the NWI if they are assessed as being significant, even in water systems that are fully allocated, over-allocated, or approaching full allocation (s 57(i) and (i)(a)), arguably exacerbating an acknowledged risk by offering a formal mechanism to ignore it.

Using tools based on methods demonstrated in this study as inappropriate for Australian conditions and water management in general (see Sections 5 and 6), both Tasmania and New South Wales were able to declare that the water use of large-scale plantation forestry was no longer an issue, largely by assessing the hydrological impacts of plantation forestry at inappropriate scales (CSIRO, 2008c; DPIPWE, 2012; DPIW, 2008; NWC, 2014c; Polglase and Benyon, 2009).

In hindsight the NWI did little to progress established jurisdictional forest water management agendas and enabled others to effectively side-step them. While its hydrological isolation makes Tasmania distant from the need to undertake further forest water reform, the Water Act 2004 (Commonwealth) contains provisions for the Murray-Darling Basin Authority to monitor and independently evaluate New South
Wales' response to forestry interception that are yet to be tested (for example: s 20(c); s 172(1b)(vi)) and may bring jurisdictions to greater account.

**Unfinished business**

The reports prepared to review the state of NWI implementation at the close of the National Water Commission identified significant unfinished business associated with interception activities (NWC, 2014c; SCEW, 2013). In the light of the qualified wording and the overarching discretion of jurisdictional water management under the Australian Constitution, further attempts to progress national management of interception under sections 55 – 57 of the NWI are likely to be unproductive. A chain of reasoning is not valid if it starts from false premises (Cicero) and without foresight may lead to unintended (Dye and Versfeld, 2007; Pittock et al., 2013; SCEW, 2013) and even paradoxical consequences (Allan and Aroney, 2008).

Other sections of the NWI not directly associated with interception have potential to deliver outcomes more in line with its objectives of ensuring secure water access entitlements (s 23(i)). Outside the NWI’s interception sections (s 55 - 57) and excluding schedules, the term *land-use change* occurs in three other places:

- s 25, Water Access Entitlements and Planning Framework (s 25-57);
- s 82, Consolidated Water Accounts (Water Resource Accounting s 80-89); and,
- s 98, Knowledge and Capacity Building (s 98-108).

Section 25 stipulates that, *water access entitlements and planning frameworks will: … xi) protect the integrity of water access entitlements from unregulated growth in interception through land-use change.* As shown in South Australia (Section 3.3.1), concerns regarding the security of water access entitlements against unregulated afforestation emerge at property scale due to the local nature of its hydrological impacts (CSIRO, 2008c; Polglase and Benyon, 2009). When land-use cover change is small compared to catchment area, associated hydrological changes in streamflow are non-linear and difficult to detect due to uncertainties in hydrometric and hydrological methods (Bosch and Hewlett, 1982; McMinn and Hewlett, 1975), and the influence of forest location and the consequent masking effects of less impacted catchment inflows (Vertessy et al., 1996 and discussion in; Vertessy et al., 2003).

Assessing the significance at catchment-scales is not consistent with Section 25 as it is unlikely to detect the impacts of unregulated growth in interception by afforestation before the integrity of water access entitlements are at risk.
NWI sections 80 to 89 relate to Water Resource Accounting and form the basis of a significant body of work undertaken by the Australian Bureau of Meteorology in producing national water accounts (BOM, 2011). Comprehensive annual accounts are currently produced for a selected number of priority areas across Australia, including the Murray-Darling Basin and the regional Western Mounty Lofty Ranges encompassing Adelaide and significant areas of plantation forestry (BOM, 2015b).

Section 82 stipulates: Recognising that robust water accounting will protect the integrity of the access entitlement system, the Parties agree to develop and implement by 2006: … iii) water resource accounts that can be reconciled annually and aggregated to produce a national water balance, including: … c) consideration of land use change, climate change and other externalities as elements of the water balance.

Notwithstanding the observations regarding large-scale assessments and detecting the risks to the integrity of access entitlement systems, the accounts in their current form do not display a capacity to consider land-use change or other interception activities as an element of the water balance. The accounting region of Adelaide contains some 160 km$^2$ of forestry (most recent account BOM, 2015a), but the Water Accounting Statements do not contain any reference to forestry land-use (as a part of the water balance, despite the significant regional forest water policy development that occurred before the account (GSA, 2009) and current water allocation planning framework which accounts for the impacts of interception by large-scale plantation forestry in the water balance (NRAMLR, 2013). Moreover liabilities in the water account are limited to water licensing information and meter readings and do not incorporate interception activities like commercial forestry or stock and domestic farm dam developments.

The effectiveness of the National Water Account in meeting section 82(c)(iii) is evidently limited, however the resources it commands could be re-focused toward collating information that was more valuable for jurisdictional management of interception.

Sections 98 to 101 are concerned with Knowledge and Capacity Building. Agreed actions include identifying the key knowledge and capacity building priorities needed to support the implementation of the NWI (s 100-101). Water management is a wicked problem (Freeman, 2000; Lach et al., 2005), as indeed is forestry (Allen and Gould, 1986; Shindler, 1999), which combine to become a doubly wicked problem with a particularly opaque science-policy interface prone to policy and scientific misalignments (Section 3.3.1), making capacity building particularly challenging.
Recognition that forest hydrology is wicked is not helpful on its own. However, the options for addressing wicked problems from a scientific perspective are constrained to quantifying uncertainty, particularly in a way which engenders confidence in stakeholders (APSC, 2007; Rittel and Webber, 1973), suggesting a philosophy for water management modelling.

Building capacity by integrating science and policy typically starts with policy-developers determining research priorities from researchers (eg Vose et al., 2011). Investment is typically in established agendas rather than new management priorities which are difficult to articulate, resulting in agendas that reflect the interests of researchers over those of policy makers. This outcome became evident in South Australia where the jurisdictional water management agency was forced to defend its advice against challenges based on outputs developed by government-funded cooperative research organisation (Section 3.3.1).

Other approaches for determining research needs are available. Two pursued in this study are a review of similar management experience with addressing a similar issue; and a more abstract approach of attempting to determine what isn’t known by the policy developers themselves. The first was available in the South African experience discussed further below while the second was addressed using a survey of South Australian agency experts (Section 3.4.1), a jurisdiction leading the nation in managing interception by large-scale plantation forestry.

The survey revealed that even agencies with significant, relevant experience may be blind to important knowledge gaps, such as climatic drivers of plantation water use, and as such not well equipped to identify and fill them, and perhaps how a leading water management agency can continue claim world firsts in developing forest water use and management legislation (GSA, 2016; WIA, 2012b), when South Africa had achieved the same milestone some 40 years before (Van der Zel, 1995).

**The South African experience**

South Africa has been grappling with the hydrological impacts of afforestation for the better part of a century. It is striking that so little of their work has been given closer attention in both water management and relevant research. Of the Australian forest hydrological research literature reviewed for this project, the greatest number of South African citations found were five in two papers: the first a review by Brown et al. (2005) (Van Lill et al., 1980; Dye 1996; Scott and Lesch, 1997; Scott and Smith, 1997; Scott et al., 2000) and the second, an article in an industry journal by Vertessy et al. (2000).

Ten different South African sources in three studies; arguably a little less than might be expected from a world leading knowledge-base but better than the efforts of studies to support water management; Zhang et al. (2010) cited three (Scott, 2000; Scott and Smith, 1997; Bosch and Hewlett, 1982), while Greenwood and Cresswell (2007) supporting South Australian policy development in integrated surface-groundwater systems cited just one (Scott and Smith, 1997).

The review of South African hydrological research undertaken for forest water management conducted in this project identified the need to:

- ensure empirical methods are defensible and robust;
- define a defensible approach to define repeatable pre-conversion/reference condition catchment yield, particularly for the use of proportional reductions;
- accommodate more flexible forest management practices;
- depict growth-dependent water use dynamics;
- address varying site conditions; and,
- address varying climatic influences.

**Empirical modelling**

The approaches used in South Australia, the only jurisdiction to regulate forest water use outside of South Africa, are empirical and lack the sophistication of any of the needs described above. The reliance of Australian forest water management on empirical data is confounded by its limited knowledge-base, requiring more be made of available information. A novel way by which this may be done is described in Section 5 of this work, the transparency of which lends itself to verifying the suitability of site data for use in regional water management.

Further progress may be made by integrating the available South African and Australian data such as shown in Figure 8.1. The empirical data used to support South Australian policy is coherent with that of South Africa, offering an empirical water planning rule that could be used as a basis for trading plantation water allocations, from a baseline evapotranspiration to a maximum water use. From a South Australian perspective that would amount to integrating the regional accounting rules used in the Lower South East (Harvey, 2009) with the deemed interception
reductions the Mount Lofty Ranges and Kangaroo Island (Greenwood and Cresswell, 2007; Prosser and Walker, 2009), with the potential to accommodate more flexible forest management practices according to demonstrable growth-dependent water use dynamics and possibly lead to refinements reflecting the influences of site versus climatic drivers during different times of the accounting cycle (see Figure 8.1 inset graphic).

**Figure 8.1:** South African age-evapotranspiration curves derived from observed data from seven sites from Jarmain et al. (2009) after Dye and Bosch (2000), with mean annual South Australian data from Greenwood and Cressell (2007), where RR = runoff reduction; MP/R/E/ΔS = mean annual precipitation/runoff/evapotranspiration/ change in storage, presented by Greenwood (2012). Maximum impacts manifest at five years in half of the cases (purple line).

A key learning from the South African experience is the adoption of a single mechanistic modelling approach to meet the demands of forest and water managers, who are expected to make decisions at different scales and under the influence of a non-stationary climate. An important consideration in the South African system is the formal recognition of need to characterise pre-afforestation yield from different landscapes, a capacity handling within the agro-hydrological platform ACRU. Current South Australian proportional reductions assume that pre-afforestation yield may be equated to that generated from a two-thirds burned and cleared, pine plantation on a
hill-face with thin soils (Greenwood and Cresswell, 2007), a landscape that is clearly not typical of the wider Mount Lofty Ranges and Kangaroo Island.

Given the jurisdictional complexities inherent in implementing national water reforms, it is questionable whether Australia will venture down the pathway to a national assessment approach. However the capacity to model yields under a range of agreed pre-afforestation conditions would be an important consideration in selecting a national platform and experience overseas suggests it is likely to be inevitable.

**Mechanistic modelling**

If a single mechanistic modelling approach is indeed inevitable, a national programme of model scrutiny will be required that meets the needs of water management in defensible decision-making and building stakeholder confidence.

Quantifying uncertainty is a key tool in demonstrating confidence in competing models, although the detail of how it should be applied is vague (Black et al., 2011; COAG, 2010b; Vaze et al., 2012). The number of uncertainty assessment frameworks is large (Matott et al., 2009). A common theme among them is targeting the assessment to the purpose (Clark et al., 2011; Clark et al., 2008; Gupta et al., 2012; Gupta et al., 2008; Jakeman et al., 2006; Refsgaard and Henriksen, 2004; Refsgaard et al., 2006; Refsgaard et al., 2007).

Exacting uncertainty analyses may arrive at the required outcome but have the clumsiness of cracking a walnut with a proverbial sledgehammer (Section 6), and while Bayesian methods provide the required rigour they are complex and challenging to implement within water management agencies with knowledge and capacity constraints (Chartres, 2006; COAG, 2004b).

Beyond rigour a range of reasons can be found to support a case for the promotion of Bayesian approaches in water management, not least the epistemic paradigm in which science is often pursued and the formal integration of expert opinion into modelling (Section 6 and citations therein). Salient points in any analysis are the judicious use of performance criteria, addressing error assumptions and targeting uncertainty quantification:

- well-established non-parametric performance criteria are available with strong diagnostic power compared to mean square criteria such as the widely used Nash-Sutcliffe efficiency (Gupta et al., 2009; Nash and Sutcliffe, 1970);
• assumptions regarding the distribution of errors which confound parameter identification can be largely addressed using a generalised formal likelihood (Schoups and Vrugt, 2010); and,
• quantifying uncertainty where it is of greatest value in creating stakeholder confidence (see Section 6 and citations therein).

When combined with learnings from South Africa regarding the need to account for the effects of growth in forest water management and the less than convincing results from some Australian forest water management studies (Feikema et al., 2008; Marcar et al., 2010), the final bullet point leads to testing whether 3-PG+ could stand a simple Bayesian test of integrity to provide confidence that further development with the aim of supporting forest water management and regulation was worthwhile.

A somewhat surprising finding of the 3-PG+ analysis was that greater complexity of the somewhat ad-hoc developments of Feikema et al. (2010b) and the generic PERFECT soil water module did indeed lead to improved confidence in model performance, as demonstrated by an improved capacity to extract information from data. It is likely that the more targeted developments of Almeida and Sands (2015) would demonstrate greater confidence than those of Feikema et al. (2010b). The findings are timely as a version of 3-PG is currently being integrated into the national modelling platform (DSEWPC, 2012), but documentation to provide further information is limited (see http://www.toolkit.net.au/Tools/CLASS-3PG).
9. CONCLUSIONS

…”…virtues are formed in man by his doing the actions”; we are what we repeatedly do. Excellence, then, is not an act but a habit.

Aristotle, De Generatione Animalium, ii, 12, in Durant (1933),

You play as you train.

Ian Maxwell “Popsie” Greenwood, personal communication (c 2009)

FOREST hydrology is a young science. The influences of forests on water resources did not receive systematic scientific attention until the early 20th century. The field of forest influences first integrated socio-economic and environmental influences, akin to the modern moves toward socio-hydrology (Sivapalan et al., 2012), and became more concerned with latter in mid-20th century. The term forest hydrology first came into use in the middle of the 20th century (Kittredge, 1948) and became a recognised discipline in the mid-1960s (Sopper and Lull, 1967). While catchment management was a consideration in early Australian forestry (Incoll, 1936), the first dedicated national forest hydrology symposium was not held until 1982 (O'Loughlin and Bren, 1982).

Forest water management represents the confluence of two wicked policy areas which has proved contentious in the two countries forest water use has been regulated. Given its complexity and the similarity of many of the well-documented issues described in South African literature over decades, it is remarkable that more was not made of South African learnings through the first stages of Australian forest water regulation. Greater engagement with South African researchers and policy makers would facilitate the progress of Australian forest water management and the development of robust hydrological assessments support it.

South Africa enjoys two distinct advantages over Australia in progressing forest water management: a unitary constitution which is conducive to implementing national policy agendas; and a prolonged commitment to applied research projects to support them. More effective national Australian forest water reform could be implemented through constitutional reform that better addresses inconsistencies between inter-jurisdictional water management arrangements; tightening the discretionary language in the NWI’s interception sections (s 55 to 57); or giving greater attention to reporting interception activities under water accounting sections (s 80 to 89).
A cohesive national regulatory framework would provide a more strategic basis for longer-term applied research that reached beyond political cycles or short-lived responses to natural disasters, no matter how well-motivated or ambitious.

Applied research underpins evidence-based policy. Policy makers develop policy agendas and researchers develop research agendas. Misalignments at the science-policy interface may arise when water managers invest in established research agendas which are designed to satisfy academic requirements rather than address pragmatic management problems. Misalignments of this sort ultimately led to a loss of confidence in South Australian planning frameworks during the first stages of forest water regulation (Section 3.3.1). While partnerships between water management agencies and research organisations offer the best opportunity to support policy development by building organisational capacity, pursuing them from within organisational silos will from limitations in scientific capacity of the clients on the one hand and insights into the business of implementing policy by research organisations on the other (Section 3.4).

It is not contradictory to assert that water managers must develop and maintain sufficient scientific capacity to become more discerning research consumers to deliver the best from their research investments. Service delivery models do not build internal capacity or create the common understanding of issues required to develop applied research programmes to implement water management and support policy development. More fluid inter-organisational movement of personnel between government and research organisations of the type seen in other parts of the world can be used promote mutual understanding and progress policy agendas (Obama, 2015). Greater understanding of jurisdictional commitments to national reforms is likely to have resulted in less unfinished business at the closure of the NWC, and significantly greater progress of interception reforms than was achieved under the NWI.

As in the early stages of South African plantation water regulation, Australian national forest water management relies on empirical data and relationships to underpin its policies. A feature of the Australian experience is the lack of effective critical evaluation of tools used to underpin a suite of forest hydrological assessments intended to inform forest water management during the early phases, the consequences of which continue to be felt in Australian natural resources management (CA, 2011; Steendam and Schreiber, 2014).
Because the paucity of Australian empirical information is greater than in South Africa, uncertainties arising from extrapolating site data to different locations and climatic conditions are greater, with greater potential to be interpreted differently by different groups. As such, two issues are of utmost concern to assessing the hydrological impacts of afforestation for Australian forest water management: robustness of the information used in extrapolation and the quantification of uncertainty.

It is possible to make greater use of regional data to transparently evaluate the robustness of empirical site data through the use of analytical tools that put variability into regional context and when combined with sound hydrological theory can improve the defensibility of scientific advice used to support decision-making (Section 5).

Quantifying uncertainty is a more vexed question, particularly as resource management’s reliance on empirical information shifts to synthetic information generated through evermore complex modelling exercises. Testing the integrity of even simple models according to standards set by contemporary research can be arduous and frankly beyond the capacity of most water management agencies in terms of expertise and time. The identification and selection of elements of rigorous uncertainty assessment approaches that are most germane to fundamental principles of water management such as confidence, can lead to targeted more efficiently targeted assessments that are capable of being extended across resource limited agencies.

Combined with sufficient awareness of learnings from relevant experience, this approach can efficiently anticipate the directions that assessing the impacts of afforestation for water management will need to follow in coming years. While hardly rocket science, this philosophy has been less than conspicuous in the activities that have accompanied forest water management to the detriment of the most recent wave of Australian national water reforms.
10. REFERENCES


REFERENCES


CCWPL, 2012. Regulation of Groundwater Interception by Forestry: Lessons from Australia & International Regimes, Second Annual Workshop, Centre for Comparative Water Policies and Laws, University of South Australia, 7 June.


COAG, 2008a. Agreement on Murray-Darling Basin Reform, Council of Australian Governments, 3 July.


CPH Water, 2001. Design of a Decision Support System and Scenario Generator for the Assessment of Land Use Impacts on Water Resources Within a Water Management Area, Department of Water Affairs and Forestry and the Department for International Development, Strategic Environmental Assessment, April.


CSIRO, 2009. Surface Water Yields in South-West Western Australia, A report to the Australian Government from the CSIRO South-West Western Australia Sustainable Yields Project. CSIRO Water for a Healthy Country Flagship, Australia.


REFERENCES


DSE, 2011c. Western Region Sustainable Water Strategy, Department of Sustainability and Environment, Victorian Government, Melbourne, November.


DW, 2008. Upper Collie Surface and Groundwater Allocation Limits: Methods and Calculations, Department of Water, Government of Western Australia, February.


REFERENCES

DWLBC-PIRSA, 2007. Agreement on a Risk Policy for Plantation Forests and Water Resources in the Mount Lofty Ranges, Department of Water, Land and Biodiversity and Department of Primary Industries and Resources South Australia, 7 June, courtesy Conservation Council of SA.


Dye, P.J. et al., 2008a. Water-Use in Relation to Biomass of Indigenous Tree Species in
Woodland, Forest and/or Plantation Conditions, WRC Report TT 361/08, Water
Research Commission, December.

in twelve Eucalyptus plantation stands in Zululand, South Africa. For. Ecol. Manage.,

Dye, P.J. et al., 2008b. Modelling Vegetation Water Use for General Application in Different
Categories of Vegetation, WRC Report No. 1319/1/08, Water Research Commission,
Republic of South Africa, September.

Dyson, M., 2005. Overview of Statutory Frameworks - Risks to Shared Water Resources,
Report prepared for the Murray-Darling Basin Commission, Publication No. 02/07,
October.

Future Flow Patterns in the River Murray System, MDB Publication No. 01/07, Murray-
Darling Basin Commission, Canberra, September.


Efstratiadis, A., Koutsoyiannis, D., 2010. One decade of multi-objective calibration
DOI:10.1080/02626660903526292


Zerger, A., Argent, R.M. (Eds.), MODSIM05, Advances and Applications for Management
and Decision Making, December, Melbourne, pp. 2484-2490.


of the United Nations, Rome.

Agriculture Organization of the United Nations, Rome.

Farrell, D., 2012. $4 million to support a national hydrologic modelling platform, Senator the Hon Don Farrell, Parliamentary Secretary for Sustainability and Urban Water, media release, DF 12/029, 29 May.


Feikema, P. et al., 2008. Predicting and Managing the Impacts of Commercial Plantations on Catchment Water Balances, Forest and Wood Products Australia, Project No: PN04.4009, August.


REFERENCES


GSA, 2013. Managing Water in the Border Zone, Lower Limestone Coast Water Allocation Plan, Factsheet 8, Natural Resources South East, Government of South Australia, November

GSA, 2014. Notice of Declaration of the Lower Limestone Coast Prescribed Wells Area to be a 'Declared Forestry Area'. The South Australian Government Gazette, 48, 1 July: 3088.


GWA, 1914. Rights in Water and Irrigation Act 1914, Government of Western Australia.


GWA, 2007a. Government Response to a Blueprint for Water Reform in Western Australia, Government of Western Australia, February.

GWA, 2007b. State Water Plan 2007, Department of Water, Department of Premier and Cabinet, Government of Western Australia, October.

GWA, 2007c. Western Australia’s NWI Implementation Plan, April 2007, Department of Water, Government of Western Australia.

GWA, 2009a. Plantation Forestry and Water Management Guideline, Department of Water, Government of Western Australia, June.


Harvey, D., 2009. Accounting for Plantation Forest Groundwater Impacts in the Lower South East of South Australia, DWLBC report no 2009/13, Department of Water, Land and Biodiversity Conservation, Adelaide.


REFERENCES


Kuczera, G., Renard, B., Thyer, M., Kavetski, D., 2010. There are no hydrological monsters, just models and observations with large uncertainties! Hydrological Sciences Journal/Journal des Sciences Hydrologiques, 55(6): 980-991. DOI:10.1080/02626667.2010.504677


Morris, J.D., 1999. Further Development of the 3PG Forest Growth Model and Application to Young Eucalypt Stands in Victoria, Report No 99/058, Centre for Forest Tree Technology, Department of Natural Resources and the Environment, Melbourne, November.


REFERENCES


REFERENCES


REFERENCES


Schirmer, J., Pirard, R., Kanowski, P., 2016. Promises and perils of plantation forestry. In:
Panwar, R., Kozak, R., Hansen, E. (Eds.), Forests, Business and Sustainability.


inference of hydrologic models with correlated, heteroscedastic, and non-Gaussian

Schulze, R.E., 1984a. Hydrological Models for Application to Small Rural Catchments in
Southern Africa: Refinements and Development, South African Water Research
Commission Report No WRC 63/2/84, University of Natal, Agricultural Catchment
Research Unit.

Water SA, 10(1): 55-64.

Schulze, R.E., 2005. Selection of a suitable agrohydrological model for climate change
impact studies over southern Africa. In: Schulze, R.E. (Ed.), Climate Change and Water
Resources in Southern Africa: Studies on Scenarios, Impacts, Vulnerabilities and

Schulze, R.E., George, W.J., 1987. A dynamic, process-based, user-oriented model of forest

Schwappach, A.F., 1904. Forestry. Translated from the German of Dr. A. Schwappach by
Fraser Story and Eric A. Nobbs. The Temple Cyclopaedic Primers. Aldine House J.M.

187-199.

Scott, D.F., Lesch, W., 1995. Low flows and flow reductions in afforested catchments,
Hydrology and Water Resources in Southern Africa: The 7th SANCAH National
Hydrology Symposium, September. South African National Committee of the International
Association of Hydrological Sciences and the Institute for Water Research, Rhodes
REFERENCES


Sutherland, W.J., Spiegelhalter, D., Mark Burgman, M., 2013. Twenty tips for interpreting scientific claims. Nature 503: 335-337. DOI:10.1038/503335a


Van der Zel, D.W., 1982. The Afforestation Permit System, Directorate of Forestry, Department of Environment Affairs, Pretoria, RSA.


Van Keulen, H., Seligman, N.G., Benjamin, R.W., 1981. Simulation of water use and herbage growth in arid regions - A re-evaluation and further development of the model 'arid crop'. Agricultural Systems, 6(3): 159-193. DOI:10.1016/0308-521x(81)90001-9


APPENDIX A: EXTRACTS FROM THE NATIONAL WATER INITIATIVE.

OBJECTIVES

KEY ELEMENTS

Water Access Entitlements and Planning Framework

Water Resource Accounting

Knowledge and Capacity Building

OBJECTIVES

23. Full implementation of this Agreement will result in a nationally-compatible, market, regulatory and planning based system of managing surface and groundwater resources for rural and urban use that optimises economic, social and environmental outcomes by achieving the following:

i) clear and nationally-compatible characteristics for secure water access entitlements;

ii) transparent, statutory-based water planning;

iii) statutory provision for environmental and other public benefit outcomes, and improved environmental management practices;

iv) complete the return of all currently overallocated or overused systems to environmentally-sustainable levels of extraction;

v) progressive removal of barriers to trade in water and meeting other requirements to facilitate the broadening and deepening of the water market, with an open trading market to be in place;

vi) clarity around the assignment of risk arising from future changes in the availability of water for the consumptive pool;

vii) water accounting which is able to meet the information needs of different water systems in respect to planning, monitoring, trading, environmental management and on-farm management;

viii) policy settings which facilitate water use efficiency and innovation in urban and rural areas;

ix) addressing future adjustment issues that may impact on water users and communities; and

x) recognition of the connectivity between surface and groundwater resources and connected systems managed as a single resource.
55. The Parties recognise that a number of land use change activities have potential to intercept significant volumes of surface and/or ground water now and in the future. Examples of such activities that are of concern, many of which are currently undertaken without a water access entitlement, include:

i) farm dams and bores;

ii) intercepting and storing of overland flows; and

iii) large-scale plantation forestry.

