The role of water quality modelling in decision-making

by

Les McNamara

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Dedication

I wish to dedicate this thesis to my mother, Olga McNamara, who passed away in May 2006. I love you and I miss you.
Declaration of Originality

I declare that the work presented in this thesis is, to the best of my knowledge and belief, original and my own work, except as acknowledged in the text. The thesis has not been submitted, either in whole, or in part, for a degree at this or any other university.

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Les McNamara
Acknowledgments

The course of this research coincided with the most turbulent and confronting period of my life. Over the last four years there have been many high and low moments that changed the way I look at myself, the people around me, and the world itself. Most of these events had nothing to do with education in the proverbial sense of the word, and most had nothing at all to do with this research. I won’t elaborate on them here, but I mention their existence so that the reader can understand the nature of these acknowledgements. The support that I received came from many sources and in many forms and the gratitude that I want to convey is heart-felt and goes way beyond what can be expressed on paper.

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Countless others, knowingly or unknowingly, made helpful contributions. Thank you all for your insights and inputs.
ABSTRACT

Catchment management organisations often use computer-based water quality models to support their decision-making needs. However, literature suggests that model use by catchment managers can be highly problematical. Commonly reported issues that negatively impact upon the effectiveness of modelling for decision-making are data quality and availability, miscommunication between analysts and decision-makers, inappropriate treatment of uncertainty, excessive model complexity (‘sophistication’) or simplicity and poor modelling practice. The challenges that beset catchment managers are usually framed as technical problems that can be overcome by using or communicating the science that underlies the models more effectively. As a result, many of the problems associated with the use of models by catchment managers have not been adequately elucidated from the standpoint of a manager.

This thesis examines the problem of modelling for decision-making from a fresh perspective. Systemic approaches to research are commonly used where the research problem is ill-defined, as it is here. ‘Action research’ is one qualitative, systemic methodology, and was used here as the guiding methodology to explore the model-related problems faced by catchment managers. Action research involves recurrent cycles of planning, action, observation and reflection. The research was undertaken with the Sydney Catchment Authority (SCA), a New South Wales government agency responsible for water quality and catchment health in the drinking water catchments of Sydney, Australia. The SCA had expressed a particular need for tools to support the assessment of development applications with respect to potential impacts on water quality, and to prioritise rural land for remedial action to improve water quality in the drinking water catchments. The research had two broad aims that were:

1. to develop two models differing in sophistication, and to use them to prioritise nutrient pollution sources and calculate nutrient loads;

And, through the participatory development of the models,

2. to learn methodological lessons that catchment managers can apply to choose and use models more effectively.

In this research, action was in the form of five planned activities, including:

i. a review of relevant literature from diverse disciplines;
ii. the holding of two workshops; the first exclusively involving SCA managers to identify their modelling needs, and the second involving water quality scientists and modelers to discuss methods for meeting the modelling needs;

iii. the development and use of two export coefficient nutrient models;

iv. a focus group discussion involving key staff in the SCA; and

v. a review of published guidelines for good modelling practice in environmental management.

Note was also taken of statements or behaviour at numerous meetings and seminars, mostly with SCA staff, that were relevant to the research questions.

AIM 1 - MODEL DEVELOPMENT AND APPLICATION

At the first workshop, SCA managers expressed a need for a conceptually simple, spatially-based approach to estimating the export of nutrients - phosphorus (P) and nitrogen (N) - from hillslopes to streams. They wanted a model that could operate at a wide range of spatial and temporal scales and allow them to prioritise mitigation actions and determine whether or not new developments would meet regulatory requirements. At the second workshop, however, it became apparent that there is fundamental disagreement amongst scientists about the relative importance of the different processes that affect nutrient mobilisation and transport, and that the expressed needs of the catchment managers were incongruent with the current data and knowledge limitations.

Given the state of the relevant science and the paucity of data at fine scales, it was determined that an export coefficient (EC) modelling approach was the most credible option for estimating potential nutrient loads delivered to streams. Modelling occurred in two stages. In stage one, EC values from studies in south-east Australia were reviewed and a set of locally-relevant EC values were derived. In stage two, the EC approach was enhanced by developing a new, simple but potentially useful method for hydrologically weighting ECs based on estimates of ‘streamflow contribution’ using a geographical information system. This method sought to overcome a major weakness of EC models, i.e., that EC values do not reflect hydrologic variations that affect nutrient exports, and concurrently meet the SCA’s need for a model that is sensitive to variations in nutrient export at finer resolutions.
Stage 1: P and N exports were estimated using the first EC model. The results for a single sub-catchment, Wingecarribee River, were compared to calculations based on water quality and flow data (Olley and Deere, 2003). The EC model estimated annual P and N loads of 29 tonnes per year (t/yr) and 243 t/yr respectively compared to calculated loads in the river of 21 t/yr and 180 t/yr.

For the entire drinking water catchment, modelled estimates showed that:

- most grazing land is a weak source of N and P per unit area, but by virtue of its extent accounts for most of the N and P generated in the catchment;
- better management of degraded pastures may provide an opportunity for significant improvement in water quality in some sub-catchments, but further studies are needed to confirm the ECs used in this study;
- dairy farms occupy a small proportion of the catchment area and their pastures are a relatively small generator of total nutrient loads, but they may be a significant source of nutrients in the Kangaroo River and Wingecarribee River sub-catchments if there are inadequate means of attenuating concentrations or flows between the farm boundary and streams;
- vegetable farms occupy a very small proportion of the total catchment area and modelling indicates that they are not a major contributor to total catchment nutrients loads, except for Werri Berri Creek sub-catchment, where they may generate one third of the nutrients. They pose a local threat to water quality if farm dams or other management features are not present to reduce delivery to streams; and
- modelled nutrient losses from urban areas are higher than from agricultural areas in the Blue Mountains and Lower Cox’s River sub-catchments.

Hydrological sub-catchments were prioritised for attention through the combined assessment of total modelled loads of P (t P/yr) and load per unit area (kg P/ha/yr). On this basis, Mulwaree River (43 t P/yr, 0.55 kg P/ha/yr) and Reedy Creek (40 t P/yr, 0.7 kg P/ha/yr) sub-catchments rank high for attention, whereas Wollondilly River ranks low because of low per unit area loads (0.22 kg P/ha/yr) despite this sub-catchment delivering the highest load overall (60 t P/yr).

Stage 2: In the literature, variation of runoff depth was shown to be a major contributor to variation in the loads of nutrients from pastures. Therefore, it was
considered that a significant improvement to the EC modelling approach would be to ‘weight’ EC values based on estimated variation in runoff using a simple approach requiring only commonly available data. In the second stage of modelling, estimates of stream flow contribution were used as a predictor (surrogate) of runoff depth and used to weight the EC values used in stage 1 (for P only) on a ‘pixel-by-pixel’ basis. When applied across the entire catchment, the hydrologically-enhanced model increased estimated P exports in humid sub-catchments (e.g., Kangaroo River, Mid Cox’s River and Upper Nepean River) and lowered P exports in drier catchments (e.g., Mulwaree River, Upper Wollondilly River and Nerrimunga River). The approach also highlighted potential differences in P generation within land-use types and sub-catchments.

**AIM 2 – CHOOSING AND USING MODELS MORE EFFECTIVELY**

At the focus group discussion that followed the modelling, it was observed that there was much variation in the way water quality issues and potential solutions were conceived by different actors. Some participants expressed strong aversion to the use of uncertain data and what they perceived to be subjective assumptions in both models. At the same time, they held a firm desire for more sophisticated models that placed much higher demands on data and knowledge. Despite criticisms, there was reluctant acknowledgement by most participants that simple models, including the simplest EC model described here, would be used in the immediate future, until better knowledge and data are available.

From the entirety of the research, including the review of literature, six issues critical to the effective implementation of modelling for decision-making were identified. Each is critical to the effective implementation of models and yet all receive scant treatment in most existing guidelines of good modelling practice:

1. **The rightful role of models and modelling in decision-making:** When selecting or evaluating models, the models should be assessed against criteria of *relevance* to the decision-making problem, and *impact* on the decision-making process. In some cases this may demonstrate the futility rather than utility of model use.

2. **Methodological tension:** Most guidelines to the selection and application of models offer no substantive support for organisations confronting challenges related to different ways that individuals and organisations perceive problems and potential
solutions. Epistemological and methodological tension amongst individuals within organisations, departments and disciplines may inhibit effective model use more than the problem of managing divides between researchers and managers.

3. Treatment of uncertainty: In general, analysts that run models should not be expected to go beyond the available data to make inferences and predictions. Scientists are subject to the same problems as lay people when they draw inferences.

4. Information generation: Information gathering is a relatively small part of the decision process and catchment managers should avoid undue time generating information. Given uncertainty in data as well as knowledge of the fundamental processes that result in nutrient runoff, information gathering may amount to little more than ‘opinion gathering’.

5. Information transformation: Before embarking on a modelling project, decision-makers need to determine what constitutes ‘evidence’ for given decision-making problems.

6. The roles and responsibilities of actors: Politics and the vested interests of different actors frequently hinder decision-making. The catchment manager should actively manage a quality assurance process for a modelling activity that is cognisant of vested interest and focused on problem resolution.

In the concluding discussion, it is proposed that resolution of these problems may be possible through the use of guidelines or ‘standard operating procedures’ that are sensitive to the sociological and organisational issues mentioned above. Large catchment management organisations may also benefit from active management of horizontal ‘boundaries’ (e.g., the transfer of information and ideas between different departments) and vertical ‘boundaries’ (e.g., to and from executive-level management) within the organisation.
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<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>ACT</td>
<td>Australian Capital Territory</td>
</tr>
<tr>
<td>aka</td>
<td>also known as</td>
</tr>
<tr>
<td>ANZECC</td>
<td>Australian and New Zealand Environment and Conservation Council</td>
</tr>
<tr>
<td>ARMCAAN</td>
<td>Agriculture and Resource Management Council of Australia and New Zealand.</td>
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<tr>
<td>ARC</td>
<td>Australian Research Council</td>
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<tr>
<td>ASAE</td>
<td>American Society of Agricultural Engineers</td>
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<td>ASTM</td>
<td>American Society for Testing and Materials</td>
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<td>AWT</td>
<td>Australian Water Technologies</td>
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<tr>
<td>BC</td>
<td>British Columbia</td>
</tr>
<tr>
<td>BDMF</td>
<td>Bay-Delta Modeling Forum</td>
</tr>
<tr>
<td>BIH</td>
<td>British Institute of Hydrology</td>
</tr>
<tr>
<td>BMW</td>
<td>Benchmark models for the Water Framework Directive</td>
</tr>
<tr>
<td>CAMASE</td>
<td>Concerted Action for the development and testing of quantitative Methods for research on Agricultural Systems and the Environment</td>
</tr>
<tr>
<td>CECIL</td>
<td>Contaminant Export for Catchments Indexed by Land-use</td>
</tr>
<tr>
<td>CEO</td>
<td>Chief Executive Officer</td>
</tr>
<tr>
<td>CMSS</td>
<td>Catchment Management Support System</td>
</tr>
<tr>
<td>COMPS</td>
<td>Catchment Operations and Major Projects Section (SCA)</td>
</tr>
<tr>
<td>conc.</td>
<td>concentration</td>
</tr>
<tr>
<td>CRC</td>
<td>Co-operative Research Centre for Catchment Hydrology</td>
</tr>
<tr>
<td>CRCCCH</td>
<td>Co-operative Research Centre for Catchment Hydrology</td>
</tr>
<tr>
<td>CSIRO</td>
<td>Commonwealth Scientific and Industrial Research</td>
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<tr>
<td>DA</td>
<td>Organisation</td>
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<tr>
<td>DEM</td>
<td>development application</td>
</tr>
<tr>
<td>DHI</td>
<td>Danish Hydraulic Institute</td>
</tr>
<tr>
<td>DIPNR</td>
<td>Department of Infrastructure, Planning and Natural Resources</td>
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<tr>
<td>DLWC</td>
<td>Department of Land and Water Conservation (NSW)</td>
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<tr>
<td>DoP</td>
<td>Department of Planning</td>
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<tr>
<td>DSS</td>
<td>decision support system</td>
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<tr>
<td>E&amp;P</td>
<td>Environment and Planning section (SCA)</td>
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<tr>
<td>EASI</td>
<td>Environmental Assessment of Sites and Infrastructure</td>
</tr>
<tr>
<td>EC</td>
<td>export coefficient</td>
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<tr>
<td>EMC</td>
<td>event mean concentration</td>
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<td>EPA</td>
<td>Environment Protection Authority</td>
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<tr>
<td>ESRI</td>
<td>Environmental Systems Research Institute</td>
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<td>est.</td>
<td>estimate</td>
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<tr>
<td>EU</td>
<td>European Union</td>
</tr>
<tr>
<td>GAO</td>
<td>General Accounting Office (now known as Government Accountability Office)   [USA]</td>
</tr>
<tr>
<td>GIS</td>
<td>Geographical Information System</td>
</tr>
<tr>
<td>GFW</td>
<td>Generic Framework Water</td>
</tr>
<tr>
<td>GM</td>
<td>General Manager</td>
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<tr>
<td>GMP</td>
<td>Good Modelling Practice</td>
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<td>ha</td>
<td>hectare</td>
</tr>
<tr>
<td>HCP</td>
<td>Healthy Catchments Program</td>
</tr>
<tr>
<td>IIASA</td>
<td>International Institute for Applied Systems Analysis (Austria)</td>
</tr>
<tr>
<td>HSPF</td>
<td>Hydrological Simulation Program - FORTRAN</td>
</tr>
<tr>
<td>kg</td>
<td>kilograms</td>
</tr>
<tr>
<td>Abbreviation</td>
<td>Description</td>
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<tr>
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</tr>
<tr>
<td>Ksat</td>
<td>saturated hydraulic conductivity</td>
</tr>
<tr>
<td>L</td>
<td>litres</td>
</tr>
<tr>
<td>LEP</td>
<td>Local Environment Plan</td>
</tr>
<tr>
<td>LWA</td>
<td>Land and Water Australia</td>
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<tr>
<td>M</td>
<td>metres</td>
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<td>mg</td>
<td>milligrams</td>
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<tr>
<td>mm</td>
<td>millimetres</td>
</tr>
<tr>
<td>MDBC</td>
<td>Murray-Darling Basin Commission</td>
</tr>
<tr>
<td>MNP</td>
<td>Netherlands Environmental Assessment Agency [Dutch: Milieu en Natuur Planbureau]</td>
</tr>
<tr>
<td>MoST</td>
<td>MOdelling Support Tool</td>
</tr>
<tr>
<td>N</td>
<td>nitrogen</td>
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<tr>
<td>NEXSYS</td>
<td>Nutrient EXpert SYStem</td>
</tr>
<tr>
<td>NOAA</td>
<td>National Oceanic and Atmospheric Administration [USA]</td>
</tr>
<tr>
<td>NorBE</td>
<td>Neutral or Beneficial Effect</td>
</tr>
<tr>
<td>NLWRA</td>
<td>National Land and Water Resources Audit</td>
</tr>
<tr>
<td>NPI</td>
<td>National Pollutant Inventory</td>
</tr>
<tr>
<td>NRC</td>
<td>National Research Council</td>
</tr>
<tr>
<td>NRCS</td>
<td>Natural Resources Conservation Service</td>
</tr>
<tr>
<td>NSW</td>
<td>New South Wales</td>
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<tr>
<td>NUSAP</td>
<td>Numeral Unit Spread Assessment Pedigree</td>
</tr>
<tr>
<td>P</td>
<td>phosphorus</td>
</tr>
<tr>
<td>PRIMA</td>
<td>Pluralistic fRamework of Integrated uncertainty Management and risk Analysis</td>
</tr>
<tr>
<td>QA</td>
<td>Quality Assurance</td>
</tr>
<tr>
<td>QDPI</td>
<td>Queensland Department of Primary Industries</td>
</tr>
<tr>
<td>RAP</td>
<td>Rectification Action Plan</td>
</tr>
<tr>
<td>REP</td>
<td>Regional Environmental Plan</td>
</tr>
<tr>
<td>RIVM</td>
<td>National Institute for Public Health and the Environment [Dutch: Rijksinstituut voor Volksgezondheid en Milieu]</td>
</tr>
<tr>
<td>SCA</td>
<td>Sydney Catchment Authority</td>
</tr>
<tr>
<td>SedNet</td>
<td>SEDiment River NETwork model</td>
</tr>
<tr>
<td>SEPP</td>
<td>State Environmental Planning Policy</td>
</tr>
<tr>
<td>SGS</td>
<td>Sustainable Grazing Systems</td>
</tr>
<tr>
<td>SLWCA</td>
<td>Strategic Land and Water Capability Assessment</td>
</tr>
<tr>
<td>SOP</td>
<td>Standard Operating Procedure</td>
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<tr>
<td>STP</td>
<td>sewage treatment plant</td>
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<tr>
<td>t</td>
<td>tonnes</td>
</tr>
<tr>
<td>TMDL</td>
<td>Total Maximum Daily Load</td>
</tr>
<tr>
<td>USDA</td>
<td>United States Department of Agriculture</td>
</tr>
<tr>
<td>US and USA</td>
<td>United States of America</td>
</tr>
<tr>
<td>USEPA</td>
<td>Environmental Protection Agency [USA]</td>
</tr>
<tr>
<td>USLE</td>
<td>Universal Soil Loss Equation</td>
</tr>
<tr>
<td>UWS</td>
<td>University of Western Sydney</td>
</tr>
<tr>
<td>VSA</td>
<td>variable source area</td>
</tr>
<tr>
<td>WA</td>
<td>Western Australia</td>
</tr>
<tr>
<td>WADE</td>
<td>Western Australia Department of the Environment</td>
</tr>
<tr>
<td>WAWRC</td>
<td>Western Australia Water and Rivers Commission</td>
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<tr>
<td>WFD</td>
<td>Water Framework Directive</td>
</tr>
<tr>
<td>WRFA</td>
<td>Water Research Foundation of Australia</td>
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<tr>
<td>yr</td>
<td>year</td>
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</table>
### Glossary

<table>
<thead>
<tr>
<th>Term</th>
<th>Definition</th>
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</thead>
<tbody>
<tr>
<td>Anthropogenic</td>
<td>Caused by human activity.</td>
</tr>
<tr>
<td>Anti-pattern</td>
<td>In software engineering, commonly reinvented problem solutions that are</td>
</tr>
<tr>
<td></td>
<td>ineffective or result in perverse outcomes and software design failures.</td>
</tr>
<tr>
<td>ArcView ©</td>
<td>A desktop GIS developed by ESRI (see <a href="http://www.esri.com/">http://www.esri.com/</a>).</td>
</tr>
<tr>
<td>Adaptive management</td>
<td>A systematic process for continually improving management policies by</td>
</tr>
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<td></td>
<td>learning from the outcomes of operational programs.</td>
</tr>
<tr>
<td>Black box model</td>
<td>A conceptual model of an input/output relationship that does not describe</td>
</tr>
<tr>
<td></td>
<td>the intervening processes.</td>
</tr>
<tr>
<td>Bootstrapping</td>
<td>In statistics, a method of estimating the sampling distribution based on</td>
</tr>
<tr>
<td></td>
<td>random resampling from observed data.</td>
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<tr>
<td>Brainstorm</td>
<td>A technique for generating a large number of creative ideas with a group</td>
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<tr>
<td></td>
<td>of people. Brainstorming involves creating an atmosphere in which people</td>
</tr>
<tr>
<td></td>
<td>feel uninhibited and free to propose solutions to problems (see Osborn,</td>
</tr>
<tr>
<td></td>
<td>1963).</td>
</tr>
<tr>
<td>Calibration</td>
<td>Adjustment of model parameters and algorithms so that model predictions</td>
</tr>
<tr>
<td></td>
<td>more closely match real-world observations.</td>
</tr>
<tr>
<td>Catchment</td>
<td>An area of land drained by a stream, lake or other water body. Also known</td>
</tr>
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<td></td>
<td>as a ‘watershed’. A large catchment may be divided into a number of smaller</td>
</tr>
<tr>
<td></td>
<td>‘sub-catchments’ that ultimately drain to a given point.</td>
</tr>
<tr>
<td>CECIL</td>
<td>A database of published work estimating contaminant exports from various</td>
</tr>
<tr>
<td></td>
<td>land uses (see AWT, 2001b).</td>
</tr>
<tr>
<td>Conceptual model</td>
<td>A model in which only the most salient processes are described. In a</td>
</tr>
<tr>
<td></td>
<td>conceptual model, several processes are often lumped into a single</td>
</tr>
<tr>
<td></td>
<td>expression.</td>
</tr>
<tr>
<td>Confidence limits</td>
<td>A numeric range based on a sample that is expected to include the</td>
</tr>
<tr>
<td></td>
<td>population mean value a specified proportion of the time (e.g., 95%).</td>
</tr>
<tr>
<td>Curve number</td>
<td>A hydrologic parameter describing the potential for storm water runoff</td>
</tr>
<tr>
<td></td>
<td>from an area of land.</td>
</tr>
<tr>
<td>Data</td>
<td>A group of measurements, facts or statistics that may be used as a basis</td>
</tr>
<tr>
<td></td>
<td>for reasoning, discussion or calculation.</td>
</tr>
<tr>
<td>Decision-maker</td>
<td>A person entrusted with the responsibility for allocating resources or</td>
</tr>
<tr>
<td></td>
<td>approving proposals.</td>
</tr>
<tr>
<td>Decision-making</td>
<td>The process of thought and action that leads to a decision.</td>
</tr>
<tr>
<td>Decision model</td>
<td>A model that is used as an aid to decision-making. Decision models</td>
</tr>
<tr>
<td></td>
<td>typically include a model describing the natural system that is the context</td>
</tr>
<tr>
<td></td>
<td>for the decisions to be made, a model of the decision maker’s objectives,</td>
</tr>
<tr>
<td></td>
<td>and a description of decisions that may be made to control or otherwise</td>
</tr>
<tr>
<td></td>
<td>exploit aspects of the system in order to achieve objectives.</td>
</tr>
<tr>
<td>Decision-space</td>
<td>The realistic range of options available to a decision-maker for a given</td>
</tr>
<tr>
<td></td>
<td>decision problem.</td>
</tr>
<tr>
<td>Decision tree</td>
<td>A type of decision model utilising a hierarchical representation of the</td>
</tr>
<tr>
<td></td>
<td>decision and chance events that make up a decision situation.</td>
</tr>
<tr>
<td>Diffuse source pollution</td>
<td>The cumulative runoff from urban and agricultural areas for which the</td>
</tr>
<tr>
<td></td>
<td>specific point of origin is not well-defined. Diffuse source pollution is</td>
</tr>
<tr>
<td></td>
<td>also known as ‘nonpoint source pollution’.</td>
</tr>
<tr>
<td><strong>Delivery Ratio</strong></td>
<td>The ratio of the amount of a pollutant reaching a water resource compared to the amount generated at its source.</td>
</tr>
<tr>
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<td>--------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td><strong>DEM</strong></td>
<td>A digital representation of topographical variation.</td>
</tr>
<tr>
<td><strong>Deterministic model</strong></td>
<td>A model that assumes that all input data is known with certainty.</td>
</tr>
<tr>
<td><strong>Discretise</strong></td>
<td>Make distinct and separable.</td>
</tr>
<tr>
<td><strong>EASI</strong></td>
<td>A pollution source mapping programme commissioned by the SCA and completed in 2002/3 to locate activities that occur in the SCA’s area of operations that have potential to cause pollution.</td>
</tr>
<tr>
<td><strong>Edaphic</strong></td>
<td>Factors relating to, or influenced by, soil conditions.</td>
</tr>
<tr>
<td><strong>Fundamental models</strong></td>
<td>A model based on fundamental scientific principles such as the conservation of material and energy.</td>
</tr>
<tr>
<td><strong>Emission models</strong></td>
<td>A model that uses leakage/export coefficients and/or empirical emission data for different contribution classes in a catchment to reveal conditions at the catchment outlet.</td>
</tr>
<tr>
<td><strong>Epistemology</strong></td>
<td>The study of how we know the things we know.</td>
</tr>
<tr>
<td><strong>Event-based model</strong></td>
<td>A model in which pollutant transport is simulated for single storms.</td>
</tr>
<tr>
<td><strong>Event mean concentration</strong></td>
<td>The mean concentration of a substance in runoff from a storm event.</td>
</tr>
<tr>
<td><strong>Expert system</strong></td>
<td>A computer program intended to embody the knowledge of experts in a given domain and emulate their problem-solving abilities.</td>
</tr>
<tr>
<td><strong>Export coefficient</strong></td>
<td>The mass of a substance exported from a given area of land over a given time period. Also known as “unit area load” and often reported as kg/ha/yr.</td>
</tr>
<tr>
<td><strong>Extensive pasture</strong></td>
<td>A large-scale, low-input farming system where land is managed to grow forage for livestock.</td>
</tr>
<tr>
<td><strong>Healthy Catchments Program</strong></td>
<td>An umbrella program instigated by the SCA to draw together its catchment protection, enhancement and community involvement programs (see <a href="http://www.sca.nsw.gov.au/catchments/hcp.html">http://www.sca.nsw.gov.au/catchments/hcp.html</a>).</td>
</tr>
<tr>
<td><strong>Hydraulic conductivity</strong></td>
<td>A soil property that describes the ease with which soil pores permit liquid water movement (expressed as cm/sec, mm/day, etc).</td>
</tr>
<tr>
<td><strong>Hysteresis</strong></td>
<td>The lagging of an effect behind its cause.</td>
</tr>
<tr>
<td><strong>Infiltration excess overland flow</strong></td>
<td>Overland flow is generated by two mechanisms: infiltration excess and saturation excess. In infiltration excess, the rainfall rate exceeds infiltration capacity and this excess rainfall moves overland depending upon the topography. This type of overland flow usually occurs at places where water table is deep.</td>
</tr>
<tr>
<td><strong>Information</strong></td>
<td>Data that has been processed to add or create meaning.</td>
</tr>
<tr>
<td><strong>Institutional</strong></td>
<td>Of or relating to the structures and mechanisms of social order and cooperation governing the behaviour of two or more individuals.</td>
</tr>
<tr>
<td><strong>Interactor</strong></td>
<td>An interactive participant in an interactive experience.</td>
</tr>
<tr>
<td><strong>Immission models</strong></td>
<td>A model that primarily uses measurements of loads or concentrations of a pollutant at a catchment outlet to describe upstream characteristics.</td>
</tr>
<tr>
<td><strong>Knowledge</strong></td>
<td>Learned information that contains guidance for action based on insight and experience.</td>
</tr>
</tbody>
</table>
**Local Environmental Plan**  
In NSW: Plans prepared by councils to guide planning decisions for local government areas. Through zoning and development controls, they allow councils to supervise the ways in which land is used (see [http://www.planning.nsw.gov.au/planningsystem/legislation_instruments.asp](http://www.planning.nsw.gov.au/planningsystem/legislation_instruments.asp)).

**Management**  
An organisational process that includes strategic planning, setting objectives, managing resources, recording and storing data and information, deploying assets needed to achieve objectives, and measuring results. Management functions are not limited to managers and supervisors in an organisation.

**Manager**  
An individual or group responsible for managing resources and processes.

**Methodology**  
A branch of philosophy that analyses the principles and procedures of inquiry.

**Model**  
A representation of ideas, mental constructs, objects or processes. A model may exist only in the human mind (mental model), be a physical representation of a larger object (physical scale model) or be a quantitative description using mathematical concepts and computers (mathematical and computer model).

**Monte Carlo simulation**  
The use of algorithms that estimate the probability of a range of outcomes from a model given uncertain input data. Monte Carlo methods rely on the random selection of input parameter values from assumed statistical distributions.

**Neutral or Beneficial Effect Test**  
A regulatory requirement under the Regional Environment Plan (REP) that proponents of developments must demonstrate a neutral or beneficial effect (NorBE) on water quality as a result of that development in order to obtain approval (see SCA, 2006).

**NEXSYS**  
An expert system used to estimate average annual export rates of N or P for several land-use types based on landscape traits including climate, topography, soil type and land management.

**Organisational**  
Of or relating to the structure, people or processes of an organisation.

**Policy-maker**  
Individuals in official bodies with the authority to make decisions about which problems within a particular sector are to be addressed and how the problems will be handled.

**Potential load**  
In the context of this research, the amount of nitrogen or phosphorus delivered to receiving waters assuming minimal attenuation during transport from source to stream or reservoir.

**Probabilistic model**  
A model that uses probability to describe aspects of uncertainty or randomness.

**Quickflow**  
Direct runoff. The portion of streamflow that is derived from channel precipitation, surface runoff and rapid subsurface flow.

**Regional Environmental Plans & The Drinking Water Catchments REP No. 1**  
REPs are plans that regulate planning and development across specified regions. REPs cover issues such as urban growth, commercial centres, extractive industries, recreational needs, rural lands, and heritage and conservation. The Drinking Water Catchments REP No. 1 is a regional plan for the environmental, social and economic future of Sydney’s drinking water catchments.

**Runoff**  
Rainfall that does not infiltrate the soil.

**Saturated hydraulic conductivity**  
The amount of water that would move vertically through saturated soil in a given time under a given hydraulic gradient. Also referred to as ‘Ksat’.

**Saturation excess overland flow**  
Runoff that occurs when soil is unable to absorb more rainfall because it is already saturated. Sometimes referred to as variable source area runoff.
<table>
<thead>
<tr>
<th>Term</th>
<th>Definition</th>
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<tbody>
<tr>
<td>Science</td>
<td>A body of knowledge based on objectivity and involving observation and experimentation.</td>
</tr>
<tr>
<td>Scientist</td>
<td>An expert in at least one area of science who uses the scientific method to do research.</td>
</tr>
<tr>
<td>Sediment rating curve</td>
<td>A regression analysis to estimate total sediment loads, with the amount of sediment described as a function of discharge.</td>
</tr>
<tr>
<td>Sensitivity analysis</td>
<td>An investigation into the variation in model outputs in response to specific variations in model input values.</td>
</tr>
<tr>
<td>Slow flow</td>
<td>The portion of streamflow that is derived from deep subsurface flows and delayed shallow subsurface flows. Also known as base flow.</td>
</tr>
<tr>
<td>Spatial Analyst ©</td>
<td>An ArcView extension that allows ArcView to import, produce, display, manipulate and save raster data.</td>
</tr>
<tr>
<td>Spreadsheet model</td>
<td>A representation of a system using an electronic spreadsheet.</td>
</tr>
<tr>
<td>State Environment Planning Policies &amp; SEPP 58</td>
<td>SEPPs deal with issues significant to the State of NSW. SEPP 58 was a planning policy to “Protect Sydney’s Water Supply”. It was repealed by the Drinking Water Catchments Regional Environmental Plan No. 1, which took effect from 01/01/07.</td>
</tr>
<tr>
<td>Steady-state model</td>
<td>A model with no time component. Steady-state models describe average temporal conditions for the period studied and do not seek to represent temporal variability.</td>
</tr>
<tr>
<td>Strategic Land and Water Capability Assessment</td>
<td>Assessments of sub-catchments designed to assist the SCA and local governments to determine the potential impacts of existing and future development on water quality. The assessments will be used to guide local governments in the preparation and review of local environmental plans.</td>
</tr>
<tr>
<td>Streamflow contribution</td>
<td>The volume of water delivered to a stream through surface flow, interflow and base flow.</td>
</tr>
<tr>
<td>Time-series</td>
<td>A series of measurements usually taken at regular intervals.</td>
</tr>
<tr>
<td>Topographic Wetness Index</td>
<td>An index calculated as the natural log of upstream contributing area/tan slope. The index is readily calculated using GIS from digital elevation models and provides a method for identifying and modelling locations likely to generate runoff by saturation excess and to quantify the variable source area concept.</td>
</tr>
<tr>
<td>Universal Soil Loss Equation</td>
<td>An equation for predicting the average annual soil loss in mass per unit area per year (A), and is defined as, A = RKLSCP, where R is a rainfall factor, K is a soil erodibility factor, L is the length of slope, S is the percent slope, C is a cropping and management factor, and P is a conservation practice factor.</td>
</tr>
<tr>
<td>Variable Source Area</td>
<td>The parts of a catchment that are the source of most runoff during storm events and therefore contribute disproportionately to the amount of pollutants transported into streams.</td>
</tr>
<tr>
<td>Water quality model</td>
<td>A numerical method for simulating fluxes of pollutants based on various theoretical assumptions and generalisations.</td>
</tr>
</tbody>
</table>
1 INTRODUCTION

1.1 Background to the Research

1.1.1 Nutrients in Catchments

Pollution of streams by nutrient-rich runoff from agricultural land is recognised as one of the most pressing environmental problems facing catchment managers throughout the developed world (Sharpley, 1995; Young et al., 1996; Novotny, 1999; Withers et al., 2000). The process by which water bodies become enriched with unusually high concentrations of nutrients, especially phosphorus (P) and, to a lesser extent, nitrogen (N), is known as ‘eutrophication’. Eutrophication can lead to algal blooms that are often toxic to humans and animals (ANZECC/ARMCAN, 2000) and can foul water used for drinking, agriculture, industry or recreation. Algal blooms may also lead to the depletion of oxygen levels in water bodies and result in adverse ecological impacts, including fish kills, shading out of other aquatic species, and reductions in biodiversity (Carpenter et al., 1998; Daniel et al., 1998; Ball et al., 2001).

In Australia, the importance of nutrient management was highlighted in 1991 when an algal bloom occurred over a 1,000 km reach of the Darling-Barwon River system in western NSW (Young et al., 1996). Since that time, the need for improved catchment management has become well-recognised and water quality models have come to play an important role in catchment management (Rizzoli and Young 1997; Young et al., 1997a; Caminiti 2004; Newham et al., 2004).

1.1.2 Water Quality Modelling and Catchment Management

Computer models play an increasingly important role in understanding human impacts on environmental systems and informing environmental management. Catchment management involves four major tasks: i) characterisation of the catchment (setting a baseline); ii) problem identification; iii) the development, evaluation and assessment of alternative management interventions and; iv) the communication of information to stakeholders. Catchment managers often use models for each of these tasks (Caminiti, 2004). The qualities that are most important
in a model will vary depending upon local circumstances and the need that the model is intended to help satisfy. A wide variety of catchment-scale nutrient models is currently available (Letcher et al., 1999) and these vary in their complexity, data requirements, precision, transparency, interactivity and explanatory power. Different models are also optimised to operate at different temporal and spatial scales.

When confronting a water quality problem, identifying which, if any, model can best inform different parts of the decision-making process is one of the earliest choices that a catchment manager must make (Xu et al., In Press).

1.1.3 The Problematic Nature of Modelling for Decision-making

The utility of models as tools to support decision-making has been constrained by issues that appear to be insufficiently resolved from the manager’s viewpoint. Some seemingly intractable issues that continue to be identified and discussed after more than three decades of computer modelling (for early examples see Grimsrud et al., 1976; OTA, 1982) include:

- inadequate quantitative knowledge of environmental processes at scales appropriate to management intervention and difficulty characterising complex hydrological and chemical processes and cultural practices (i.e., land-use and management) using spatially and temporally sparse data (e.g., Young et al., 1996; Davis et al., 1998; McDonnell, 2003; Dougherty et al., 2004; Harris and Heathwaite, 2005);
- disciplinary divides between scientists (who typically develop and run models) and managers (who typically use model outputs to inform decision-making) (e.g., Cullen, 1990; Loucks, 1992; Kinzig, 2001; Bosch et al., 2003; Cash et al., 2003; Caminiti, 2004); and
- difficulty reconciling socio-political, economic and environmental values (e.g., Cullen, 1990; Loucks, 1992; Rizzoli and Young, 1997; Bernknopf and Karl, 1998; Jakku, 2003; Sarewitz; 2004).

Managers who wish to use models for decision-support should therefore: i) choose models that are appropriate given the decision that must be made, the complexity of the system being modelled and the limitations of available data and knowledge; and ii) use a model and/or modelling ‘process’ that facilitates the transfer of scientific
knowledge into decisions and actions that are appropriate given the range of alternative issues that decision-makers need to consider.

1.1.4 Modelling in the Sydney Drinking Water Catchments
The Sydney Catchment Authority (SCA) is a NSW State government agency responsible for the health of the catchments that supply drinking water to four million residents of Sydney, Australia. The SCA’s area of operations is extensive, comprising an area of 16,000 km$^2$ and a population of 110,000 people (SCA and DoP, 2006) (Figure 1-1)

![Figure 1-1. The Sydney Catchment Authority’s area of operations.](image)

A plan for the future management of Sydney’s drinking water catchments is outlined in a regional environmental planning document entitled *Sustaining the Catchments*
(DIPNR and SCA, 2004). The plan specifies that the SCA will use models and decision support systems as assessment tools to support consideration of development proposals. More generally, it says that models will also be used to test alternative land management scenarios and support catchment management decision-making. According to the plan, an important role for such models will be to identify diffuse sources of nutrient pollution (N and P), such as rural and urban land, and estimate and predict nutrient loads delivered to streams.

The SCA and other organisations have been undertaking computer modelling of water quantity and quality in Sydney’s drinking water catchments for several years. Some examples of predictive models that are capable of modelling non-point sources of pollution, and have been used in SCA catchments, include CMSS (Cuddy et al., 1994), ANSWERS (Armstrong et al., 1995), HSPF (AWT, 2001a), IQQM (Young et al., 2000), AnnAGNPS (AWT, 2003), and SedNet (Olley and Deere 2003). A brief overview of most of these models is found in (Letcher et al., 1999).

Although considerable effort has been expended modelling water quality in the drinking water catchments, there appeared to be limited integration of previous research with current research and management planning activities. Models were seldom re-used and many potential users of the model outputs in the SCA expressed a general lack of confidence in their findings. Unarticulated ‘barriers’ seemed to have limited user confidence and the utility of the models and effectively inhibited the transfer of usable information or knowledge produced by the scientists that developed and used the models to the managers wishing to use the models to inform catchment management actions.

In 2005, the SCA engaged researchers from the University of Western Sydney (UWS) and other organisations to undertake collaborative research aimed at improving knowledge about various issues that potentially affect water quality. Perhaps due to the ‘barriers’ previously mentioned, some of the researchers (including the author of this thesis) were tasked with identifying nutrient sources and quantifying nutrient loads, in spite of earlier research with similar objectives (Cuddy et al., 1994; AWT, 2001a, 2001b, 2003; Long, 2003; Sherman and Orr, 2003 p.69). The research that is the subject of this thesis was initiated to both undertake the modelling required of the collaborative research project and, at the same time, seek to understand why previous
nutrient modelling was apparently unsuccessful in meeting the SCA’s needs. This thesis therefore includes the modelling component required by the SCA and identifies and explores the issues that affect confidence in models.

1.2 Research Problem and Research Objectives

The research has two broad aims that are to:

1. develop models of varying sophistication to identify nutrient pollution sources and calculate nutrient loads in the drinking water catchments of Sydney, Australia; and
2. identify key criteria that catchment managers can apply to choose and use models more effectively.

Through a cyclical process of action research (Carr and Kemmis, 1983) centred upon the development of the EC models, the thesis examines the unique needs of catchment managers and explores how those needs are different to the needs often ascribed to model users by most workers in science and research-based disciplines.

To achieve the two research aims, four specific objectives of this thesis were identified:

1. Identify barriers that inhibit the effective use of models in decision-making, and, in particular, the transfer of scientific knowledge into management action;
2. Develop evaluation criteria or guidelines to assess the utility of catchment models in different decision-making and policy development situations;
3. Assess the impact of model sophistication on catchment managers’ confidence in model predictions; and
4. Assess the impact of participation in modelling on catchment managers’ confidence in model predictions.

The conclusions that are derived from this thesis can be used to enable model developers and analysts to better meet the needs of catchment managers, and empower catchment managers who wish to maximise the utility of models in decision-making.
1.3 Research Justification

Despite the limitations of models, most managers and scientists agree that they play an important role in catchment management (Rizzoli and Young 1997; Young et al., 1997a; Caminiti 2004; Newham et al., 2004). There are a plethora of catchment-scale water quality and quantity models, and these are often reviewed and compared against various criteria (e.g., Ghadiri and Rose 1992; Singh and Woolhiser, 1995; Shoemaker et al., 1997; Letcher et al., 1999; Borah and Bera 2003 and Merritt et al., 2003). Most reviews are written in the ‘language’ of science, and focus on technical aspects of model development and use, including model structure, parameterisation, calibration, sensitivity analysis and more recently validation and uncertainty analysis. In contrast, there has been relatively little study into the utility of scientific modelling and prediction in environmental and water management (Cullen 1990; Beck 2002). For catchment managers, technical aspects of model design and use may not reflect the value of a model as a tool to support catchment management and planning processes (Loucks 1992; Pielke Jr. et al., 2000).