56. The Parties also recognise that if these activities are not subject to some form of planning and regulation, they present a risk to the future integrity of water access entitlements and the achievement of environmental objectives for water systems. The intention is therefore to assess the significance of such activities on catchments and aquifers, based on an understanding of the total water cycle, the economic and environmental costs and benefits of the activities of concern, and to apply appropriate planning, management and/or regulatory measures where necessary to protect the integrity of the water access entitlements system and the achievement of environmental objectives.

57. Accordingly, the Parties agree to implement the following measures in relation to water interception on a priority basis in accordance with the timetable contained in their implementation plans and no later than 2011:

i) in water systems that are fully allocated, overallocated, or approaching full allocation:

   a) interception activities that are assessed as being significant should be recorded (for example, through a licensing system),

   b) any proposals for additional interception activities above an agreed threshold size, will require a water access entitlement-
      - the threshold size will be determined for the entire water system covered by a water plan, having regard to regional circumstances and taking account of both the positive and negative impacts of water interception on regional (including crossborder) natural resource management outcomes (for example, the control of rising water tables by plantations), and
      - the threshold may not apply to activities for restricted purposes, such as contaminated water from intensive livestock operations,

   c) a robust compliance monitoring regime will be implemented; and
ii) in water systems that are not yet fully allocated, or approaching full allocation:–

a) significant interception activities should be identified and estimates made of the amount of water likely to be intercepted by those activities over the life of the relevant water plan,

b) an appropriate threshold level will be calculated of water interception by the significant interception activities that is allowable without a water access entitlement across the entire water system covered by the plan—
   - this threshold level should be determined as per paragraph 57 (i)b) above, and

c) progress of the catchment or aquifer towards either full allocation or the threshold level of interception should be regularly monitored and publicly reported—
   - once the threshold level of interception is reached, or the system is approaching full allocation, all additional proposals for significant interception activities will require a water access entitlement unless for activities for restricted purposes, such as contaminated water from intensive livestock operations.
Water Resource Accounting (s 80 – s 89)

Outcome

80. The Parties agree that the outcome of water resource accounting is to ensure that adequate measurement, monitoring and reporting systems are in place in all jurisdictions, to support public and investor confidence in the amount of water being traded, extracted for consumptive use, and recovered and managed for environmental and other public benefit outcomes.

Actions

Benchmarking of Accounting Systems

81. Recognising that a national framework for comparison of water accounting systems can encourage continuous improvement leading to adoption of best practice, the Parties agree to benchmark jurisdictional water accounting systems on a national scale by June 2005, including:

i) state based water entitlement registering systems;

ii) water service provider water accounting systems;

iii) water service provider water use/delivery efficiency; and

iv) jurisdictional/system water and related data bases.

Consolidated Water Accounts

82. Recognising that robust water accounting will protect the integrity of the access entitlement system, the Parties agree to develop and implement by 2006:

i) accounting system standards, particularly where jurisdictions share the resources of river systems and where water markets are operating;

ii) standardised reporting formats to enable ready comparison of water use, compliance against entitlements and trading information;

iii) water resource accounts that can be reconciled annually and aggregated to produce a national water balance, including:

a) a water balance covering all significant water use, for all managed water resource systems;

b) systems to integrate the accounting of groundwater and surface water use where close interaction between groundwater aquifers and streamflow exist; and

c) consideration of land use change, climate change and other externalities as elements of the water balance.
States and Territories agree to identify by end 2005 situations where close interaction between groundwater aquifers and streamflow exist and implement by 2008 systems to integrate the accounting of groundwater and surface water use.

Environmental Water Accounting

The Parties agree that principles for environmental water accounting will be developed and applied in the context of consolidated water accounts in paragraph 82.

The Parties further agree to develop by mid 2005 and apply by mid 2006:

i) a compatible register of new and existing environmental water (consistent with paragraph 35) showing all relevant details of source, location, volume, security, use, environmental outcomes sought and type; and

ii) annual reporting arrangements to include reporting on the environmental water rules, whether or not they were activated in a particular year, the extent to which rules were implemented and the overall effectiveness of the use of resources in the context of the environmental and other public benefit outcomes sought and achieved.

Information

States and Territories agree to:

i) improve the coordination of data collection and management systems to facilitate better sharing of this information;

ii) develop partnerships in data collection and storage; and

iii) identify best practice in data management systems for broad adoption.

Metering and Measuring

The Parties agree that generally metering should be undertaken on a consistent basis in the following circumstances:

i) for categories of entitlements identified in a water planning process as requiring metering;

ii) where water access entitlements are traded;

iii) where in an area where there are disputes over the sharing of available water;

iv) where new entitlements are issued; or

v) where there is a community demand.

Recognising that information available from metering needs to be practical, credible and reliable, the Parties agree to develop by 2006 and apply by 2007:

i) a national meter specification;
ii) national meter standards specifying the installation of meters in conjunction with the meter specification; and

iii) national standards for ancillary data collection systems associated with meters.

**Reporting**

89. The Parties agree to develop by mid 2005 and apply national guidelines by 2007 covering the application, scale, detail and frequency for open reporting addressing:

i) metered water use and associated compliance and enforcement actions;

ii) trade outcomes;

iii) environmental water releases and management actions; and

iv) availability of water access entitlement against the rules for availability and use.

**Knowledge and Capacity Building (s 98 – s 101)**

98. This Agreement identifies a number of areas where there are significant knowledge and capacity building needs for its ongoing implementation. These include: regional water accounts and assessment of availability through time and across catchments; changes to water availability from climate and land use change; interaction between surface and groundwater components of the water cycle; demonstrating ecological outcomes from environmental flow management; improvements in farm, irrigation system and catchment water use efficiency; catchment processes that impact on water quality; improvements in urban water use efficiency; and independent reviews of the knowledge base.

99. There are significant national investments in knowledge and capacity building in water, including through the Cooperative Research programme, CSIRO Water Flagship and Land and Water Australia, State agencies, local government and higher education institutions. Scientific, technical and social aspects of water management are multidisciplinary and extend beyond the capacity of any single research institution.
Outcome

100. Parties agree that the outcome of knowledge and capacity building will assist in underpinning implementation of this Agreement.

Actions

101. Parties agree to:

i) identify the key knowledge and capacity building priorities needed to support ongoing implementation of this Agreement; and

ii) identify and implement proposals to more effectively coordinate the national water knowledge effort.
APPENDIX B: SURVEY
A 10-question survey to provide information to support a general discussion on the scientific capacity of a water management agency to assess the hydrological impacts of interception by afforestation and the contribution of the NWI.

ALL RESPONSES CONFIDENTIAL

CONTEXT

National Water Initiative

Knowledge and Capacity Building is a key element of the National Water Initiative (s24(vii)).

Knowledge and Capacity Building

98. This Agreement identifies a number of areas where there are significant knowledge and capacity building needs for its ongoing implementation. These include: regional water accounts and assessment of availability through time and across catchments; changes to water availability from climate and land use change; interaction between surface and groundwater components of the water cycle; demonstrating ecological outcomes from environmental flow management; improvements in farm, irrigation system and catchment water use efficiency; catchment processes that impact on water quality; improvements in urban water use efficiency; and independent reviews of the knowledge base.

99. There are significant national investments in knowledge and capacity building in water, including through the Cooperative Research programme, CSIRO Water Flagship and Land and Water Australia, State agencies, local government and higher education institutions. Scientific, technical and social aspects of water management are multidisciplinary and extend beyond the capacity of any single research institution.

Outcome

100. Parties agree that the outcome of knowledge and capacity building will assist in underpinning implementation of this Agreement.

Actions

101. Parties agree to:

i) identify the key knowledge and capacity building priorities needed to support ongoing implementation of this Agreement; and

ii) identify and implement proposals to more effectively coordinate the national water knowledge effort.
NAME (optional): ____________________________________________________________

BACKGROUND/ROLE:
☐ HYDROLOGIST  ☐ HYDROGEOLOGIST  ☐ ECOLOGIST  ☐ OTHER

Please specify: ________________________________________________________________

PROFESSIONAL QUALIFICATIONS: ____________________________________________

1. What experience do you have in impact assessment or supporting initiatives to address the ecohydrological impacts of afforestation by plantation forestry?
   eg: water resource impact assessment: modelling, monitoring; ecological impact assessment; provision of expert advice; planning; supporting policy development etc

   Was it:  ☐ mostly in research?  ☐ mostly for management?

2. Please rate your knowledge of issues associated with assessing/managing the ecohydrological impacts of afforestation.

<table>
<thead>
<tr>
<th>low knowledge</th>
<th>general knowledge</th>
<th>detailed knowledge</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 2 3 4 5 6 7 8 9 10</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

   Please rate your knowledge of forest processes.

3. Radiation; radiation interception and canopy dynamics

<table>
<thead>
<tr>
<th>low knowledge</th>
<th>general knowledge</th>
<th>detailed knowledge</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 2 3 4 5 6 7 8 9 10</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

4. Canopy processes and conductance

<table>
<thead>
<tr>
<th>low knowledge</th>
<th>general knowledge</th>
<th>detailed knowledge</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 2 3 4 5 6 7 8 9 10</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

5. Photosynthetic production and canopy light efficiency

<table>
<thead>
<tr>
<th>low knowledge</th>
<th>general knowledge</th>
<th>detailed knowledge</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 2 3 4 5 6 7 8 9 10</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

6. Stomatal conductance and processes

<table>
<thead>
<tr>
<th>low knowledge</th>
<th>general knowledge</th>
<th>detailed knowledge</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 2 3 4 5 6 7 8 9 10</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
7. Plant-root-soil-water interactions and processes:

<table>
<thead>
<tr>
<th>low knowledge</th>
<th>general knowledge</th>
<th>detailed knowledge</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>2</td>
<td>3</td>
</tr>
<tr>
<td>4</td>
<td>5</td>
<td>6</td>
</tr>
<tr>
<td>7</td>
<td>8</td>
<td>9</td>
</tr>
<tr>
<td>10</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

8. Soil-water (balance) dynamics:

<table>
<thead>
<tr>
<th>low knowledge</th>
<th>general knowledge</th>
<th>detailed knowledge</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>2</td>
<td>3</td>
</tr>
<tr>
<td>4</td>
<td>5</td>
<td>6</td>
</tr>
<tr>
<td>7</td>
<td>8</td>
<td>9</td>
</tr>
<tr>
<td>10</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

9(a) What tools/approaches have you used to assess impacts?

☐ EMPIRICAL MODELS/DATA  ☐ MECHANISTIC MODELS
☐ PLANT PROCESS MODELS  ☐ EXPERT OPINION
☐ FIELD SURVEYS        ☐ OTHER

Please specify: ____________________________________________________________

9(b) Can you name other tools/approaches?

Please specify: ____________________________________________________________

10. With specific regard to assessing the impacts of interception by plantation forestry:

(a) Do you know of any agency initiatives targeted at identifying the key knowledge and capacity building priorities needed to support the NWI? If so what?

(b) Can you identify any agency initiatives targeted at more effectively coordinating the national water knowledge effort?

Thank you for your time.

A

March 2015
Review

The first stages of Australian forest water regulation: National reform and regional implementation

Ashley J.B. Greenwood a,b,*

a Department of Forest and Ecosystem Science, University of Melbourne, Parkville, Victoria 3010, Australia
b CRC for Forestry, College Road, Sandy Bay, Tasmania 7050, Australia

ARTICLE INFO

Article history:
Received 17 August 2012
Received in revised form
7 December 2012
Accepted 25 January 2013
Published on line 1 March 2013

Keywords:
Forest hydrology
Regulation
Policy
Models
Water management

ABSTRACT

Australia has recently become the second country behind the Republic of South Africa to regulate plantation forest water use. To date South Australia is the only jurisdiction to have enacted state-wide forest water legislation which was introduced in accord with national water reforms, but stemmed from a series of earlier regional initiatives initially prompted by water accounting issues and the concerns of existing water users regarding the ongoing security of their entitlements. National reforms were driven by the effects of drought, which also highlighted the potential impacts of plantation forestry on water availability over its benefits for water quality and managing dryland salinity. Competing policy agendas emerged at national and jurisdictional scales, leading to a misalignment between the objectives of initiatives and management responses in some South Australian regions. Different assessment methods were used to support different policy agendas, raising concerns among planners and eventually prompting amendments to regional planning frameworks to provide greater consistency in decision-making. The relationships finally adopted to underpin state-wide forest water regulation differ according to regional hydrology while their integrity reflects the effectiveness of management processes surrounding their development. Additional investigations will be required in some regions to augment the limited knowledge base and ensure assessment frameworks are able to properly account for the hydrological impacts of plantation forestry and manage risk. Continual critical review is required to ensure policy is based on reliable and robust evidence. A national research agenda in forest water management is required to underpin the implementation of national water reforms at regional scales.

© 2013 Elsevier Ltd. All rights reserved.

1. Introduction

The scientific basis of forest water regulation may be traced to a series of investigations conducted in the early 20th century: in Switzerland (Engler 1919, in McCulloch and Robinson, 1993), the United States (Bates and Henry, 1928) and South Africa. The South African work commenced in 1935 and unlike the earlier studies focused on the impacts of afforestation rather than clear-felling, motivated by the need to manage limited water resources (Wicht, 1967). Commercial plantation development had commenced in 1876 and by 1915 farmers were...
complaining that water availability downstream of afforested areas had declined (Van der Zel, 1995). Following decades of research which featured a high level of integration with water resources management (see Bosch, 1982; Bosch and Von Gadow, 1990; Dye and Versfeld, 2007; Dye and Bosch, 2000; Wicht, 1967 among others), commercial forestry was regulated under a permit system in the early 1970s (Van der Zel, 1995) and subject to water licensing in the late 1990s (Dye and Versfeld, 2007).

In 2011 South Australia introduced state-wide legislation to regulate the water use of commercial forestry, making Australia only the second country behind South Africa to do so. A number of features of the Australian experience provide an interesting contrast against that of South Africa: regulation has only been enacted within a single jurisdiction, a consequence of Australia’s federal system of government and the legal instruments used to implement national reform; legislation was introduced within a decade of issues first being formally considered by the jurisdictional government; accounting models were developed independently of national research programmes to address regional issues; the knowledge base is notably limited and to date, no comprehensive description of the experience has been published in international research literature.

This paper brings the first Australian experience of forest water regulation to the attention of the international community. It serves as a convenient point of reflection before reforms progress in other Australian jurisdictions, identifying issues associated with competing government agendas, the impacts of different scales of regulatory administration and misalignments between research activities and management needs. An overview of national reforms is presented in Section 2, which shows how drought suddenly changed the agenda from water quality to quantity, eventually leading to forestry water regulation. Section 3 describes the role of jurisdictional and regional processes in implementing the inter-governmental approach to national reform. As the only jurisdiction to implement national reforms to the stage of forest regulation, the South Australian experience is discussed in some detail, illustrating the dynamics between national-, jurisdictional- and regional-scale initiatives. Section 4 presents the scientific basis of the South Australian forest water accounting tools which is followed by a discussion of a number of consequences of the regulation process in Section 5, before final conclusions are drawn in Section 6.

### 2. Australian water reform

The federated nature of the Australian constitutional system represents a complex institutional environment in which to implement national water reform. Inter-governmental agreements (IGAs) struck between federal and jurisdictional administrations have been the most common instrument used in national water management initiatives (Kildea and Williams, 2010). Two particular IGAs, one in 1994 and the other in 2004, were to lead directly to forest water regulation.

In early 1994, the Council of Australian Governments (COAG), the nation’s peak inter-governmental body, adopted a wide reaching series of reforms under the Water Reform Framework Agreement to address a range of natural resource management issues, including the degradation of the quality and/or quantity of the nation’s water resources (COAG, 1994). The reforms resulted in an ambitious programme of investigations and policy initiatives, including a cap on water diversions in the Murray–Darling Basin, the nation’s largest shared water resource (IAG, 1996), and a suite of national projects under the Natural Heritage Trust of Australia Act 1997, such as the National Land and Water Resources Audit (NLWRA, 2001). Land and water degradation, specifically salinity and deteriorating water quality, were regarded as a national crisis (Natural Heritage Trust of Australia Act 1997) which saw the initiation of the National Action Plan for Salinity and Water Quality (COAG, 2000, also implemented by IGAs). Salinity targets were set across the Murray–Darling Basin and investment in plantation and farm forestry was actively supported as a mitigation strategy (MDBMC, 2001), providing impetus to the expansion of plantations already stimulated by the National Forest Policy Statement (CA, 1995), the 2020 Vision (Roberts, 2005) and managed investment schemes supported by Australian taxation rulings (Brown and Beadle, 2008). Salinity was the problem and trees were the answer (Passioura, 2005).

However, in the late 1990s water availability in south east Australia began to decline under the so-called Millennium Drought of 1997–2009 (Kendall, 2010). In 2001 the Intergovernmental Panel on Climate Change published its third report warning of changing water availability (Arnell et al., 2001), adding a global context to Australia’s big dry (Gergis et al., 2012). Water availability declined even further in 2002, which became one of the most severe drought years in Australian recorded history and culminated in widespread bushfires (Ellis et al., 2004).

The impacts of 2002 prompted a number of key responses. In 2003, significant risks were identified to water availability in the Murray–Darling Basin including climate change; re-growth of native vegetation following the bushfires and plantation expansion (Dyson, 2005; Earth Tech Engineering, 2003). COAG acknowledged a pressing need to refresh its 1994 agenda (COAG, 2003) and committed to develop an inter-governmental agreement on a series of sweeping water reforms under the National Water Initiative (NWI, see NWC, 2004) which reflected priorities and risks identified within the Murray–Darling Basin, including plantation forestry.

Meanwhile, concerns regarding the risks of dryland salinity were proving unduly pessimistic (Passioura, 2005) and were overwhelmed by those associated with drought, declining water availability and climate change (Arnell et al., 2001). The balance between the potential benefits of plantation forestry in the management of dryland salinity and its potential impacts in reducing water security shifted in the national policy agenda. In 2004 large scale plantation forestry became an interception activity subject to controls under the NWI (s55–57 NWC, 2004), paving the way for jurisdictional water regulation and creating challenges for competing national forestry agendas. In 2009 South Australia released a state-wide forest water policy framework (GSA, 2009) followed by state-wide legislation in 2011. Victoria released a state-wide policy in 2011 (DSE, 2011), while progress in other jurisdictions is mixed (NWC, 2011).
3. South Australia: a Landmark in Australian Forest Water Regulation

The implementation of IGAs such as COAG (1994) or NWC (2004) require the alignment of national policy with jurisdictional legislation and regional-scale planning systems (Fig. 1). Prior to its landmark state-wide legislation, South Australian forest water management was being implemented at a regional scale, in response to a series of plantation forestry expansion initiatives in the lower South-East, the Mount Lofty Ranges and Kangaroo Island (Fig. 2). The character of different regional water management responses have proved fundamental to the integrity of policies at a jurisdictional level, with the potential to colour the national response as South Australia’s accomplishments are lauded in national and international forums (CCWPL, 2012; WIA, 2012).

3.1. Legislative context

The South Australian Natural Resources Management Act 2004 (the Act) provides for the regulation and management of activities which impact upon water resources. Certain activities are scheduled as requiring a permit or licence if they are conducted in areas that are formally declared or prescribed under the Act (s127). Permits are used to control the extent and nature of an activity, while licences enable a specified volume of water to be taken from a prescribed water resource (GSA, 2009).

Prescription of a water resource under the Act is generally associated with restricting or prohibiting further water use at a catchment or regional scale due to concerns about current levels of use and the risks they represent to the integrity of water resources (s132). Regional natural resource management (NRM) boards are then required to prepare a water allocation plan for the prescribed area (see outline of planning frameworks in Fig. 1).

Outside prescribed areas, activities with the potential to impact water resources are only subject to indirect control through policies within regional NRM plans and development plans prepared by local authorities under the relevant legislation (district councils under the Development Act 1993 in South Australia, see the local government planning framework in Fig. 1). In these areas, planning authorities may send development applications to the responsible agency for advice or direction subject to regional planning requirements. When an area is prescribed or development is subject to prohibition under the Act, councils are obliged to seek advice from the responsible agency in regard to certain scheduled development activities including commercial forestry, and can not give approval without regard for the advice (Development Regulations 1993, Schedule 8), a mechanism which was to have significant consequences in the Mount Lofty Ranges and Kangaroo Island discussed in below.

In November 2011 the Natural Resources Management (Commercial Forests) Amendment Act 2011 was given assent. The amendments provide the legislative tools necessary to implement the South Australian government’s state-wide policy framework (GSA, 2009) and enable the responsible agency to declare forestry areas in accordance with water allocation plans and grant forest water licences to commercial forest operations.

3.2. The South East

Water resources of the South East region are relatively abundant, but the region’s low topography makes surface water difficult to manage and its most productive resource is groundwater (GSA, 2000). Plantation forestry is a significant, mature regional industry. In the year 2000, around 100,000 ha of established plantations (mostly pine), were contained in an area of the South East known as the Green Triangle. The period 1999–2000 saw a sudden increase in blue gum forestry with development applications totaling 35,000 ha, over 70% of which were planted by 2002 (Harvey, 2009).

This dramatic expansion was marked by two sequential parliamentary enquiries which considered its impacts on water resources management. The first was conducted before the expansion had been fully realised, as part of a wider ranging enquiry into regional water management and allocation practices (PSA-SCWA, 1999). Water licence holders were concerned that the anticipated expansion of groundwater use by blue gums would introduce an unaccounted demand on regional water resources and threaten the security of their entitlements. Recommendations of the 1999 enquiry included accounting for the impacts of plantations within regional water planning processes.
The second enquiry considered specific concerns regarding the blue gum expansion (PSA-SCGR, 2001). Stakeholder discussions during 2001 were productive and consensus was found in a government proposal that included legislation and, contrary to trends in the Murray-Darling Basin and other drought-striken parts of south-eastern Australia, identifying additional water resources to support over 50,000 ha of plantation expansion (PSA-SCGR, 2001).

In 2004 commercial forestry became subject to regional regulation as a water affecting activity (CSA, 2004b), some 22 days before the NWI was signed on the 25th of June (NWC, 2004), which required planning authorities to refer forestry development applications to the water management agency for direction (Development Regulations, Schedule 8).

3.3 Mount Lofty Ranges

Competing state government agendas, risks to significant ecosystems and limited water availability made the situation in the Mount Lofty Ranges and Fleurieu Peninsula more turbulent. The regional water resources of the Mount Lofty Ranges are particularly significant to South Australia, providing 60% of greater Adelaide’s water supply in an average year, with the balance being sourced from the River Murray (CSA, 2000). Their isolation within an otherwise arid region makes the Mount Lofty Ranges a biodiversity hotspot (DEH, 2009), with a range of nationally recognised ecological assets, including the Swamps of the Fleurieu Peninsula (CA, 2003).

Plantation forestry represents less than 5% of the region’s land-use. The largest developments reside in the government’s mostly pine estate totaling 10,000 ha, (AMLRNRM, 2008). Expansion of the regional estate was regarded as a state government strategic priority (PIRSA, 2003) and after a number of years in negotiation and planning a private international consortium signed a joint venture to develop 10,000 ha of grassland as eucalypt plantations within 10 years (NYK, 2002). The project commenced in 2003 and was facilitated by state government service providers (PIRSA, 2003).

Before the project had progressed, regional water resources came under tighter state regulation due to concerns regarding overuse and its impacts (CSA, 2003, 2004a). In accordance with tighter state regulations, the water resources management agency became responsible for providing local planning authorities with advice on the risks and likely impacts of commercial forestry development applications (see Section 3.1), marking the beginning of formal forest water management processes in the region, some months ahead of the NWI. In 2005, regulations were varied to ensure water could not be taken directly from Fleurieu swamps (CSA, 2005) and the risks of their degradation due to the hydrological impacts of plantation forestry became a contentious issue (DWLBC-PIRSA, 2007).