The approach used in this thesis offers a fresh perspective. The research is aimed at empowering managers by offering a more robust exploration of their modelling needs. It is appropriate that catchment managers who use models to support decision-making evaluate models according to criteria that emphasise aspects of model selection and use that are important in the field of planning and management.
2 APPROACH AND SCOPE OF THE THESIS

2.1 Research Philosophy and Methodological Framework

In this research, the author has sought to avoid methodological monism, \(i.e.,\) the reliance on using a single research method. The view of the author is that several alternative methodologies are potentially useful research tools. The research is interdisciplinary in the sense that it has elements that involve disciplines that are traditionally studied using ‘hard’ science approaches (\(i.e.,\) biophysical science and mathematical modelling using positivist, experimental approaches) and elements that involve disciplines that are often studied using ‘softer’ social science approaches (\(i.e.,\) knowledge management and decision-making using interpretivist research approaches).

Although avoiding methodological monism, the paradigm that dominates this study is that of qualitative research. A strength of qualitative research methods is that they are particularly useful for explaining what happens in organisations (Avison et al., 1999). Likewise, action research, the main qualitative method used in the thesis, is also suited to exploring issues at the boundary of science and management because it combines theory and practice (and therefore researchers and practitioners) using methods that are generally perceived to be mutually acceptable to both sides (Avison et al., 1999).

Action research was chosen as the primary methodological instrument for this study. The research sought methodological learning that is more applied than academic in nature. Action research is a form of applied research in which the researcher seeks to achieve learning outcomes that are of practical value to the individuals, groups or organisations with whom the researcher is working. At the same time, action researchers seek to re-inform current theory in the domain studied (Carr and Kemmis, 1983).

Action research involves active intervention and places more emphasis on what practitioners do than what they say. As such, action research can be more insightful than less participative approaches, such as case studies, that rely on interviews or
other methods of observing people in situations (Avison et al., 1999). The most salient aspects of the chosen methodology are elaborated in chapter 4 of this thesis.

The learnings of action research are usually achieved through a cyclical process, sometimes divided into four steps: planning; action; observation; and reflection. Future research activities are typically adjusted as the researcher reflects upon past activities and uses the findings of these earlier activities to inform (i.e. plan) the next activity (Carr and Kemmis, 1983). In this research, the activities (actions) undertaken within this action research framework were:

- a review of relevant literature from diverse disciplines;
- a workshop involving SCA catchment managers;
- a workshop involving water quality scientists and modellers;
- the selection of nutrient ECs and their application in a simple EC model;
- the enhancement of an EC model by weighting ECs using estimates of streamflow contribution;
- a focus group discussion to review the models involving SCA catchment managers; and
- a follow-up review of over 20 papers containing guidelines for good modelling practice in environmental management.

Each action involved participation and observation by the researcher. Reflection upon the process and the outcomes of the activity subsequently led to a planning phase that informed the development of the future activities.

### 2.2 Thesis Outline

The following describes the layout of the remainder of the thesis chapter-by-chapter:

*Chapter Three: Literature Review*

Chapter three contains a review of literature covering major aspects of the research including water quality modelling, decision-making by catchment management organisations, decision-making under uncertainty, barriers between science and management, barriers to the effective use of models and a brief introduction to studies of good modelling practice.
Chapter Four: Methodology

Chapter four describes the research strategy used in the study. The methodology used is contrasted against competing methodologies.

Chapter Five: Workshops

Chapter five presents the results of two workshops. The first workshop (5.1) involved SCA managers and sought to identify the modelling needs of the SCA. The second workshop (5.2) involved water quality scientists and modellers and sought an understanding of the current state of knowledge about nutrient modelling at catchment scale in an effort to identify the most appropriate modelling approach to meet the SCA’s modelling need.

Chapter Six: Modelling

Chapter six is divided into two parts. Part one (6.1) describes and discusses the development of a simple EC model. The findings are presented and discussed in terms of their applicability to catchment management. Part two (6.2) describes the development of an enhanced EC model in which ECs are weighted according to relative variations in streamflow contribution across the drinking water catchments. The author specified the scope of data inputs and outputs for the model, the degree of complexity expected, and the broad conceptual basis for the model, which was then developed in collaboration with Dr Barry Croke, Australian National University, Canberra.

Chapter Seven: Focus Group Discussion

Chapter seven describes and presents the outcomes of a focus group discussion involving SCA managers. The purpose of the discussion was to describe the models to the SCA and understand how they felt about the utility of each modelling approach.

Chapter Eight: Review of Modelling Practice Guidelines

In chapter eight, guidelines for good modelling practice are reviewed and discussed in relation to the methodological lessons learned throughout this research.
Chapter Nine: Improving Guidelines for Modelling Practice

In chapter nine, new criteria for better modelling practice are developed and discussed in relation to the experiential and theoretical aspects of the research. The chapter includes suggestions for implementing the methodological learnings of this thesis.

Chapter Ten: Conclusion

Chapter ten provides a concluding summary of the research dealing with each of the underlying research questions, and the two overall aims.

The thesis closes with references and appendices.

2.3 Definitions

2.3.1 Models

In this thesis, the term ‘model’ refers only to mathematical computer models. The OTA (1982) definition of a model is used in this research:

“a ‘numerical representation’ of how the real world or some part of it -a lake, a dam, or a community- works. More precisely, a model uses numbers or symbols to represent relationships among the components of these real-world systems” (p. 27).

2.3.2 Water Quality Models

An important but often unexpressed fact is that most water quality models do not provide ‘water quality’ information. Most water quality models simply provide an estimate of pollutants delivered to the edge of a field, the edge of a watercourse or to some point in a catchment. The assimilation of pollutants within a water body is highly variable and not a feature of many ‘water quality models’. Nevertheless, throughout this thesis, the term, ‘water quality model’ will be used. Readers should understand that most water quality models would more correctly be described as ‘pollutant loading models’.
2.3.2.1 Simulation and Prediction Models

Although many workers speak of most water quality models as tools that simulate and predict pollutant loading or ‘water quality’ impacts, most models are not reliable predictors of pollutant loading, but rather they are a crude indicator of the magnitude of the impact and provide comparative analyses that allow the model user to compare and contrast one scenario versus another, or to compare a hypothetical future scenario to present or baseline conditions (NRCS, 2006).

2.3.2.2 Export Coefficients and Export Coefficient Models

Whilst the discussion and findings of this thesis are not limited to any one type of water quality model, the type of model developed and used in this study is an ‘export coefficient model’. An EC is the mass of a substance exported from a given area of land over a given time period, and is often expressed as kilograms per hectare per year (kg/ha/yr). Gross potential loads for a pollutant exported from a given land-use are calculated by multiplying the land-use and pollutant-specific EC by the geographical area of the land-use. Total potential catchment loads of the pollutant can be calculated by summing the load for each land-use in the catchment.

The estimated loads are sometimes referred to as potential loads because the approach does not usually take account of assimilation of nutrients that may occur along the transport pathway in larger catchments (unless the EC value has been determined at this scale), which may reduce the actual amount of the nutrient that reaches a stream, reservoir or other water body.

ECs are sometimes referred to as ‘unit loads’ or ‘unit area loads’ or ‘generation rates’. The terms are usually used interchangeably.

2.3.3 Model Users

Rizzoli and Young (1997) identified three main categories of users of computer models and other environmental decision support systems. These are:

- environmental scientists (or system analysts);
- environmental managers (or decision-makers); and
- environmental stakeholders (e.g., special interest groups or landholders).
In this thesis, ‘model users’ may refer to any or all of these three groups. Most of the focus of the thesis is on environmental scientists, also referred to simply as ‘scientists’ or ‘analysts’ and environmental managers, who may also be referred to more specifically as ‘water resource managers’, ‘catchment managers’ or more broadly as ‘managers’ or ‘decision-makers’.

2.3.4 Institutions and Organisations

The thesis contains several references to ‘institutional’ and ‘organisational’ issues that affect decision-making. In general, the term ‘institutional’ refers to issues relating to structures and mechanisms of social order and cooperation that govern the behaviour of groups of individuals; and ‘organisational’ refers to issues relating to the structure, people or processes of an organisation (in this context, usually the SCA).

A major problem in setting consistent definitions is that the terms ‘organisational’ and ‘institutional’ are often used interchangeably in literature, particularly in relation to government departments that play a role in both policy-making and management. The issue is complicated in the thesis because the research came to be concerned with both the institutions that govern the organisation and the organisation as an institution in and of itself. Where possible, the author sought to use the correct usage of each term.

2.4 Limitations of scope and key assumptions

A significant part of this research involved developing an enhanced EC model using robust ECs for key land uses in the SCA’s area of operations developed from a comprehensive review of literature. However, the modelling itself was not the overarching priority of the research. Rather, modelling was used primarily as a vehicle from which methodological lessons about model development, selection and use in catchment management could be learned.

At the outset of the research, the expectation of the researcher was that issues relating to model sophistication, uncertainty and manager-participation in model development and use would be vital to the success of a modelling activity. As the research progressed, however, the focus of attention moved gradually from the models themselves and the modelling process towards issues focused primarily on the organisational and decision-making context. Although this change was not
anticipated at the beginning of the research, it is the nature of action research that the focus of inquiry may change as a study progresses. This is seen as strength of action research.
3 LITERATURE REVIEW

3.1 Environmental Modelling

In developed nations, public concern about environmental matters increased with the emergence of the environmental movement in the 1960s and 1970s, e.g., Silent Spring (Carson, 1962); Limits to Growth (Meadows et al., 1972); A Blueprint for Survival (The Ecologist, 1972); The 1972 United Nations Conference on the Human Environment (United Nations, 1973). At the time, political pressure from the public, special interest groups and concerned scientists motivated governments to address environmental problems, and a wave of environmental legislation was enacted to protect water quality and conserve natural resources, e.g., Australia (NSW): Clean Waters Act (1970); and Pollution Control Act (1970); USA: National Environmental Policy Act (1969); Great Lakes Water Quality Agreement [with Canada] (1972); and Clean Water Act (1972); Europe (EC/EU): Surface Water Directive (1975); and Bathing Water Directive (1976).

In response to the requirements of the new environmental laws, research organisations and government agencies of the 1970s and 1980s increasingly began to use mathematical models to quantify pollution problems or find optimal solutions (Grimsrud et al., 1976; OTA, 1982; Reckhow et al., 1985). The US Office of Technology Assessment (OTA) conducted an assessment of water resources models in 1982. They reported that:

“The technical capabilities of models vary greatly...however,... models capable of analyzing many pressing water resource management issues are currently available and have significant potential for increasing the accuracy and effectiveness of information available to managers, decision-makers and scientists” (OTA, 1982 p. 3).

The OTA also recognised that there were a number of constraints to the effective use of models as management tools. The major impediments identified by the assessors were described as “institutional constraints”, and included:

“...lack of information about available models, lack of training in model use and interpretation, lack of communication between decisionmakers and modelers and lack of general support services” (ibid).
Additionally, the authors noted a number of technical barriers to effective model use, especially “data constraints” and inadequate quantitative knowledge (i.e. “theoretical formulations” p. 143) of environmental processes.

In the 1970s and early 1980s computers were expensive and relatively inaccessible and possessed an air of “mystique” (OTA, 1982 p. iii). Since then, computing power has increased and computing costs have decreased dramatically. Twenty years after the OTA assessment, Orlob (1992) described the personal computer as a “ubiquitous” water resource management tool. He noted:

“... [computer-based] mathematical modelling has become an accepted part of the process of establishing and evaluating alternative scenarios for water-quality management” (p. 295).

Orlob was optimistic that with the continued acceleration in computer capability, in particular the ease of use, visualisation, and pre- and post-processing capabilities of modern personal computers, computer-based models of water quality were “…certain to become truly useful tools in the environmental management process” (p. 305).

From the mid-1980s and through the 1990s research-oriented models became more complex and featured more user-friendly interfaces. Also, management-oriented modelling tools, including Expert Systems (Lein, 1989), Environmental Decision Support Systems (Guarisco and Werthner, 1989), Group Decision-Support Systems (Gray, 1987) and Geographical Information Systems (Burrough, 1986) evolved and became more widespread. Each of these genres of computer-based models and tools were often promoted as holding the promise of overcoming many of the impediments that faced early environmental model users. In relation to the software developments of the time, Loucks (1992 p. 220) wrote:

“This software is aimed at facilitating model use and, more importantly, interaction and communication between the analysts or modelers and their clients...[and]...giving planners and managers improved opportunities for increasing their understanding of water resource systems.”

The advances that have been made in recent decades are well-documented. Computers, at least in developed nations, are inexpensive and powerful. There have been consistent improvements in scientific knowledge of the processes associated
with mobilisation, transport and delivery of all kinds of pollutants, including nutrients (Reckhow and Chapra 1999; McDowell et al., 2004). Increasingly sophisticated modelling tools have been developed (Silberstein, 2006), and relatively detailed spatial datasets are readily available (Grayson and Blöschl 2000a). Nevertheless, many of the problems experienced by model users in the 1970s and 1980s continue to trouble today’s resource managers. For example, twenty years after OTA (1982 p. 143) identified that inadequate “theoretical formulations” were a significant barrier to the effective use of models in water resource management, Beven (2002) drew a similar conclusion:

“Computing constraints are now much less critical for many problems of great practical interest, but with hindsight there would appear to be more important problems in theoretical hydrology than [in] increasing mathematical complexity.”

3.2 Catchment Scale Nutrient Modelling

3.2.1 Background

In the 1960s and 1970s, models of nutrient fluxes in lake catchments were developed in Europe and North America using relatively simple, empirically based EC approaches. Nutrient losses from the land surface to lakes, and later other water bodies, were computed using ECs (unit loads) for different nutrient sources that were derived from literature (Vollenweider, 1968; Jørgensen, 1980; Reckhow and Simpson, 1980). Although EC approaches continue to be used by researchers and catchment managers (e.g., Johnes, 1996; Davis and Farley, 1997; Hanrahan et al., 2001; Endreny and Wood, 2003; McNamara and Cornish, 2004), an ever-increasing body of international and Australian research has led to the development of a very wide variety of models that purport to predict or estimate (i.e. simulate) nutrient loads delivered to water bodies at catchment-scales (Table 3-1).
<table>
<thead>
<tr>
<th>Name/ Acronym</th>
<th>Long Name</th>
<th>Origin</th>
<th>References</th>
<th>Detailed Description</th>
<th>Overview</th>
</tr>
</thead>
<tbody>
<tr>
<td>AEAM</td>
<td>Adaptive Environment Assessment and Management</td>
<td>Environment Canada &amp; University of BC (Canada); IISA (Austria)</td>
<td>Holling, 1978</td>
<td>3,6</td>
<td></td>
</tr>
<tr>
<td>AGNPS</td>
<td>Agricultural NonPoint Source pollution model</td>
<td>USDA (USA)</td>
<td>Young et al., 1987</td>
<td>1,2,3,4,5</td>
<td></td>
</tr>
<tr>
<td>AnnAGNPS</td>
<td>Annualized Agricultural NonPoint Source model</td>
<td>USDA (USA)</td>
<td>Cronshay and Theurer, 1998</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>ANSWERS</td>
<td>Areal Nonpoint Source Watershed Env. Response Simulation</td>
<td>Purdue University (USA)</td>
<td>Beasley et al., 1980</td>
<td>1,2,3,4,5</td>
<td></td>
</tr>
<tr>
<td>AQUALM</td>
<td>XP-AQUALM</td>
<td>XP-Software (Australia)</td>
<td>Phillips et al., 1993</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>CMSS</td>
<td>Catchment Management Support System</td>
<td>CSIRO (Australia)</td>
<td>Davis et al., 1991</td>
<td>3,6</td>
<td></td>
</tr>
<tr>
<td>EMSS</td>
<td>Environmental Management Support System</td>
<td>CSIRO and CRCCH (Australia)</td>
<td>Vertessy et al., 2001</td>
<td>5,6</td>
<td></td>
</tr>
<tr>
<td>HSPF</td>
<td>Hydrological Simulation Program - Fortran</td>
<td>USEPA (USA)</td>
<td>Bicknell et al., 1993</td>
<td>1,2,3,4,5</td>
<td></td>
</tr>
<tr>
<td>IQQM</td>
<td>Integrated Water Quantity and Quality Model</td>
<td>NSW DLWC (Australia)</td>
<td>DLWC, 1995</td>
<td>3,5,6</td>
<td></td>
</tr>
<tr>
<td>LASCAM</td>
<td>Large-scale Catchment Model</td>
<td>University of WA (Australia)</td>
<td>Viney et al., 2000</td>
<td>3,5</td>
<td></td>
</tr>
<tr>
<td>MIKE SHE</td>
<td>European Hydrological System model</td>
<td>BIH (UK); SOGREAH (France); DHI (Denmark)</td>
<td>Refsgaard and Storm, 1995</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>MUSIC</td>
<td>Model for Urban Stormwater Improvement Conceptualisation</td>
<td>CRCCH (Australia)</td>
<td>Wong et al., 2001</td>
<td>6</td>
<td></td>
</tr>
<tr>
<td>SWAT</td>
<td>Soil and Water Assessment Tool</td>
<td>USDA (USA)</td>
<td>Arnold et al., 1998</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>SWMM</td>
<td>Storm Water Management Model</td>
<td>USEPA (USA)</td>
<td>Metcalf and Eddy Inc. et al., 1971</td>
<td>2,4</td>
<td></td>
</tr>
<tr>
<td>SWRRB</td>
<td>Simulator for Water Resources in Rural Basins</td>
<td>USDA (USA)</td>
<td>Williams et al., 1985</td>
<td>2,4,5</td>
<td></td>
</tr>
</tbody>
</table>

(1) Borah and Bera (2003); (2) Donigian and Huber (1991); (3) Letcher et al. (1999); (4) Ghadiri and Rose (1992); (5) Merritt et al. (2003); (6) CRCCH (2005b)
3.2.2 Model Classification

For most water resource models, including those that simulate nutrient fluxes, model classification is generally based on how the model represents the hydrological system. One of the most popular methods for classifying computer-based mathematical water resource models is to group models according whether the models represent environmental processes using an empirical, conceptual or physical schema (Refsgaard, 1996).

The simplest category of model is the empirical model. Empirical models are based on observed relationships between pollutant generation and related catchment characteristics. Little or no attempt is made to describe physical processes. Conceptual models are based on a generalised concept of internal storages through which flows and pollutants pass. These models vary in their level of sophistication in their representation of hydrological systems, but are generally more complex than empirical models. Physical models are usually the most complex and often feature detailed representations of the processes that drive pollutant transport (Refsgaard, 1996; Donelley et al., 1998; Martin and McCutcheon, 1999; Letcher et al., 1999; Grayson and Blöschl, 2001b; Borah and Bera, 2003).

The main characteristics of each modelling schema are described below:

Empirical models

Other names: Black box-, metric models

Complexity: Low

Input data requirements: Low

Examples: AEAM, CMSS, MUSIC

Empirical models are usually the simplest quantitative models. They are based on mathematical equations that reflect statistical relationships between a particular process (or groups of processes) and the various factors that influence the process (Donelley et al., 1998). The equations are derived from repeated experiments and observations that are often undertaken at large temporal or spatial scales (e.g., catchment scale, annual time step). Empirical models do not usually attempt to simulate the numerous biophysical or geochemical interactions that create the
statistical relationships they use, and therefore need to be calibrated using local (or locally-relevant) datasets (Letcher et al., 1999).

**Conceptual models**

**Other names:** Grey box-, stocks and flows-, bucket models  
**Complexity:** Medium  
**Input data requirements:** Medium  
**Examples:** AGNPS, AnnAGNPS, AQUALM, EMSS, HSPF, IQQM, LASCAM

In a conceptual water resource model the system being modelled is usually more elaborately described than for their empirical counterparts. A conceptual water resource model is comprised of several separate hydrological components or ‘quasi-physical storages and interactions’ (Donnelly et al., 1999 p. 36) that are based on a broad and essentially qualitative a priori understanding of how water and hydraulically mobilised materials move through the system. The transfers between the storage components are defined by parameters and equations that are calibrated using locally-relevant experiments and observations.