In mid-2005 a local council approved a forestry development surrounding a Fleurieu wetland, conditional on the proponent increasing their proposed buffer widths as a precautionary measure to protect the wetland ecosystem (Yankalilla DC, 2005). The proponent successfully challenged the decision in the South Australian Environment, Resources and Development Court (ERDCSA, 2006), who ordered a compromise (Yankalilla DC, 2006). In 2006 the responsible Ministers for water and forestry met to resolve issues associated with plantation forest development applications including the assessment of impacts on water resources and Fleurieu Peninsula swamps (DWLBC-PIRSA, 2007), however tensions persisted at operational levels within the respective departments. Water managers maintained that water should be passed to meet all wetland requirements in accordance with state-wide guidelines, while forest proponents held that...
ecosystems could be maintained at acceptable but unspecified levels of risk of degradation through the provision of buffers, estimated using a concept of effective runoff areas based on soil properties (Greenwood et al., 2007a). By 2007 an independent review affirmed the assessment methods employed by water managers (Casanova and Zhang, 2007), who began documenting their approach in a series of government and internationally peer-reviewed publications (Greenwood, 2007; Greenwood and Cresswell, 2007; Greenwood et al., 2007a,b, 2008), while forest managers began developing their own water management tools (DWLBC-PIRSA, 2007), discussed further in Section 4.2.

The situation created sufficient uncertainty among planning authorities to prompt amendments to regional planning frameworks to address inconsistencies in the processing and assessment of commercial forestry development proposals (GSA, 2008).

3.4 Kangaroo Island

Similar tensions emerged on Kangaroo Island, a temperate region off the coast of the Fleurieu Peninsula covering an area of approximately 4400 km² (Fig. 2). Island water demand is met by a small reservoir and a desalination plant (SA Water, 2009). In the year 2000 plantation forestry occupied 3200 ha. By 2008 it had expanded to 20,300 ha, almost exclusively blue gums, while another 60,000 ha is considered suitable by industry (KINRMB, 2009).

In 2007, concerns regarding water security had prompted the Kangaroo Island district council planning authorities to refer commercial forestry development applications within the Middle River water supply catchment (Fig. 2) to water management agencies for advice (Greenwood, 2007). The severity of the 2006–2007 drought was such that the South Australian public water utility purchased water from a neighbouring private farm dam, ironically owned by the commercial forest proponent, to secure the island’s domestic supply (PSA-NRC, 2008b). The vulnerability of Middle River water resources prompted the water utility to commission hydrological modelling to evaluate the risks presented by forestry to its security of supply under prevailing drought conditions (Greenwood, 2007; SA Water, 2009).

Advice from the South Australian water resource management agency was provided according to state government precautionary guidelines and included recommendations that the development be limited to less than the proposed areal extent (see Greenwood, 2007 Appendix B). Despite this advice, initial proposals were approved by council planning authorities in their original form (KIC, 2007a), supported by an independent assessment (Bren 2004 in Greenwood, 2007, see Section 4.2). However regional planning authorities sought further advice (KIC, 2007a), and a subsequent application within the water supply catchment was refused (KIC, 2007b).

Uncertainty persisted until regional water planning processes recognised commercial forestry as a water affecting activity requiring a permit (KINRMB, 2009). More recently changes have been proposed to the Kangaroo Island council development planning framework to address issues associated with assessing commercial forestry development applications and legal challenges relating to the impacts of forestry on the sustainability of water resources (KIC, 2012).

4 Water accounting models

While the field of modern forest hydrology encompasses a global range of enquiry too vast to cite here (McCulloch and Robinson, 1993), investigations directly relating to accounting for the hydrological impacts of land-use conversion to plantation forestry for the purposes of regulation are limited to a range of South African studies (Gush et al., 2002; Nänni, 1970; Scott and Smith, 1997; Van der Zel, 1995 among others) and a more limited number of South Australian assessments (described below). Despite being at the forefront of national water regulation, South Australian water accounting models have a regional character, reflecting differences in regional hydrology and administration under regional planning frameworks. All models represent simplifications of natural processes and while scientific, industry and government reviews have noted their limitations (Harrington et al., 2011; Prosser and Walker, 2009), they have been found sufficiently robust to underpin regulation given the available technical information and administrative constraints (Harvey, 2009; Prosser and Walker, 2009).

4.1 South East – annualised recharge interception and direct groundwater use models

The distinctive unconfined groundwater systems of the South East rely heavily on recharge from seasonal rainfall to maintain the viability of regional water resources. Initial groundwater impact models were designed to assess the effects of forestry on annual vertical recharge, based on the knowledge that little or no groundwater recharge occurred under pines with closed canopies (Allison and Hughes, 1972; Colville and Holmes, 1972; Holmes and Colville, 1970a,b cited in both preceeding papers). As knowledge grew, groundwater extraction models were developed for areas with shallower water tables and both sorts of models were refined in consultation with stakeholders (Harvey, 2009).

Annual vertical recharge may be defined as the water remaining from annual rainfall entering the water table after accounting for losses to evapotranspiration due to the pre-plantation land-cover and run-off (PSA-SCWA, 1999). For resource management purposes, it is regarded as a fixed annual baseline for the pre-forest, dryland agricultural landscape in any given management area (Brown et al., 2006). Interception impact models were first developed in 2001 (Fig. 3a) and were directly linked to forest management practices. At the beginning of the plantation cycle, recharge is deemed to proceed at baseline levels until such a time as the canopy is assumed to be closed, whereupon 100% of recharge is intercepted until harvest. Changes between recharge and interception states are assumed to be linear and impacts are averaged on an annual time-step or annualised over the forest management cycle (Harvey, 2009) (Fig. 3a). Recharge models were revised as part of a formal consultation process in 2006, at the same time direct extraction models were developed (discussed below). Most changes related to the forestry
management cycle and had little effect on the final impacts (Harvey, 2009) (Fig. 3b). Annualised interception rates for blue gums under the 2006 revisions are deemed to be 78% per year (Fig. 3b) and 83% for pines (not shown), as determined by forest management practices including rotation length and thinning schedules.

Investigations into direct groundwater extraction commenced soon after the 2001 parliamentary enquiry in a joint state-government funded study by Benyon and Doody (2004). The study reported that plantations reached an average maximum extraction rate of 4.35 mega-litres (ML) per hectare per year in areas where the average depth to the water table was 6 m or less, while the maximum extraction rate appeared to be more a function of site influences than tree species.

This information was used as a basis for developing direct water use accounting models in 2006 (see blue gum example in Fig. 4): maximum direct groundwater use rates by pines and blue gums were considered to be the same; plantations could only access groundwater directly in areas with a median water table of less than 6 m; direct groundwater use could only commence following the attainment of a notional canopy closure after three years, after which it increased in a linear fashion towards the maximum extraction rate until harvesting (Fig. 4). The maximum direct groundwater use rate used in the 2006 model was determined using a subset of the sites in Benyon and Doody (2004) which were considered to represent commercially managed plantations, which averaged a water use of 3.64 rather than 4.35 ML/ha/year (Harvey, 2009; Latcham et al., 2007). Like the recharge interception models, impacts are annualised over the length of the management cycle. For the blue gum example in Fig. 4, direct groundwater use begins in year four, the maximum annual extraction rate is achieved in year seven where it remains until harvest at the end of year 10. Clean-up occurs in year 11 without any direct groundwater use. The annualised impacts over the entire 11 year management cycle amount to 1.82 ML/ha/year (Fig. 4).

In the case of pines, canopy closure is assumed to occur after six years and annualised impacts are averaged over a 36-year management cycle which after accounting for regular periods of thinning, results in an annualised extraction rate of 1.66 ML/ha/year (see detailed descriptions in Brown et al., 2006; Harvey, 2009; Latcham et al., 2007). The annualised approach to accounting for water use in both the recharge interception and direct extraction models underestimates actual annual impacts for significant portions of a rotation. However, it is considered appropriate to the regional conditions of the South East, with its distinctive groundwater systems and a mature timber industry with mixed plantation age profiles arising from continuous harvesting schedules (Harvey, 2009).

4.2. Mount Lofty Ranges

The first model to be formally adopted for use in the Mount Lofty Ranges was developed in mid-2007, within the agency responsible for primary industries (including forestry) to facilitate blue gum expansion around Fleurieu Swamps (DWLSC-PIRSA, 2007). Runoff reductions due to plantation development were assessed using a method based on Zhang et al. (1999, 2001) curves and sustainable levels of forest expansion were estimated relative to a conceptual native
woodland runoff, intermediate between forest and pasture (pre-forest) curves. Limitations of the approach meant that it was never used by water managers, and included: a lack of consideration of the water requirements of established or downstream users (Scholz and Greenwood, 2007); the tendency of Zhang curves to overestimate runoff from both grassland and forested land-cover in South Australian conditions and underestimate runoff reductions due to afforestation (Greenwood, 2007; Greenwood and Cresswell, 2007); while the use of a conceptual native woodland reference runoff was inconsistent with water allocation planning conducted throughout the region (see further discussion in Section 5.1.3) (GSA, 2012).

The accounting model ultimately adopted in South Australian state policy was developed within the agency responsible for water resources management, using local data from Burnt Out Creek, a small gauged pine plantation on the slopes of Mt Bold, a Mount Lofty Ranges water supply reservoir (Greenwood and Cresswell, 2007) (Fig. 2). Separate groundwater impacts were not explicitly considered, but are grouped with surface runoff as an integrated total reduction in water yield. Rainfall–runoff curves based on hyperbolic tangent (tanh) functions (Grayson et al., 1996) were fitted to observed pre- and post-afforestation data (Fig. 5a). The curves were then used to construct proportional runoff-reduction curves following the approach of Scott and Smith (1997) in South African forest water regulation (Fig. 5b). Observed maximum annual reductions relative to the tanh pre-forest modelled runoff varied between 73 and 93%, averaging 85% per annum (Greenwood and Cresswell, 2007), the figure eventually adopted for South Australian regulation in the Mount Lofty Ranges and Kangaroo Island for both blue gums and pines (GSA, 2009).

Unlike the accounting tools developed in the South East, forest management practices were not used to derive an annualised impact and aim to provide a higher level of protection to downstream entitlements over a longer portion of the rotation (Greenwood and Cresswell, 2007; GSA, 2009). Consequently the Mount Lofty Ranges model overestimates impacts in the early years of a rotation, but provides a closer account of maximum impacts than the South East models (Fig. 5b). The superficial resemblance between integrated yield reductions in the Mount Lofty Ranges (85% per annum) and recharge interception rates under pine plantations in the South East (83% per annum see discussion in Section 4.1) is due entirely to forest management practices and not to any similarity in regional hydrological characteristics.

4.3 Kangaroo Island

Zhang curves were also the first tools to be used on Kangaroo Island (see Bren 2004 in Greenwood, 2007). Ultimately the 85% runoff reduction developed in the Mount Lofty Ranges (see Section 4.2) was incorporated into the interim Kangaroo Island water management policy (KINRMB, 2009). However, like the initial Mount Lofty Ranges forest model, the policy sets impacts against a native vegetation reference runoff, giving rise to a number of consequences discussed further in Section 5.1.3.

5 Discussion

The implementation of national water reforms in Australia is determined by its federal constitution. The success of reforms is reflected in the success of their implementation across its independent jurisdictions, in-turn determined by the effectiveness of regional water management and planning frameworks. The experience of South Australian forest water regulation demonstrates how the progress of a significant national reform has been shaped by alignments and misalignments with underlying regional processes and systems. Plantation forestry expansion in regional South Australia and other parts of the country prior to the NWI was driven by a national policy initiative (CA, 1995) and facilitated by managed investment schemes. Competing forest-water government agendas emerged at a jurisdictional scale with regional water management initiatives, and at a national scale with the NWI. However, the NWI played an important role in resolving South Australian jurisdictional and regional issues. The development of a state-wide policy under the NWI (GSA, 2009) brought greater coherence to regional planning frameworks and enabled administrative dynamics to be more effectively managed.

Jurisdictional parliamentary enquiries facilitated state-wide regulation by articulating and analyzing public concerns, but their fundamentally regional perspective resulted in mixed outcomes. Achievements in the South East included the effective bridging of silos (PSA-SCGR, 2001; PSA-SCWA, 1999) while the Kangaroo Island enquiry foresaw the need to list commercial forestry as a water affecting activity ahead of

Fig. 5 – (a) Pre- and post afforestation annual runoff. (b) Proportional runoff reduction models fitted to Burnt Out Creek data after South African work by Scott and Smith (1997) (both figures after Greenwood and Cresswell, 2007). Note the term runoff is used here to describe areally normalised water yield, incorporating all elements of catchment yield including groundwater derived baseflow, rather than surface runoff alone.
state-wide regulation (PSA-NRC, 2008b). In contrast, the sub-regional focus adopted by two enquiries in the Mount Lofty Ranges saw energetic consideration of the impacts of established pine plantations within a single catchment, while significant regional issues associated with blue gum expansion were largely ignored (PSA-NRC, 2007, 2008a).

The emergence of a national scientific research agenda in forest water management in the early 1990s presented an opportunity to expedite the implementation of the national policy agenda and contribute to the development of regional assessment tools. Prior to the 1990s, the science of forest hydrology lacked the consistency of purpose required to support a national scientific agenda (Costin and Slater, 1967) due to a lack of consensus on its objectives (O’Loughlin, 1988) and the volatility of government investment (Bren and McGuire, 2007). In 1992 the Cooperative Research Centre for Catchment Hydrology (CRCCCH) was formed with a mission to provide resource managers with the capability to assess the impacts of land-use and water management decision making (see eWater, 2011 for a description of the history of the current CRC). Its inception immediately before the period of COAG reforms saw it ideally positioned to support water management outcomes. However its research did not directly contribute to development of forest water accounting and assessment tools.

CRCCCH hydrological plant process models used in the mid to late 1990s (Vertessy et al., 1996; Watson et al., 1998), appeared to be heading towards the type of platform currently used in South African forest water regulation (Gush et al., 2002), but empirical models (Vertessy et al., 2000; Zhang et al., 1999, 2001), sustained greater interest (Vertessy, 2001), particularly the Zhang et al. (1999, 2001) curves which grew in stature through support in a number of influential publications (Vertessy, 2001; Vertessy et al., 2003) and were eventually endorsed by leading scientific agencies as established relationships which could be used to resolve conflicting conclusions arising from different methodologies (BRS, 2003).

Zhang curves were developed using global-scale data and show distinct limitations when used in drier areas, such as Australia (Greenwood et al., 2011), particularly South Australia (Greenwood, 2007). Nonetheless support for them continued with the successor CRC, eWater established in 2005 (see Zhang et al., 2007). In hindsight, the participatory support of the South Australian water management agency represented an opportunity for eWater to support the development of tools required by national forest water regulation in an arena with an emerging need for them. Unfortunately the opportunity was missed and water managers in the Mount Lofty Ranges and Kangaroo Island were required to defend regional methods against those endorsed by their own CRC (see Greenwood et al., 2007a and related work).

5.1. Water accounting models

The assessment methods that emerged from this period were subject to independent scrutiny before their final adoption in state-wide policy (Aryal, 2010; DFW, 2010 and references therein, including: Bren, 2008, Gippel, 2008, Gippel and Watson, 2006, Smerdon, 2009; Prosser and Walker, 2009). Despite the intensity of review, a number of issues remain in some regions with implications for their integrity and application.

5.1.1. Basis

The tools that underpin South Australian forest water regulation are effectively based on three discrete investigations: a series of interception studies; a study of direct groundwater use and a description of observed water yield reductions. Additional information is available, particularly regarding recharge and groundwater extraction (for example Benyon et al., 2008 and a range of government reports), but little is directly relevant to the data that support the accounting tools and even less has appeared in international peer-reviewed scientific literature. Consequently the knowledge base is limited. By comparison, the number of South African scientific reviews written in the context of water management outcomes (Bosch, 1982; Bosch and Von Gadow, 1990; Dye and Versfeld, 2007; Dye and Bosch, 2000; Van der Zel, 1995; Wicht, 1967), is comparable to the total number of South Australian peer-reviewed research articles (see below).

Most of the articles relate to interception studies, arguably the least complex of the hydrological processes subject to forest water regulation (Allison and Hughes, 1972; Holmes and Colville 1970a,b cited in both preceding papers; Colville and Holmes, 1972). Direct groundwater use studies (Benyon and Doody, 2004; Benyon et al., 2006) represent a more sophisticated level of investigation, but the data used in estimating direct impacts for water accounting purposes were limited to five blue gum sites, monitored for a period of 3-years (Harvey, 2009). Water yield reductions described by Greenwood and Cresswell (2007) were determined using gauged catchment data from a site established to study water quality from a burned pine plantation, and monitored over a period of some 29 years. Only 14 complete years of data were available for analysis due to temporary site closure.

Access to a more substantial knowledge base such as the South African work, would have created greater confidence in the tools developed in the Mount Lofty Ranges and resulted in less uncertainty among developers and more efficient policy implementation.

5.1.2. Limitations

The South East accounting models are arguably the most developed and have been validated using computer modelling of regional groundwater levels (Aquaterra, 2010). However the nature of annualised impacts determined by fixed forest management practices effectively underestimates the actual impacts in any year post-canopy closure and makes for an accounting system which can not readily accommodate departures from the agreed silviculture and known spatial and temporal variations in groundwater dynamics (Harrington et al., 2011).

In contrast to the South East models, annual impacts estimated using the Mount Lofty Ranges model are independent of forest management. A single impact is accounted each year, for all sites across the Mount Lofty Ranges and Kangaroo Island. This simple rule aims to ensure water security to downstream users is maintained at times of maximum plantation water use (GSA, 2009). It also presents the possibility of making a water resource available to foresters
before maximum impacts are realised. However, the adopted 85% reduction in annual water yield (Greenwood and Cresswell, 2007; GSA, 2009) is based purely on empirical observations at a single site, while the impacts of afforestation are known to vary according to a site’s climatic and physical characteristics and the tree species (see discussion in Prosser and Walker, 2009). Assuming that all areas within the Mount Lofty Ranges and Kangaroo Island region will experience an 85% reduction in annual water yield, effectively assumes all sites across the region are hydrologically identical to the reference conditions at Burnt Out Creek (Fig. 2): a steep, partially cleared hillside with shallow soils and one third of its area vegetated with mature *Pinus radiata* (Greenwood and Cresswell, 2007). It is reasonable to expect greater impacts at sites with more modest topography and deeper soils than Burnt Out Creek, pointing to a need for additional investigations, particularly in heavily developed water management areas (Prosser and Walker, 2009).

In South Africa, similar questions around the robustness of empirical models (Van der Zel, 1995) led to the development of improved empirical models (Scott et al., 1998; Scott and Smith, 1997) until they were replaced by deterministic agro-hydrological modelling (Gush et al., 2002). The approach has distinct advantages in simulating a wider range of situations not covered by experimental catchments but concerns remain (Dye and Versfeld, 2007), including limitations associated with modelling water use through the catchment scale representation of plant processes (Gush et al., 2002) which are fundamentally influenced by site-scale dynamics (Soares and Almeida, 2001). Recent developments in plant process modelling have presented an opportunity to assess the impacts of site, site-scale management and growth on water use (Dye, 2001) but have yet to be applied in developing the tools required in forest water management.

### 5.1.3. Risks

The maximum annual impacts accounted for using the South East models include 100% of recharge interception and a transparent maximum annual rate of direct groundwater use. Impacts are then annualised to a lower figure by averaging annual impacts over the duration of a forest management cycle (see Figs. 3 and 4). In the Mount Lofty Ranges annual reductions in yield were observed to reach 93% (Greenwood and Cresswell, 2007), while impacts up to 100% are considered likely (GSA, 2009). The regional accounting rate is 85% per annum (GSA, 2009).

Maximum water use rates were adopted in the state-wide policy for new plantations in the Mount Lofty Ranges and Kangaroo Island to ensure water security at times of maximum water use (GSA, 2009, p. 16). However, maximum impacts in the region may exceed 85% in any year or at different locations (GSA, 2009) and the protection of downstream entitlements may be weaker than anticipated. Fig. 6 shows that impacts can be expected to equal or exceed the 85% threshold 45% of the time, indicating that unaccounted risks to downstream entitlements may emerge every 2–3 years, more frequently at sites where impacts are likely to be greater than those observed at Burnt Out Creek.

The systemic underestimation of impacts was also evident in the first Mount Lofty Ranges forest water accounting model (introduced at the beginning of Section 4.2), which was adopted to facilitate plantation expansion around wetlands of the Fleurieu Peninsula (DWLBC-PIRSA, 2007). However unaccounted risks to downstream water entitlements were exacerbated by the use of a conceptual native vegetation landcover as a baseline against which to assess the hydrological impacts of plantation forestry, an approach currently used in Kangaroo Island (KINRMB, 2009).

The annual water yield from a catchment with intact native vegetation can be expected to be less than that of a cleared catchment, due to broad-scale removal of vegetation with higher interception rates than grassland or pasture. On face value it may seem reasonable to allocate water to pre-European levels, effectively increasing the amount of water extraction possible while maintaining water yield at apparently natural levels. The approach adopted on Kangaroo Island stipulates that total annual water use across a cleared or partially cleared catchment must not result in water yield falling below levels equivalent to the mean annual runoff measured in the undisturbed Rocky River reference catchment (Fig. 2).

Recent work on Kangaroo Island has highlighted the significance of sub-annual flow regimes and the role of intact native vegetation in maintaining pre-European flow regimes (Banks et al., 2011). The absence of significant hydrological vegetation dynamics in areas with disturbed native vegetation means that flow regimes from an undisturbed *reference* catchment are not relevant to water management in altered, post-European landscapes. Allocating water to conceptual native reference levels runs the risk of desiccating remnant ecosystems which have adjusted to post-European hydrological regimes (Bunn and Arthington, 2002), particularly in the case of plantation developments which exert greater impacts on critical low flows (Lane et al., 2005).

The use of a conceptual native reference runoff is inconsistent with established water planning practices (see Barossa Valley, Clare Valley, East and West Mount Lofty Ranges, Marne River and Saunders Creek and other plans at GSA, 2012). In consideration of the additional unaccounted risks the approach presents to downstream users including the environment, it is unlikely to survive comprehensive jurisdictional water planning processes.
6. Conclusions

The South Australian experience of forest-water regulation demonstrates that the effectiveness of Australian national reforms depends on how competing government agendas are managed and whether regional management frameworks can be aligned with national policy objectives.

The Australian water reform agenda of 1994 initially resulted in a programme of assessments which focused on resource degradation and promoted the beneficial role of plantation forestry in addressing water quality and dryland salinity, complementing contemporary national forest policies which advocated expansion of the national forest estate. Managed investment schemes stimulated plantation expansion, which eventually resulted in jurisdictional initiatives to account for the water use of plantations in the South East of South Australia. Plantation development also expanded in the Mount Lofty Ranges and Fleurieu Peninsula followed shortly thereafter by jurisdictional water management processes arising from concerns about existing levels of (non-forest) water use. The processes resulted in water managers assessing the impacts of proposed plantation developments on water resources, particularly around significant wetlands and created a competition between forest expansion and water management policy agendas at a jurisdictional level.

An intensification of prolonged drought conditions suddenly brought the impacts of plantations on water availability to the attention of national water managers, providing the impetus for further national reforms under the NWI. Risks to water resources identified in the Murray–Darling Basin fed into the NWI and included forest water regulation, highlighting the influence of both climatic disaster and Murray–Darling issues on Australian national water resources policy development.

Plantation controls under the NWI effectively established competing priorities with the national forest policy agenda at a national level. However, it drove the development of a state-wide policy and regulation in South Australia, which resolved competing jurisdictional agendas that had resulted in uncertainty in Mount Lofty Ranges and Kangaroo Island planning systems. Nonetheless jurisdictional policies developed under national reforms display varying integrity at regional scales. Accounting systems developed for distinctive ground water systems and established mixed-aged plantations in the South East appear robust and reflect healthy stakeholder engagement and management processes. Limitations of the current procedure for assessing deemed impacts may not be significant compared to the additional data required to develop more sophisticated approaches. More fundamental shortcomings are evident in the Mount Lofty Ranges and Kangaroo Island accounting methods which were developed in a more contentious atmosphere and incorporate unacknowledged risks associated with a limited knowledge base and broader requirements for consistency in water management planning. It is likely that significant additional work will be required to develop an accounting system that satisfactorily meets future regional water management needs.

The pursuit of competing policy priorities resulted in poor coordination between South Australian government agencies, an issue recognised as a major national weakness in the context of implementing NWI reforms (Chartres, 2006). A genuinely national water management research agenda is required that is developed and implemented collaboratively by both scientists and policy makers to underpin the requirements of national water reforms at a regional scale. While socio-economic issues may challenge on-ground implementation, lessons may still be learned from the South African experience, which features decades of integrated scientific investigation and water regulation.

Acknowledgements

This work was completed with resources provided under a special studentship through the CRC for Forestry, Tasmania. The author would like to acknowledge the support and advice of Assoc Prof Patrick Lane and Dr Richard Benyon of the University of Melbourne, Victoria; Messrs Michael Good, Jason Vanlaarhoven and particularly Mr Darryl Harvey of the South Australian Department of Environment, Water and Natural Resources and the advice of two anonymous reviewers.

REFERENCES


Aquaterra, 2010. Modelling forestry effects on groundwater resources in the Southeast of SA. Report prepared for Department of Water, Land and Biodiversity Conservation, 7 June.


DWLBC-PIRSA, 2007. Agreement on a Risk Policy for Plantation Forests and Water Resources in the Mount Lofty Ranges. Department of Water, Land and Biodiversity and Department of Primary Industries and Resources South Australia, 7 June, Courtesy Conservation Council of SA.


Harvey, D., 2009. Accounting for plantation forest groundwater impacts in the lower South East of South Australia, DWLBC report no 2009/13, Department of Water, Land and Biodiversity Conservation, Adelaide.


KIC, 2007. Minutes of the Development Assessment Panel Meeting held at the Kangaroo Island Council Chambers, Dunaecy Street, Kingscote on Monday 4th June 2007 at 1.30 pm. Kangaroo Island Council, South Australia, Kingscote, p. 25.


APPENDIX D: SECTION 5

A method for assessing the hydrological impact of afforestation using regional mean annual data and empirical rainfall–runoff curves

Ashley J.B. Greenwood, Richard G. Benyon, Patrick N.J. Lane

Department of Forest and Ecosystem Science, University of Melbourne, 221 Bouverie Street, Parkville, Victoria 3010, Australia

Abstract

Using a case study from Australia, a method was developed to integrate regional-scale rainfall-runoff data with site-scale information to predict the likely magnitude of mean annual runoff reductions due to afforestation. It was hypothesised that large, long-term, mean annual rainfall–runoff datasets can be used to provide statistical limits around the variability of regional hydrology and that data from plantation conversion experiments can be used to identify quantiles within such data that correspond to pre- and post-plantation land-cover, enabling the transparent quantification of regional changes in runoff due to afforestation. Three simple empirical functions with flexible parameter structures and a documented precedence in the assessment of the hydrological impacts of land-cover change were used to perform quantile regression on the regional data. Mean annual data and empirical rainfall–runoff curves were in turn used to estimate proportional runoff reductions for regional Australia. USDA curves provided the closest agreement with results reported by a number of independent studies compared to a simple tanh function and Zhang curves. The USDA function had a more flexible structure in describing runoff from dry areas than either tanh or Zhang curves. Inflexibility in the Zhang curves was attributed to limitations in the model’s specification.

Keywords:
Forest hydrology
Water resource assessment
Afforestation
Empirical rainfall–runoff curves
Quantile regression
Runoff reductions

1. Introduction

Predicted reductions in water availability due to climate change (Bates et al., 2008) and the adaptive responses of policy makers (Kundzewicz et al., 2007), suggest that the hydrological impacts of plantation forestry are likely to become an issue of even greater interest to water managers in coming years. Afforestation, the intentional establishment of any tree-dominated vegetation on land that was formerly not tree-dominated (FAO, 2005; Van Dijk and Keenan, 2007), is recognised as having a beneficial role in mitigating the impacts of climate change by assisting in the removal of atmospheric CO₂ (UNFCCC, 2009). This role is being stimulated by a number of independent initiatives around the world (Görgens, 2003; HarvestPlus, 2008; Benyon et al., 2005), policy frameworks to account for risks to water availability presented by afforestation were first developed in South Africa in the early 1970s (Van der Zel, 1982). Similar initiatives have recently emerged in Australia (GSA, 2009; NWC, 2004). These are underpinnings of climate change, by assisting in the removal of atmospheric CO₂ (UNFCCC, 2009) to offset greenhouse gas emissions (UNFCCC, 1998). However, the impacts of afforestation have been found to be easy to use, provide transparently repeatable results and capable of accommodating new information (Görgens, 2003), although legitimate concerns have also been expressed around their predictive ability under conditions that vary from those of the original studies (Bosch and Von Gadaw, 1990; Dye and Bosch, 2000).

Empirical curves have been superceded in South African water management by deterministic computer modelling (Gush et al., 2002). In Australia, where there are fewer data available to support detailed modelling over large areas, more sophisticated assessments have integrated empirical runoff reductions into hydrological modelling with useful results (Brown et al., 2007; Chiew et al., 2008; Zhang et al., 2010), highlighting the potential to use empirical relationships to infill knowledge gaps until more information becomes available. However it is unlikely that the number of relevant hydrological afforestation datasets in Australia will increase substantially in the immediate future. If greenhouse gas abatement or carbon offset initiatives stimulate the expansion of plantation developments into areas of...
limited water availability, the required water resource assessments are likely to be supported, at least in part, by empirical methods.

The integrity of any empirically-based assessment is ultimately governed by the quality of the information on which it is based and the uncertainty inherent in extrapolating its findings. The aim of this study is to determine whether long-term regional rainfall–runoff data can be used to reduce, or at least quantify this uncertainty.

Australian regional runoff patterns have been recognised as distinctive among global data (McMahon et al., 1992), driven by its position in global climatic circulation patterns (Ummenhofer et al., 2011), modest topography (Gentilli, 1971) and exposure to mid-latitude radiation levels (Budyko, 1974). A suitably representative regional dataset should consequently provide statistical bounds on rainfall–runoff patterns that may be reasonably expected under regional hydro-climatic influences, with potential to provide a transparent context for any other data collected within the region. When long-term mean–annual rainfall–runoff data are used, changes in soil moisture maybe neglected, so that differences evident in regional mean annual runoff for a given rainfall may be attributed to differences in mean annual evapotranspiration (Budyko, 1974). Variations due to local influences should fall within the range of variability evident in the regional dataset. Sites within the lower quantiles of the regional rainfall–runoff data should be those with high mean annual evapotranspiration (for example ‘forest’), while sites in higher quantiles should be those with lower evapotranspiration (for example ‘grassland’).

It was hypothesised that data from plantation conversion experiments could be used to identify quantiles in regional mean annual rainfall–runoff datasets corresponding to pre-conversion (‘grass’) and post-conversion (‘forest’) land-cover, enabling quantification of changes in runoff due to afforestation. Quantile rainfall–runoff curves were constructed through a long-term regional Australian mean annual rainfall–runoff dataset (Peel et al., 2000) using non-linear quantile regression (Koenker and Park, 1996) and three simple empirical rainfall–runoff functions. Empirical functions were chosen which had flexible parameter structure and a documented precedence in the assessment of the hydrological impacts of land-use change: USDA curves (USDA-NSRC, 2004), Zhang curves (Zhang et al., 2001) and tanh curves (Boughton, 1966; McMahon et al., 1992). Conceptual ‘grassland’ (vegetation not dominated by trees) and ‘forest’ (intensively planted, tree-dominated vegetation) quantile curves were then selected from the quantile curve array using statistical association with Australian plantation conversion data from publicly available literature. The selected quantile curves were then used to derive runoff reduction curves after Scott et al. (2000) which were compared to results of plantation conversion assessments conducted by independent methods.

2. Theory

2.1. Regional patterns of runoff variability

Streamflow (or runoff) is known to exhibit patterns of large-scale regional variability. Budyko (1974) identified global-scale geobotanic zones according to their mean annual heat and water balances using his “equation of relation” (Eq. (1), Table 1). Regional climatic influences were found to be dominant in controlling regional hydrological variability while local discrepancies could be attributed to site-dependent influences. In a review of the Budyko (1974) model, Donohue et al. (2007) concluded that Eq. (1) reliably predicted catchment water balances when applied over large catchment areas and long periods of time. When catchments were smaller (<1000 km²) and time scales shorter (<1–5 years) the dynamics of sub-annual processes and non-climatic influences could not be ignored. The analysis of Budyko’s function and four other simple rainfall–runoff functions by Oudin et al. (2008) found that introducing (non-climatic) land-cover terms improved estimates of mean annual runoff from 1508 small catchments (<10 km²) in France, Sweden, the UK and the USA.

McMahon (1979) found continental-scale patterns in the variability of runoff from arid and humid climatic zones. Using a global dataset which included sufficient information from the southern hemisphere to enable questions of intercontinental-scale variability to be addressed, McMahon et al. (1982, 1987, 1992) concluded that global annual runoff varied at a continental scale, and that runoff from Australian and southern African streams was the most variable in the world. Differences could not be accounted for by hemispherical characteristics nor completely ascribed to the variability of total rainfall, and was likely to be a result of the broader variability of regional hydrological and climatic variability (see also McMahon et al., 2007). Probst and Tardy (1987) found anti-synchronous patterns in North American and European runoff at a continental-scale in response to large-scale climatic fluctuations. Dettlinger and Diaz (2000) identified regional patterns of variability which were not limited to hemisphere or continent. Poff et al. (2006) identified intercontinental differences in streamflow variability in a study of 463 sites around the world; Australian streams showed greater inter-annual variability than New Zealand, South Africa, Europe, and the United States. Sub-continental scale regions could be defined hydro-climatically, while smaller sub-regions could be defined by (non-climatic) geomorphic characteristics.

Peel et al. (2001, 2004) attributed differences in continental runoff patterns to continental differences in the variability of annual precipitation, the impacts of evapotranspiration from regionally distributed evergreen and deciduous vegetation and higher levels of solar radiation in the southern hemisphere. Global climate circulation patterns were identified as playing an important role in increasing the variability of rainfall patterns. In a review of studies into catchment vegetation and water yield, Peel (2009) concluded that catchment vegetation appeared to have a second-order impact on runoff at large scales, relative to aridity (the ratio of potential evapotranspiration to precipitation, E/P), while at small scales its impact on runoff can be significant. Based on a large global dataset, Peel et al. (2010) found that in large catchments (>1000 km²), global-scale climatic zonation needed to be taken into account when assessing the impacts of vegetation type on mean annual evapotranspiration. They observed that in catchments with areas <1000 km², the influence of vegetation type was significant in temperate climatic zones.

2.1.1. Regional climatic influences in Australia

Hydrological variability in south-east Australia has been linked to the El Niño–Southern Oscillation (ENSO) teleconnection across the equatorial Pacific Ocean (Chiew et al., 1998; Dettlinger and Diaz, 2000; Kuhnel et al., 1990; McBride and Nicholls, 1983; Piechota et al., 1998), seasonally low pressure over the Southern Ocean (Maheshwari et al., 1995; Whetton, 1988) and surface sea temperatures and sea level pressures in the equatorial eastern Indian Ocean (Kuhnel, 1990; Nicholls, 1989; Wright, 1988a).

Unlike continents of the northern hemisphere, Australia’s landscape is characterised by modest orographic relief. It does not exert a major influence on its weather, but is subject to regional climatic circulation patterns (Gentilli, 1971), which interact over south and south-eastern Australia (Kuhnel, 1990; Ummenhofer et al., 2011; Wright, 1988b) to produce a distinctive regional hydro-climatic environment.

2.2. Rainfall–runoff curves

Semi-empirical relationships have been used to assess hydrological systems since the beginning of the 20th Century (Schrierer, 1904 in Budyko (1974)). Sophisticated rainfall–runoff (rainfall used here synonymously with precipitation) curve systems for assessing
Table 1
Rainfall–runoff functions.

**Budyko (1974)**

1 \[ RO = P - \left( \frac{RP}{ET} \tanh \left( \frac{LP}{ET} \right) \right) \left( 1 - \cosh \left( \frac{R}{ET} \right) + \sinh \left( \frac{R}{ET} \right) \right)^{1/2} \]

RO = mean annual runoff (mm/year)

ET = mean annual net radiation (energy/area/year)

P = mean annual precipitation (mm/year)

L = mean annual latent heat of vaporization (energy/area/year)

USDA-NRCS (2010)

2 \[ RO = \frac{(P - L)S}{P + S}; \quad S = \frac{1000}{CN} - 10 \]

RO = runoff (inches/event)

\( L \) = initial abstraction (inches/event)

\( S \) = initial abstraction ratio \(( S/L )\)

P = precipitation (inches/event)

S = maximum potential infiltration (inches/event)

CN = curve number \(( CN = 1 - 100 )\)

**Zhang et al. (1999, 2001)**

4 \[ ET = \frac{1 + \omega E_o}{P} \]

\( \omega \) = plant available water coefficient

ET = evapotranspiration

\( E_o \) = potential evapotranspiration

5 \[ RO = P - P \left( \frac{1 + \omega E_o}{P + \omega E_o} \right) \]

RO = mean annual runoff (mm/year)

\( E_o \) = mean annual potential evapotranspiration (mm/year)

**Boughton (1966) and McMahon et al. (1992)**

6 \[ RO = P - F \tanh \left( \frac{P}{F} \right) \]

Boughton (1966)

7 \[ RO = P - IL - F \tanh \left( \frac{P - IL}{F} \right) \]

McMahon et al. (1992). IL and F defined after Grayson et al. (1996)

\( F \) = precipitation (mm)

IL = infiltration (mm)

\( P \) = precipitation (mm)

\( F \) = infiltration (mm)

**USDA curves**

USDA curves were initially developed in the late 1940s (Ponce and Hawkins, 1996; Rallison, 1980) and are still supported by the US government as part of the National Engineering Handbook (USDA-NRCS, 2010). USDA curves have been widely applied across the United States and other parts of the world and their development and application have been documented by a range of authors (Hawkins et al., 2009; Miller and Cronshey, 1989; Mishra and Singh, 1999; Plummer and Woodward, 1998; Rallison and Miller, 1982).

The USDA rainfall–runoff equation defines a relationship between runoff, rainfall and catchment losses (Eqs. (2) and (3), Table 1). It assumes that the ratio of actual infiltration (termed “retention”) to maximum potential retention could be equated to the ratio of actual direct runoff to maximum potential runoff; and that an empirical, linear relationship could be formed between initial abstraction (or initial loss, \( I_a \) and maximum potential retention (5) (Mishra and Singh, 1999), termed the initial abstraction ratio \(( I_a / S )\) This latter assumption was compounded by setting \( I_a \) equal to 0.2 for all catchments across the United States to simplify field calculations (Ponce, 1996).

**2.2.2. Zhang curves**

Zhang et al. (1999, 2001) reviewed the impact of vegetation changes on mean annual evapotranspiration and developed a simple water balance model that described the effect of vegetation change on mean annual evapotranspiration. A rational relationship between evapotranspiration and precipitation was proposed to satisfy boundary conditions postulated by Budyko (1948 in Budyko (1974)) for very high and low rainfall. An additional plant available water parameter \(( \omega )\) was introduced to

the hydrological impacts of land-cover change have been available since the 1950s (Ponce and Hawkins, 1996). Curves designed specifically for assessing the impacts of grassland conversion to plantations were developed in the early 1970s in South Africa (Nanni, 1970; Van der Zel, 1982), while similar systems emerged later in Australia (Holmes and Sinclair, 1986; Zhang et al., 2001). Generic rainfall–runoff curves used in early computer models and global runoff assessments (Boughton, 1966; McMahon et al., 1992) have also been applied in assessing land-cover change (see below).

The curve systems used in this study were: USDA curves or “curve numbers” (USDA-NRCS, 2010) (Eqs. (2) and (3), Table 1), Zhang curves (Zhang et al., 2001) (Eqs. (4) and (5), Table 1) and a simple single parameter hyperbolic tanh function used by Boughton (1966) (Eq. (6), Table 1).

Figs. 1a–c show the functional forms obtained when varying one-parameter according to procedures followed in many studies. USDA curves (Fig. 1b) show a greater potential to model runoff from catchments with higher levels of pre-runoff losses. Figs. 1d–f show the additional freedom gained when two parameters are varied (Fig. 1d shows the two parameter tanh function of McMahon et al. (1992); see Eq. (7), Table 1), but high levels of parameter interaction prohibited their use in deriving coherent quantile curves for this study (see Section 5).
reflect the mediative role of soil water capacity (Eq. (4), Table 1),
analogous in effect to the soil water capacity-precipitation ratio of Milly (1994). The authors postulated that \( \omega \) should lie around 0.1 for bare soil and range between 0.5 and 2.0 for short grass and forest respectively. Estimates of mean annual potential evapotranspiration values for grass and forest were determined by fixing \( \omega \) to 0.5 and 2.0, and performing non-linear least squares regression between Eq. (4) (Table 1) and data from 256 catchments around the world. The analysis yielded 1100 mm/year for herbaceous plants and 1410 mm/year for trees which were used in Eq. 4 (Table 1) to generate curves to compare mean annual evapotranspiration from grassland and forest for a range of mean annual rainfall.

Subsequent authors have used the precipitation–evapotranspiration curves to derive rainfall–runoff curves by assuming zero change storage of a simple conceptual water balance model (Eq. (5), Table 1). In this form they have been employed in a wide range of land-use change assessments in Australia (Bray et al., 2001; Bren and Hopmans, 2007; Bren et al., 2006; Brown et al., 2007; Vertessy and Bessard, 1999; Zhang et al., 2003), the United States (Sun et al., 2006) and China (Wang et al., 2008). Their most far reaching impact is probably in adjusting flow duration curves to assess impacts of plantation conversion (Brown et al., 2006), which have been used in a number of significant Australian water resource assessments (Chiew et al., 2008; Zhang et al., 2010).

### 2.2.3. Tanh curves

Budyko (1974) used hyperbolic functions to model mean annual runoff as a function of climatic parameters (Eq. (1), Table 1). The equation was based on precipitation–evapotranspiration relationships derived by Schreiber (1904) and Ol’dekop (1911) (in Budyko (1974)). Boughton (1966) used a tanh function to estimate daily runoff in an early computer model (Eq. (6), Table 1). It was chosen ahead of the widely accepted USDA curve number equation, as it better replicated the rate at which runoff approached its maximum limit during large rainfall events (Boughton, 1966). McMahon et al. (1992) used a modified version of the Boughton equation to model annual runoff in a study of global runoff variability. A term was added to account for initial loss (Eq. (7), Table 1), which exerts a similar influence to initial abstraction in the USDA system, creating a more generic function with intuitive parameters and greater flexibility. In a comparison with complex conceptual rainfall–runoff models, Chiew et al. (1993) found that Eq. (7) (Table 1) gave satisfactory results for monthly and annual data in wetter catchments. Their simplicity and flexibility has seen tanh functions used for a range of hydrological applications: infilling data gaps (Grayson et al., 1996); characterising rainfall–runoff and land-cover relationships in global data (McMahon et al., 1992; Peel et al., 2010), estimating runoff in un-gauged catchments (Boughton, 2004; Boughton and Chiew, 2003), water resources risk assessment (McMurray, 2007) and assessing the impacts of land-use change (Greenwood and Cresswell, 2007; Siriwardena et al., 2006) and climate change (Gardner, 2009; Guo et al., 2002).

### 2.3. Proportional rainfall–runoff reductions

Hydrological impacts of afforestation have been quantified as a proportion of pre-treatment runoff for assessment and management purposes (Brown et al., 2005; Greenwood and Cresswell, 2007; GSA, 2009; Scott and Smith, 1997). Proportional reductions have been reported to produce better results than absolute values in assessing the impacts of afforestation (Brown et al., 2006; Podger, 2005) and facilitate inter-regional comparisons (Scott et al., 2008).

![Fig. 1. Empirical curves systems used in this study. Note multiple variable parameter curves were not used in this study due to difficulties in generating coherent quantile arrays and are shown here for discussion. Functions used in this study are shown in panels 1a–1c and 1f.](image-url)
Proportional runoff reduction curves may be constructed using Eq. (8) where ‘treatment runoff’ is runoff from any plantation conversion and ‘reference runoff’ that from any pre-treatment land-cover, (usually ‘grassland’), to which the change in runoff is being compared. The calculation provides an estimate of maximum or long term impacts under hydrological steady-state conditions.

2.4. Quantile regression

Quantiles represent the proportions of a dataset below a given value. The more familiar ‘percentiles’ are quantiles expressed as one hundredth proportion of a dataset, for example the 80th percentile or 0.8 quantile (denoted here by the symbol τ) is the value at which 80% of the data are equal to or smaller than that value and 20% are equal to or greater than it (NIST/SEMATECH, 2003).

Conventional regression provides an indication of the average relationship between explanatory and response variables by fitting a function to a covariate dataset. Quantile regression provides a means by which a function may be fitted to different proportions within a covariate dataset simultaneously, yielding a more complete picture of the data, with the potential to yield greater insight into underling processes modelled by the function.

The principles underlying quantile regression may be described analogously to least squares linear regression. Linear regression involves fitting a straight line function to some data and assumes that the mean function responses lie along a straight line for all values of the explanatory variable. If the data are independent and the function is unbiased, that is, capable of generating normally distributed errors from the data, the parameters which provide the best estimate of the line are those that minimise the variance between the data and modelled estimates (Press et al., 2007), also known as the mean parameter values.

In the same way that the mean represents the solution to the problem of minimising the residuals generated by the given function, the median (or 0.5 quantile see below) can be defined as the solution to the problem of minimising the sum of absolute residuals (Press et al., 2007). The symmetry of the absolute value function obtained when calculating absolute residuals, implies that the minimisation of their sum must require an equal number of positive and negative residuals, requiring the same (symmetrical) number of data above and below the median estimate (Koenker and Hallock, 2001). As the symmetry of the absolute residuals leads to an estimate of the median, it can also be shown that an optimised sum of absolute residuals that are asymmetrically weighted to yield different numbers of positive and negative values, will yield other quantiles (Koenker and Hallock, 2001).

A method of implementing quantile regression for functions that are linear in their parameters, was developed by Koenker and Bassett (1978). For a set of n random variables \( r_1, r_2, r_3, \ldots, r_n \), the objective function of absolute residuals to be minimised by symmetrical regression around the median may be written:

\[
\min \sum_{i=1}^{n} |(r_i - \hat{r}_m)|
\]  

where \( \hat{r}_m \) is the estimate of the median. For asymmetrical regression around other quantiles, the absolute residuals must be weighted, the median replaced by the parametric model function used to model the data, \( R(x, \hat{\lambda}) \):

\[
\min \sum_{i=1}^{n} \rho_r (r_i - R(x_i, \hat{\lambda}))
\]  

where \( \hat{\lambda} \) is the set of model parameters and \( \rho_r \) is an asymmetric or tilted absolute value function which provides the required weights to the residuals for the quantile of interest (Koenker and Hallock, 2001). A method for non-linear quantile regression was developed for functions that were non-linear in their parameters by Koenker and Park (1996), which uses a variant of iteratively re-weighted least squares based on interior point methods for linear programming (Nocedal and Wright, 2006). Further information may be found in Koenker (2005).

3. Material and methods

3.1. Data

3.1.1. Regional data

Peel et al. (2000) published a large, long-term rainfall–runoff dataset that can be considered representative of regional Australia, particularly across temperate areas where plantation conversion is likely (Keenan et al., 2004) (Fig. 2). The data were compiled to provide an extended time series of unimpaired streamflow data for use in both research and management of Australia’s hydrological and ecological systems (Peel et al., 2000). Appendix 1 (Peel et al., 2000) comprises mean annual rainfall–runoff data pairs and runoff coefficients from 331 catchments across Australia. Each data pair is recorded with its location (including administrative jurisdiction), catchment area, gauge number and station name. Land-cover information was not recorded in the dataset.

Most of the sites in Peel et al. (2000) are located in eastern and south-east Australia (Fig. 2 and Table 2). They were selected to be unimpaired by regulation or diversions; have at least 10 years of historic monthly streamflow data and have catchment areas between 50 km² and 2000 km², so that the lumped daily rainfall used for the modelling had similar meaning and optimised model parameter values could be compared across catchments (Peel et al., 2000).

Mean annual rainfall–runoff data are based on long-term modelled data (1901–1998), generated using the SIMHYD rainfall–runoff model with an optimised parameter set derived from calibration against the entire observed monthly streamflow record for each site (Peel et al., 2000). Rainfall was based on the gridded 5 km × 5 km daily data described by Jeffrey et al. (2001). Monthly evapotranspiration were from Wang et al. (2001). More information on the modelling procedure and data used are available in Peel et al. (2000).

A subset of 313 of the sites were used in this assessment, to provide a dataset that was more representative of the regions identified by Keenan et al. (2004) as having potential for development of new plantations (Fig. 2). Most sites may be expected to be dominated by winter precipitation in accordance with their location in southern Australia (Fig. 2). The number featuring significant amounts of snowfall was not recorded by Peel et al. (2000), but is expected to be small. Data removed comprised: all sites north of the Tropic of Capricorn (10 from Queensland and four from the Northern Territory); Todd River in the central Australian desert; Kanyaka Creek in South Australia’s arid Flinder’s Ranges and two sites in western Tasmania with runoff greater than 2000 mm/year (Fig. 2, Table 2).

3.1.2. Plantation conversion data

Australian hydrological plantation conversion data are limited. A set was compiled for this study from a number of afforestation studies (Bari et al., 1996; Borg et al., 1988; Bren and Hopmans, 2007; Greenwood and Cresswell, 2007; Hickel, 2001; Lane et al., 2005; Putuhena and Corderoy, 2000, see Table 3). Here ‘grassland’ runoff was taken as pre-treatment runoff, which ranged from true grassland or pasture, to cleared forests. None of these data were contained within the Peel et al. (2000) information.
3.1.3. Other data

Due to the limited conversion data, a number of data from native vegetation sites were used to provide an indication of the validity of the selection of quantile ‘forest’ curves, which proved particularly informative in higher rainfall areas (Fig. 6). Data were sourced from publically available literature, many from studies in West Australia (see Table 4 in Section 4).

Proportional reductions in mean annual runoff have been reported in a number of Australian studies and were used here to compare with the results obtained using quantile curves (Fig. 7 in Section 4). Greenwood and Cresswell (2007) used pre- and post-afforestation tanh rainfall–runoff curves at Burnt Out Creek, South Australia. Reductions at Stewart’s Creek in Victoria and Redhill in New South Wales were taken from Lane et al. (2005) who used a flow duration curve adjustment technique, while those reported for Pine Creek (Victoria) were based on data in Lane et al. (2003).