**Physical Models**

**Other names:** White box-, mechanistic-, physics-based-, process models  
**Complexity:** High  
**Input data requirements:** High  
**Examples:** ANSWERS, MIKE SHE, SWAT, SWMM, SWRRB

Physical models are more complex than empirical or conceptual models. They describe the behaviour of a system by combining quantitative descriptions of all significant physical processes and their interactions at relatively small spatial or temporal scales (e.g., plot scale, small time-step). Physical models use the numerical solutions of fundamental equations to describe fluxes of materials, usually water, but also sediment, nutrients, salts, pesticides and pathogens, and energy (e.g., temperature, erosivity) (Borah and Bera, 2003; Martin and McCutcheon, 1999). Physical models therefore reflect a predominately “reductionist modelling philosophy” (Donnelly et al., 1998 p.33), and because they offer a more complete characterisation of catchment processes, are (in theory at least) less reliant on local calibration data.
Sometimes *material* models, which are constructions made of wood, metal or concrete for example, are also called *physical* models (e.g., wave machines, globes, mechanical representations of planetary motion), but should not be confused with computer-based (*i.e.* mathematical) physical (*i.e.* physics-based) models.

*Other Schema*

Models are also characterised on the basis of their level of detail and their treatment of time, space or other factors (Argent *et al.*, 2006). Models can thus be described as ‘discrete’ or ‘continuous’, ‘coarse’ or ‘fine’, ‘dynamic’ or ‘static’, ‘lumped’ or ‘distributed’, ‘deterministic’ or ‘stochastic’, and ‘descriptive’ or ‘mechanistic’. The most commonly used model descriptors are shown in Table 3-2.

<table>
<thead>
<tr>
<th>Table 3-2. Some commonly used model classifications</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Discrete</strong></td>
</tr>
<tr>
<td>A model in which time (or other variables) are described in incremental steps (or categories). The variables in a discrete model change at separated points in time.</td>
</tr>
<tr>
<td><strong>Coarse</strong></td>
</tr>
<tr>
<td>A coarse model generally operates over coarse temporal and spatial scales. Coarse models are generally less accurate but more efficient in terms of data, knowledge and computing requirements, than fine models</td>
</tr>
<tr>
<td><strong>Static</strong></td>
</tr>
<tr>
<td>A model that has no time component. Model outputs usually describe average temporal conditions or ‘snapshots’.</td>
</tr>
<tr>
<td><strong>Lumped</strong></td>
</tr>
<tr>
<td>A model in which a system is described in terms of average quantities. Model states and parameters do not vary throughout the system being modelled.</td>
</tr>
<tr>
<td><strong>Deterministic</strong></td>
</tr>
<tr>
<td>A ‘non-probabilistic’ model in which all parameter values are fixed and which yields a ‘determined’ single-value result for each output.</td>
</tr>
<tr>
<td><strong>Descriptive</strong></td>
</tr>
<tr>
<td>A model that represent relationships between variables by drawing on values in a database that are based on experiments or observations.</td>
</tr>
</tbody>
</table>
Of the hundreds of models available, many feature components that fit more than one schema and attempts to classify models are inherently problematic (Ghadiri and Rose, 1992; Letcher et al., 1999).

Approaches to model classification can also be described as ‘top-down’ or ‘bottom-up’ (CRCCH 2005a) and ‘soft’ or ‘hard’ (Nijkamp 1980; Checkland and Scholes 1990) (Table 3-3).

**Table 3-3. Two alternative approaches to model classification.**

<table>
<thead>
<tr>
<th>Top-down (downward)</th>
<th>Bottom-up (upward)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Top-down models are designed to be as simple as possible. Complexity is only added when there is sufficient supporting data to confirm the modelled assumptions and when there is sufficient field data to confirm that the added model component will increase the accuracy of the model outputs.</td>
<td>In the bottom-up approach, the modeller seeks to include routines that represent all relevant physical processes. The validity of this kind of model is mainly dependant on the identification and faithful representation of key system processes. Bottom-up models tend to be more complex and have higher input data requirements than top-down models.</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Soft</th>
<th>Hard</th>
</tr>
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<tbody>
<tr>
<td>Soft models are ‘people-focussed’ and rely on qualitative assessments. Model inputs may be agreed upon using participatory processes that include affected stakeholders. Soft models are usually less structured than hard models.</td>
<td>Hard models are ‘thing-focussed’ and rely on thorough knowledge of quantitative measures. Hard models are often used in technical domains that emphasise the value of the traditional scientific method and peer review.</td>
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### 3.3 Choosing and Using Models

In a recent Australian survey (Marston et al., 2002), catchment managers were asked questions that sought to prioritise key catchment management issues and list models and modelling approaches that may be used to address those issues. Despite the perceived importance of models in catchment management and the increasing use of models, or at least the outputs from models, by catchment managers, few respondents addressed any of the questions in the survey that focussed specifically on models and modelling approaches. The complexity associated with using complex water quality models may put distance between analysts and decision-makers, since complex models are much less transparent to non-specialists than simpler models (Packman and Old, 2005). Managers may therefore entrust model selection to the consultants and scientists who use the models.
3.3.1 Model Complexity

Many workers argue that it is very important to choose and use a model with an appropriate level of complexity or sophistication (e.g., Grayson et al., 1992; Jakeman and Hornberger, 1993; Barnes, 1995; Letcher et al., 1999; Perrin et al., 2001; Argent and Mitchell, 2003; Wasson et al., 2003; CRCCH, 2005b). The general view of workers in this area is that appropriate model complexity should be dictated by objective criteria relating to the nature of the problem to be resolved and the availability of relevant data and knowledge, although there is considerable evidence that subjective criteria, including expertise and personal preference also strongly influence model choice (Wagener and Kollat, In Press).

Letcher et al. (1999 p.80) point out that complex models are most often used by researchers aiming to further “knowledge of some of the processes involved in sediment and nutrient generation”, whilst simpler, empirical models are generally used in catchment management programs to support the implementation of so-called best management practice. The authors state that these simpler models are useful even though they do not “give a definitive solution to a problem” (p. 80). Readers should not draw from this the implication that more complicated models can, at least potentially, provide a definitive solution. Many authors have highlighted the limitations of environmental models and believe that they will never provide indisputable results (e.g., Oreskes et al., 1994; Pielke Jr et al., 2000).

Letcher et al. (1999) advocate the use of simple models and consider complex physics-based and conceptual models as being unsuitable for the estimation of catchment exports of nutrients and sediment because of a lack of input and calibration data, and because their interpretation is overly dependent on the experience of the model user. Nevertheless, they argue that there is no single optimal estimation technique, and that the choice of model should depend upon the availability of water quality data and specific catchment characteristics. They also conclude that users must consider the preferred accuracy and complexity of any proposed estimation technique.
3.3.2 Prediction Products and Processes

Pielke Jr et al. (2000) point out that modelling for decision-making must be viewed in a wider framework in which models are considered not only in the context of the quality of the predictions that are produced, but also based on the quality of modelling as a tool for facilitating better decisions. They divide these assessment criteria into two categories based on; the quality of the “product” they produce (usually a prediction); and the quality of the “process”.

Prediction as a product

According to Pielke Jr et al. (2000), there are three primary considerations when considering prediction as a product. These are accuracy, sophistication and experience. ‘Accuracy’, they argue, needs to be measured in terms of the improvement of a forecast over some naïve standard. The authors point out that ‘sophistication’ in a model does not necessarily improve the accuracy or quality of the predictions and note that simple models can be run more times and can produce more output than a complex model that requires a lot of resources to run and analyse. ‘Experience’ refers especially to the experience of managers in analysing model outputs.

Prediction as a Process

When considering prediction as a process, managers need to consider three parallel, sub-processes (Pielke Jr et al., 2000). These are the Research Process, the Communication Process, and the Use or Choice Process (see also Pielke Jr, 2003). Pielke Jr et al. (2000) describe these as follows:

1. Research. Including science, observations, modelling, etc., as well as forecasters’ judgments and the organisational structure - all of which go into the production of predictions for decision makers.
2. Communication. Both the sending and receiving of information - e.g., who says what to whom, how it is said, and with what effect.
3. Use. The incorporation of predictive information into decision-making, acknowledging that decisions are typically contingent on many factors other than predictions.
Although most literature focuses on the research sub-process, and in particular the physical science that backs it, Pielke Jr et al. (2000) consider that each activity is important for good decision-making, and that the research activity may be the least critical. They demonstrate this by pointing out that:

“open communication and consideration of alternative policy approaches can lead to successful decisions in the face of unsuccessful prediction products, but the opposite is unlikely to occur” (p. 377).

3.3.3 Transparency and Participation

Another issue highlighted by many workers is the importance of model ‘transparency’ and stakeholder ‘participation’. Haag and Kaupenjohann (2001), for example, believe that models should be transparent, and that framing of models and model choice and the evaluation of models should involve stakeholders and “local actors”.

3.3.3.1 User Confidence

Rizzoli and Young (1996) believe that many decision-makers do not trust the algorithms of the most “trustworthy” environmental models, and that the expert opinion of the decision-maker is sometimes more influential than information from models.

Trust in models and the conclusions of scientists and decision-makers can be a major concern among stakeholders. Many authors stress the need for stakeholders to be involved in modelling for decision-making (e.g., Simonovic and Bender, 1996; Yearly, 1999; Kloprogge and van der Sluijs, 2002). Caminiti (2004) discusses this from the perspective of a catchment manager. Expectations of stakeholders and the wider community need to be carefully managed from the outset of a modelling exercise and all parties need to be aware that models do not provide “the answer” because they cannot take into account all relevant factors (Caminiti, 2004).

A significant body of literature suggests that complex models inhibit participation of stakeholders in decision-making. In order for decision-makers to be better served by models, processes need to be transparent, thereby allowing stakeholders an opportunity to scrutinise the assumptions used in the model (Caminiti, 2004). A black box approach should be avoided. Non-experts should be able to understand the basic
components and limitations of a model. As tools in decision-making, complex simulation models are referred to as ‘black boxes’ because their operation is generally opaque to outsiders, inhibiting participative decision processes (Haag and Kaupenjohann, 2001).

There is increasing acknowledgement that stakeholders need to be included in decision-making in a meaningful way. According to Loucks (1992 p. 216):

“Planning and managing involves not only decision-making, but also developing among all interested and influential individuals an understanding and consensus that legitimises the decisions and enhances their successful implementation”

Modelling can either help or hinder the development of understanding and consensus. To support decision-making, models and DSS should ideally support the political, organisational and social dimensions of the decision-making process (Reitsma, 1996).

The involvement of stakeholders and the incorporation of their values in decision-making can help a resource manager decide which actions are feasible and how much risk is acceptable. The aim is to find a “consensus solution” (Simonovic and Bender, 1996 p. 250). Models can be used as a tool to facilitate this, but it is important that stakeholders be allowed to express their preferences using their own language conventions (Simonovic and Bender, 1996). Stakeholders can be involved in modelling during model development, while running the model, and while evaluating the model, and stakeholder involvement can take the form of elicitation, consultation and participation (Kloprogge and van der Sluijs, 2002).

Another reason to engage a wider group of stakeholders is to expand the knowledge base and foster creative solutions (Simonovic and Bender, 1996). Models generally do not have the ability to accurately measure the impact of management scenarios that are not a part of the base model, or measures that are part of a treatment train (Caminiti, 2004). Traditional decision-making approaches based on physical modelling alone are unlikely to represent the multidimensionality of environmental problems (Reitsma, 1996) and may constrain creativity and limit choice of feasible solutions.
In addition to greater participation of stakeholders, Caminiti (2004) and Loucks (1992) argue that there should be more meaningful interaction between the managers who must make decisions, and analysts who run the models on behalf of the managers. Uncertainties, assumptions and limitations of models and their outputs need to be meticulously documented by modellers and borne in mind by resource managers when using models and model outputs to set priorities and make decisions. This is best achieved throughout model development and use rather than at the end of the process (Caminiti, 2004). Loucks (1992 p.217) argues that analysts need to work on the issues of concern to planners and managers, and need to be prepared to interact with the political and social structure of the institutions they are attempting to assist, and contribute towards the improvement of the planning and management process. This can be achieved through a consultative approach, giving the resource manager and the steering committee a better understanding of the model, and comfort with the model outputs and associated uncertainties (Caminiti, 2004).

Mackenzie (1999) believes that there is a relationship between “proximity to knowledge” and confidence in that knowledge. He speculates that groups who are committed to the use of a technology, but not directly involved in producing that knowledge seem to have more certainty about the reliability, safety or predictability of a technology than “insiders” who are directly involved in producing the knowledge. Those who are alienated from the source of knowledge production, usually the general public, have the least certainty (i.e. in this case referring to ‘confidence’) in the technology. Mackenzie speculates that the uncertainty trough is important for policy because when decision-makers commit to a technology but do not have an intimate knowledge of it they may have undue confidence. In the public, he speculates that the surfacing of “insider” uncertainty can lead to over-reaction controversy by a public that expects science to yield certain knowledge.

If this relationship holds true in the context of catchment modelling for decision-making, it may be the case that some environmental managers/decision-makers have more confidence in catchment models than the original developers or a sceptical public.
3.3.4 What is a ‘Good’ Model?

Beck (2002) considers that three questions are important when evaluating a model:

1. **Peer Review**: Has the model been constructed of approved materials?
2. **Matching History**: Does the behaviour approximate well that observed in respect of the real thing?
3. **Fulfilling a Designated Task**: Does it work?

Barnes (1995) lists some generally accepted criteria answering his question: “*What is a good model?*” These include that modelling should be focussed, and that models should be as simple as possible, “…generalisable and adaptable, and preferably with wide scope” (p. 751). Barnes also highlights the need to consider data availability and quality when parametising the model, and to be explicit and consistent about limitations and assumptions. Importantly, Barnes points out the heuristic value of models:

> “a good model will suggest further developments and give rise to deeper questions and insights, with implications for future data gathering” (p. 751).

Rizzoli and Young (1997) point out six desirable features of computer-based environmental decision support systems. These are summarised below:

1. The ability to acquire, represent and structure the knowledge in the domain under study;
2. The ability to separate data from the model for model re-usability and prototyping;
3. The ability to deal with spatial data (a GIS component);
4. The ability to provide expert knowledge specific to the domain of interest;
5. The ability to be used effectively for diagnosis, planning, management and optimisation; and
6. The ability to assist the user during problem formulation and selecting the solution methods.

Conventional approaches to model evaluation can be summed up by four guiding principles (Hillel, 1986):
• *Parsimony:* A model should not be any more complex than it needs to be and should include only the smallest number of parameters whose values must be obtained from data;

• *Modesty:* A model should not pretend to do too much, there is no such thing as ‘the’ model;

• *Accuracy:* We need not have our model depict a phenomenon much more accurately than our ability to measure it; and

• *Testability:* A model must be testable and we need to know if it is valid or not, and what are the limits of its validity.

### 3.4 Modelling from the Perspective of the Manager

For managers, modelling can be an onerous, time consuming, expensive and bewildering exercise (Safai-Amini, 2000; Caminiti, 2004). An important challenge for managers and planners is to decide which, if any, models should be used to assist planning and decision-making in catchments, and, if they are used, managers need to know how to implement them within an effective decision-making framework (Reitsma, 1996; Safai-Amini, 2000).

Issues relating to the implementation of models in a decision-making context are important because environmental decision-making is essentially an ill-structured process, especially for complex issues where there are imperfect data with which to characterise the problem (Reitsma, 1996; Simonovic and Bender, 1996). Most models used in resource management are simplifications of very complex environmental systems that are poorly understood and/or difficult to quantify (Faucheux and Froger, 1995). Yet computer programs can only be effective in solving problems if those problems are formulated as algorithms (*i.e.* well structured). As a result, models can provide a process or a product (see Pielke Jr et al., 2000 – discussed later) with which to frame or analyse a decision-making problem, but they will rarely, if ever, be adequate to replace the judgement and intuition of the catchment manager (Loucks, 1992; Simonovic and Bender, 1996).

The abundance of catchment-based models available to decision-makers is highlighted by Marston *et al.* (2002). In a survey responded to by 46 model users, 36 different catchment models were cited and only 14 models were used by more than
one respondent. Although most of these models were developed and written by users at considerable investment of time and effort, over 90% of the models used could be substituted by an alternative model. Choosing the model that is best suited to any given purpose is often the subject of personal or philosophical preference of the analyst who uses the model (frequently based on their skill and experience using different types of models) to produce the product (predictions) for the decision-maker (Barnes, 1995; Dahl and Wilson, 2001; Marston et al., 2002). This can “promote the attitude of bending the task to suit the model” (Barnes, 1995 p. 748).

3.4.1 Key Elements Catchment Managers Need to Consider When Selecting Models

3.4.1.1 Data and Knowledge
The most basic requirement for the estimation of catchment exports using models is data. Data are important for model parameterisation, calibration and validation. Young et al. (1996) reviewed Australian research that attempted to measure nutrient export from different land use types and compared these data with North American results. The authors concluded that the Australian dataset was incomplete and that the process of estimating nutrient loads in Australia was hampered by “a poorly defined conceptual model of the nutrient export process” (p. 180). They also concluded that the North American dataset, which is more complete, is not comparable to Australian conditions and therefore unsuitable for use in Australia.

Letcher et al. (2002) compared nutrient and sediment loads estimated by four different models. Although the catchments that were modelled in their study were selected because there was thought to be an adequate supply of water quality data for modelling purposes, this assumption was subsequently found to be incorrect. The authors found that, even for simpler conceptual and/or empirical models, more strategic and long term water quality monitoring was required to estimate loads. They also stated that “further work on regionalisation studies that relate model parameter values to landscape attributes is required to improve confidence in the results of models for which the prediction of the effects of land use change is required” (p. 10).
Other authors have also highlighted the problem of modelling nutrients with sparse datasets. For example, Barnes (1995) stated that “without high quality data sets with which to guide and constrain the modelling effort, it is hard to foresee much real progress towards the goal of ungauged catchment modelling” (p. 751). The observation that existing and pre-existing data-gathering programmes are inappropriate and inadequate for connecting to realistic and meaningful management objectives is common amongst water quality professionals. Ongley (1999) refers to this as the “data crisis”.

Some workers argue that water quality monitoring should be coordinated to help calibrate and validate modelling efforts. This coordination includes ensuring that monitoring is sensitive to the internal structure of catchments and provides a sufficient number of samples and data covering a wide range of flow conditions and different spatial scales (Christophersen et al., 1993; Letcher et al., 2002).

However, other workers note that there is a danger that the public and decision-makers may interpret a quest for more data and better understanding as a declaration that the scientific basis for decision-making is inadequate. Reckhow (1994) believes that this interpretation is generally incorrect. Several authors warn that research needs must be tempered by the immediate and practical needs of decision-makers who will always need to make decisions and develop policy without perfect information. The fact that there is inadequate data to run some models does not necessarily mean that more data is required to make policy decisions about land use and its affect on current or future water quality. Ongley (1999) described the “data paradigm”, in which institutions misguidedely seek “more and better” data without recognising that natural systems are so highly variable that monitoring programmes may be unable to capture this variability.

3.4.1.2 Measuring and Managing Uncertainty

For modellers, a better understanding of model behaviour, especially parameter sensitivity and uncertainty, can lead directly to model improvements by identifying those parameters, model processes or constraints that most need refinement. Most workers agree that decision-makers should know the level of uncertainty associated with modelling results and modellers need to develop understandable ways of
communicating uncertainty to those decision-makers (e.g., Loucks, 1992; Burgman, 2001). However, Sarewitz and Pielke Jr (2000) argue that both the scientific community and decision-makers have a poor record of understanding the uncertainty associated with model predictions.

Understanding uncertainty helps decision-makers to know when there is a need for more research and when a management intervention is justifiable. In the absence of a measurement of uncertainty, decision-makers will not have a full appreciation of the risk of failure of a planned management action and may call for more research or even more modelling in an attempt to reduce the perceived risk. As Pielke Jr et al. (2000 p. 362) observed, “Knowing when to depend on predictions is itself a challenge of the prediction process”.

The increasing use of water quality models to guide management and decision-making has stimulated increasing interest in model ‘goodness of fit’, and led to studies in sensitivity analysis, uncertainty analysis and model validation (Reckhow and Chapra, 1999). Each is summarised below.

**Sensitivity Analysis:** Sensitivity analysis is conducted by varying each parameter in the model by a small, fixed amount (e.g., 10% of the nominal value). Model output is then regressed against the input parameters, and the regression coefficients for each parameter indicate their relative sensitivities. A sensitive parameter is one that elicits a large change in model output for a small change in the input parameter (Reckhow and Chapra, 1999).