3.2. Curves

USDA, Zhang and tanh curves all have a well-documented history in Australia, but functional forms with multiple variable parameters shown in Fig. 1 generated quantile curves that were prone to crossing (not coherent), or contained parameters outside the feasible range.

Consequently simpler forms with single variable parameters were used. In conventional application USDA and Zhang curves employ single variable parameters (USDA-NRCS, 2010; Zhang et al., 2001) (see Fig. 1b for conventional parameter variation in USDA curves and Fig. 4 for conventional or default parameter Zhang curves). The tanh function used was the simple form of Boughton (1966) (Eq. (6), Table 1, see Fig. 1a), which does not have the flexibility of replicating high levels of pre-runoff catchment losses as that of McMahon et al. (1992) (Eq. (7), Table 1, see Fig. 1d).

Conventional fixed $\omega$ Zhang curves effectively represent two different functions: one each for ‘grassland’ ($\omega = 0.5$) and ‘forest’ ($\omega = 2.0$). Quantile regression was completed for both values of $\omega$ despite the conceptual incongruity of deriving forest curves in higher quantiles and vice versa. The overall fit of the Zhang model to the Peel et al. (2000) data was assessed using an $E_2$ parameter derived from a hypothetical ‘mixed vegetation’ value half way between those for ‘grassland’ and ‘forest’ ($\omega = 1.25$, see discussion of Fig. 3 in Section 4 and Fig. 5c).
Table 3
Summary of additional Australian plantation conversion data. Note: data from sites marked with an asterisk (*) were provided by CSIRO and University of Melbourne Department of Forest and Ecosystem Science.

<table>
<thead>
<tr>
<th>Site</th>
<th>Reference/control</th>
<th>Vegetation</th>
<th>Period</th>
<th>Treatment</th>
<th>Vegetation</th>
<th>Period</th>
<th>Key References</th>
<th>Comments</th>
</tr>
</thead>
</table>

Table 4
Summary of Australian native vegetation data (also see Fig. 6).

<table>
<thead>
<tr>
<th>Site</th>
<th>Period</th>
<th>Vegetation/treatment</th>
<th>Canopy closure</th>
<th>Key references</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>a</td>
<td>Thomson Brook, WA</td>
<td>1978–1986</td>
<td>Mature jarrah, 45% cleared, ceased</td>
<td>No data</td>
<td>Borg et al. (1988)</td>
</tr>
<tr>
<td>d</td>
<td>Ernies, WA</td>
<td>1974–1983</td>
<td>As above, no clearing</td>
<td>11%</td>
<td>Williamson et al. (1987)</td>
</tr>
<tr>
<td>g</td>
<td>Ella, Vic</td>
<td>1997–2005</td>
<td>Intact mixed eucalypt bush</td>
<td>No data</td>
<td>Bren and Hopmans (2007)</td>
</tr>
<tr>
<td>i</td>
<td>L6, NSW</td>
<td>1967–1977</td>
<td>Dry sclerophyll eucalypt forest, 100%</td>
<td>34% (L5)</td>
<td>Putuhena and Cordery (2000)</td>
</tr>
<tr>
<td>j</td>
<td>Crabapple, NSW</td>
<td>1976–1999</td>
<td>Wet sclerophyll eucalypt forest, 100%</td>
<td>Continuous</td>
<td>Cornish and Vertessy (2001)</td>
</tr>
<tr>
<td>k</td>
<td>Sassafras, NSW</td>
<td>1976–1999</td>
<td>As above</td>
<td>Continuous</td>
<td>Cornish and Vertessy (2001)</td>
</tr>
</tbody>
</table>

3.3. Quantile regression and R

Non-linear quantile regression was implemented in the R open source statistical analysis environment (R Development Core Team, 2003) using nlrq, a function within in the quantreg package (Koenker, 2005). Objects returned by nlrq are fitted model objects of class ‘‘nlrq’. They have methods for a number of generic R functions such as summary which can be used to report optimised parameters, standard errors, t-statistics, and p-values, but lack methods for some others, most notably for this study anova. The default optimising algorithm in nlrq is a quasi–Newton method that uses function values and gradients to define the optimisation surface (Byrd et al., 1995), and was used in all analyses. Starting parameter values for nlrq were determined using non-linear least squares regression, nls, in R (Bates and Chambers, 1992).

3.4. Other statistical tests

The Kolmogorov–Smirnov (K–S) test (ks.test in R) was used to assess similarity between data sets and goodness of fit of the rain–fall–runoff curve models to data in this analysis. It tests the null hypothesis that two samples are drawn from the same continuous distribution: the higher the p-value of the K–S statistic, the higher the probability that both samples were drawn from the same distribution. Based on the absolute difference between the cumulative probability distributions of the datasets of interest, the K–S statistic is non-parametric, so its significance is not affected by transformation of units. The accuracy of significance estimates is relatively high for small sample sizes, and may be considered “quite good” for data sets comprising as few as eight sample pairs (Press et al., 2007).

The small number of data resulted in some tied K–S statistics. In these instances quantile curves were identified according to the lowest variance as indicated by the root mean squared error (RMSE), a simple goodness of fit test that may be used to evaluate the relative merit of a number of competing models (NIST/SEMATECH, 2003).

4. Results

Regional Australian data fitted with quantile curves conditional on the three empirical rainfall–runoff curve systems are shown in
Log–log scales were used to facilitate examination of low rainfall data. The theoretical linear limit of runoff equal to precipitation is shown in each panel for comparison.

Quantile curve arrays produced were coherent. That is, the curves did not cross, maintaining the requirement that quantiles be monotone increasing (Koenker, 2005), more of which will be discussed below. The USDA curve quantile array showed much greater sensitivity to the entire dataset compared to Zhang and tanh functions, providing a much better description of its variability at low rainfall (Fig. 3b). Its median curve provided the strongest agreement with the data of all models: the results of the Kolmogorov–Smirnov test that data generated by the curve could be considered drawn from the same distribution as the observed data could not be disproved (K–S p > 0.3).

Fig. 3. Selected data from Peel et al. (2000) and quantile curves conditional on the three empirical curve systems. The same quantiles are plotted in each panel. Default Zhang curves (ω = 0.5, Ez = 1100; ω = 0.5, Ez = 1400) are shown as broken lines for comparison.

Zhang and simple tanh curves produced very similar quantile curve arrays, the latter (Fig. 3a) produced a slightly narrower range of variability at low rainfall and a slightly better fit to the trend of the data (median curve K–S p > 0.1). Median Zhang quantile curves for ω = 0.5, 1.25 (see also Fig. 5c) and 2.0 showed the weakest accord with data of all functions: data generated by the median curves were significantly different to the observed data (K–S p < 0.05).

Australian plantation conversion data collated for this study (see Table 3) are shown plotted in Fig. 4 with the regional data and default parameter Zhang curves for comparison. Both pre-treatment ‘grassland’ and plantation ‘forest’ data were located as expected within the...
upper and lower quantiles of the regional data, rather than consistently representing the extremes of regional runoff variability, providing optimism around the prospects of identifying regional quantile curves which could be used to represent them.

Fig. 5. Kolmogorov–Smirnov p-values and root-mean-square-errors (normalised by the maximum RMSE to plot on a 0–1 scale) of fits between quantile curves and (a) pre-treatment ‘grassland’ data (b) plantation data.
Fig. 4 also shows the paucity of plantation conversion data in wetter areas (>1000 mm/year). Sites 5, 7 and 8, representing a native re-growth forest in West Australia and two Victorian pine forests respectively (see Table 3), showed a wide range of runoff for a relatively narrow range of rainfall. According to the studies cited above (see Section 2) sites with a given level of evapotranspiration exhibiting noisy levels of runoff were likely to be inconsistent with regional runoff patterns and influenced by local physical factors, such as unusually steep slopes or deep soils. The presence of noise or outliers is capable of distorting an analysis, particularly where data are limited (see Press et al., 2007 and other standard references). Consequently it was decided to exclude conversion data with mean annual rainfall greater than an arbitrarily chosen 1000 mm/year.

It is germane to later discussion to note that default parameter Zhang curves showed much closer agreement with the wetter conversion sites (Fig. 4) than the drier ones. Without the additional context provided by the regional data and the quantile curves of Fig. 3, the limitations of the data points and the Zhang curves would be much more difficult to detect.

Regional ‘grassland’ and ‘forest’ quantile curves were selected as those with the highest agreement with the wetter conversion sites (Fig. 4) than the drier ones. Without the additional context provided by the regional data and the quantile curves of Fig. 3, the limitations of the data points and the Zhang curves would be much more difficult to detect.

Regional ‘grassland’ and ‘forest’ quantile curves were selected as those with the highest agreement with the conversion data (mean annual rainfall <1000 mm/year) using the K–S test and RMSE. Where K–S were tied, the quantile that yielded the lowest RMSE was considered the best fit. Results are shown in Fig. 5.

The selected regional ‘grassland’ and ‘forest’ quantile curves are shown in Fig. 6, along with median curves for comparison. The
difficulty Zhang and tanh functions encountered when attempting to simultaneously describe rainfall–runoff relationships in both wet and dry areas in Fig. 3 is again apparent. It should be noted that quantile curves were derived from the information rich regional data and regional ‘grassland’ and ‘forest’ curves were only selected using the limited conversion data from dry areas. However it was possible that this selection process biased the identification of quantile curves away from those that provided a useful description of ‘forest’ runoff in wet areas, particularly in the light of the apparent discrepancy between Zhang and tanh curves and regional data. As publically available Australian plantation data were effectively exhausted, information from native forest sites was used for validation (Borg et al., 1988; Cornish and Vertessy, 2001; Ruprecht and Schofield, 1989; Silberstein et al., 1999; Williamson et al., 1987, see Table 4).

These sites are plotted in each panel of Fig. 6 as open squares and coded in Fig. 6b with letter identifiers corresponding to those listed in Table 4. The additional points support the hypothesis that all Australian rainfall–runoff data should conform to the regional distribution and that high evapotranspiration sites should generally be expected to be distributed within its lower quantiles. Two points ‘a’ and ‘i’ show distinctly high runoff. Point ‘a’ Thomson Brook, represents a 45% cleared jarrah forest form West Australia (Table 4). It sits on the median quantile curve (τ = 0.5), as may be expected if regional evapotranspiration was driving the long-term mean annual runoff. Point ‘i’ L6 from Lismore NSW (Table 4, see also Table 3), was 100% dry sclerophyll eucalypt forest. Information available in the study is insufficient to account for the high levels of runoff. Forest canopy was relatively open (34%), soils were less than 2 m in depth, and effectively confined all root growth (Putuhena and Corderoy, 2000). Nonetheless the site appears inconsistent with patterns of regional runoff. Runoff was also notably high after the site was cleared (Fig. 4, point ‘3’), but reduced to levels common to other sites following plantation conversion. Although local runnoff drivers may remain unexplained, regional data may still be used to identify the site as prone to larger runoff reductions than may be expected from regional trends.

USDA curves showed the best agreement with the native bush data (Fig. 6b) (K–S p > 0.8 compared with p = 0.5 for tanh and Zhang quantile forest curves and RMSE approximately 50% lower than the other models). The trend of the USDA ‘forest’ curve selected using the low rainfall conversion data showed close agreement with native bush data in both wet and dry areas, suggesting that the quantile curve selection process was not unduly biased by the use of low rainfall conversion data.

The wisdom in showing caution toward the scattered plantation conversion data from wetter areas was further supported by the addition of the native bush information. Points ‘g’ and ‘h’ in Fig. 6b represent the bush controls for the pine plantation at Clem Creek (Bren and Hopmans, 2007), which is depicted by an open triangle lying directly above them (also see point ‘8’ in Fig. 4). Evapotranspiration and canopy and forest floor interception have been identified as higher in pine plantations than native bush (Putuhena and Corderoy, 2000) and runnoff lower. Clem Creek appears to be unusual in this regard. The topography at Clem Creek is steeper and the stream perennial compared to the gently sloping intermittent streams of the native bush control catchments (Bren and Papworth, 1991), suggesting that the unusually high runoff from the plantation at Clem Creek is likely to be a result of peculiar site influences rather than indicative of regional hydrological processes.

The ability to use quantile curves to predict runnoff reductions was tested by constructing proportional runoff curves using Eq. (8) (see Section 2) and plotting them with reductions calculated by other authors using independent methods. Results are shown in Fig. 7. Runoff reductions due to Zhang default curves are shown for comparison. Data are limited but USDA runoff reduction curves display the best fit (K–S p = 1) and demonstrate the greatest capacity to model runoff reductions across the range of rainfall. Zhang quantile and tanh curves show a reasonable accord with data (K–S p > 0.7 for both), but their lack of sensitivity to the regional data appears to have translated into a limited ability to represent the range of variation in runoff reductions apparent in the USDA runoff reduction curve or the data. Runoff reductions modelled by the default Zhang curves were poorly estimated and did not bear any significant resemblance to the independent data (K–S p < 0.05).

Future plantation development in Australia is likely to be in the 600–800 mm/year rainfall zone (Keenan et al., 2004). Using the best performing empirical method of this study, afforestation within these areas may be expected to reduce mean annual runoff by between 75% and 90%, 10–20% more than reductions estimated using default Zhang curves.

5. Discussion

5.1. Value of regional data to smaller-scale studies

Recent studies of regional comparative hydrology have been able to take advantage of modern datasets to more clearly resolve the driving influences on regional–scale patterns of runoff variability identified by earlier authors (Budyko, 1974). At large scales regional climatic patterns exhibit a dominant influence (Peel et al., 2010), while at small scales non-climatic influences become more significant (Donohue et al., 2007; Milly, 1994; Oudin et al., 2008; Peel et al., 2010; Poff et al., 2006). Potential exists to use this information for applied outcomes where regional climatic patterns can be identified as distinctive.

The regional data used here were filtered to reflect areas with potential for plantation development, which represent the more temperate parts of the continent (Fig. 2). It is arguable that the same approach may be taken to identify sub-continental regions with distinctive climate to conduct similar analyses at sub-regional scales. Peel et al. (2000) noted a significant correlation between some of their model parameters and sub-regional climatic and catchment characteristics. Boughton and Chiew (2003, 2007), found that 11 of the 16 Tasmanian Peel et al. (2000) sites displayed higher levels of runnoff than mainland sites. A sub-regional scale analysis (Boughton and Chiew, 2007) showed significant improvement in predictions of mean annual runoff in the Murray-Darling Basin compared to the pooled mainland dataset.

The question of scale is important here. A distinct hydro-climatic region is required to provide statistical context of site-scale data. Smaller areas may be biased toward a number of non-climatic influences and likely to contain fewer data. Boughton and Chiew (2007) noted that smaller sample sizes from Peel et al. (2000) data in sub-regions outside the Murray-Darling Basin produced less convincing results. Comments by Donohue et al. (2007) and analyses by Peel et al. (2010) suggest that non-climatic influences become subordinate to regional climate over areas greater than 1000 km²; Budyko (1974) areas greater than 10,000 km². A number of systems for delineating geo-climatic zones are available (Budyko, 1974; Sanderson, 1999). Köppen-type systems have a history of successful application (Peel et al., 2007) but have been identified as limited due to their reliance on mean annual air temperature to characterise potential evapotranspiration without taking the effect of the earth’s radiative heat balance into account in forming geographic zones (Budyko, 1974).

5.2. Australian plantation conversion data

The amount of publically available hydrological plantation conversion data in Australia is limited as reflected in the sources
identified for this study (Tables 3 and 4) (see also Brown et al., 2005) although more may be available from other sources (Bren and McGuire, 2007). An assumption underpinning this work is that the regional data are sufficiently long–term to negate the influence of changes in catchment storage. For regional data this ensures high and low transpiration sites are located in the lower and upper quantiles. The dataset of Peel et al. (2000) meets this requirement ensuring that quantiles are identified which reflect evapotranspiration as the dominant influence on runoff. Zhang et al. (1999, 2001) considered 5 to 10 years a sufficient period to achieve steady–state hydrology, Donohue et al. (2007) greater than five. Many sites in Tables 3 and 4 have a period of record less than this, suggesting that sub-regional processes may be capable of influencing the quantile curve selection. Additional conversion data from sites that reflect patterns of regional geomorphology would improve confidence in identifying regional ‘grassland’ and ‘forest’ runoff quantile curves.

5.3. Parameter structure of rainfall–runoff functions

Single variable parameter rainfall–runoff curves were used because they produced coherent quantile curve arrays (Fig. 3). Allowing both model parameters to vary would provide greater freedom to describe the data (Fig. 1). To explore this further, quantile curves were generated by allowing both model parameters to vary in each rainfall–runoff function. Results are shown in Fig. 8. Quantile curve arrays derived using USDA and tanh curves were regular but not necessarily coherent: quantile curves conditional on the same function crossed (Fig. 8a and c). Quantile curves derived using variable $\omega$ and $E_z$ Zhang curves were more erratic.

5.3.1. Parameter interaction

ANOVA of one and two parameter regression models indicated that complexity provided by the additional parameter was warranted for USDA and Zhang curves ($p < 0.05$ and $< 0.001$ respectively), but evidently at the cost of incoherent quantiles and/or $E_z$ estimates. This was particularly true of the Zhang curves. Of the 15 quantile curves in Fig. 8b 13 represented incoherent parameter estimates, that is, plant available water coefficients were greater than the limit posited for ‘forest’ ($\omega = 2.0$) by Zhang et al. (2001) and/or $E_z$ estimates were negative.

Data or reasonable conceptual models of hydrological processes should inform the identification of parameters when competing estimates become apparent to ensure they lie within an acceptably feasible range (Franks et al., 1997; Schoups et al., 2008; Troutman, 1985). Families of quantile functions must be monotonically increasing (Koenker, 2005): for example, no part of a $\tau = 0.5$ quantile curve should lie below a $\tau = 0.4$ curve if they are conditional on the same function. Problems generated by the additional degrees of freedom may be attributed to high degrees of parameter interaction and the structure of the two-parameter models (Table 5). The effects of parameter interaction are well known, particularly

![Fig. 8. Selected data from Peel et al. (2000) and quantile curves conditional on the three empirical curve systems, with both parameters allowed to vary through the quantile regression. The same quantiles are plotted in each panel. Default Zhang curves ($\omega = 0.5, E_z = 1100$; $\omega = 0.5, E_z = 1400$) are shown as broken lines for comparison. Note results shown in (d) are identical to those in Fig. 3b.](image-url)
among more complex hydrological models (Duan et al., 1992; Franks et al., 1997; Jakeman and Hornberger, 1993; Kuczera, 1997; Schoups et al., 2008) but were surprising in these simple functions. Where model parameters are highly correlated, confidence in their simultaneous identification tends to be low (Schoups et al., 2008; Troutman, 1985), increasing the likelihood of multiple solutions being identified by optimisation algorithms (Duan et al., 1992; Kuczera, 1997), resulting in incoherent quantile models. Functions suffering from an inability to provide uniquely determined solutions can be described as ‘ill-posed’ and may be solved using methods that are beyond the scope of this paper (Arsenin, 1976).

The issue is likely to be compounded by the default algorithm used by nlsq. Newton-type algorithms have been found to be poorly suited to the objective function response surfaces generated by catchment rainfall–runoff models, which may be plagued with discontinuities and poor smoothness (Hendrickson et al., 1988), resulting in the identification of multiple optimal parameter estimates (Duan et al., 1992).

5.3.2. Structure

Conventional USDA fixed $\lambda$ quantile curves performed better than simple tanh and Zhang curves in describing the regional data, particularly at low rainfall (Fig. 6). This can be attributed to the structure of its parameters. Assuming a constant ratio between initial abstraction and retention for any catchment ($\lambda = 0.2$) may seem an unrealistic imposition, particularly as developers considered that $\lambda$ could theoretically adopt any reasonable value if the data under consideration warranted it (Ponce, 1996). But by constraining $\lambda$ around an assumed average level of interacting catchment losses, model specification proved sufficiently complex to provide an acceptable representation of runoff response while maintaining an analytically convenient and well-posed parameter structure. Results here may conceivably be improved through the development of an Australian initial abstraction ratio, but it is likely that $\lambda = 0.2$ is a good first estimate. Fu et al. (2011) found that varying $\lambda$ between 0.05 and 0.2 did not improve predictions of annual runoff, consistent with the results of variable $\lambda$ quantile regression completed here. Of the 15 curves in Fig. 8c over 50% featured $\lambda$ values ranging between 0.1 and 0.3; 80% were between 0.1 and 0.4.

The structure of single parameter tanh and Zhang curves showed an insensitivity to the regional data (Figs. 3 and 6), providing a reasonable description of runoff in either high or low rainfall areas but not both. ANOVA of the one and two parameter nls tanh models indicated that the increased complexity (Eq. (7), Table 1) was not warranted ($p > 0.3$), however the fit of the two parameter function to the regional data using both nls and nlsq ($t = 0.5$) was better than the one parameter model ($K-S p = 0.01$ versus 0.09 and $K-S p = 0.1$ versus 0.4 respectively). Clearly additional complexity in the tanh parameter structure would assist in its description of Australian data (compare Figs. 3a and 8a), consistent with a need for greater model complexity in arid areas (Atkinson et al., 2002), but re-parameterisation similar to that of USDA curves would be required to enable the function to form coherent quantile curves.

ANOVA of the Zhang function indicated that additional structural complexity was justified ($p < 0.001$, see above). But results of ordinary non-linear and non-linear quantile regression allowing both $\omega$ and $E_z$ to vary, resulted in parameter estimates outside the feasible range of $0.5 < \omega < 2.0$ posited by Zhang et al. (2001) (nls estimates: $\omega = 36$, $E_z = 313$ mm/year; nlrq ($t = 0.5$): $\omega = 7.1$, $E_z = 176$ mm/year). Goodness of fit to regional data did not improve compared to the single variable parameter ‘mixed vegetation’ $\omega = 1.25$ model described above ($K-S p = 0.05$ and <0.001 for nls and nlrq ($t = 0.5$), compared to $p < 0.05$). The perfect negative parameter correlation of the Zhang function (Table 5) also enabled a continuous range of solutions for $E_z$ to be identified for any limiting estimate of $\omega$ over its feasible range ($0.5 < \omega < 2.0$). A need for re-parameterisation is quite evident.

5.3.3. Misspecification

Single parameter, default ($\omega = 0.5$ and 2.0) Zhang curves also showed notably poor agreement with regional data. Figs. 3, 4, 6 and 8 show model bias in the consistent overestimation of runoff in areas of low rainfall, while performance in wetter areas is more consistent with regional data and quantile curves. This is not a feature of the regional data (after Peel et al., 2000), as the same bias is evident when Zhang curves are compared to their original information. Fig. 9 shows the data of Zhang et al. (1999) grouped according to continent with Zhang curves and USDA quantile curves from Fig. 6b for comparison. Unlike tanh curves, Zhang curves exhibit a similar level of bias and insensitivity to rainfall–runoff data at low rainfall, irrespective of whether one or two parameters are allowed to vary. Consistent bias indicates model ‘misspecification’, suggesting that model conceptualisation may be inadequate to describe physical processes (Troutman, 1985).

The conceptualisation of Zhang curves is different to the other functions studied here which model runoff from rainfall and catchment losses. Zhang curves follow the Budyko (1974) paradigm using potential evapotranspiration, requiring additional climatic information to model regional-scale runoff (see Eq. (1), Table 1). A fundamental principle underlying the Budyko approach is the integration of the radiative heat and water balances of the earth. Zhang et al. (1999, 2001) simplified Budyko’s boundary conditions by equating potential evapotranspiration to the ratio of net radiation and latent heat of vaporisation ($E_o = R/L$). Budyko (1974) noted that the simplification enabled the equations of Schrieber (1904) and Ol’dekop (1911) (in Budyko (1974)) to satisfy his postulated boundary conditions (assumed to be those cited by Zhang et al. (1999, 2001), but were only valid under conditions of sufficient moisture, where heat flux between the land and atmosphere may be neglected. In dry climates, turbulent surface-atmospheric heat
flux can not be ignored, amplifying the radiative heat flux term \( R \) in Eq. (1), resulting in increasing potential evapotranspiration with increasing aridity (Budyko, 1974).

Zhang et al. (1999, 2001) derived their ‘grassland’ and ‘forest’ equations by fixing \( \omega \) and performing non-linear regression against a global dataset to estimate a fixed estimate of a parameter that reflected potential evapotranspiration \( (E_o = E_z \) in Eqs. (4) and (5), Table 1). Comments by Budyko (1974) would suggest the specification of the model in this way would lead to an inherent insensitivity to potential evapotranspiration data from drier regions. Optimised estimates of \( E_z \) derived from a dataset with wide range of mean annual rainfall are likely to reflect the influence of information from higher rainfall regions, where the effects of turbulent heat flux between the earth’s surface and its atmosphere can be ignored. When the same values of \( E_z \) are used to estimate mean annual runoff in drier areas, model insensitivity is likely to result in a consistently biased underestimation of \( E_z \) and an overestimation of runoff. Better model specification may be possible with the inclusion of a term reflecting radiative heat balance.