**Uncertainty Analysis:** The results of most environmental models usually consist of single deterministic predictions of the values of selected output variables. Uncertainty analysis is conducted by varying each parameter by the standard error of its estimate. Model output is then regressed on the input parameters (as with sensitivity analysis), and uncertainty is indexed as the amount of variation in the output accounted by each of the parameters (i.e. their partial $R^2$). An uncertain parameter is one that is sensitive within the range of precision with which it can be estimated. There are two basic methods for undertaking an uncertainty analysis: first order error analysis; and Monte Carlo simulation (Reckhow and Chapra, 1999).
First order error analysis: is based on linear approximation of the model with 
error terms represented by variance; covarying errors can be accounted for 
with a correlation term. The approximation means that first order analysis is 
not exact for nonlinear models and for models with error terms that are not 
fully characterised by variance alone. In principle, first order error analysis 
can be applied to any model, although from a practical standpoint, highly non-
linear models with skewed error distributions are not good candidates 
(Reckhow and Chapra, 1999).

Monte Carlo Simulation became feasible with the advent of modern computing. 
Monte Carlo simulation begins with the selection of a probability density 
function characterising the uncertainty in each term in the model, while the 
model itself is unchanged from standard deterministic applications. Then, 
when the model is applied, instead of a single run of the model, hundreds or 
thousands of model runs are made. For each model run, the program draws a 
sample from the probability distribution for each uncertain parameter and uses 
those sampled values to compute a single solution. After this 
sampling/modelling process is repeated many times, a distribution of 
responses is generated, representing the combined effect of all uncertain 
terms. Parameter covariance may be accounted for with correlated sampling 
between distributions (Reckhow and Chapra, 1999).

Uncertainty analyses have not been widely used for nutrient models, but where they 
have been performed, indications are that predictions are not very precise, with 
prediction errors unlikely to be less than ± 30%, and sometimes well above ± 100% 
(Reckhow and Chapra, 1999).

Model Validation: Model validation refers to the ‘goodness of fit’ between observed 
(i.e. measured, historical) and modelled (i.e. predicted) results. Model validation is 
more rigorous if the data used to validate a model are substantially different from that 
used to calibrate the model. Model validation is not straightforward. A model may be 
right for the wrong reasons if data used for calibrating a model do not contain enough 
information to uniquely determine model parameters, or if state variables contained 
within the model can be difficult to relate to field observations because of spatial (and 
temporal: Oreskes, 1998) heterogeneity or the limitations of the “conceptual” model
structure (Christophersen et al., 1993). Tumeo and Orlob (1986) also highlight the fact that model output may not fit with observed results not because the model or its parameters contain errors, but simply because the model is predicting a single possible value within a wide range of probable values (i.e. environmental stochasticity).

Oreskes et al. (1994) and Oreskes (1998) argue that stochasticity and empirical uncertainties mean that validation of models of complex natural systems is technically impossible, and that we should view quality control of models as a process of evaluation rather than validation.

3.4.1.3 Bias

The inherent subjectivity in models (see Barnes, 1995; Kloprogge and van der Sluijs, 2002) often results in criticism by scientists and stakeholders when model results are used in decision-making (Haag and Kaupenjohann, 2001; Kloprogge and van der Sluijs, 2002). Bias may be introduced at many stages in the course of modelling for decision-making (Kloprogge and van der Sluijs, 2002), including when framing a problem for analysis; when making an inventory of options to be chosen from or modelled; when choosing variables and indicators; when making actual choices; and when evaluating the choice. Without some acknowledgement of the influence of these choices on the results of modelling, models may serve only to “confirm our own biases and support incorrect intuitions” (Oreskes et al., 1994 p. 644).

Kloprogge and van der Sluijs (2002) believe that an analysis of “choice processes” is useful in modelling for decision-making because it assists model users to assess “what choices to make transparent to users, stakeholders and peers, and assists them in incorporating multiple views in the model” (p. 101). According to Oreskes et al. (1994), questions to ask of a model are “how much is based on measurement of accessible phenomena, how much is based on informed judgment and how much is convenience?” (p. 643).

Burgman (2001) argued that the assessment of uncertainty in ecological risk assessments should be formalised. If uncertainty analysis is not formalised as part of a modelling process, risks are likely to be assessed subjectively by the resource manager or stakeholder. According to Burgman (2001), subjective assessment of uncertainty can be influenced by cognitive biases including:
• Judgement bias: Overconfidence in one’s ability to predict, or the ability of the model to predict;
• Framing effects: Where judgements of risk are sensitive to the prospect of personal gain or loss; and
• Anchoring: The tendency to be influenced by initial estimates, and insensitivity to sample size.

3.5 Modelling and Decision-making

According to Brezonik and Renwick (2003), scientific understanding and technical knowledge may not be the main limiting factors in solving non-point source pollution problems. They state that “the principal limiting factors seem to be related to the socio-cultural, economic, and political environments in which technical solutions need to be implemented” (p. 151). They suggest that limitations in existing legal authorities, economic constraints, conflicting attitudes and priorities among key stakeholders, and the highly disaggregated nature of the pollutant contributors have slowed the pace of progress in solving non-point source pollution problems in rural America. Whilst the science of measuring and conceptually understanding the problem of diffuse pollution Australia may not be as advanced as it is in the USA (Young et al., 1996), many of these limiting factors may also be operating here.

Conventional (‘normal’ or ‘reductionist’) scientific approaches value concepts of peer review, statistical precision, completeness, observability and experimental validity, but for policy-makers, the value of research is more often measured in terms of feasibility, utility, efficiency and stakeholder acceptance. The focus for decision-makers is not so much on demonstrable proof or accuracy of models, but on the demonstrability of the legitimacy of the outcome (Haag and Kaupenjohann, 2001).

Different principles may be said to apply when models are used for policy development and decision support. For example, Simonovic and Bender (1996) emphasise these characteristics:
 • Problem identification;
 • Problem formulation (a social, learning process that helps clarify the issues);
 • Adaptability (opportunity to ask ‘what if’ questions);
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- Facilitation (the use of evaluation tools, artificial intelligence techniques and visualisation capabilities); and
- Interaction (the ability to manipulate and understand the output of the DSS in a way that is appropriate to potential users).

It could be said that for better decision-making, the use of models and prediction needs to be process-oriented. Pielke Jr et al. (2000) believe that managers and those affected by models must question predictions, and even the need for prediction.

Yet conventional scientific models and modelling approaches continue to appeal to decision–makers who wish to make predictions that can be used to confirm the need for action, set priorities and legitimise policy decisions. The expectation of decision-makers is that scientific modelling will help them understand environmental systems and reduce the level of uncertainty associated with proposed management actions. In the mind of the decision-maker, the utility of scientific models for decision-making is confirmed by scientists who support the use of their models in the policy arena in order to justify public investment in their research (Pielke Jr and Stohlgren, 2002).

However, the needs of decision-makers may not be well-served by the current suite of scientific tools. Pool (1990 p. 673) contends that the questions that are “most interesting scientifically may not be the questions that are important to setting policy”. When a topic becomes a public policy issue, a new suite of non-scientific factors enter policy-making, especially when scientific evidence is inconclusive.

3.5.1 Barriers to Modelling for Decision-making

Much scholarship from writers of different disciplinary backgrounds has been dedicated to exploring ‘barriers’ to the effective use of scientific information or models in environmental management. (e.g., Cullen, 1990; NRC, 1990; Loucks, 1992; Roots, 1992; Barnes, 1995; Somlyódy, 1997; Bernknopf and Herman, 1998; Berry et al., 1998; Cortner et al., 1998; Bouyssou et al, 2000; Dent, 2000; Argent and Mitchell, 2003; Bosch et al., 2003; Brezonik and Renwick, 2003; Cash et al., 2003; Landry et al., 2003; Caminiti, 2004; McIntosh et al., 2004; van Kerkhoff, 2005; Saloranta, In Press). Roots (1992 p. 89) defined such a barrier as:
“an identifiable thing or problem - physical, scientific, technological, institutional, or economic and cultural - that can be built up, broken down, changed in position or otherwise altered to change the relationships or flows between the two entities that are kept apart by the barrier.”

Barriers can be classified a number of ways. In March 1990, the Institute for Research on Environment and Economy sponsored a Workshop on the topic of “Breaking the Barriers to Environmental Information”. The Workshop explored a number of barriers to the “production, distribution and dissemination of environmental information” (Needham and Rapport, 1992 p. 85) which were published in Volume 20 of Environmental Monitoring and Assessment. At the workshop, and in this chapter, barriers have been classified into 3 major types: Scientific barriers; technological barriers; and political, social and institutional barriers. Each category of barrier was described by Roots (1992 p. 92):

“The scientific limitations - the problems that arise because communicable knowledge and understanding is, for one reason or another, simply insufficient to enable satisfactory information about the environment, in all its complexity, both natural and human, to be obtained or passed along to others. Some important environmental questions are, at present, scientifically unanswerable.

The technological problems - the questions of method, scale, quantity and complexity, to say nothing of problems of expense, organization, and communicability that stand between observation and measurement of environmental characteristics or processes at the one end of the information transfer process and the possession of environmental knowledge by the public or policy-makers at the other.

The institutional and political problems - the difficult issues surrounding the institutional systems that are necessary if systematic information is to be obtained, and the fact that all such systems necessarily involve constraints, biases, conflicts of purpose, and selectivity concerning the information obtained and its interpretation or dissemination.”
3.5.1.1 Scientific Barriers

Findlay (1992) divided scientific barriers into two kinds: 1) barriers related to the nature of environmental science itself, which are data-related; and 2) barriers associated with scientists as the practitioners of environmental science.

1. Data-related Scientific Barriers

Within environmental science itself, Findlay (1992) identified four sub-categories: i) Data limitations, ii) Data dissemination limitations; iii) Data synthesis limitations; and iv) Limitations on the integration of data into environmental management programs.

i. Data limitations: Those barriers relating to quality or quantity of data that can be used to support environmental management.

ii. Data dissemination limitations: The effectiveness of environmental information in resource management is dependant on the efficiency of the dissemination of data and information. Data dissemination limitations often relate to the willingness or ability of individuals or agencies to share data, and the medium within which data and information is stored or published.

Scattered or inaccessible information is not useful for planning and decision-making. Crispin and Dupuis (1992) suggest that scientific information can be more useful if accurate and specific information from scattered sources is centralised and organised for easy retrieval. However, they argue that this is often overlooked “for more theoretically or technologically glamorous alternatives” (p. 181) that are not always conclusive enough to be useful for planners and managers.

iii. Data synthesis limitations: Data synthesis involves identifying data and information that is relevant to management from the ever-increasing amount of data that is being generated by researchers, and presenting the data in a form that is useful to managers.

iv. Limitations on the integration of data into environmental management programs: Much of the data that is collected by researchers is aimed at better understanding environmental processes, but to be useful in management, data needs to have a larger applied component, and be aimed at solving or better
managing an environmental problem and informing management program objectives.

Of the four sub-categories, Findlay (1992) highlighted the integration of data into environmental management programs as “…perhaps the major impediment to advances in environmental science” (p. 122). According to Roots (1992), “we should be aware of the problems as well as the needs of translating information into forms others can use” (p. 91). He points out that “all translation involves interpretation, with some loss or distortion of cultural content or significance” (p. 91).

2. Practitioner-related Scientific Barriers

In relation to the practitioner-related scientific barriers, Findlay (1992) argued, “scientists are either unwilling or unable (or perhaps both) to address themselves to the larger social and political implications of their work” (p.122). He argued that scientists are very comfortable in the role of investigating and analysing “what is”, but are largely unwilling to “help decide what should be” (p.123), and regard this role as one that is outside their purview.

Manning (1992) elaborates on this issue, noting that scientific recognition is most often achieved through focussed research on very specific and isolated variables. Our institutions, he argued, “conspire to reinforce these scientific barriers, rewarding precision and often penalizing broader and more integrative [and applied] approaches” (Manning 1992 p.125).

3.5.1.2 Technological Barriers

According to Rapport (1992), the key technological barrier or limitation may not be technology itself, but the problem of developing appropriate technology. He postulates that the nature of this barrier may therefore be more conceptual than technological. Two issues that have proven difficult technological barriers are related to 1) the limits of current scientific understanding; and 2) temporal and spatial scales. Each is discussed below:
1. Limits of Scientific Knowledge

Science’s quantitative understanding of the elements needed to model water quality is relatively immature when compared with other disciplines in which modelling also plays an important role, such as flood forecasting (Refsgaard et al., 2005a). The integration of the science into decision-making is therefore less mature and more problematic than for some other disciplines.

2. Temporal and Spatial Scales

Roots (1992 p.91) contended, “the problem of change of scale, and relationships between different scales in both space and time, is perhaps the most difficult, technically and conceptually, in the whole field of environmental information”. The consideration of multiple scales (temporal and spatial), which is a relatively new phenomenon in environmental management, means that “advocating a single perspective that encompasses everything in a system becomes increasingly difficult – plus less effective (Poch et al., 2004 p. 858).

3.5.1.3 Political, Social and Institutional Barriers

Catchment management is both scientifically and institutionally complex. Gunderson et al. (1995) argued that most management proposals can be threatening to organisations or interest groups. The perceived threats relating to management proposals can create a climate where individuals become acutely aware of uncertainty, and administrative interests may be best served by avoiding decision-making. Modelling or other scientific work, in best cases, offers environmental managers a possibility of greater certitude, but, at worst, creates a need for more research that can be used to avoid potentially controversial decisions.

Cortner et al. (1998) examined political and institutional barriers (and “incentives”), including organisational structures, co-operation across institutional and land boundaries, the large scales and broad focuses for management and research. Other barriers relate to legislative arrangements, relationships between departments, agencies and between public and private interests.

Bosch et al. (2003) lists a number of other institutional issues that might sometimes represent institutional barriers to the use of science and models, including mistrust in
sharing information, lack of funding for long-term knowledge-building approaches (such as adaptive management), slow responses to new knowledge and conflicting ecological, economic and social values.

Such cultural barriers are typically viewed as ‘soft’ systems problems (see Checkland 1994). Cultural barriers are those that relate to socio-cultural problems (i.e. problems relating to people, groups and organisations and the way they think and interact). These barriers are usually ill-defined and difficult to quantify.

### 3.5.2 Evaluating the need for modelling

Based on much of the published literature, it could be assumed that models that predict nutrient export to water bodies are essential for the good management of catchments. But some authors question the role of prediction as a useful tool for good decision-making. Loucks (1992) and Pielke Jr (2003) argued that the use of models as tools for prediction may make the decision-maker’s job more difficult. Whilst it may be the original intention of a decision-maker to use a model to quantify aspects of a management problem and minimise the uncertainty associated with proposed policy and management actions, advancing scientific knowledge may in fact have the confounding effect of making obvious the “vast complexities associated with phenomena” (Pielke Jr, 2003 p. 121), thereby increasing political controversy and producing a perception of a need for more research rather than management action.

Oreskes et al. (1994) and Oreskes (1998) argued that verification and validation of numerical models of natural systems is impossible, essentially because they require input parameters that are never completely known. They contend that the predictive value of models will always be open to question, and that the primary value of models is that they can promote debate and organisational and individual learning.

Pielke Jr et al. (2000) contended that the conventional practice, whereby a policy-maker recognises a problem, asks scientists to do research to predict natural behaviour associated with the problem then deliver predictions to decision-makers with the expectation that they will be useful and well used, is a practice that rarely functions well in practice.
Nevertheless, some authors have remained optimistic. For example, Rizzoli and Young (1996) consider that the development of computer-based environmental decision support system is “an extremely worthy research goal” (p. 239), although they concede that this ambition will not be realised in the near future.

3.5.2.1 Alternatives to Simulation and Prediction

Relatively few workers attempt to list alternatives to simulation or prediction using models. However, Pielke Jr and Stohlgren (2002) listed three alternatives; [i]“…no-regrets public policies, [ii] adaptation [i.e. adaptive management], and [iii] better planning and engineering. The authors argue that “alternatives to prediction must be evaluated as a part of the prediction process”.

3.6 Reflection

Decisions must be made despite uncertainty. In the case of diffuse nutrient management, uncertainty relates not only to the environmental factors, such as climate and other natural processes, but also the social and economic factors that affect land use and management.

Managers often use predictive models with little understanding of their real utility (or otherwise) in solving policy problems. Although counter-intuitive, it cannot be assumed that a predictive model designed for scientific research is a useful basis for decision-making and policy development (Oreskes et al., 1994; Pielke Jr et al., 2000; Pielke Jr and Conant, 2003). This is partly because advances in knowledge that may come from model-based research do not necessarily reduce uncertainty and may not increase the confidence that managers and stakeholders have in any proposed management action. Sometimes complexity in models will improve accuracy, but this may also increase uncertainty since each variable added to a model raises questions and provides another source of uncertainty to the model prediction. This is referred to by Oreskes et al. (1994) as the “complexity paradox”.

Whilst this may be desirable from a scientific or heuristic perspective where the aim is to discover the ‘truth’, and better understand how things work, in a decision setting this can have the perverse effect of increasing controversy (Oreskes et al., 1994; Pielke Jr et al., 2000). Decision-makers should seek to understand uncertainty,
including its sources and potential reducibility as it affects the decision-making process (Pielke Jr et al., 2000). This will allow the decision-maker to determine if the model results provide a solid basis for decision-making, or if more research is needed. Sometimes the uncertainty may be so great and irreducible that decision-making should turn to alternatives to prediction.

To determine the utility of a model for decision-making requires methods of evaluation that are more multi-dimensional than scientific evaluation. In the scientific arena, the validity of a model and its outputs is traditionally determined by peer review, and based on the pedigree of the model and statistical evaluation of modelled versus observed results. However, in the socio-political arena occupied by managers and decision-makers, validation becomes a negotiation process involving an extended peer group of all parties potentially affected by decisions and management actions (Haag and Kaupenjohann, 2001). Essentially, each stakeholder develops their own validation criteria based on their value systems, interests and insecurities.

The consequence of this is that modelling for decision-making requires that models be evaluated for specific decision problems. Special attention should be paid to the framing and transparency of models, and for communicating uncertainty. The information conveyed by models should help organise knowledge in a way that facilitates better decision-making. Unfortunately though, it is relatively easy to invalidate a predictive model of a complex natural system, but very difficult or impossible to validate the same model.

Uncertainty and the management of uncertainty is a recurrent theme in literature about modelling for decision-making. The management of uncertainty should be viewed as both a scientific and cultural issue.
4 METHODOLOGY

4.1 Introduction

This chapter provides a rationale for the research methodology used in the thesis. Data collection and analytical methods are also discussed.

4.2 Background

This research began with the fuzzy premise that ‘barriers’ prevented a catchment management organisation, the SCA, from choosing and using models effectively. A combination of literature review and experience (see McNamara and Cornish, 2004) suggested that issues such as model complexity and transparency and participation between modellers and decision-makers might provide clues to better model selection and function. This type of problem is well-suited to research using qualitative methods such as action research.

4.3 Qualitative Research

Qualitative research is essentially an inductive research methodology that allows the researcher to develop a holistic understanding of complex and ill-defined problems. The methodology relies on transcripts, documents and observation as raw data, rather than the quantitative measurements that are relied on in quantitative research.

The research method used in the current research bears some similarity to “Grounded theory” (Glaser and Strauss, 1967), which is an explicitly ‘emergent’ methodology. Rather than focus solely on one or more premise or hypothesis, in this research, the goal was to develop concepts from patterns that emerge as the research progresses through different stages or activities.

4.3.1 Experiential Learning

Experiential learning, or ‘learning by doing’ is a learning tool that allows the user to challenge existing theories and practices, against personal experience through active involvement (i.e. ‘immersion’) in the phenomenon being studied, or through role play.
The most significant pioneer in this branch of pedagogy was Dewey (1933), his work influencing many later proponents, notably Lewin (1946) and Kolb (1984).

A feature of experiential learning given high importance by all influential practitioners is the notion of ‘reflection’. Dewey (1938) is credited with the phrase “Experience plus Reflection equals Learning”.