This discussion provides a contrast to other studies which have found that Zhang curves represent a practical and robust tool for assessing the hydrological impacts of afforestation (Brown et al., 2005; Vertessy et al., 2003; Zhang et al., 2010 among others) and have ultimately resulted in leading Australian scientific organisations recommending them to policy-makers and industry (BRS, 2003; Zhang et al., 2007). The discrepancy between proportional run off reductions estimated by Zhang default parameter curves and USDA quantile curves (Fig. 7) suggests that recent important regional assessments in Australia (Chiew et al., 2008 and related reports) may have underestimated the potential impacts of afforestation in drier areas by 10–20%. In the context of large-scale assessments, this uncertainty may not be readily apparent given the others inherent in catchment hydrology. However the continuing uncritical application of the Zhang curves imbues them with an integrity that is not supported by Australian data with the potential of creating confusion in the minds of policy-makers.

5.4. Further work

This approach has the potential to establish standard regional or even sub-regional runoff reductions for Australian forestry planning in the context of emerging national water management initiatives (NWC, 2004). It would require a comprehensive database of all hydrological plantation conversion data available in Australia made accessible to jurisdictional water management agencies.

6. Conclusions

The hydrological variability of a coherent continental-scale region is driven by its regional climate. Departures from regional rainfall–runoff patterns may be attributed to sub-regional processes. Variance of these departures may be significant but data must still adhere to patterns of variability controlled by regional climate. This observation can be surprisingly powerful in defining regional hydrological relationships and providing statistical context to site scale studies. The distinctive hydro-climatic conditions of Australia are likely to be particularly useful in this regard, providing opportunities to supplement local data with regional knowledge. In principle, sub-regions with demonstrably distinctive conditions may be used in a similar way to provide insights at smaller-scales.
Quantile curves represent a method of identifying regional evapotranspiration-driven runoff trends by transparently integrating regional and site-scale data. The results obtained provide a graphical indication of the uncertainty associated with extrapolating data to other sites or across regions. Additional statistical context gained by adopting this approach provides potential for further identification of influential catchment processes at particular sites or even infer the likely magnitude of long-term impacts in the context of regional relationships.

It is possible to use any variable parameter function to derive quantile curves, but sensible, coherent results are only possible if the model is well-posed. USDA curves proved the most successful of the empirical systems used here due to their superior structure, which provided flexible specification in a single parameter model. Single parameter tanh and Zhang curves were less flexible in simultaneously describing runoff from both dry and wet areas across regional Australia. Tanh curves showed a capacity to improve in this area with the addition of a loss parameter, but the existing two-parameter structure (Eq. (7), Table 1) would require transformation into a more well-posed form to produce coherent quantile curves.

All two-parameter functions were found to suffer from high levels of parameter correlation. Zhang curves also appeared to be misspecified with regard to rainfall–runoff data. Re-parameterisation of the Zhang function (Eqs. (4) and (5), Table 1) may reduce the effects of parameter interaction but it is likely that limitations in their conceptualisation may continue to limit their success in drier areas.

Runoff reductions reported by other studies were best described by USDA runoff reduction curves, compared to widely accepted methods which underestimated impacts by 10–20% in the 600–800 mm/year rainfall range, where plantation development in Australia is likely.

Rather than collecting more data, this approach seeks to obtain more information from available data. Catchment responses to land-cover changes may be extremely variable and unpredictable (Hibbert, 1967), but regardless of the problems of drawing statistical inference from a scattered set of small-scale studies, valuable information on the hydrological impacts of afforestation is available and ready for use in planning (Bosch and Hewlett, 1982).

Acknowledgements

This study was funded by a University of Melbourne scholarship made available through the Cooperative Research Centre for Forestry. Additional Australian time series data were provided by Dr. Lu Zhang and Prof. Ian Gordon. I would like to thank Ms Rachel Adams of the Morris Miller Library, University of Auckland for timely advice on R and Prof. Ian Gordon of the University of Melbourne Statistical Consulting Service.

References


APPENDIX E: SECTION 6

Bayesian scrutiny of simple rainfall–runoff models used in forest water management

Ashley J.B. Greenwood\textsuperscript{a,b,*}, Gerrit Schoups\textsuperscript{c}, Edward P. Campbell\textsuperscript{d}, Patrick N.J. Lane\textsuperscript{a,b}

\textsuperscript{a}Department of Forest and Ecosystem Science, University of Melbourne, 221 Bouverie Street, Parkville, Victoria 3010, Australia
\textsuperscript{b}CRC for Forestry, College Road, Sandy Bay, Tasmania 7050, Australia
\textsuperscript{c}Department of Water Management, Delft University of Technology, PO Box 5048, 2600 GA Delft, The Netherlands
\textsuperscript{d}CSIRO Mathematics, Informatics and Statistics, Floreat, WA 6014, Australia

A R T I C L E   I N F O

Article history:
Received 26 August 2013
Received in revised form 19 December 2013
Accepted 31 January 2014
Available online 13 February 2014
This manuscript was handled by Geoff Syme, Editor-in-Chief

Keywords:
Water management
Forest hydrology
Hydrological modelling
Model selection
Bayesian analysis
Markov chain Monte Carlo

S U M M A R Y

Simple rainfall–runoff models used in the assessment of land-use change and to support forest water management are subjected to a selection process which scrutinises their veracity and integrity. Veracity, the ability of a model to provide meaningful information is assessed using performance criteria, incorporating: a popular mean square error (MSE) approach; empirical distribution functions and information criteria. Integrity, a model's plausibility reflected in its ability to extract information from data, is assessed using a Bayesian approach. A delayed rejection, adaptive Metropolis algorithm is used with a generalised likelihood to calibrate the models. Predictive uncertainty is assessed using a split sample procedure which uses high runoff data for calibration and drier data for validation. A simple multiplicative latent variable is used to accommodate input uncertainty in rainfall data, enabling a distinction to be made between uncertainty associated with data, parameters and the models themselves. The study demonstrates: the focus provided by setting model evaluation in a philosophical context; the benefits of using a more meaningful range of performance criteria than MSE-based approaches and the insights into integrity provided by Bayesian analyses. A hyperbolic tangent model is selected as the best of five candidates for its superior veracity and integrity under Australian conditions. Models with extensive application in South Africa, Australia and USA are rejected. Challenges to applying this approach in water management are identified in the pragmatic nature of the sector, its capacity constraints and a tendency of researchers to place confidence in accepted methods at the expense of rigour.

1. Introduction

The scrutiny of hydrological model structures has received close attention in recent years (Clark et al., 2011; Fencia et al., 2011; Gupta et al., 2012; Krueger et al., 2010; Martinez and Gupta, 2010). The problem is particularly important in water resources management, where hydrological models are used to underpin the security of access entitlements (Etchells and Malano, 2005). There are now a plethora of models available which are capable of supporting differing assertions regarding water availability (Clark et al., 2011 and citations therein) due to their emphasis on different catchment processes and the difficulties in clearly defining the mathematics of hydrological processes (Grayson et al., 1992), ultimately making regulatory frameworks vulnerable to challenge. Modelling and associated decision-making processes were challenged in the first stages of forest water regulation in Australia, prompting the amendment of regional planning frameworks to ensure greater consistency in the assessment of commercial forestry development applications (Greenwood, 2013).

At the heart of the matter was a loss of confidence precipitated by competing agendas (Greenwood, 2013). Confidence, or the degree of belief in any proposition or theory is fundamental to building scientific hypotheses and particularly germane to model choice (Gupta et al., 2012). A survey conducted by the ASCE (1985) concluded that confidence in a model may be based on: (1) scientific insight, or if insight is lacking; (2) an acceptance of simplified, sometimes empirical approaches with reduced expectations; or (3) in the absence of insight or acceptable alternatives, faith in the scientific authority associated with the model. The mixture of faith and science can become particularly heady in
decision-making, leading to unrecognised uncertainties and un-founded optimism (Ascher, 1993; Oreskes and Belitz, 2001).

Principles of model selection have been articulated in a range of publications that have considered methods (Laio et al., 2009; Marshall et al., 2005; Schoups et al., 2008); frameworks (Clark et al., 2008, 2011; Gupta et al., 2008, 2012; Jakeman et al., 2006; Martinez and Gupta, 2010; Matott et al., 2009) and guidelines (Black et al., 2011; Liu et al., 2008; Refsgaard and Henriksen, 2004; Vaze et al., 2012). If models represent hypotheses or theories (Clark et al., 2011; Gupta et al., 2012), the underlying motivation of these works is to identify a good theory. Hypothesis testing has traditionally followed either deductive logic, where weakly informative models are falsified, or inductive logic, where the most plausible proposition is identified among a number of competing alternatives. Both philosophies are well represented in science, but neither on their own is sufficient to conclude anything about the acceptability of a theory (Brodie, 1993). The two approaches were combined by Huber (2008) using formal logic, who concluded that a theory must be both informative and more plausible than other theories; a concise expression of the concept of strong induction in which alternative hypotheses are devised and experiments conducted to exclude one or more of them on the pathway toward inference (Platt, 1964).

Strong induction is evident in many of the systematic model selection approaches cited above, the key elements of which include: the identification of candidate models; an evaluation of model performance in replicating observed data and generating reasonable predictions; and the quantification of uncertainty. The first step encompasses the development of conceptual models reflecting available knowledge, theory and beliefs (Gupta et al., 2012). A philosophy of scrutiny should then be established (Burnham and Anderson, 2004) before model performance is evaluated using appropriate performance criteria. This establishes a model’s veracity or ability to provide meaningful information. After Huber (2008), an additional step is required to establish its integrity, or whether a model is plausible given the data used and any other contentions raised in presenting the conceptual model for scrutiny. A well understood feature of a plausible model is its identifiability or its ability to access sufficient information from the data to constrain model parameters given its specification and structure (Jakeman and Hornberger, 1993; Schoups et al., 2008; Soroooshian and Gupta, 1985). If a model provides a good description of processes the data can be expected to provide robust parameter estimation (Sivia and Skilling, 2006). Models that have difficulty in this regard are likely to be ill-posed and prone to implausible or meaningless predictions; a condition readily diagnosed from the character of the posterior distribution (Spiegelhalter et al., 2002). The use of information criteria (BIC) approximation to the Bayes’ factor, a purely Bayesian approach which quantifies the weight of evidence in favour of one model hypothesis over another (Kass and Raftery, 1995). For Bayesian purists, little else is required, however difficulties in calculating Bayes’ factors and philosophical concerns regarding underlying assumptions has limited their use (Gelman et al., 2004).

A Bayesian analogue of AIC is available in the deviance information criterion (DIC), which like its frequentist counterpart comprises a goodness of fit term based on the deviance of modelled data from observations and a penalty term to account for model complexity (Spiegelhalter et al., 2002). The use of information criteria in both hydrology (Diamantopoulos et al., 2012; Laio et al., 2009; Marshall et al., 2005) and forest hydrology has become more common in recent times (MacKay et al., 2012).

Quantifying uncertainty is fundamental to model selection (Matott et al., 2009; Sivapalan et al., 2003). Uncertainties arise from data, parameter estimates and the model itself (Renard et al., 2010). Numerous approaches for quantifying uncertainty have been proposed in hydrological modelling, all emphasise its role in improving confidence in the modelling process and its outputs (Gupta et al., 2008; Matott et al., 2009; McMillan et al., 2011; Nott et al., 2012; Renard et al., 2010; Vrugt et al., 2009 among many others).

Bayesian approaches are now more common in hydrology (Koskela et al., 2012; Laloy and Vrugt, 2012; Vrugt et al., 2009 and many others), and have the benefit of simultaneously enabling uncertainty analysis and scrutiny of model integrity (Renard et al., 2010). Analyses include assessments of predictive uncertainty that lump uncertainty due to data and model structure together (Kavetski et al., 2011; Laloy et al., 2010; Schoups and Vrugt, 2010) or those that explicitly represent separate sources of uncertainty (Koskela et al., 2012; McMillan et al., 2011; Renard et al., 2010; Renard et al., 2011) according to the requirements of the study (Kavetski et al., 2011; Laloy et al., 2010). Accounting for different sources of uncertainty, particularly those associated with input data leads to reduced bias, more consistent parameter estimates and more reliable estimates of predictive uncertainty (Renard et al., 2010; 2011).
Different hydrological models and assessment approaches were advocated under competing agendas in the first stages of Australian forest water regulation. While one approach was adopted to support state-wide policy in South Australia (Greenwood, 2013), questions raised regarding the veracity of the other (Greenwood et al., 2011, and citations therein) have not interrupted its continued use in important assessments (Post et al., 2012), leaving the situation unresolved and demanding further systematic scrutiny in the light of alternatives.

This paper addresses the matter by subjecting three similarly simple curve models used to underpin forest water regulation in both South Africa and Australia, along with the widely used USDA curve number function to a formal selection process. A linear trend line is included for comparison. Model selection is undertaken in a four step process: (1) candidate models are selected according to their structural similarity and use in forest water management and/or land-use change assessment; (2) a philosophy of evaluation is adopted which addresses both model veracity and integrity/plausibility; (3) veracity is evaluated by an MSE variant augmented by EDFs and information criteria; (4) model integrity is evaluated using a Bayesian approach, encompassing uncertainty assessment.

The analysis demonstrates the rigour and confidence gained by: the explicit adoption of a formal philosophy in model evaluation; the robustness attained when assessing model veracity using meaningful performance criteria; and the insights into model integrity possible in a Bayesian analysis. The paper commences with a description of the models in Section 2, followed by the data in Section 3. Section 4 describes the performance criteria while Section 5 provides an overview of the Bayesian methodology. Results are presented in Section 6 followed by a discussion and conclusions in Sections 7 and 8 respectively.

2. Rainfall–runoff curves

Plantation forest water use is currently regulated in South Africa and parts of Australia. Its hydrological impacts were accounted for in South Africa with empirical rainfall–runoff relationships which were eventually superseded by mechanistic modelling (Dye and Versfeld, 2007). The early stages of regulation in Australia saw the use of competing curve systems, one of which was ultimately used to support South Australian state policy (Greenwood, 2013).

2.1. Midgley & Pitman

South African curves were based on a number of studies. Nänni (1970) derived rainfall–runoff relationships based on a power function by Midgley and Pitman (1969), modified to reflect empirical observations from the Natal uplands (Nänni, 1970). South African government officers then modified Nänni curves by incorporating additional data from international studies to underpin forest water management (Van der Zel, 1995). Their purely empirical nature makes it impossible to re-calibrate Van der Zel or Nänni curves using independent data, consequently this study examined the underlying parametric relationship.

The Midgley and Pitman (1969) equation comprises a power function below a critical threshold value of rainfall and a linear function at higher rainfall (Eqs. (1) and (2)).

\[ R = \beta P; \quad P \leq P_t \]

\[ R = P - I; \quad P > P_t \]

where \( R \) is mean annual runoff (mm/year); \( P \) is mean annual precipitation (mm/year); \( P_t \) is the threshold rainfall above which the relationship between mean annual rainfall and runoff is linear (gradient equal to unity) and \( I \) is a constant loss rate for higher \( P \). The value of \( P_t \) and \( I \) is determined by differentiating Eq. (1) and equating \( \frac{dR}{dP} \) to unity, while \( \beta \) and \( \gamma \) are parameters to be determined (Midgley and Pitman, 1969).

2.2. Hyperbolic tangent

A two parameter hyperbolic tangent or tanh function was developed by McMahon et al. (1992) to generate effective precipitation before routing it through a linear store to generate yield for a study on global runoff patterns:

\[ R = P - L - F \tanh \left( \frac{P - L}{F} \right) \]

where \( R \) is runoff or yield (mm); \( P \) is precipitation (mm); \( tanh \) is the hyperbolic tangent function, while \( L \) and \( F \) are notional loss parameters (mm) to be determined.

Grayson et al. (1996) advocated its use to infill incomplete rainfall–runoff data series, while Sirirawdona et al. (2006) used it to model the impacts of land-use change on annual catchment yield. Greenwood and Cresswell (2007) calibrated Eq. (3) with empirical data to derive runoff reductions currently used to support forest water regulation in South Australia (Greenwood, 2013).

2.3. USDA curves

USDA curve numbers are arguably the best supported empirical curve system in the world. Initially developed in the 1950s by the US Department of Agriculture (USDA) to model direct runoff from different land-cover (Ponce and Hawkins, 1996), the system has evolved into a sophisticated decision-making framework incorporating different rainfall–runoff relationships for soil-land-cover complexes with differing hydrological properties under different antecedent conditions (USDA-NRCS, 2004). While originally developed for event-based runoff the function has also been used to model yield at annual and mean annual time steps (Fu et al., 2011; Greenwood et al., 2011).

The USDA relationship is given by the equation:

\[ R = \left( \frac{P - L}{P + (1 - L)S} \right) \cdot \frac{S}{S + 10}; \quad S = \frac{1000}{CN} \cdot \frac{1 - \lambda}{\lambda}; \quad \lambda = \frac{I}{S} \]

\[ R = \frac{\varepsilon CN (P + 10\varepsilon - 100\varepsilon^2)}{CN (P + 10\varepsilon - 100\varepsilon^2)} \]

where \( R \) is runoff or yield (inches/event in Eq. (4), mm/event in Eq. (5)) if \( \varepsilon = 25.5 \text{ mm} \), after Ponce and Hawkins, 1996); \( P \) is precipitation (inches or mm/event in Eq. (4) or (5) as for \( R \); \( CN \) is the curve number (a number corresponding to a rainfall–runoff curve ranging between 1 and 100 for low and high yielding land-cover respectively, dimensionless) and \( \lambda \) is a measure of loss known as the initial abstraction ratio, the quotient of initial abstraction \( I_\lambda \) (inches) and estimated catchment retention \( S \) (inches). The initial abstraction ratio is generally set by convention to 0.2 but is known to exceed 1.5 (see Fig. 10-1 in USDA-NRCS, 2004).

2.4. Zhang curves

Zhang curves are the most widely used empirical curve system in Australian forest hydrology. Confidence inspired by their reputation among researchers lead them to be adopted to support water management decision making in the first stages of forest water regulation in South Australia (Greenwood, 2013) and they continue to be used in a range of important water resource assessments (Chiew et al., 2008; Post et al., 2012).

Zhang et al. (2001) developed a rational relationship between evapotranspiration and precipitation using boundary conditions postulated by Budyko (1974):
\[
\frac{E_t}{P} = \frac{1 + \omega K}{1 + \omega K + (\frac{\omega}{P})^4}
\]

where \(E_t\) is the evapotranspiration from a given land-cover (mm); \(E_o\) is potential evapotranspiration (mm); \(P\) is precipitation (mm) and \(\omega\) is a plant available water coefficient, representing the ability of plants to store water in the root zone for transpiration and posed to vary between 0.5 and 2.0 for short grass and forest respectively (Zhang et al., 2001). Eq. (6) was used to estimate evapotranspiration for grass and forest by fixing \(\omega\) to 0.5 and 2.0 respectively, and conducting non-linear, least squares regression with data from 256 catchments around the world. Average evapotranspiration from trees was 1410 mm/year and 1100 mm/year from herbaceous plants. By assuming that changes in soil moisture could be regarded as zero in a long-term mean annual catchment water balance, a function describing mean annual runoff for a given land-cover could be derived:

\[
R = P - E_t; \quad R = P - P \left[ \frac{1 + \omega K}{1 + \omega K + (\frac{\omega}{P})^4} \right]
\]

where variables are as described for Eq. (6) above, but on a mean annual time step (mm/year) and \(E_t\) is the mean annual evapotranspiration optimised for a given land-cover using a global data set (mm/year).

3. Data

Key to the scrutiny of models is the use of appropriate data. An independent dataset was required that was consistent with the conditions required in the derivation of the functions. A particular constraint of the study is that both Midgley and Pitman and Zhang curves were derived using mean annual data. Furthermore, Zhang et al. (2001) required rainfall to be the dominant form of precipitation, catchment gradients be gentle, soil be relatively thick and records of annual rainfall and streamflow data be sufficiently long to ensure a zero change soil water storage when calculating the mean average annual water balance. Moreover, the regions where curves have been used in forest water regulation (South Africa and Australia) are dry by world standards and have distinctively variable hydrology (McMahon et al., 1992).

A publically available, independent dataset that meets the criteria above with regard to rainfall, length of record and representation of highly variable hydrology is the Australian regional dataset of Peel et al. (2000). As Australia's topography is relatively modest, most catchments are likely to be gently sloping, however no information is available on soil depth (Peel et al., 2000). The complete dataset comprise mean annual rainfall–runoff data pairs and runoff coefficients from 331 catchments across Australia. Each data pair is recorded with its location (including administrative jurisdiction), catchment area, gauge number and station name. Sites were selected by Peel et al. (2000) to be unimpaired by regulation or diversions; have at least 10 years of historic monthly streamflow data and catchment areas between 50 km² and 2000 km². Additional information regarding the data’s compilation may be found in Peel et al. (2000).

The data selected were a subset of 313 temperate sites used by Greenwood et al. (2011), which represent the regions of Australia considered prospective for plantation expansion by Keenan et al. (2004) (Fig. 1). For the purposes of uncertainty analysis the 313 point dataset was split into 200 point “wet” or high runoff subset and a 113 “dry” or low runoff set (see further discussion in Section 5.3 below).

4. Performance criteria

4.1. Nash–Sutcliffe efficiency

Functions based on mean squared error (MSE), particularly the normalised Nash–Sutcliffe efficiency (NSE), are the most widely used criteria in evaluating hydrological model performance against observed data (Gupta et al., 2009):

\[
MSE = \frac{1}{n} \sum_{i=1}^{n} (y_{mi} - y_{o,i})^2
\]

\[
NSE = \frac{\sum_{i=1}^{n} (y_{mi} - y_{o,i})^2}{\sum_{i=1}^{n} (y_{mi} - \mu_o)^2} = 1 - \frac{MSE}{\sigma_o^2}
\]

where \(n\) is the total number of data; \(y_{mi}\) is the \(i\)th modelled response (at \(n = i\)); \(y_{o,i}\) is the \(i\)th observed response and \(\mu_o\), \(\sigma_o\) are the mean and standard deviation of the observations respectively. NSE requires that the assumptions for ordinary least squares (OLS) hold; that errors are independent with constant variance. Unfortunately the assumptions are usually invalid for hydrological datasets, making MSE and its variants poor performance measures (Gupta et al., 2009). The squared residuals in MSE/NSE also make them sensitive to the units of measurement and sensitivity to large values (high flows) comes at the expense of performance under low flow conditions (Krause et al., 2005).

4.2. Empirical distribution functions

Like flow duration curves, empirical distribution functions (EDFs) are based on an ordered sample of observations \(y = \{y_1 < y_2 < \ldots < y_{n-1} < y_n\}\). If the cumulative distribution function (CDF) of \(y\) is denoted by \(F_n(y)\), EDFs test the null hypothesis of whether \(F_n(y)\) belongs to a theoretical parametric distribution \(F_n(y, \theta)\) where \(\theta\) is a parameter vector, or a competing empirical distribution \(G_m(y)\). Different EDF performance criteria are defined according to how the discrepancy between two distributions is measured. The Kolmogorov–Smirnov (KS) test measures the maximum absolute difference between two distributions and the test statistic \(D_{KS}\) may be written in the form:

\[
D_{KS} = \text{max} |F_n(y) - G_m(y)|
\]

KS tests are most sensitive around the median values and less so around distribution extremes, giving it strength in identifying shifts in central regions of distributions but less discrimination in distinguishing more dispersed differences (Arnold and Emerson, 2011).

Other EDFs are regarded as more powerful. A number of quadratic statistics have been derived which can be written in a general form:

\[
Q^2 = n \int_{-\infty}^{\infty} |F_n(y) - F(y, \theta)|^2 \Psi(y)dydF(y)
\]

where \(\Psi(y)\) is a weight function (Laio, 2004). When \(\Psi(y) = 1, Q^2\) becomes the Cramer-von Mises statistic \(W^2\): when \(\Psi(y) = |F(y, \theta) - F(y, \theta)|^2\), \(Q^2\) becomes the Anderson–Darling (AD) statistic \(A^2\). Eq. (12) shows the AD test against a theoretical parametric distribution, while Eq. (13) shows the two sample version between two empirical distributions:

\[
A^2 = n \int_{-\infty}^{\infty} |F_n(y) - F(y, \theta)|^2 dydF(y)
\]

\[
A^2 = \frac{mm}{N} \int_{-\infty}^{\infty} |H_n(y) - H_o(y)|^2 dH_n(y)
\]

where \(H_n(y) = (nF_n(y) + mG_m(y))/N\), is the EDF of the pooled sample with \(N = m + n\). Unlike the KS statistic, quadratic criteria inte-
grate differences over entire distributions and are better suited to identifying more modest deviations over a wider data range (Arnold and Emerson, 2011). The weighting of the AD statistic provides greater discriminatory strength in the tail regions of the candidate distributions and the capacity to distinguish more dispersed differences (Laio, 2004; Laio et al., 2009).

While EDFs are robust and can deal with small sample sizes, their limitations include an inability to effectively penalise models to account for complexity (Laio et al., 2009), although approximations have been developed to address some situations (Laio, 2004).

4.3. Information criteria

NSE, EDFs and other classical performance criteria are applied under the assumption that the best model has been identified (Burnham and Anderson, 2004). This assumption is being confronted in hydrology by admitting competing modelling hypotheses into model evaluation and development (Clark et al., 2011; Gupta et al., 2012). Model choice has moved from purely performance based considerations to more philosophical paradigms, such as those which expound principles of parsimony to manage uncertainties arising from unwarranted structural complexity (Son and Sivapalan, 2007).