4.3.2 Action Research

Whereas experiential learning can refer to any kind of learning through experience, action research is a specific type of experiential learning that was conceptualised in the 1940s by Lewin (1946) and further developed by Kolb (1984) and other practitioners (e.g., Carr and Kemmis, 1983; Bawden, 1990; Zuber-Skerritt, 2000; Coghlan and Brannick, 2001). Action research is regarded as a non-traditional, pluralist form of applied research that differs from conventional (quantitative and disciplinary) research (and some kinds of experiential learning) in that the researcher not only seeks to provide a contribution to knowledge, but also seeks to develop opportunities to facilitate change, or ‘situation improvement’ through the direct involvement of the researcher in the issue or problem being studied (Williams, 1992).

Action research, which involves active intervention, can be more insightful than non-participative case study-style approaches that include interviewing and/or observing people in certain situations. Action research places more emphasis on what practitioners actually do, rather than what they say they do (Avison et al., 1999) and is suited to developing theories that explain what happens in organisations. Action research is particularly useful for exploring issues at the boundary of science and management because action research combines theory and practice (and therefore researchers and practitioners) using methods that are generally mutually acceptable to both sides (Avison et al., 1999).

The main premise of action research is that the active involvement of a researcher in a complex human system can lead to better understanding and beneficial outcomes that may not be obtained through traditional research methods that involve recording events and formulating theses whilst attempting to remain disparate from the phenomenon under study (Stringer, 1999).
Action research typically involves a four-step process of ‘planning’, ‘acting’, ‘observing’ and ‘reflecting’ (Zuber-Skerritt, 2000) (Figure 4-1). Each phase of the cycle is described in Table 4-1.

![Four-step cyclical action research process](image)

**Figure 4-1. The four-step cyclical action research process.**

<table>
<thead>
<tr>
<th>Phase</th>
<th>Typical activities</th>
</tr>
</thead>
<tbody>
<tr>
<td>Plan</td>
<td>Planning new actions, including planning steps for monitoring and review of outcomes.</td>
</tr>
<tr>
<td>Act</td>
<td>Acting is where the researcher carry out a plan and makes use of prior learning.</td>
</tr>
<tr>
<td>Observe</td>
<td>The researcher seeks input from participants and observers, seeking observations or measurements that are relevant and unbiased. This stage may also include document and sharing observations.</td>
</tr>
<tr>
<td>Reflect</td>
<td>In this stage, the researcher reflects on what worked and what didn’t and thinks about what the results mean for changed practice and incorporate learning into the next plan. New conceptual models are developed.</td>
</tr>
</tbody>
</table>

Subtle variations of this schema have been advanced by other workers. For example, Carr and Kemmis (1983) argued that ‘reflection’ should occur throughout the cycle, not just at the end of the planning, acting and observing stages, and therefore used the terms ‘plan’, ‘act’, ‘monitor’ and ‘evaluate’. Juch (1983) described the four activities as ‘addressing’, ‘doing’, ‘sensing’ and ‘thinking’; Kolb’s cycle (Kolb, 1984) referred to ‘active experimentation’, ‘concrete experience’, ‘reflective observation’ and ‘abstract conceptualisation’; and Coghlan and Brannick, 2001 used the terms ‘action planning’, ‘action taking’, ‘action evaluation’ and ‘diagnosis’.

Authoritative practitioners of action research, including Lewin (1946), agree that positive outcomes from action research are more likely when research involves multiple iterations of the action research cycle. Barr and Sharda (1997) found that single iterations of the action research cycle may reduce the effectiveness of decision-
making or have negative consequences. Argyris and Schön (1974) differentiated learning activities that involved single iterations of the cycle from activities that involved multiple iterations using the terms “single loop learning” and “double loop learning”. Throughout most action research studies, an investigation is typically reinforced as more sophisticated understandings of the important issues emerged through a reiterative cyclical process commonly referred to as the “action learning spiral” (Lewin, 1946; Paton, 2001) (Figure 4-2).

![Figure 4-2. A spiral of action research cycles (Zuber-Skerritt, 1995 p. 13: reproduced in Hatten et al., 1997).](image)

The repeated instances of critical reflection that occur in action research projects with multiple iterations of the action research cycle can also be seen as a technique for ensuring rigour in the researcher's analyses, since each cycle provides an opportunity to test assumptions and “search for disconfirming evidence” (Dick, 2002).

In an action research project, there may be cycles within cycles. Some cycles may last only a matter of minutes, whilst others may last through the entire period of the study (Dick, 2002).

Some salient features of action research are listed below:

- Action research is ‘context-bound’ and addresses real-life problems;
- Action research deals more with language than numbers;
• Action research is a cyclical series of steps that tend to recur in a similar sequence;
• Critical reflection on the process and the outcomes of each cycle are important;
• The action research approach is responsive and allows refinements to ‘fuzzy’ questions, ‘course corrections’ and ever-improving understanding. Each cycle helps the researcher to identify the important questions and clarify the issue at hand;
• As with all forms of qualitative research, the researcher is the primary instrument for data collection and analysis;
• Clients are involved in the research process;
• The focus of action research is ‘situation improvement’. Change is made possible because of improved understanding of the researcher(s), stakeholders, or both; and
• Action research involves the interdependence and integration of theory and practice, research and development, thought and action.

(see Dick, 1993, 1997; Levin and Greenwood, 2000; Smith, 2001; Zuber-Skerritt, 2000, Merriam, 2002)

4.3.3 Limitations of Action Research

All methods of inquiry suffer from limitations that can impair the quality of research outcomes. The main limitations ascribed to action research are listed below:

• Action research is typically small-scale and limited to a single organisation. The results of the research may not be representative or generalisable;
• The research is undertaken in the real-world. Some variables that affect outcomes cannot be controlled as they would be in a conventional, experimental research;
• Active involvement with others is a key feature of the research, but can be constrained by ethical considerations;
• Action research is a partnership between the researcher and an organisation. Leadership of research and ownership of findings can be contested;
• Action research relies heavily on qualitative data and action researchers aren’t detached or impartial. Proponents of positivistic approaches may be uncomfortable with action research;
• The research is often of lesser interest to disciplinary scientists and the research may not be as widely publishable as conventional forms of research;
• There are no generally agreed criteria for assessing the quality of action research.

(see Adelman, 1989; Hammersley, 1992)

In an effort to give the research global relevance, the literature review in the current study is comprehensive, and the conclusions drawn at the end of the research draw heavily from the review.

4.3.4 Other Qualitative Methods

Other “theory-creating” (see Järvinen, 2000) qualitative methods that may be viewed as alternatives to action research are: 1) Case Study research; 2) Ethnographic research; and 3) Grounded Theory.

1. Case Studies: Case studies are intensive empirical inquiries that are used to investigate contemporary phenomena within their real-life context, especially when the boundaries between the phenomena and their context are not clearly evident (Yin, 2002). Case studies typically use a range of methods, including interviews and observations. Action research is sometimes considered to be a form of case study (Cunningham, 1997).

2. Ethnographic Research: Ethnographic researchers (ethnographers) immerse themselves in the lives of the people they study over an extended period of time and seek to place the phenomena studied in their social and cultural context (Lewis, 1985). The greatest strength of ethnographic research is that the researcher develops an intimate familiarity with the phenomenon being studied.

3. Grounded Theory: In grounded theory studies, researchers seek to develop a theoretical account of a research topic that is grounded in qualitative and qualitative data that has been systematically or iteratively gathered and analysed. Martin and Turner (1986 p. 141) describe grounded theory as “an inductive, theory discovery
methodology that allows the researcher to develop a theoretical account of the general features of a topic while simultaneously grounding the account in empirical observations or data”. Grounded theory has thus been described as a means of ‘theory generation’ rather than ‘theory testing’ (Glaser and Strauss, 1967). Grounded theorists seek objectivity through reliance on empirical validation of theoretical interpretations and adherence to rigid procedures that include codifying data and delaying the literature review until after theories emerge.

Each approach has weaknesses that would impact negatively on the quality of research aimed at influencing and improving practice. Conventional case study research focuses primarily on what practitioners say rather than what they do, and does not give the researcher access to the insights that can arise from immersion and intervention in the phenomenon being studied (Avison et al., 1999). Ethnographers achieve an in-depth, situation-specific understanding of a phenomenon, but ethnographic studies tend to lack generalisability and often fail to inform or influence practice (Hammersley, 1992). Grounded theory has been criticised because the method fails to integrate emergent theories with existing knowledge and, because of reliance on codified data and empirical validation, the method may obscure the broad context or ‘fullness’ of the phenomenon being studied from the researcher (Charmaz, 2000).

4.4 Research Procedure and Methods

As discussed in chapter 1.2, this research had two broad aims that were:

1. to develop two models differing in sophistication, and to use them to prioritise nutrient pollution sources and calculate nutrient loads;

And, through the participatory development of the models,

2. to learn methodological lessons that catchment managers can apply to choose and use models more effectively.

In order to address these aims, ‘action’ was in the form of five planned activities, including:

1. a review of relevant literature from diverse disciplines;
2. the holding of two workshops; the first exclusively involving SCA managers to identify their modelling needs, and the second involving water quality scientists and modelers to discuss methods for meeting the modelling needs;
3. the development and use of two export coefficient nutrient models;
4. a focus group discussion involving key staff in the SCA; and
5. a review of published guidelines for good modelling practice in environmental management.

Note was also taken of statements or behaviour at numerous meetings and seminars, mostly with SCA staff, that were relevant to the research questions.

Each of these actions provided opportunities for data gathering using documents, observations and transcripts. It is typical in an action research thesis to access certain specialised literature as the research progresses (Dick, 1997). In this thesis, the review of good modelling practice guidelines (Action 5 above and Chapter 8 in this thesis) is an example of this and was not foreshadowed at the outset of the research and was planned after the focus group (Action 4) had been conducted.

Methods describing the workshops, the development of the models and the functioning of the focus group are contained in chapters 5, 6 and 7 respectively. Although the modelling aspect of the research produced a standalone outcome, which is discussed in detail in chapter 6, the process of selecting, using, enhancing and presenting model outputs was, in effect, used as a vehicle for engaging with the SCA and the modelling community. Active engagement provided opportunities for the researcher to learn the methodological lessons that catchment managers can apply to choose and use models more effectively and opportunities to communicate the author’s ideas back to the SCA.

Participation of the author in this research was both active (i.e. as an interactor) and passive (as an observer). Active participation occurred primarily through the involvement of the author in the modelling activity, workshops and focus group, which were action research activities. However, other activities also provided valuable data and information that helped inform the author’s understanding of relevant issues. Examples of some useful activities included attendance at model training and model development workshops, occasional SCA and other seminars and
meetings of the SCA/UWS collaborative research group and other collaborative research teams.
5 WORKSHOPS

This chapter describes and presents the results of two workshops. The first workshop involved SCA managers and sought to identify the modelling needs of the SCA. The second workshop involved water quality scientists and modellers and sought an understanding of the current state of knowledge about nutrient modelling at catchment scale in an effort to identify the most appropriate modelling approach to meet the SCA’s modelling need. At the time of the second workshop, an ‘enhanced export coefficient model’ was being developed by the author based on the literature review and the outcomes of the first workshop. Further details about the development of the model are given in chapter 6.2.

5.1 Workshop One: Catchment Managers

5.1.1 Introduction

Workshop One was organised to identify the modelling needs of the SCA and seek information on how models might be more usefully applied to planning and decision-making in the SCA. This the first of two workshops that were to eventually lead to the development of a catchment-scale diffuse nutrient model intended to meet the needs of the SCA’s planners and decision-makers.

5.1.2 Structure of the Workshop

The half-day morning workshop commenced with a one hour introductory discussion to describe the project and present and clarify the intended meaning of each workshop question (See Chapter 5.1.4). After the introduction, participants divided into three ‘break out’ groups for 90 minutes to ‘brainstorm’ and answer three questions (see 5.1.4) and record their answers on a flipchart. At the end of the group session a nominated speaker from each group presented their answers to all participants. The workshop was facilitated by Dr. Richard Davis, an experienced water resource scientist. The author was present to take notes and participate in the proceedings. A brief paper summarising the main issues raised and inviting comment was sent to each participant after the conclusion of the workshop.
5.1.3 Attendance

Attendance at the first workshop was open to all SCA staff responsible for choosing or using models that were intended to inform SCA planning or decision-making. Most interest came from staff involved in the development and use of various catchment-scale water quality models (i.e., predominately in the Environment and Planning Section). In order to enhance both the representativeness and diversity of the workshop group, the participation of staff from other sections was actively sought. A small number of external researchers involved in collaborative projects with the SCA also attended. All non-SCA participants were actively engaged with the SCA and had interest and experience in the application of scientific models in decision-support for SCA planning or policy development. Twenty-two participants attended the first workshop. There were 16 SCA attendees, comprising staff from Environment and Planning (11), Catchment Operations and Major Projects (3), Policy and Governance (1) and Bulk Water (1). Other attendees came from University of Technology Sydney (1), CRC for Freshwater Ecology (1), Hawkesbury Nepean Catchment Management Authority (1), State Forests (NSW) (1) and University of Western Sydney (2). Names and other personal details of the attendees have been suppressed to protect their privacy.

5.1.4 Questions and Introductory Discussion

Questions

In order to elicit discussion that would inform the two aims of the research (i.e., i) to develop models that identify nutrient pollution sources and calculate nutrient loads; and ii) to identify criteria that catchment managers can use to choose and use models more effectively), three questions were posed of participants:

1. What are the varied modelling needs of SCA managers and planners, and associated users including other government agencies and consultants?
2. What barriers obstruct the embedding of models in the management and planning processes of the SCA and related external decision makers (e.g., local governments)? How can these barriers be overcome?
3. What general principles (evaluation criteria) can be used by model developers and users to ensure that a model or ‘modelling framework’ meets the needs of decision-makers?
The questions were developed based on early observations and interactions with the SCA (see Chapter 1.1.4), where it appeared that unarticulated ‘barriers’ were limiting the effective use of environmental models as decision-support tools in the SCA. The literature reviewed in this research contains numerous references to ‘barriers’ that inhibit the effective use of science and modelling in environmental decision-making (see Chapter 3.5.1), but at this stage of the research, its relevance to the SCA was unknown. The questions were open-ended and designed to trigger broad-ranging discussion and surface issues relevant to effective model selection and use by the SCA, which may or may not have been discovered in the literature review.

To ensure that the questions were cogent and relevant to participants, key staff in the SCA who knew the purpose of the study and the characteristics of participants were given an opportunity to review the questions. The moderator, who is experienced in guiding group discussions in the field of natural resource management, assessed the order and content of questions, room arrangement, and group composition.

Discussion Issues
Although the focus of the research was diffuse source nutrient models, workshop participants were asked to draw upon their experience with a wider range of models in answering these questions. It was also agreed that the term ‘model’ should be interpreted broadly to include non-numeric decision aids such as decision-trees and checklists, because these could be more appropriate ‘models’ for some decisions than classical, numerical models. The concept of a ‘modelling framework’ was questioned by participants and discussed. A modelling framework was agreed to include all aspects of a model and its use, and therefore at a broad level, could include the support structure that surrounds the use of a model, including training, acceptance by senior managers, technical support and data availability, or, at a narrow level, the approach or concepts upon which a model is based, for example an ‘EC’ approach.

5.1.5 Results

**Question 1.** What are the varied modelling ‘needs’ of SCA managers and planners and other associated users?

Participants described the modelling needs of the SCA in two ways. The first focussed on the ‘need’ to manage pollution, and identified four aspects ways that
models could be used to improve catchment manager’s knowledge of the pollution problem and how to address it. The aspects were:

- estimation of current and/or predict future pollution fluxes from source to stream and reservoir at a range of spatial and temporal scales;
- better understanding of the transport and fate of pollutants at a range of spatial and temporal scales;
- identification of pollution sources; and
- identification and prioritisation of abatement actions.

The second way of describing the modelling needs of the SCA was to describe these in terms of the major programs managed by the SCA. These were said to include:

- Assessment of future land uses for the Neutral or Beneficial Effect Test (NorBE)
- Land use planning, including the Strategic Land and Water Capability Assessment (SLWCA) and Local Environment Plans (LEPs)
- Implementation of Rectification Action Plans (RAP)s
- The Healthy Catchments Program (HCP)
- Environmental reporting
- Economic assessments and pricing
- Regulatory enforcement and pollution monitoring
- Bulk water management

Participants also discussed many of the technical requirements of models, such as issues relating to temporal and spatial scale, the natural and cultural features that needed to be included in a model, model accuracy, and a concept that several participants came to refer to as “consistency”. Each issue is discussed below:

**Scale Issues**

It was widely acknowledged that the spatial and/or temporal scale in which modelling should be undertaken depends on the task for which the model is being used. In general however, the feeling was expressed that assessment of DAs for the NorBE test required farm-scale assessments, while RAP required sub-catchment scale assessments. Nevertheless, most participants preferred the idea that there should be a basic physical continuity in calculations across temporal and spatial scales.
Processes Modelled

Some felt that models should cover the full range of transport pathways, especially for assessing DAs, and that models should be able to assess the negative or positive impact of different management practices.

Accuracy

The general view was that in order for models to be effectively implemented by the SCA, they needed to be data-efficient and yet produce results that were “robust” or “defensible”. In the introductory discussion, one participant stated that:

“Models need to be defensible...not 100%, but uncertainty must be within ‘reasonable’ bounds.”

Another argued that:

“[Models] should be based on good science, use limited data...and be usable in court.”

Participants did not offer suggestions as to how “reasonable” should be defined, or how or even if uncertainty should be measured. It was unclear what level of accuracy or model sophistication was necessary to facilitate model-based decision-making of any kind. As one participate said “We need a guide to what level of sophistication is necessary”.

Consistency

Another view that was widely held and perceived as being very important to the effective use of models was that models used within the SCA, and by associated users such as contracted analysts and local government planners (i.e. for LEPs), should be “consistent”. The term ‘consistency’ took on a number of meanings, depending upon the context of the discussion. When talking about consistency, participants referred to:

- The use of a suite of accredited models by all parties engaged in activities for which SCA compliance was required (local governments, developers);
- The use of a suite of models that were not based on conflicting assumptions and maintained physical continuity across scales;
• Interoperability of models and efficient data-transfer between models;
• A consistent understanding, at all levels of the organisation, of how models should be used to inform decision-making.

One participant explained:

“The SCA needs models that have an internal consistency of assumptions across scales.”

When comments similar to this were teased out, the argument seemed to be that although different models may be used for different purposes, there should be a high degree of methodological consistency between the different approaches, and that this should apply across different spatial and temporal scales.

Some participants argued that it should be possible to aggregate predictions from models used at a lower spatial or temporal scale to a higher spatial or temporal scale, such that the aggregated predictions are comparable to those that would be obtained by a model operating at the higher scale.

Another aspect of consistency that was discussed was a perceived need to ensure that external organisations, such as consultants reporting for DAs, use models that are “consistent” with those used by SCA. This implication that was discussed in relation to this was that the SCA may need to promulgate a set of accredited models for external users.

There was also a strong feeling that there needed to be some level of consistency throughout the hierarchy of the organisation. In the introductory session, one participant stated:

“We need a language and understanding of [model] use that extends up to the executive”

**Question 2.** What are the barriers to ‘embedding’ models in the SCA and related management and planning processes? How do you overcome those barriers?

The barriers that participants mentioned can be divided into two broad and sometimes overlapping categories: 1) Scientific/Technical barriers and; 2) Institutional barriers. Scientific/Technical barriers are those that may be mitigated, although often not
completely resolved, through additional research, information gathering or training, while institutional barriers are those that may be mitigated through cultural or structural change within the institution.

The barriers that were identified by participants are listed below. For clarity, they have been grouped into Scientific/technical barriers and institutional barriers; although no such distinction was made at the workshop.

**Scientific and Technical Barriers**

- **Data deficiencies**: It was commonly argued that the lack of monitoring and experimental data (at relevant spatial and temporal scales) meant that models were not as accurate as they could or should be;
- **Inconsistency**: Several participants argued that the use of models with different underlying assumptions or methodological underpinnings weakened the SCA’s confidence in models and was therefore a barrier to the effective use of models;
- **Lack of expertise**: Model users inside and outside the SCA often have insufficient expertise to select appropriate models and are unable to understand, interpret or critique modelling approaches and model outputs;
- **Uncertainty**: There is a lack of data at the temporal and spatial scale required to validate models. The accuracy of models is generally unknown;
- **Complexity, transparency and confidence**: It is sometimes difficult to use model outputs because the assumptions used in some models are not clearly articulated by model developers or the analysts that run models. Users can be sceptical and lack confidence in model outputs;
- **Model availability**: Sometimes the required model is simply unavailable (the absence of a useable pathogen transport and decay model was given as an example).