Information criteria represent a trade-off between the deviance of candidate models from the true model and their complexity (Burnham and Anderson, 2004; Spiegelhalter et al., 2002). For any set of $j$ parameteric models $\mathcal{M}_j(\theta)$, the deviance of the $j$th model $D_j$ can be defined as the distance between the logarithms of the “true” and candidate model probability distributions:

$$D_j(\hat{\theta}) = -2\log_e(L(\hat{\theta})) + 2\log_e(p(y))$$  \hspace{1cm} (14)

where $L(\hat{\theta}) = \prod_{i=1}^n p(M_j(\theta_i))$ is the likelihood function defined through the model at its maximum likelihood parameter estimate $\theta = \hat{\theta}$ and $p(y)$ is a standardising term based on observed data $y$, which remains constant across all models (Burnham and Anderson, 2004; Laio et al., 2009). The familiar Akaike information criterion (AIC), when applied to multi-model comparison and simplified, may be written:

$$AIC_j = -2\log_e(L(\hat{\theta})) + 2p_j$$  \hspace{1cm} (15)

where $p_j$ is the number of parameters $p$ in model $j$ (Laio et al., 2009, Appendix A).

A similar relationship can be derived from a Bayesian perspective, where comparison with an ontologically true probability distribution can be replaced by a direct comparison between the observed and modelled response data. The Bayesian information criterion (BIC) relates the discrepancy between the model $\mathcal{M}_j$ and the data $D$, to the highest posterior probability, $p(M_j|D)$:

$$BIC_j = -\phi(p(M_j|D))$$  \hspace{1cm} (16)

where $\phi(.)$ is a generic monotonically increasing function (Laio et al., 2009, Appendix B). If uniform priors are assumed and $\phi(.) = 2\log_e(.)$, the relationship simplifies to the log-likelihood and
a constant which penalises for the number of parameters or model complexity:

\[ \text{BIC}_j = -2 \log(L(\hat{\theta})) + p_j \log(n) \]  

(17)

where \( n \) is sample size. The \( \log_e \) in the penalty term results in BIC tending toward simpler (lower-dimensional) models. The model chosen is that maximises \( p(M|D) \), producing the lowest BIC, while the relative merit of competing models may be compared in an analogous manner to Bayes factors and AIC, by the difference in BIC (ΔBIC). If ΔBIC > 2, the models are similar; within the range 3–10 they are different (with the lowest preferred) and very strongly different when ΔBIC exceeds 10 (Burnham and Anderson, 2004; Kass and Raftery, 1995; Laio et al., 2009).

The deviance information criteria (DIC), is a well-established Bayesian criteria less commonly seen in hydrological literature (a recent example may be found in Diamantopoulos et al., 2012). Its derivation commences with an estimate of deviance or fit (Eq. (14)), and incorporates a penalty term for complexity in models of arbitrary structure (Spiegelhalter et al., 2002). The measure of fit used is the posterior mean deviance \( D_j(\hat{\theta}) \) with \( p(y) = 1 \) for simplicity. The penalty term \( p_p \) is estimated through the reduction in uncertainty attained during parameter estimation, as the difference between \( D_j(\hat{\theta}) \) posterior mean deviance and the deviance at the maximum likelihood parameter estimate \( D(\hat{\theta}) \), which may be chosen as the posterior mean, mode or median:

\[ p_p = D_j(\hat{\theta}) - D(\hat{\theta}) \]  

(18)

Adding fit and the penalty gives the DIC:

\[ \text{DIC}_j = D_j(\hat{\theta}) + p_p = D_j(\hat{\theta}) - D(\hat{\theta}) \]  

(19)

The resulting criterion is trivial to compute with access to a posterior parameter distribution derived using MCMC methods (see section 5.1 below) and there is no need to consider exact or limiting forms. Like BIC, the preferred model has the lowest DIC. Spiegelhalter et al. (2002) asserted that rules of thumb used in evaluating the significance of AIC tests be used for DIC. Models with ΔDIC exceeding 3–10 of the lowest have considerably less support; models with DIC exceeding 10 of the lowest have essentially no support (Burnham and Anderson, 2004).

The Bayesian approach used in this study prompted the use of BIC and DIC, although the uniformly distributed priors and identical number of parameters in each model would have generated AIC results similar to BIC. DIC \( \hat{\theta} \) was set as the modal maximum likelihood.

5. Bayesian analysis

Bayesian techniques are becoming more common in both research and applied hydrological fields (De Roo et al., 2011; Vrugt and Ter Braak, 2011). However, they are not widely used in routine water management despite the recognised need for uncertainty assessment (Black et al., 2011) and the capacity of Bayesian analysis to facilitate it.

All quantities subject to inference in a Bayesian analysis are assigned probability distributions. Parameters are identified within a posterior probability distribution of possible alternatives (see below). A proper posterior distribution enables the direct estimation of confidence/credible intervals, and with little further work, can be used to estimate prediction intervals in a split sample analysis. Computational difficulties in evaluating complex integrals associated with probability distributions have been largely circumvented with the development of MCMC techniques, which allow direct simulation of the posterior distribution (see Chib and Greenberg, 1995).

Any parametric model \( M(x|\theta) \), with input data \( x = \{x_1, \ldots, x_i\} \), parameters \( \theta = \{\theta_1, \ldots, \theta_n\} \) and observed responses \( y = \{y_1, \ldots, y_i\} \) may be written in terms of a non-linear regression relationship comprising responses and additive, random errors \( e = \{e_1, \ldots, e_i\} \):

\[ y = M(x|\theta) + e \]  

(20)

Given some parameters and forcing data, Eq. (20) can be used to generate modelled responses. In hydrological modelling the problem is termed “inverse”; the data and responses are known but the parameters need to be identified. Bayes’ theorem reverses the inverse problem, allowing parameters to be identified from data:

\[ p(\theta|y) = \frac{p(y|\theta) \times p(\theta)}{p(y)} \]  

(21)

where “\( \times \)” indicates conditionality on the ensuing entities and \( p(\cdot) \) their associated probability distribution. Conventional nomenclature denotes \( p(\cdot) \) as the prior probability distribution; \( p(y|\theta) \) the likelihood arising from the modelled and observed outputs; \( p(y) \) the probability of the data, marginal likelihood or evidence and \( p(\theta|y) \) is the posterior probability of the parameters (Sivia and Skilling, 2006).

5.1. MCMC

Markov chain Monte Carlo (MCMC) uses a Markov chain to generate a sequence of parameter values under conditions which ensure the chain’s limiting distribution is the distribution of interest. The limiting or equilibrium distribution of a Markov chain, is one to which an associated chain will converge under certain mathematical conditions (see Chib and Greenberg, 1995; Gilks et al., 1996 and citations therein for further information). Given transitions between sequential, random realisations \( (\theta_0, \ldots, \theta_t) \), the general MCMC algorithm may be written descriptively as:

1. Initialize the chain with a realisation of \( \theta_0 \) and set \( t = 1 \). This will often be done by drawing a realisation from a prior distribution of parameters.
2. Generate a proposal \( \theta' \) for \( \theta_t \) by drawing from a proposal probability density function \( q(\theta_{t-1}, \theta') \), and evaluating the posterior probabilities \( p(\theta_{t-1}) \) and \( p(\theta') \) using Eq. (21).
3. With random probability a \( \alpha(\theta_{t-1}, \theta') \), accept \( \theta' \) as \( \theta_{t+1} \), otherwise retain \( \theta_t \) as \( \theta_{t+1} \).
4. Increment \( t \) and return to step 2 until sufficient realisations have been generated.

The rapidity with which the chain traverses the state space toward equilibrium is termed mixing and is related to the efficiency of the algorithm. In the algorithm above the proposals and acceptance probability at steps 2 and 3 are constructed to ensure that the equilibrium distribution is the target distribution of interest, the posterior distribution of Bayesian inference.

Mixing depends on the choice of proposal distribution variance. A wide variance is likely to result in slow mixing due to a large number of samples being proposed in low probability areas, away from modal states. Under these conditions a large number of rejections will be generated and giving the chain an appearance of becoming stuck. Slow mixing is a characteristic of high parameter dependence where narrow, eccentric joint posterior distributions are readily spanned by the proposal distribution. It is also evident when a proposal variance is too narrow, making exploration of the parameter space slow, generating large numbers of acceptances (Gelman et al., 1996; Gilks and Roberts, 1996; Roberts et al., 1997).

Adaptive MCMC algorithms facilitate mixing by refining the proposal variance–covariance using information collected by the Markov chain as it progresses through parameter space (Haario et al., 2006).
et al., 2001). However, overall efficiency gains may be undermined by excessive rejections before the benefits of tuning are felt. If the number of rejections starts to become unacceptably large, mixing can be improved by introducing steps into the MCMC algorithm which delay rejection and tighten the proposal covariance, thereby increasing the likelihood of acceptance (Haario et al., 2006).

The convergence of an MCMC simulation is effectively attained when inferences do not depend on the starting point of the chain(s); a state that can be difficult to diagnose (Gelman and Rubin, 1992; Geyer, 1992; Gilks et al., 1996). The early sections of Markov chains termed burn-in, are strongly dependent on the starting values, may be dominated by low probability realisations and are typically discarded.

The length of burn-in may vary from a few percent (Geyer, 1992) to half the chain (Gelman and Rubin, 1992), according to the simulation context. The total number of iterations required to achieve convergence depends on the complexity of the model (Gilks and Roberts, 1996) and the scaling of the proposal distribution as discussed above (Gelman et al., 1996; Roberts et al., 1997). If the chain is too short, the posterior may not be representative of the target stationary distribution (Geyer, 1992).

A key indicator of convergence and algorithm efficiency is the acceptance rate: the total number of post burn-in acceptances divided by the number of post burn-in iterations. A high acceptance rate indicates poor mixing, arising from the chain moving very slowly through higher probability state space (Chib and Greenberg, 1995). Lower acceptance rates are generally preferred, although unacceptably low acceptance rates can arise from high parameter interaction. Gelman et al. (1996) suggest that acceptance rates should range between 44% for symmetric 1-dimensional proposal distributions, declining asymptotically to 23% multivariate problems of very large dimensions.

Multiple Markov chains may facilitate the diagnosis of convergence (Gelman and Rubin, 1992), and seem to be the norm in hydrology, where complex response surfaces are common (Laloy and Vrugt, 2012; Vrugt and Ter Braak, 2011). However, criteria based on multiple chains do not guarantee convergence (Brooks and Gelman, 1998) and if runs are sufficiently long, then one chain is sufficient (Geyer, 1992).

This study employed a simple Metropolis algorithm (see Appendix 4 Van Oijen, 2008) with a delayed-rejection, adaptive routine (after Haario et al., 2006, 2001).

The adaptive period was set to commence at 10% of the chain, while adaptation itself commenced after 33% of the chain had been realised. If 10 or more consecutive rejections were generated during the first 10% of the chain, the proposal covariance was multiplied by 0.1. Starting proposal covariances were initially set at $2.4\sqrt{d}$ after Gelman et al. (1996) where $d$ is the number of parameters, and refined in short runs. The simple parameter structures prompted the use of long ($\geq 1 \times 10^5$), single chains (see Results). Half the chains were relegated to burn-in. Convergence was diagnosed when acceptance rates stabilized between 20% and 25%.

5.2. Assigning distributions

Uniform prior distributions were assigned to all parameters, including latent rainfall multipliers (Table 1). Prior ranges for the hyperbolic tangent function had to be expanded to accommodate the posterior distribution generated during the wet calibration (discussed below), while those of the Zhang model were varied a number of times during the analysis to further explore parameter identifiability (see Results, Fig. 8).

Assigning a formal likelihood to the model and data involves making assumptions about the distribution of residual errors in the joint posterior distribution, usually the existence of Gaussian errors (Schoups and Vrugt, 2010). However inappropriate assumptions regarding the independence and variance of residuals may be violated in hydrological modelling, with implications for parameter identification and the estimation of predictive uncertainty (Kavetski et al., 2006; Kavetski et al., 2003; Thyer et al., 2009). This study employs the generalised likelihood (GL) of Schoups and Vrugt (2010), which enables the use of a formal Bayesian approach in the presence of non-stationary, non-Gaussian errors. The GL

### Table 1

Parameters and prior ranges used in calibration with total and wet data sets. Note Zhang model prior ranges were varied during the analysis to further explore parameter identifiability.

<table>
<thead>
<tr>
<th>Model/parameters</th>
<th>Symbol</th>
<th>Min</th>
<th>Max</th>
<th>Units</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Linear</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Gradient</td>
<td>$m$</td>
<td>0</td>
<td>1</td>
<td>None</td>
<td>Ranges based on preliminary regression analysis</td>
</tr>
<tr>
<td>Intercept</td>
<td>$c$</td>
<td>$-1000$</td>
<td>200</td>
<td>mm/year</td>
<td></td>
</tr>
<tr>
<td><strong>Midgley &amp; Pitman</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Coefficient</td>
<td>$\beta$</td>
<td>$10^{-5}$</td>
<td>$10^{-1}$</td>
<td>None</td>
<td>Based on preliminary regression analysis</td>
</tr>
<tr>
<td>Exponent</td>
<td>$\gamma$</td>
<td>1</td>
<td>3</td>
<td>None</td>
<td></td>
</tr>
<tr>
<td><strong>Hyperbolic tangent</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Notional loss</td>
<td>$L$</td>
<td>0</td>
<td>1000</td>
<td>mm/year</td>
<td></td>
</tr>
<tr>
<td>Notional infiltration</td>
<td>$F$</td>
<td>$-500$</td>
<td>1000</td>
<td>mm/year</td>
<td>Required for “wet” calibration</td>
</tr>
<tr>
<td><strong>USDA</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Abstraction ratio</td>
<td>$\lambda$</td>
<td>0</td>
<td>3</td>
<td>None</td>
<td>Based on USDA-NRCS (2004)</td>
</tr>
<tr>
<td>Curve number</td>
<td>$CN$</td>
<td>0</td>
<td>100</td>
<td>None</td>
<td></td>
</tr>
<tr>
<td><strong>Zhang</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Plant water coeff.</td>
<td>$\omega$</td>
<td>0</td>
<td>5</td>
<td>None</td>
<td>Based on Zhang et al. (2001): $\omega \in [0, 2]; \ E_{\omega} = [1000, 1410]$</td>
</tr>
<tr>
<td>Evapotranspiration</td>
<td>$E_{\omega}$</td>
<td>100</td>
<td>3000</td>
<td>mm/year</td>
<td></td>
</tr>
<tr>
<td><strong>Rainfall multiplier</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>$M$</td>
<td></td>
<td>0.5</td>
<td>1.5</td>
<td>None</td>
<td>Based on reasonable expectation</td>
</tr>
<tr>
<td><strong>Schoups &amp; Vrugt likelihood</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>– Intercept</td>
<td>$\sigma_2$</td>
<td>0</td>
<td>200</td>
<td>mm/year</td>
<td>Based on reasonable expectation</td>
</tr>
<tr>
<td>– Gradient</td>
<td>$\sigma_1$</td>
<td>0.0</td>
<td>1.0</td>
<td>None</td>
<td>Based on reasonable expectation</td>
</tr>
<tr>
<td>Kurtosis</td>
<td>$\beta$</td>
<td>$-1$</td>
<td>1</td>
<td>None</td>
<td>After Schoups and Vrugt (2010)</td>
</tr>
<tr>
<td>Skewness</td>
<td>$\psi$</td>
<td>0.1</td>
<td>10</td>
<td>None</td>
<td>After Schoups and Vrugt (2010)</td>
</tr>
</tbody>
</table>
uses an autoregressive model and a skew exponential power (SEP) distribution to model errors:

$$
\Phi_p(B)e_t = \sigma_a a_t \sim \text{SEP}(0, 1, \xi, \beta)
$$

where $\Phi_p(B)$ is an autoregressive polynomial with $p$ parameters; $B$ is the backshift operator; $\sigma_a$ is the standard deviation at time $t$ and $a_t$ is an independent identically distributed random error with zero mean unit standard deviation described by an SEP density (Schoups and Vrugt, 2010).

Non-constant variance can be addressed using Box-Cox transformations or by modelling it explicitly using a linear relationship with modelled output:

$$
\sigma_a = \sigma_0 + \sigma_1 y_t
$$

where $y_t$ is model response at time $t$ and $\sigma_0$ and $\sigma_1$ are determined during the calibration. If autoregressive dependence can be ignored, Eqs. (22) and (23) simply reduce to:

$$
e_t = (\sigma_0 + \sigma_1 y_t)a_t
$$

The SEP$(0, l, \xi, \beta)$ probability density function (pdf) in Eq. (22) is written:

$$
p(a_t | \xi, \beta) = \frac{2\sigma^2 \xi}{\xi + \xi + \beta} \exp\left(-c_0 \xi | a_t |^{2(1+\beta)}\right)
$$

where $a_t = \xi^{-\text{sign}(\mu, \sigma, a_t)}(\mu + \sigma a_t)$ and $\xi$ and $\beta$ are skewness and kurtosis parameters respectively. Other parameters $\sigma_0$, $\omega_1$, $\omega_2$, $\mu_1$ are computed as a function of $\xi$ and $\beta$ as per Schoups and Vrugt (2010, Appendix A). The pdf in Eq. (25) is symmetric for $\xi = 1$ and positively (negatively) skewed for $\xi > 1(\xi < 1)$, while kurtosis is Gaussian (bell-shaped) for $\beta = 0$ and peaked/double exponential/Laplace (uniform/flat) for $\beta = 1(\beta = -1)$. Consequently SEP$(0, 1, \xi = 1, \beta = 0)$ corresponds to a standard normal Gaussian density.

The temporal independence of the mean annual data in this study suggested that autoregressive modelling was not necessary, leaving errors $e$ defined according to Eq. (24).

5.3. Uncertainty

The recognition of uncertainty in data has been a central theme in recent hydrological modelling studies (Koskela et al., 2012; McMillan et al., 2011; Renard et al., 2010; Renard et al., 2011; Thyer et al., 2009). Efforts to characterise all sources of uncertainty in Bayesian model calibration have tended to use event-based rainfall multipliers estimated as latent variables during the calibration (Kavetski et al., 2003; Kuczera et al., 2006). Independent, multiplicative errors have been found useful where minimum intensity...
thresholds are respected, however assumptions of independence can be violated, particularly where the definition of an event is not straightforward (McMillan et al., 2011). Furthermore, the use of uninformative prior distributions on multiplicative errors can have consequences for parameter identification when attempting to explicitly account for structural errors (Kavetski et al., 2006; Koskela et al., 2012; Kuczera et al., 2010; Renard et al., 2010), although others have found success with sophisticated MCMC algorithms (Vrugt et al., 2009) and/or approaches which quantify input errors alone (Vrugt et al., 2008).

Multiple rainfall multipliers were calibrated along with model parameters (see Kavetski et al., 2006; Kavetski et al., 2003; Vrugt et al., 2008; Vrugt et al., 2009) across different rainfall ranges, but without any improvement in the results returned using a single multiplier. Consequently input uncertainty was quantified as a single rainfall multiplier for each model, which was considered consistent with the simple model structures, likelihood and the independent character of the data, and no further attempt was made to decompose uncertainty.

\[
y_i = M(x_i | \theta_j) + \epsilon_i, \quad i = 1, 2, \ldots, 200
\]
\[
j = 1, 2, \ldots, 5000
\]

Predictive uncertainty was estimated by re-calibrating the models to a 200 point, “wet” subset of the data (see Fig. 10) and re-running the models (both with and without rainfall uncertainty) in a Monte Carlo framework.

\[
\begin{align*}
\sigma_0 & (\text{mm/year}) \\
\sigma_1 & \\
\beta & \\
\xi & \\
\end{align*}
\]

Fig. 3. Marginal distributions for generalised likelihood statistical parameters.
Carlo simulation using the remaining 113 point “dry” rainfall subset and a sample of 5000 parameter sets from the posterior distribution generated by the calibration data subset (see Mroczkowski et al., 1997; Schoups and Vrugt, 2010). The procedure relies on being able to generate random response errors as per Eq. (26) (see earlier discussion of generalised likelihood in Schoups and Vrugt, 2010).

If the models are run without rainfall uncertainty, then the 100 and 100(1−x) percentiles define the 100(1−2x)% confidence limits and rainfall uncertainty. If random errors are added to the output of the rainfall multiplier models, the stipulated percentiles define the 100(1−2x)% prediction limits, encompassing uncertainty due to parameters, the data and the model itself (Eq. (26)).

If a model is sound and the validation data are consistent with those used in the calibration, then 100(1−x)% of a ‘large’ validation dataset should fall within the 100(1−2x)% prediction intervals (Hahn and Meeker, 1991), although some authors have...
Fig. 6. Results of performance criteria tests.

Fig. 7. Joint posterior probability distributions of model parameters. All show eccentricity associated with parameter correlation. A positive inclination or slope reflects positive parameter correlation and vice versa. Light grey lines are burn-in data, dark grey circles post-burn-in equilibrium. Light grey lines are burn-in data, dark grey circles post-burn-in equilibrium. Chain lengths for (a) to (d) are $1 \times 10^5$ and $2 \times 10^5$ for (e).
adopted a less prescriptive interpretation, rejecting a model hypothesis if a significant fraction of the data fall outside prediction limits (Mroczkowski et al., 1997).

6. Results

The 313 point rainfall–runoff dataset were first explored using OLS regression (Fig. 2a, \texttt{lm(stats)} in R, R Core Team, 2012). The linear relationship appears to fit data reasonably well $R^2 = 0.79$, although a curvilinear relationship is evident in low rainfall data, matched by bias in the residuals (<100 mm/year runoff in Fig. 2a). The heteroscedastic residuals (Fig. 2b) and their observed density (Fig. 2c) indicate the assumptions of a Gaussian error model are clearly violated. The independence of the (mean annual) data is confirmed by the insignificant autocorrelation (Fig. 2d).

All models were then calibrated using the 313 point dataset, the SEP generalised likelihood and 10$^5$ iteration chains, without introducing rainfall uncertainty. The Zhang model gave indeterminate results using a 1$^5$ chain, but provided consistent parameter estimation when it was doubled in length (2$^5$ iterations).

The integrity of the calibration is reflected in the well identified statistical parameters of the generalised likelihood, with symmetrical, marginal posterior distributions (Fig. 3). Heteroscedasticity was corrected as is evident in the standardised residuals from all models (Fig. 4a), while the close agreement between observed

Fig. 8. Marginal posterior probability distributions. All show good parameter identifiability independent of the prior distribution with the exception the Zhang model, which is controlled by the selection of prior (i–p).
and theoretical residual densities support the use of the generalised likelihood as a generic solution to the analysis of data with non-Gaussian errors (Fig. 4b).

6.1. Model veracity

Model veracity, the ability to provide meaningful information, was tested using the selected performance criteria to compare calibrated output with observed data. Quantile–quantile plots are shown in Fig. 5 (below) (qqplot.stats in R). The hyperbolic tangent model gave the strongest performance across the entire data distribution (Fig. 5c), as indicated by the close agreement between the 25:75 modelled percentile and the theoretical 1:1 trend. KS (ks.test(dfog)) and AD (adk.test(adk)) tests showed consistent results. Data generated by linear, MP and Zhang models were significantly different to the observed data (p < 0.05), while those from the hyperbolic tangent and USDA models were indistinguishable (Fig. 6a).

Results from DIC and BIC tests showed strong coherence due to their derivation using the same likelihood function, uniform priors and the same number of model parameters (Fig. 6b). The tanh model had lowest DIC/BIC (3499/3498), followed by the USDA (3502/3502) and Zhang (3511/3511) models. Thresholds proposed by Spiegelhalter et al. (2002) indicate that the linear and Midgley and Pitman (MP) models performed significantly worse than other candidates (ΔBIC, ΔDIC > 10), while the USDA model had less support than the tanh function (ΔBIC, ΔDIC > 3). Although the Zhang model had far greater support than the linear or MP models, it had essentially no support compared to the tanh function (ΔBIC, ΔDIC = 12, 13).

NSE results were strong (>0.70) for all models, underlining its general weakness in discriminatory power (Fig. 6c). Given its sensitivity to large data values, the high NSE returned by the Zhang model is consistent with a similar performance to the tanh and USDA models for high runoff data.

6.2. Model integrity

Model integrity, reflected in the ability to remain plausible when extracting information from the data, was evaluated using
the results of the Bayesian calibration. A highly plausible, two parameter model should generate a tight, circular joint posterior distribution, reflecting parameters that can be identified from the data with a very high level of confidence. Eccentricity or other symmetries evident in the posterior distribution indicate inter-parametric dependence or parameter correlation, reflecting structural weakness and resulting in difficulty in uniquely identifying parameters (see discussions in Sivia and Skilling, 2006 or similar texts). Pronounced symmetries point toward low model plausibility given the available data.