**Institutional Barriers**

- **Data acquisition cost**: The cost of acquiring data to calibrate and validate models was understood to be high. Funds might be available if senior management appreciated the link or ‘trade-off’ between data and model reliability;
• **Lack of procedure**: There are no standards with which to assess model performance before using the model;

• **Lack of co-ordination**: There is no framework within the SCA for using models. For example, one group highlighted the lack of any plan for training or the provision of technical assistance to model users;

• **Lack of vision**: There is a lack of support or understanding of the modelling needs of the SCA at the General Manager level and higher levels in the SCA (senior management were often referred to as “the Executive”). Several participants felt that the SCA needed a high-ranking manager to “champion” the cause of modelling, or to help achieve “Executive-level buy-in”;

• **Poorly-defined objectives**: There is a lack of clarity regarding which models are needed and why. Some participants felt that there needed to be clearer and more consistent modelling objectives;

*Overcoming the Barriers*

In order to use models more efficiently and effectively in the SCA, a number of actions were proposed by participants:

• **More data**: There needs to be a more concerted effort to collect monitoring data (and where necessary, experimental data) at the appropriate temporal and spatial scales for calibrating and validating models;

• **Identify expertise and define roles and responsibilities**: Some participants felt that the SCA should invest in identifying staff that have expertise in different aspects of modelling, including coding/programming, running models, modifying models and applying model outputs. It was also argued that the SCA needed to invest more in training and education of SCA staff;

• **Be adaptive**: A participant felt that the SCA could manage predictive uncertainty in models by adopting an approach focussed on continual improvement akin to ‘adaptive management’;

• **Increased focus on sensitivity analysis and validation**: It was argued that confidence in models could be improved if more attention was paid to appropriate testing of models through sensitivity analysis and formal model validation;
• **Standardise operating procedures**: The idea of having a “**Standard Operating Procedure**” (SOP) was discussed and widely supported. The SOP could take the form of a manual, flow chart, decision tree or other tool that would give formal guidance to the use of models to support decision-making in the SCA; and

• **Support from senior management**: Participants re-iterated the idea that senior management “buy-in” (e.g., from the GM and Chief Executive Officer) would lead to more efficient and effective use of models.

**Question 3. What general principles (evaluation criteria) can be used by model developers and users to ensure that a model or modelling framework meets the needs of decision-makers?**

The answers given in question 3 reinforced many of the ideas discussed in questions one and two. Although the responses were diverse, the criteria raised by participants fit into 3 broad categories that Sargent (1982) identified as being useful for validating the accuracy and usefulness of models: i) Conceptual criteria, which test the conceptual soundness of a model; ii) Quantitative criteria, which test the accuracy of a model, or how well the model outputs fit ‘real world’ observations; and iii) Operational criteria that test how well a model answers the questions it is supposed to answer. The criteria mentioned by participants included:

**Conceptual criteria**

• The assumptions used by the model and the operation of the model should be explicit (i.e. transparent), so that they can be understood by users and open to critique;

• Flexibility for upgrading and modifying the model based on new knowledge about how the system works; and

• Models should be “consistent” across scales. Some participants saw this as an important test of conceptual soundness.

**Operational criteria**

• Models should be accompanied by documentation that includes a clear statement of the scope of applicability of model;
• Models would be more useful if they included documentation that identified sources of required data;
• Model outputs should match the tasks for which they have been used and should be useful at the appropriate spatial and temporal scale;
• Models should be easy-to-use;
• A model is more useful if there is an ability to link inputs/outputs with other models;
• Flexibility for upgrading and modifying the model to meet new operational circumstances;
• Cost-effectiveness;
• Outputs should ideally be expressed as a range rather than a single value;
• Effectiveness in informing (or changing) manager’s decisions; and
• Not only should ‘good’ models be consistent within today’s tasks of the SCA but they should also allow the organisation to address long-term issues of water management.

Quantitative criteria

• A model is more useful if there is an ability to verify the outputs/predictions; and
• Providing an assessment of economic value or impact.

5.1.6 Reflection

The consensus of opinion amongst workshop participants was that computer models can and should play an important role in many aspects of the SCA’s business. However, for the models to be useful to the SCA, models need to produce “robust” and “defensible” outputs and, at the same time, they should be able to operate within the constraints of available data, or data that can realistically be acquired.

Many participants argued that users could not have reasonable confidence in model results if a suite of models was used that relied upon competing sets of assumptions. Participants believed that a methodological consensus of opinion within the SCA’s suite of models could avoid a situation where the SCA appeared to use or support “duelling models”. Participants also wanted a level of congruence amongst the suite
of models that made it possible for users to link model outputs across domains and
temporal and spatial scales. At the workshop, these concepts of consensus and
congruence were phrased as “consistency”.

Participants suggested two overarching ideas about how to improve the efficiency and
effectiveness of model use for decision-support in the SCA. The first was that there
should be more and better-focussed data collection to improve scientific
understanding of the important biophysical processes and produce data that could be
used as inputs for parameterising, calibrating and validating models. The second was
that modelling in the SCA should be guided by formal procedures when using models,
rather than use models in a less co-ordinated, ad hoc fashion.

*Scientific/Technical Barriers*

The technical barriers that participants raised are common to all institutions that use
models. These include lack of expertise, limited data availability, difficulty choosing
appropriate models, lack of suitable modelling tools, difficulty understanding complex
models, inadequate or user-unfriendly interfaces, difficulty linking models and model
outputs across temporal and spatial scales and research domains, and difficulty
quantifying uncertainty in model input data, algorithms and outputs.

Potential methods for overcoming some technical barriers seem relatively clear, but
are often expensive or very difficult to implement. Suggestions included better/more
monitoring and field experimentation, more training of staff or the identification of
staff with the appropriate skills and more thorough model verification. Improving
aspects of model design might also help, by having better user interfaces, better
integrating capability and more open structures that allow assumptions and algorithms
to be understood and critiqued by users and other stakeholders.

The limitations associated with the quality and quantity of data available for
parameterising, calibrating and validating models were a popular discussion topic at
the workshop and were also routinely expressed at other SCA workshops and
meetings attended by all of the collaborative researchers.

Much scientific discourse has also served to highlight the lack of data available to
support models. In their series on model choice, CRCCH describe data limitations as
“a fundamental problem in water quality modelling” (2005b p. 14) and contend that they are “…the single biggest constraint to model choice and confidence in results.” (2005a p. 17). Letcher et al. (2002) modelled sediment and nutrient exports in several catchments across Australia using four different water quality models. Although each of the modelled catchments was selected because they were thought to have the best available data in the each State or Territory in which they were located, the researchers reported that there was insufficient data to calibrate any of the models in most catchments, or provide a “robust estimate” of annualised loads using monitoring data and direct estimation techniques in any catchment. They were subsequently unable to conclude which model provided the most accurate prediction.

Many other researchers have highlighted the lack of data available for water quality modelling in Australian catchments (e.g., Grayson et al., 1992; Young et al., 1996; Donnelly et al., 1998; Ball et al., 2001; Vertessy, 2001). Similar difficulties have also been expressed in the USA and Europe, where such data are often more abundant than in Australia (e.g., Reckhow, 1994; Anderson and Bates, 2001; da Silva et al., 2002; Singh and Woolhisser; 2002).

It is ironic but understandable then that although there is a dearth of data and limited quantitative scientific knowledge to support models, many workers have noted that there are a “plethora” of models to choose from (Simonovic and Bender, 1996; Letcher et al., 1999; Chiew et al., 2002; Letcher et al., 2002; Perrin et al., 2002; Caminiti, 2004). Where calibration data and/or scientific knowledge is very limited, the choice of model parameters or important model processes becomes little more than “guesswork” (Reckhow, 1994).

**Institutional Barriers**

Based on participant’s responses, it would seem that the major barrier to embedding models in decision-making was the absence of a consistent SCA policy on the selection and use of models (i.e. an SOP). Lack of adequate training programs and inadequate organisation of the SCA’s in-house modelling capability were also cited as institutional barriers.
One group of participants suggested that the SCA’s modelling capability could be improved or reorganised according to a three-tiered structure based on modelling expertise. Staff that used models could be divided into: i) advanced users/analysts who are able to develop new models, calibrate and analyse model inputs, algorithms and outputs; ii) intermediate users who have sufficient expertise to follow instructions for setting up, running and possibly calibrating pre-existing models; and iii) low-end users who can run queries or scenarios on models without changing their inputs or algorithms. High-level expertise, they said, would be expensive for the SCA to acquire and maintain, and the SCA needs to consider how much of this capacity needs to be in-house.

Other participants held the view that for out-sourced modelling, there could be better informal and formal liaison between modellers and SCA managers. Procedures for accessing external expertise could be documented or formalised as part of an SOP.

The SOP was seen as a panacea for overcoming organisational issues. The form and function that an SOP should take was unclear, but there was consensus that a SOP could and should be used, and that it could conceivably take the form of a manual, flow chart, decision-tree or something similar. The SOP might contain procedures covering the selection and use of models by SCA staff, consultants and others, and some guidelines on how information derived from models should be represented and used.

However, participants felt that there would not be a systemic approach to model use in the SCA unless a motivated “champion” emerged from senior management in the organisation. Participants were generally pessimistic about the possibility of senior management support for a systematic approach to modelling due to the lack of understanding of the difficulties associated with model use at senior management level.

Of the barrier ‘breakers’ proposed by participants, the development of a SOP caused the most discussion and was widely supported. The SOP was seen as a potential aid to selecting appropriate models in diverse circumstances. The use of a decision-tree for example led to a discussion about how the ‘branches’ of the tree might be used to identify where the SCA might need to develop new models.
The task of developing a credible SOP was seen as a difficult challenge that might best be achieved if the SCA focussed on identifying what it already knows about the usefulness of models for its purposes. One participant suggested that an SOP could go beyond identifying appropriate models to suggesting other SCA actions such as education programs, posting web-based material, or anything else relating to the selection, use and application of models and model outputs.

Although not a major topic of discussion at the workshop, the idea that models can be used in an adaptive management framework, as a way of managing uncertainty, has been a topic of academic inquiry for almost 30 years (Holling, 1978). BDMF (2000) define adaptive management as “a systematic process for continually improving management policies by learning from the outcomes of operational programs” (p. 10). Adaptive management is therefore a tool that catchment managers can use to ‘learn-by-doing’, rather than be “paralyzed by indecision”. Further, because adaptive management views interventions as systematic and rigorous management experiments (Walters and Holling, 1990), catchment managers can reduce a perception amongst stakeholders that they are “charging ahead blindly” (BDMF, 2000 p. 10).

5.2 Workshop Two: Scientists and Analysts

5.2.1 Introduction

The aim of the second workshop was to seek input that would enable the author to develop a modelling approach that will deliver the best possible estimates of diffuse nutrient sources in the SCA catchments given the SCA’s modelling needs and the constraints of data, scientific knowledge, time, resources and expertise.

Before this workshop, an SCA participant was asked to provide a statement as to the intended purpose of the modelling tool being developed by UWS. This statement is provided in full in Appendix 1. In summary, SCA planners and managers require a tool to assess the relative strength of different rural diffuse sources of nutrients. The approach taken needs to be conceptually simple, easy to use, based on currently available data, applicable across sub-catchments and preferably GIS-based. The primary purpose of the tool is to assist the SCA with the assessment of development applications (DAs) in relation to the Neutral or Beneficial Effect test (NorBE) and the
prioritisation of land uses and parts of the catchment for Rectification Action Planning (RAP).

5.2.2 Proposed Modelling Tool

Given the needs of the SCA, the author proposed an idea for modelling nutrient sources *hydrologically sensible* export coefficients linked to land use, soil phosphorus (P) concentrations, runoff volumes and, if practicable, other landscape characteristics (referred to as the ‘enhanced EC approach’). In particular, the method would use the relationship between soil P concentration and P concentration in runoff, together with runoff coefficient modelling (to provide annual estimates of runoff depth based on long-term runoff modelling on a daily time-step) to derive generation rates.

The tool is conceptually simple, easy to use, based on currently available data, applicable across sub-catchments and GIS-based. This second workshop was used to obtain expert input on how to best develop the enhanced EC approach, identify its limitations and, if necessary identify potentially useful alternative approaches.

5.2.3 Structure of the Workshop

This full-day workshop commenced with a half hour introductory discussion to describe the project, present the SCA’s ‘modelling need’ statement, and present and clarify the intended meaning of each question (See Chapter 5.2.5). The introduction also included a discussion about the results of the first workshop and a presentation from the SCA about the type and quality of spatial data available to modellers.

After the introduction, participants ‘brainstormed’ the first question as one group for half an hour and then split into three ‘breakout groups’ for one hour to brainstorm questions two, three and four, recording the key points raised by each group on flipcharts. After a lunch break, a nominated speaker from each group presented their answers to all participants. Question five was answered by all participants at the end of the workshop. As for workshop one, the second workshop was facilitated by Dr Richard Davis. A brief paper summarising the main issues raised and inviting comment was sent to each participant after the conclusion of the workshop.
5.2.4 Attendance

Thirteen experienced water quality scientists, most with doctorates and extensive modelling experience, participated in the second workshop. There were seven SCA scientists present. Other attendees came from the Australian National University (2), Environment Protection Authority (NSW) (1), CSIRO (1) and University of Western Sydney (2). The author was present to take notes and participate in the proceedings. The names and other personal details of the attendees have been suppressed to protect their privacy.

5.2.5 Questions and Introductory Discussion

Workshop participants worked through five questions aimed at identifying the limitations of EC-based approaches, finding the best ways to enhance the basic export coefficient approach and make it more hydrologically sensible and more credible, and to identify alternative approaches. The questions were:

1. What are the assumptions and limitations of the export coefficient approach and this application of the approach?;
2. How can we provide locally relevant and hydrologically sensible export coefficients for each land use to maximise the value and credibility of the model? (focussing on nutrient generation);
3. How do we simply estimate/model delivery from a point on a hillslope to the stream, using available data?;
4. What further enhancements to an EC model should be considered?; and
5. Having reviewed EC models and enhancements, what alternative approaches, if any, may have more appeal (for specific SCA activities/tasks) given the data and other constraints?

Discussion Issues

In early discussion, key SCA participants advised the attendees that, for NorBE testing and RAP, the SCA was only interested in nutrient deliveries to the edge of proposed developments (for NorBE) or to the edge of receiving stream waters (for RAP). In-stream processes were not to be the focus of this discussion or any resultant modelling.
The SCA ‘needs statement’ (i.e. Appendix 1), which came to be called a “wish list”, generated some pessimistic comments from two SCA participants. The first participant argued that “meeting all of the needs in the ‘wish list’ would require a huge investment of SCA resources”, beyond that available in the current project, and later added that modelling would be: “a waste of time without calibration data”.

Another SCA participant expressed concern that:

“There is an expectation in the SCA that a model should do everything, but NorBE and RAP are two vastly different approaches”

In response to these types of statements, an SCA participant responded:

“This is a stepwise process. We should be able to address some of the issues in the wish list quickly and easily”.

5.2.6 Results

Question 1. What are the assumptions and limitations of the export coefficient approach and this application of the approach?

At this point in the workshop, participants focussed only on the basic EC model and how it might be used to support NorBE or RAP.

Assumptions

The only assumption mentioned by participants was that the EC approach assumed that land use is the best integrator of the factors that control nutrient loss. One participant questioned the reliability of this assumption, and another argued that land “cover” was a better indicator than land “use”.

Limitations

The limitations of basic EC approaches that were identified by participants are listed below:

- The EC approach is “imprecise”;
- EC approaches may be less suitable at larger scales where a number of transformation processes affect the delivery of nutrients to the stream bank;
- The EC approach estimates annual averages and does not consider “events”;
• The EC approach does not explicitly recognise the processes involved in nutrient mobilisation, transport and delivery, especially hydrologic response;
• The effects of land management on nutrient exports are not explicit; and
• The EC approach usually lacks error estimates.

Question 2. How can we provide locally relevant and ‘hydrologically sensible’ export coefficients for each land use to maximise the value and credibility of the model? (focussing on nutrient generation)

In response to both parts of this question, participants mentioned a wide range of factors that could or should be added to a model to make it ‘hydrologically sensible’, useful and credible. These are listed below:

• The model should adjust EC values based on:
  ▪ Many different land-use categories (one participant stated that 13 was “not enough to be useful” for RAP);
  ▪ vegetation cover;
  ▪ land management (especially changes in cover over time);
  ▪ phosphorus content of the soil;
  ▪ variable source areas;
  ▪ rainfall variability (temporal and spatial) across the catchment (i.e. it should be event-based); and
  ▪ antecedent soil moisture.

• Concentration-based approaches are likely to be more hydrologically sensible than EC-based approaches, as they explicitly take account of individual events;
• Parts of the SCA’s catchment areas that have intensive use and are likely to be “hotspots” for diffuse nutrients could be ground-truthed for accuracy;
• The SCA could regularly update its GIS database using aerial photographs or satellite images so that new information can be obtained showing changes in land cover/condition (which might be used as a replacement or surrogate for land management), and that rainfall and other temporarily variable factors such as rainfall are also regularly updated;
• A categorisation of land use, topography, soil-landscape, soil-P could be developed and linked to a monitoring program inside and outside the catchment to develop and validate the modelling approach used in this study;

• Research and sensitivity analysis should be undertaken to identify the parameters that are most important in determining nutrient exports. Future modelling, monitoring and experimentation efforts should concentrate on these. Some participants felt that the most important factors were land cover, rainfall and rainfall intensity; and

• The USLE is a good first estimator of erosion and hence adsorbed nutrients.

There was much debate about Variable Source Area (VSA) approaches. The VSA concept identifies parts of catchments that are the source of substantial runoff during storm events, and therefore contribute more pollutants to streams. VSAs are a function of topography, soils, geology, climate, and management and may contract and expand within and between events. A limitation of this approach is that it may only be applicable in cool, humid areas with reasonably reliable rainfall.

Some participants felt that simple and generic modelling was inappropriate for NorBE assessment because it cannot take into account very specific site-dependent factors and peculiarities of a development application, however SCA participants that work in this area considered that a simple approach can be used as a first-cut tool to assess the level of risk associated with proposed land use changes and their outputs might trigger the use of more appropriate tools. It was widely agreed that an enhanced EC approach could be suitable for RAP prioritisation.

Whilst one participant argued that land-use needed to be discretised into more than 13 land-use types, another participant argued that “refining land use categories too much would not be a useful investment. It may better to try to incorporate other factors”.

Question 3. How do we simply estimate/model delivery from a point on a hillslope to the stream, using available data?

The factors identified by participants that could be included in a diffuse nutrient model were:

• proximity to stream (confirmed by all groups), or stream density (e.g., the number of channel metres per hectare)
cover (as a surrogate for roughness and overland interception)
- land use
- slope
- soil drainage and infiltration characteristics
- rainfall variability
- interception of flow (e.g., farm dams)
- Soil hydrologic grouping (i.e. the infiltration characteristics of the soil)

There were no substantive comments made as to how these parameters could be used in a conceptually simple modelling tool. One of the break out groups reported that no existing model was appropriate for this task.

Question 4. What further enhancements to an EC model should be considered?

Suggested enhancements and other comments are listed below:

- Sensitivity analysis;
- A simple but enhanced EC model could be linked to other models such as HSPF (Bicknell et al., 1993), which is used for SCA bulk water modelling;
- The range of pollutants could be expanded beyond nutrients to include sediments, pesticides and pathogens;
- The model could be expanded to include in-stream transport processes;
- An expert system such as NEXSYS (Young et al., 1997b) could be used to make the EC sensitive to a variety of landscape factors;
- Channel erosion as well as surface erosion could be process-modelled;
- Farms dams, contour banking and effects of vegetation cover and other management related factors could be included;
- Experiments could be undertaken to obtain data for SCA conditions. This could include production of a soil-P database for the SCA area given relationship revealed between soil-P and nutrient concentration. The existing soil-landscape maps could possibly be interpreted to provide soil-P maps if combined with some soil testing experimental work;
- A literature search could be made for a soil or other parameter that shows a good a relationship with N export. This would also need to be supported by local research; and
Fieldwork should be undertaken to acquire data for the model, particularly infiltration and saturation excess overland flow and hillslope delivery ratios. Existing models, such as SedNet (Prosser et al., 2001) usually assume delivery ratios of 5-15% but there is little evidence to support their assumptions.