Joint posterior parameter distributions are shown in Fig. 7. All models exhibited strong parameter dependency. Greatest eccentricity was displayed by the MP and USDA distributions (Fig. 7b and d). Linear and hyperbolic tangent distributions were more oblate than other models, indicating better parameter identification (Fig. 7a and c). Both MP and Zhang joint distributions showed distinct non-linear dependencies (Fig. 7b and e). The Zhang model required a longer chain ($2 \times 10^5$) to reach equilibrium (converge) suggesting poorer parameter identifiability, while the posterior distribution was controlled by the upper range of the prior (Fig. 7e).

Model parameter identifiability can be evaluated from marginal posterior distributions (Fig. 8). Most models displayed good identifiability, as evident in the Gaussian symmetry although the MP model exhibited weak skew (Fig. 8c and d). The Zhang distributions for the original prior range ($\omega [0.5]$ Fig. 8m and n) were strongly skewed and controlled by the upper prior limit, as evident in Fig. 7e (above). The effect of prior ranges on the identification of $\omega$ was explored further by performing additional calibrations for a number of different ranges within and beyond the feasible range. In each case the form of the posterior was determined by prior knowledge (Fig. 8i–p), with a proportionately smaller estimate of $E_z$ as the upper limit of $\omega$ was increased, consistent with the negative correlation evident in the joint posterior distribution (Fig. 7e). The results indicate that the model is unable to identify both parameters using the available data without explicitly relying on prior information concerning $\omega$. While this is implicit in the development of the Zhang et al. (2001) model, it suggests limitations in the structure not evident in the other candidates.

The curves generated by the range of additional calibrated parameters (Fig. 8i–p) were plotted with the data and conventional Zhang grassland and forest curves for comparison in Fig. 9a. Performance criteria indicated a consistently better fit to regional Australian data as the plant-available water coefficient $\omega$ increased (Fig. 9b–d). However the optimal $\omega$ estimated by calibration ($\omega, E_z = [9.7, 585]$ Fig. 8a), is well outside the theoretical range
Fig. 11. Marginal distributions for generalised likelihood statistical parameters for models with latent variable rainfall multipliers.

Fig. 12. Latent variable rainfall multiplier marginal probability distributions. Greater compensation in the latent variable is evident in models with greater parameter interaction.
for plants posited by Zhang et al. (2001) \( \omega = [0.5, 2.0] \), creating a dilemma of plausibility.

Furthermore, Fig. 9a reveals that the standard Zhang curves are also inconsistent with Australian data. In dry areas (<1000 mm/year rainfall) the Zhang grassland curve is situated well above the body of the rainfall–runoff data, while the forest curve which may be expected to skirt along lower margin of the data cluster (corresponding to high evapotranspiration and therefore low runoff), forms a line of best fit through its centre, close to the calibrated curves. Inconsistencies with observed data and structural weakness reveal conceptual difficulties with the Zhang model which undermine its integrity.

6.2.1. Uncertainty

Uncertainty was assessed using a split sample validation. Split sample tests using similar data to the calibration can represent little more than interpolation (Mroczkowski et al., 1997). Consequently data were split to reflect wet and dry regions, or more correctly high and low runoff sites, as wet data extend well into the dry rainfall range (Fig. 10).

Models modified with a rainfall multiplier were calibrated against the wet (high runoff) data. The added dimension required longer chains to reach equilibrium and so were extended to \( 2 \times 10^5 \) iterations, \( 5 \times 10^5 \) in the case of the Zhang model. Half were discarded as burn-in. With the exception of the MP model, all marginal statistical parameter distributions for the generalised likelihood were again close to symmetrical (Fig. 11).

Marginal rainfall multiplier distributions for the Zhang and tanh models showed modal estimates close to unity (Fig. 12c and e below). The MP rainfall multiplier reduced rainfall by approximately half (Fig. 12b). Such a severe correction suggests the addition of the latent variable degraded the model’s integrity rather than produced a reasonable description of input data error. The additional dimension exacerbated the strong parameter dependence evident in the eccentricity of the posterior distribution in Fig. 7b. A similar but less pronounced effect was produced by the USDA model.

Fig. 13. Joint posterior probability distributions of model parameters, with the inclusion of a latent variable rainfall multiplier. Light grey lines are burn-in data, dark grey circles post-burn-in equilibrium. Chain lengths for (a) to (d) are \( 2 \times 10^5 \), and \( 5 \times 10^5 \) for (e).
multiplier, where rainfall was reduced by 20% (Fig. 12d). The rainfall multiplier of the linear model showed pronounced positive compensation, controlled by the upper limit of the prior range [0.5,1.5] (Fig. 12a). Unique estimation may have been achieved by extending the prior range, but at the cost of unfeasible compensation.

Joint posterior distributions of the model parameters are shown in Fig. 13 (below). In contrast to those in Fig. 7, they are broader, indicating weaker parameter identifiability, as expected from the increase in dimensions, crudeness of the error model and possibly the uniform priors (Renard et al., 2010; Thyer et al., 2009). The prior range on the tanh \( L \) parameter was extended (from [0, 1000] to \([−500, 1000]\) mm/year, see Table 1) to obtain consistent estimation (Fig. 13c). While physical interpretation of loss parameters in simple models can only be regarded as indicative, negative losses are consistent with modelling anomalously high runoff levels represented by the wet calibration dataset. The symmetry of the tanh posterior distribution was otherwise the most conducive to unique parameter identification, showing regularly contoured probability intervals around the mode (Fig. 13c).

The addition of the rainfall multiplier apparently corrected the dependence of the Zhang model on its prior range as indicated by the position of maximum likelihood parameter estimates (Fig. 13e). However, non-linear dependencies remain and optimal parameters are weakly defined, lying within a broad mass of confused probability contours. Both \( \omega \) and \( E_c \) lie within 95% credible intervals spanning an order of magnitude, while the highest probability estimate of \( \omega = 3.4 \) is again outside the conceptual range for plants posited by its developers (\( \omega = [0.5, 2.0]\)).

The MP model showed particularly high, nonlinear parameter interaction and very poor identifiability on \( \beta \) (Fig. 13b). High levels of parameter dependence were also evident in the USDA model, but identifiability was superior to that of the MP model (Fig. 13d). The linear model showed a well-defined estimate of the intercept \( C \), but a broad range of viable gradient estimates within \( m = [0.4, 0.8] \) (Fig. 13a).

The use of a uniform prior distribution in an MCMC analysis means that the likelihood is directly proportional to the posterior distribution and may be used to draw inference in much the same way. Figs. 14 and 15 reveal the impact of the addition of the rainfall

---

**Fig. 14.** Model parameter marginal likelihood distributions.

**Fig. 15.** Model parameter marginal likelihood distributions with the inclusion of a latent variable rainfall multiplier.
multiplier on the marginal likelihoods. The loss of parameter identifiability in all models is marked. Only the $C$ parameter of the linear model retains a symmetrical form (compare Figs. 14 and 15b), demonstrating its independence from the gradient parameter $m$, and the structural strength of the relationship. However the addition of the multiplier $M$ (as $y = mM + C$) obliterated the identifiability of $m$ (Fig. 15a). For a well-identified parameter $C$, any estimate of $m$ would provide as good a description of the data as any other in the presence of the rainfall uncertainty model. Impacts on the MP and USDA models were not as severe, but nonetheless became heavily skewed, suggesting high levels of interaction between the latent variable and model parameters (Fig. 15c and d, and g and h).

The hyperbolic tangent retained the best identifiability on both parameters, suggesting a structure more robust to multiplicative rainfall uncertainty estimation than other models (Figs. 14 and 15e and f). In contrast, the Zhang model suffered the heaviest. Despite the encouraging independence to prior distributions apparent in Figs. 13e, 15i and j show how identifiability has been completely lost in the modified Zhang model. The maximum likelihood estimate of $E_t$ is very weakly defined around a modest (possibly local) maxima between 600 and 700 mm/year, for any value of $\alpha$. The severity of the impact probably reflects the already limited ability of the Zhang model to extract information from the data, readily compounded by additional structural dependencies introduced by the latent variable.

Total predictive uncertainty is shown in Fig. 16, along with intervals corresponding to uncertainty due to the parameters; uncertainty due to rainfall and the parameters and the calibration and validation data. Confidence limits - uncertainty due to parameters alone, should represent the smallest interval, followed by a broader envelope corresponding to uncertainty in both parameters and input data, followed by a yet broader interval encompassing parameters, input and model uncertainty (negating uncertainty associated with output data). The intervals in Fig. 16 followed this general pattern but the inclusion of the latent rainfall multiplier added very little uncertainty to the confidence limits. Negating output uncertainty, most predictive uncertainty could be attributed to the models themselves, an expected consequence of fitting curves to noisy data whose variance is driven by a range of catchment-scale influences not taken into account by simple two-parameter model structures (see discussion in Budyko, 1974, chapter VI).

If the models were sound, 5% of the validation data (or 6 of the 113 points) would fall outside of the 95% prediction limits. Table 2 shows how far this is from the case. None of the candidate models were able to adequately predict runoff responses in dry conditions when calibrated in wetter conditions.

Table 2

<table>
<thead>
<tr>
<th>Model</th>
<th>&gt;97.5%</th>
<th>&lt;2.5%</th>
<th>Total</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Linear</td>
<td>0</td>
<td>21</td>
<td>22</td>
<td>19</td>
</tr>
<tr>
<td>Midgley &amp; Pitman</td>
<td>0</td>
<td>102</td>
<td>102</td>
<td>90</td>
</tr>
<tr>
<td>Hyperbolic tangent</td>
<td>0</td>
<td>18</td>
<td>18</td>
<td>16</td>
</tr>
<tr>
<td>USDA</td>
<td>0</td>
<td>19</td>
<td>19</td>
<td>17</td>
</tr>
<tr>
<td>Zhang</td>
<td>0</td>
<td>49</td>
<td>49</td>
<td>43</td>
</tr>
</tbody>
</table>

Fig. 16. Uncertainty intervals for all models. Strongest predictors of dry data given a calibration using wet are the linear, tanh and USDA models. Midgley and Pitman and Zhang models are weakest, particularly in very dry areas.

Most of the areas considered prospective for forestry expansion in Australia lie in areas with <1000 mm/year rainfall (Keenan et al.,...
Given its importance to Australian forest hydrology assessment, the integrity of the Zhang model was examined further by recalibrating it using both wet and dry datasets, with and without a rainfall multiplier (Fig. 17). Notwithstanding the non-linear, banana shaped joint probability distribution, Fig. 17a shows a modest but well-defined ridge of high probability around the modal values, suggesting improved parameter estimation when using wet data and independence from the prior distribution. In contrast Fig. 16b shows control by the prior when using dry data (similar to results obtained when calibrating against the entire dataset in Fig. 7e).

Fig. 17c re-presents Fig. 13e showing the results of calibration using the wet dataset with the marginal posterior distribution of the rainfall multiplier inset (see Fig. 12e). Calibration against the dry dataset exacerbated the impact of the rainfall multiplier, obliterating remnant parameter identifiability shown in Fig. 15i and j (Fig. 17d). Parameter estimation was not only determined by the choice of the prior range on $\omega$ but on the range on the rainfall multiplier, revealing a conspicuous insensitivity to low runoff data.

**7. Discussion**

Sustainable water resources management aspires to account for a widening pool of legitimate users in an environment of changing water security due to competing demand and/or reduced availability under a changing climate (NWC, 2004). Increasing competition requires robust accounting systems (Young and McColl, 2009), underpinned by quality assured modelling (Black et al., 2011; Jakeman et al., 2006 among others). However, organisational considerations in water management result in assessment approaches which favour functional adequacy that may fall short of research aspirations (Etchells and Malano, 2005; Gupta et al., 2012).

In recent times researchers have advocated the comparative evaluation of multiple hypothesis in the systematic development of hydrological models (Clark et al., 2011; Clark et al., 2008; Gupta et al., 2012; Sivapalan et al., 2003; Son and Sivapalan, 2007) and the use of more sophisticated performance criteria in their selection and optimisation (Gupta et al., 2009; Gupta et al., 2008; Laio, 2004; Laio et al., 2009). But recent experience in Australia reveals
how the scope of even well-resourced studies can be constrained by the practical realities of water management. Assessments undertaken to support the high-profile Murray-Darling Basin (MDB) Plan (MDBA, 2011) were limited to models developed by water agencies to ensure work linked with existing systems (Pogger et al., 2010; Van Dijk et al., 2008). When wider scrutiny was possible through a need to complete additional modelling, the range of potential candidates was also distinctly limited (Chiew et al., 2008). The entire programme of investigation was heavily reliant on MSE criteria, popular but weak indicators of hydrological performance (Chiew et al., 2008; Post et al., 2012; Van Dijk et al., 2008). Beyond pragmatic constraints induced by management, the approach also reflects the little discussed tendency of scientists to adopt techniques in which they have confidence, arising from personal experience or associations with respected professionals and organizations, rather than the objective consideration of viable alternatives (ASCE, 1985).

A similar mindset is evident in the continued use of Zhang curves in assessing the hydrological impacts of plantation forestry (Chiew et al., 2008; Post et al., 2012). While Zhang curves represent a pragmatic choice with wide-spread professional support from influential organisations (BRS, 2003), they have been subject to very little critical scrutiny against competing alternatives, particularly in dry conditions. Brown et al. (2005) found Zhang curves to be robust, but only by comparison with data dominated by mean annual rainfall > 1000 mm/year, not typical of areas prospective for forestry in Australia (Keenan et al., 2004). CSIRO (2007) found Zhang curves gave similar results to those based on the empirical hyperbolic tangent relationship of Greenwood and Cresswell (2007), but the contrasting method of application of each model made direct comparison meaningless.

Concerns regarding the suitability of Zhang curves for use in supporting forest developments were raised by water managers, but were not sufficiently influential to discourage their inappropriate use during the first stages of Australian forest water regulation (Greenwood, 2007, 2013). Recent research has now identified that Zhang curves are insensitive to data from low rainfall areas, consistent with model assumptions that do not address the radiative heat-water balance in dry conditions (Greenwood et al., 2011), a contention supported by the results of this study.

This discussion highlights two significant issues for water management: that operational considerations are a fundamental constraint in the application of research; and that even the most reputable scientific advice may carry hidden risks. The way forward involves finding the middle ground by either getting researchers more involved in policy development or upgrading the technical capacity of applied water scientists. The former is unrealistic (Sutherland et al., 2013) but moreover, lacks insight into the way water management works.

Resource management agencies commonly maintain teams of applied scientists to provide the expertise required to support decision-making. They provide an interface between research and policy on a day-to-day basis, giving them far deeper insights into water management than their colleagues in research and a clearer view of their organisation’s business needs. However, their roles typically limit their capacity for developing innovative approaches, leaving water agencies reliant on historical precedents (Etchells and Malano, 2005). Knowledge and capacity building have been recognised as key to implementing effective water management (NWC, 2004). The challenge to water management agencies is to ensure technicians have the skills required to identify the limitations of advice provided by researchers and the ability to lead scientific agendas and influence policy development.

In the light of the above, it is interesting to reflect that the two worst performing models evaluated in this study have received the most enduring support. Curves based on the MP model were used to underpin forest water management in South Africa for decades, although their structure was augmented by empirical observations (Van der Zel, 1995). Zhang curves swiftly became a feature of Australian forest hydrological assessment and later a tool used in the early stages of forest water regulation (Greenwood, 2013). The adoption of these models for forest water management did not feature a convincing evaluation of competing alternatives which would have no doubt proved efficacious in facilitating policy development and generating confidence in decision-making.

8. Conclusions

Simple rainfall–runoff models were subjected to scrutiny according to a philosophy which evaluated their veracity and integrity. Veracity, a model’s ability to provide meaningful information, was assessed by comparing their performance against observed data. Optimal parameter sets were identified in a simple Bayesian calibration using a generalised likelihood which addressed the problem of non-Gaussian errors. Performance criteria encompassed NSE, EDFs and information criteria. In contrast to NSE, the non-parameteric nature of the EDFs and use of the generalised likelihood in the information criteria provided methods which were not dependent on the underlying data or Gaussian error assumptions. EDFs consistently rejected the linear, M&P and Zhang models, which according to the information criteria had effectively no support compared to the tanh and USDA models. NSE could not reject any of the models and produced favourable support for the Zhang model which, given the results of other tests probably reflected NSE’s vulnerability to influence by large data values than strong model performance.

Integrity was assessed according to how well the models could extract information from the data using a Bayesian approach. Model (parameter) identifyability was assessed by scrutinising joint posterior distributions. All models showed strong parameter dependencies. The linear and hyperbolic tangent functions provided the best identifyability, while the USDA and M&P models featured extreme parameter interaction and weaker structure, indicating lower integrity.

The Zhang model exhibited non-linear parameter dependencies and the weakest structure, being too ill-posed to uniquely identify both parameters given the data. Prior information was required on $\omega$ in order to estimate $E_z$. The integrity of the Zhang model as a single parameter model (in accordance with its derivation) was also questionable, as optimal estimates of $E_z$ required setting $\omega$ values beyond the feasible range posited by developers.

Uncertainty was evaluated by introducing an additional multiplicative parameter into the models to explicitly account for uncertainty in the input data. The modified models were recalibrated using a high runoff (wet) data subset and validated against low runoff (dry) data. Recalibration using shorter data and an additional dimension resulted in poorer identifyability for all models. The hyperbolic tangent function retained the best identifyability of all models and generated a prediction envelope that had the greatest coherence with dry validation data. The Zhang model demonstrated weaker coherence and identifyability in dry conditions, consistent with an insensitivity to data from dry regions (see discussion by Greenwood et al., 2011).

Of the two models not rejected by the performance criteria the tanh model was the most the plausible under Australian conditions.

The analysis here is exhaustive, taking pains to put flesh on the bones of model selection processes articulated in a number of guidelines emerging from research. It demonstrates the focus provided by setting model evaluation in a philosophical context; the
364


benefits of explicitly addressing non-Gaussian assumptions and
the insights provided by Bayesian analyses. Operational realities
will inevitably constrain how much can be implemented in routine
water management assessments, particularly knowledge and
capacity constraints within agencies and more particularly in gaining competence in Bayesian analysis, where the literature has been
described as ‘‘a forbidding morass of extreme technicalities of
doubtful importance presented in impenetrable jargon’’ ([sic] Sivia
and Skilling, 2006) and providing ‘‘conflicting advice about the
most elementary aspects of running a Markov chain simulation’’

Acknowledgements
This study was funded by a University of Melbourne scholarship
made available through the Cooperative Research Centre for Forestry. I would like to thank Mr. Francesco Minunno of the Instituto
Superior de Agronomia, Portugal and Dr. Craig Beverly of the
Department of Environment and Primary Industries, Victoria, for
advice and input to an earlier phase of this study and Dr. Richard
Benyon, of the University of Melbourne for editorial and academic
support.

References
ASCE, 1985. Evaluation of hydrologic models used to quantify major land-use
Ascher, W., 1993. The ambiguous nature of forecasts in project evaluation:
diagnosing the over-optimism of rate-of-return analysis. Int. J. Forecast. 9 (1),
109–115.
Black, D. et al., 2011. Guidelines for Water Management Modelling: Towards BestPractice Model Application. eWater Cooperative Research Centre, Canberra,
Australia.
Springer, Berlin, Heidelberg.
of paired catchment studies for determining changes in water yield resulting
from alterations in vegetation. J. Hydrol. 310, 28–61.
Key Scientific Issues. A statement of agreed outcomes from a meeting of
scientists representing CSIRO, Murray Darling Basin Commission and CRC for
Catchment Hydrology, 24 October, Canberra.
Chiew, F.H.S. et al., 2008. Rainfall–runoff modelling across the Murray-Darling
Basin. A report to the Australian Government from the CSIRO Murray-Darling
Basin Sustainable Yields Project, CSIRO, Australia.
Clark, M.P. et al., 2008. Framework for understanding structural errors (FUSE): a
modular framework to diagnose differences between hydrological models.
Australian Government from the CSIRO Murray-Darling Basin Sustainable
Yields Project, CSIRO, Australia.
De Roo, A. et al., 2011. Quality control, validation and user feedback of the European
flood alert system (EFAS). Int. J. Digital Earth 4 (Suppl. 1), 77–90.
nonequilibrium in water flow with an effective approach. Water Resour. Res. 48
(3), W03503.
In: Zerger, A., Argent, R.M. (Eds.), MODSIM05, Advances and Applications for
Management and Decision Making, December, Melbourne, pp. 2484–2490.
conceptual hydrological modeling: 1. Motivation and theoretical development.

169.
in Statistical Science Series, second ed. Chapman & Hall/CRC, Boca Raton,
Florida, 668 pp.
Bernado, J.M., Berger, J.O., Dawid, A.P., Smith, A.F.M. (Eds.), Bayesian Statistics 5
– Fifth Valencia International Meeting on Bayesian Statistics, Alicante Spain,
Monte Carlo. In: Gilks, W.R., Richardson, S., Spiegelhalter, D.J. (Eds.), Markov
Richardson, S., Spiegelhalter, D.J. (Eds.), Markov Chain Monte Carlo in Practice.
function for infilling rainfall–runoff data, Hydrologic Recipes. Estimation
Techniques in Australian Hydrology. Cooperative Research Centre for
Hydrology, Monash University, Clayton, Victoria, pp. 81–83.
Evapotranspiration and Rainfall in the Assessment of the Impact of Plantation
Forestry on Runoff. DWLBC Technical Note 2007/14, Department of Water, Land
and Biodiversity Conservation, Government of South Australia, November.
the Mount Lofty Ranges, Case Study: Burnt Out Creek. DWLBC Technical Note
2007/11 Department of Water, Land and Biodiversity Conservation,
Government of South Australia, November.
Greenwood, A.J.B., 2013. The first stages of Australian forest water regulation:
hydrological impact of afforestation using regional mean annual data and
comprehensive assessment of model structural adequacy. Water Resour. Res. 48
(8), W08301.
Gupta, H.V., Kling, H., Yilmaz, K.K., Martinez, G.F., 2009. Decomposition of the mean
squared error and NSE performance criteria: implications for improving
Gupta, H.V., Wagener, T., Liu, Y., 2008. Reconciling theory with observations:
elements of a diagnostic approach to model evaluation. Hydrol. Process. 22 (18),
3802–3813.
Bernoulli 7 (2), 223–242.
Wiley & Sons, New York.
and evaluation of environmental models. Environ. Model. Softw. 21 (5), 602–
614.
Kavetski, D., Fenicia, F., Clark, M.P., 2011. Impact of temporal data resolution on
parameter inference and model identification in conceptual hydrological
modeling: insights from an experimental catchment. Water Resour. Res. 47
(5), 25.
Kavetski, D., Kuczera, G., Franks, S.W., 2006. Bayesian analysis of input uncertainty
Kavetski, D.N., Franks, S.W., Kuczera, G., 2003. Confronting input uncertainty in
rainfall–runoff modelling. In: Duan, Q., Gupta, H.V., Sorooshian, S., Rousseau,
A.N., Turcotte, R. (Eds.), Calibration of Watershed Models. American
Geophysical Union, Washington, DC, pp. 49–68.
the Forest and Wood Products, Research and Development Corporation, Project
No. PN04.4005, Bureau of Rural Sciences, Canberra.
inference of uncertainties in precipitation-streamflow modeling in a snow
affected catchment. Water Resour. Res. 48 (11), W11513.
Krause, P., Boyle, D.P., Base, F., 2005. Comparison of different efficiency criteria for
Water Resour. Res. 46 (7), W07516.
Kuczera, G., Kavetski, D., Franks, S., Thyer, M., 2006. Towards a Bayesian total error
analysis of conceptual rainfall–runoff models: characterising model error using
Kuczera, G., Renard, B., Thyer, M., Kavetski, D., 2010. There are no hydrological
monsters, just models and observations with large uncertainties! Hydrol. Sci. J./


APPENDIX F: OTHER PUBLICATIONS


Abstract

South Australia has recently become the second jurisdiction behind the Republic of South Africa to regulate plantation forest water use. State-wide regulation was enacted in accord with national water reforms, but stemmed from a series of earlier regional initiatives which were initially concerned with a need to account for the impacts of expanding plantation developments on water entitlements. The effects of severe drought changed the emphasis of the national policy agenda from water quality to water availability and highlighted the potential impacts of plantation forestry on water resources over its benefits for salinity management. The progress of South Australian forest water management was hindered by competing state-government agendas in some regions, leading to confusion among decision-makers and eventually prompting amendments to regional planning frameworks. The integrity of models finally adopted to underpin regulation reflects the effectiveness of management processes surrounding their development. Additional investigations are likely to be required to augment the limited knowledge base and ensure plantation forestry water accounting systems are able to maintain the security of established water access entitlements at an acceptable level of risk. A genuinely national research agenda in water forest management is required to underpin the implementation of national water reforms.
Author/s: Greenwood, A. J.
Title: Assessing hydrological interception by plantation forestry for application in water resources management
Date: 2017
Persistent Link: http://hdl.handle.net/11343/124187
File Description: Assessing hydrological interception by plantation forestry for application in water resources management
Terms and Conditions: Copyright in works deposited in Minerva Access is retained by the copyright owner. The work may not be altered without permission from the copyright owner. Readers may only download, print and save electronic copies of whole works for their own personal non-commercial use. Any use that exceeds these limits requires permission from the copyright owner. Attribution is essential when quoting or paraphrasing from these works.