**Question 5.** *Having reviewed EC models and enhancements, what alternative approaches, if any, may have more appeal (for specific SCA activities/tasks) given the data and other constraints?*

Most participants showed an obvious affinity for models that were familiar to them given their area of expertise and this created a kind of ‘methodological tension’ distinguishing participant’s different disciplinary backgrounds. Participants with a hydrology background, for example, tended to favour models with a strong hydrologic component, whilst participants with a background in soil and erosion modelling conceptually linked nutrient export with erosion and displayed a preference for models that had a strong sediment component. There were also differences in the amount of model complexity that participant’s felt should be included in the model, and this seemed to reflect the amount of fundamental research that they had been engaged in during their careers. Breaking entrenched world views and sorting through the favoured assumptions of the participants to achieve some kind of consensus about the way forward did not seem an achievable outcome of this one day workshop.

Altogether, five alternative approaches were discussed over the course of the workshop: Approaches based on mean concentrations of nutrients in streams; the L-THIA model; USLE-based approaches; the SedNet model and the HSPF model. Each is discussed below:

*Approaches based on Event Mean Concentrations (EMCs)*

An EMC-based approach was suggested as an alternative. An advantage of this approach is that it allows the user to explicitly take account of large events. However, this approach has the same fundamental data requirements that are needed to properly parameterise the EC approach. In order to satisfy the SCA’s ‘wish list’, the approach will need to relate EMCs to land use and other landscape characteristics.
Examples of models already using this approach are EMSS (Vertessy et al., 2001) and a model developed by the NSW EPA (Baginska and Ruffio, 2003). EMC-based models that are under development at the moment include the Catchment Contaminant Cycle Model (CCCM) (Rutherford et al., 2005) and E2 (Argent et al., 2005). One participant noted that the input data requirements for EMSS, CCCM and E2 are substantial, although the underlying approaches for determining nutrient exports are relatively simple.

*Long-Term Hydrologic Impact Assessment (L-THIA)*

L-THIA\(^1\) is a GIS, Excel and web-based tool developed at Purdue University, USA. L-THIA is used to provide estimates of changes in runoff, recharge and diffuse (nonpoint) source pollution resulting from past or proposed land use changes. It gives long-term average annual runoff for a land use configuration, based on actual long-term climate data for that area. The only data required by L-THIA is location (to identify the appropriate rainfall data from the software’s rainfall database), land use category, hydrologic soil group and land use area. The current database only contains rainfall information for the US.

Runoff volume is computed by runoff depth multiplied by the land use area. Event Mean Concentration data are used to predict diffuse pollutant masses by multiplying the runoff volume of a grid cell by the EMC value.

*Universal Soil Loss Equation (USLE, (M)USLE, and (R)USLE)*

A group of participants felt that approaches based on USLE\(^2\) are a viable alternative, especially where soil erosion is the dominant process involved in nutrient mobilisation and delivery to streams. However, there is compelling evidence that P loss from well covered pasture is predominantly in the soluble form (Nash and Halliwell, 1999, 2000). A USLE based sub-model could be used in conjunction with a hydrologic sub-model.

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1 See: http://www.ecn.purdue.edu/runoff/index.html.
The Modified Universal Soil Loss Equation (MUSLE) is similar to USLE except that
it operates for an individual storm rather than the average of a 20-year period. A
rainfall factor in USLE is replaced with a runoff factor in MUSLE.

*SedNet*

SedNet (Prosser et al., 2001) was suggested by several participants as the most
suitable alternative existing model. SedNet is a GIS-based software package
originally developed to assess water quality in major Australian catchments as part of
the Australian National Land and Water Resources Audit. SedNet estimates river
sediment loads by constructing material budgets that account for the main sources and
stores of sediment. AWT (2001b) found that “The basis for soluble components is
however very generalised and is unlikely to be appropriate to the scale on which DAs
or RAPs need to be conducted” (p. 52).

*HSPF*

HSPF (Bicknell et al., 1993) was briefly discussed as an alternative approach, mainly
because HSPF has already been operationalised by the SCA for bulk water quality and
quantity modelling. This approach was ruled out because, in its current use, many
parameters that are important in NorBE testing and RAP prioritisation are not
spatially explicit (i.e. they are lumped). It was considered impractical to operate
HSPF at the required scale.

5.2.7 Reflection

Many of the suggestions made at this workshop mirror those from a series of
workshops organised by Australian Water Technologies (see AWT, 2001b).

There is a lack of quantitative knowledge of basic physical processes, or at least the
relative importance of different processes, at different spatial and temporal scales.
New, locally relevant data are needed to calibrate and validate any model that
operates at the hillslope scale. Participants identified a large number of potentially
useful parameters, but there was no consensus or clarity as to how these might
practically be applied (i.e. how the model could be parameterised). This appears to
be a reflection of the lack of scientific knowledge of relevant processes.
The group seemed to be split into two ideological camps: those advocating a *horses for courses* approach, who typically argued that too much was being asked of this one model, and another camp that was optimistic about the possibilities offered by one model that would operate at a range of scales and incorporate a wide range of landscape factors that affect nutrient export. The ‘horses for courses approach’ contrasts against manager’s desire in the first workshop for “conceptually consistent” approaches, which appears to be more aligned with the ‘optimist’s’ viewpoint.

Mixed land uses, short monitoring records and lack of monitoring data at the small catchment scale make it difficult to establish long-term areal loading rates (*i.e.* ECs) or average EMCs. The proposed approach differs from other EC and EMC-based approaches in that it will be less dependent on monitoring data and more strongly related to discrete land uses and landscape characteristics.

Whilst soil test phosphorus might be a good indicator of when an appreciable concentration of dissolved P may be in runoff water, it does not offer any indication of the amount (rate) of runoff water that may be generated for a given set of conditions. A reliable method of runoff modelling is important.

Some potentially useful models were not discussed, such as SWAT (Arnold *et al.*, 1998), PLOAD (USEPA, 2001), and AGNPS (Young *et al.*, 1987) - a model that was recently used for Sydney Water and the SCA to estimate nutrient exports (AWT, 2003).

Upon reflection, it was decided that there was no convincing argument for using any of the alternative approaches in place of the enhanced EC model. A number of opportunities are provided by the enhanced EC model. These include:

- Surrogates may be used in place of the key parameters. For example soil landscape and/or land use may be used as a surrogate for soil P;
- Soil P/runoff P relationships have been reported widely in recent years. ECs based on these and runoff modelling may be more readily attainable and just as reliable for certain land uses as EMCs based on monitoring.
- The soil P/runoff P relationship is a simple and verifiable relationship that does not attempt to model processes. This may be a good option to use in some situations, when the important processes in the catchment are unknown,
and monitoring data is unavailable, or unable to resolve different ECs for different land uses.

- The method provides an opportunity to undertake some runoff modelling, which may have additional value to the SCA;

- Although the model is likely to be first and foremost a scoping tool to calculate annual loads, the enhanced EC approach will be more versatile than the EC approach. Further development and refinement and validation, based on monitoring, may allow an upgraded model to include daily estimates, or other enhancements that may give the model more usability and credibility.

Notwithstanding these benefits, the approach suffers from some limitations, which are outlined below:

- The enhanced EC approach may not be suitable for all parts of the SCA’s management areas. For example, the approach may be unsuitable for basalt-derived soils where assumptions about the relationships between soil P and runoff P (the ‘extraction coefficient’) may not hold, or badly degraded pastures where serious soil erosion can occur and USLE might be the best predictor of P loss (with sediment). Where cultivation is regularly practised as part of a land use, those lands would also be better simulated by USLE or its derivatives (RUSLE and MUSLE). Nevertheless, the approach may be suitable for most extensive pastures in the SCA’s area of operations.

- The enhanced EC approach assumes that soil P is the main determinant of runoff P, however, there is good evidence that recent additions of fertiliser, manure or effluent will elevate P concentrations in runoff. This effect may need to be accounted for in more intensive land uses. Further work is needed to establish ‘extraction coefficients’ for both P and N in both cultivated and non-cultivated systems.

- Most focus has been on P. Ways to model N need to be investigated.
6 MODELLING

This chapter describes the modelling that was undertaken. The chapter is divided into two sections. The first section describes a simple EC model that was produced to provide the SCA with a quick ‘first-cut’ model of nutrient source strengths. The second section presents an ‘enhanced’ EC model, which was developed to meet the needs of the SCA, as per the catchment manager’s workshop, but within the limitations of available data and knowledge that were highlighted at the scientists and analysts workshop. Both sections conclude with some reflection on the learning gained from the modelling activity in terms of the issues of modelling for decision-making. The simple EC model is discussed in Chapter 6.1 and the enhanced EC model is discussed in Chapter 6.2.

6.1 Simple Export Coefficient Model

6.1.1 The Export Coefficient Approach

Diffuse loads of total P and total N exported from major land-uses in the Sydney drinking water catchments were estimated using a simple EC approach. An EC is the mass of a substance exported from a given area of land over a given time period, and is often expressed as kilograms per hectare per year (kg/ha/yr). Export coefficients are typically estimated by monitoring pollutant loads in runoff from plots, fields or small single-use catchments over time and then dividing the total amount of the pollutant export from the plot or catchment by the catchment area. The results are usually normalised to produce the annual export rate.

Gross potential loads for a pollutant exported from a given land-use are calculated by multiplying the land-use and pollutant-specific EC by the geographical area of the land-use. Total potential catchment loads of the pollutant can be calculated by summing the load for each land-use in the catchment. The estimated loads are sometimes referred to as potential loads because the approach does not usually take account of assimilation of nutrients that may occur along the transport pathway in larger catchments, which may reduce the actual amount of the nutrient that reaches a stream, reservoir or other water body.
6.1.2 Use of Export Coefficient Models

Although EC modelling was pioneered in Europe 35 years ago (Vollenweider 1968), the continued popularity and utility of the approach is reflected in the widespread use of EC approaches by catchment management organisations and others throughout Australia (e.g., Atech, 2000; Joo et al., 2000; McMurray, 2000; WAWRC, 2002; WADE, 2004) and around the world (Johnes, 1996; Mattikalli and Richards, 1996; Johnes and Heathwaite, 1997; McGuckin et al., 1999; Worrall and Burt, 1999; Irvine et al., 2000; Hanrahan et al., 2001; Endreny and Wood, 2003; Ierodiaconou et al., 2005; Khadam and Kaluarachchi, 2006).

Catchment managers and others often use EC models to quantify catchment nutrient loads and prioritise land-uses for rectification action. Export coefficient models seem to remain popular because they have input data needs that are commensurate with data availability. Export coefficient models require only land-use data and a generation coefficient for major land-use types. They also have simple hardware requirements which typically include a GIS and/or spreadsheet program to calculate land areas and potential pollutant loads and display the results as graphs, tables or maps. In some cases, EC models have been produced as stand-alone software packages, such as CMSS (Davis et al., 1991; Davis and Farley, 1997) and PLOAD (USEPA, 2001).

6.1.3 Export Coefficient Models and Sydney’s Drinking Water Catchments

Export coefficient models have been used to estimate nutrient loads in the drinking water catchments previously. The most notable application of the approach was undertaken in the early 1990s using CMSS in the Hawkesbury-Nepean catchment area, including upstream from Warragamba Dam (Cuddy et al., 1994). In recent years, Long (2003) used a simple spreadsheet-based EC model of the catchments. In other applications, the EPA has relied on the results of EC modelling for the 2003 and 2005 Audits of the Sydney Drinking Water Catchments (DEC 2003; 2005) and Sherman and Orr (2003) used an EC approach to compare nutrient generation rates for the sub-catchments of the Shoalhaven River.
6.1.4 Method

6.1.4.1 Overview of the Approach
The drinking water catchments were divided into 12 different land-use categories. The area of each land-use was multiplied by land-use specific ECs for total N and total P to give an estimate of the potential N and P export from each land-use. Summing the results gave an estimate of the total potential nutrient load generated within the drinking water catchments and sub-catchments for an annual time step. The calculations are described in Equation 1.

Equation 1. Algorithm used to estimate pollutant load in a simple EC model.

\[ L_p = \sum_u \left( \hat{E}_{pu} \times A_u \right) \]

where:  
- \( L_p \) = the annual catchment pollutant load for pollutant \( p \) (kilograms/year);  
- \( \hat{E}_{pu} \) = the export coefficient (kilograms/hectare/year) for pollutant \( p \) and land-use type \( u \);  
- \( A_u \) = the area of land-use type \( u \) (hectares)

6.1.4.2 Sub-catchment Divisions
For the purpose of presenting the results of modelling, the Sydney drinking water catchments were divided into 27 hydrological sub-catchment (Figure 6-1). The sub-catchments are often used by the SCA when reporting on environmental health and water quality in the SCA’s area of operations (e.g., Long 2003) and must be prioritised as part of the Catchment Audit process (CSIRO 1999, 2002; DEC, 2003, 2005). The sub-catchments are also the basic management unit for Rectification Action Planning and are mentioned in s. 5 of the Drinking Water Catchments Regional Environmental Plan No 1 (2006-289 NSW)\(^3\).

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Figure 6-1. Hydrological sub-catchments of the Sydney drinking water catchments.
6.1.4.3 Land-use Divisions

The SCA’s ‘Broad Land Cover/Use 2002’ dataset was used to derive most of the land-use categories used in this study. The dataset is a compilation of datasets and satellite imagery classifications that divides the drinking water catchments into 38 different classes. The land-use classes were reclassified (Appendix 2) into ten land-use categories:

1. Water; 6. Horticulture (orchards);
2. Forest; 7. Urban Residential;
3. Degraded Pasture; 8. Urban Industrial;
4. Unimproved Pasture; 9. Rural Residential; and
5. Improved Pasture; 10. Other

Land areas that could not be assigned to any of these categories (e.g., roads and mines) were recorded as ‘Other’. Two land-uses that are not identifiable from the Land Cover/Use map, but are likely to be important in terms of nutrient management in the drinking water catchments are vegetable and dairy farms. In order to include these land-uses in the potential nutrient export estimates, the SCA’s ‘EASI data Horticulture’ and ‘EASI data Livestock’ datasets were used.

The Horticulture dataset gives the location of known vegetable farms (and other horticultural sites) as point features and gives an estimate of the area occupied by each farm. These data were used to tabulate the area occupied by vegetable farms in each sub-catchment. Similarly, the Livestock dataset was used to identify the location of dairy farms. The Livestock dataset does not give the area of land occupied by each farm or provide other data that might be useful to gauge the size of the farm or estimate nutrient exports, such as the herd size. To make the data useful for EC modelling, the area of land occupied by dairies was arbitrarily assumed to be 100ha per farm. This size is a conservative value, since average dairy farm sizes in NSW are reported to be 260 ha in NSW and 170 ha in Victoria (Kompas and Che, 2004). The total estimated area of dairy and vegetable farms was subtracted from land-use previously classified as ‘Other’.

Forest is the most widespread land-use, occupying 65% of the total catchment area. Unimproved pastures, degraded and improved pastures cover 22%, 4% and 2%
respectively. Water covers just over 1% of the catchment. All other land-uses occupy less than 1% of the total catchment area each (Figure 2). The area of each land-use category in the SCA’S drinking water catchments is shown in Table 6-1.

<table>
<thead>
<tr>
<th>Land-use category</th>
<th>Area (ha)</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest</td>
<td>1,013,304</td>
<td>64.7%</td>
</tr>
<tr>
<td>Unimproved pasture</td>
<td>386,610</td>
<td>24.7%</td>
</tr>
<tr>
<td>Degraded pasture</td>
<td>68,536</td>
<td>4.4%</td>
</tr>
<tr>
<td>Improved pasture</td>
<td>35,113</td>
<td>2.2%</td>
</tr>
<tr>
<td>Water</td>
<td>20,019</td>
<td>1.3%</td>
</tr>
<tr>
<td>Urban - residential</td>
<td>13,045</td>
<td>0.8%</td>
</tr>
<tr>
<td>Urban - industrial</td>
<td>1,570</td>
<td>0.1%</td>
</tr>
<tr>
<td>Rural residential</td>
<td>1,738</td>
<td>0.1%</td>
</tr>
<tr>
<td>Dairy farms</td>
<td>3,200</td>
<td>0.2%</td>
</tr>
<tr>
<td>Horticulture (orchards)</td>
<td>603</td>
<td>0.04%</td>
</tr>
<tr>
<td>Vegetable farms</td>
<td>310</td>
<td>0.02%</td>
</tr>
<tr>
<td>Other</td>
<td>25,021</td>
<td>1.6%</td>
</tr>
<tr>
<td>Total</td>
<td>1,567,330</td>
<td>100%</td>
</tr>
</tbody>
</table>

### 6.1.4.4 Land use data limitations

An improved land-use map was under preparation, but unavailable for use in time for this study. Considerable effort was put into interpretation of the existing dataset, including some ground-truthing, but weakness in the available data limit the utility of the map for any EC approach. The weaknesses relate to both the land-use classifications and the spatial extent of those classifications. Specific areas of concern relating to the land-use datasets are listed below:

- Ground-truthing of lands listed as “Agriculture – Intensive Pasture” in the “Broad Land Cover/Use 2002” dataset showed them to be generally associated with degraded extensive pasture, as would be expected in the lower-rainfall (and drought-stricken) areas west and south of Goulburn and around Braidwood, where this land-use classification is dominant. Long (2003) interpreted the same classification as “cropping/intensive plants”, although this interpretation was accompanied by a question mark, suggesting
uncertainty with this classification. For this study, Agriculture – Intensive Pasture was re-designated as degraded pasture.

- Rural residential land in the “Broad Land Cover/Use 2002” dataset may be under-represented.
- Grazed forest areas are not distinguishable from other forest types.
- Some data points in the EASI database appear to be incorrectly labelled or georeferenced. Many of the data points labelled as occurring in Lake Burrarorang sub-catchment are situated in Werri Berri sub-catchment. Several other points also appear to be incorrectly labelled, especially when they occur near catchment boundaries.

6.1.4.5 Setting Export Coefficients

A list of the export rates used in the study for each land-use category is given in Table 6-2. Uncategorised land (i.e. ‘Other’) was assigned an EC equivalent to the average spatially weighted EC of N and P of the categorised land-uses (including vegetable farms and dairies) in each sub-catchment.

<table>
<thead>
<tr>
<th>Land-use category</th>
<th>Export Coefficient (kg/ha/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>P</td>
</tr>
<tr>
<td>Forest</td>
<td>0.05</td>
</tr>
<tr>
<td>Unimproved pasture</td>
<td>0.3</td>
</tr>
<tr>
<td>Degraded pasture</td>
<td>2</td>
</tr>
<tr>
<td>Improved pasture</td>
<td>0.7</td>
</tr>
<tr>
<td>Urban - residential</td>
<td>0.4</td>
</tr>
<tr>
<td>Urban - industrial</td>
<td>3</td>
</tr>
<tr>
<td>Rural residential*</td>
<td>0.7</td>
</tr>
<tr>
<td>Dairy farms</td>
<td>5</td>
</tr>
<tr>
<td>Horticulture (orchards)</td>
<td>0.6</td>
</tr>
<tr>
<td>Vegetable farms</td>
<td>10</td>
</tr>
</tbody>
</table>

* Rural residential land is assumed to have the same export rate as improved pasture, based on the observation that most such land carries horses, cattle or other livestock and appears to be fertilised (Baginska et al., 1998).

Export coefficients were primarily derived from CECIL, a database of annualised export rates derived from published studies (see AWT, 2001b) and NEXSYS (Young
et al., 1997), an Expert System that can be used to estimate average annual export rates of N and P for major land-uses based on landscape traits including climate, topography, soil type and land management. Recently published studies, which are not included in CECIL, were also used. Important new research contributions that were used to refine ECs included Ridley et al. (2003) and McCaskill et al. (2003), who reported nutrient export and runoff data from several sites in NSW and Victoria with native, naturalised and improved pasture on soil types relevant to the drinking water catchments, and Cornish et al. (2002), who reported P exports from dairy pasture near Camden. ECs for forest were based mainly on medium-term monitoring of native forest at Little River and Reedy Creek4 in the Blue Mountains, both of which are situated within the drinking water catchments (Hollinger and Cornish, 2001).

Relatively few experimental studies of nutrient exports have been undertaken in the Sydney drinking water catchments that can be used to derive ECs. ECs must therefore be based primarily on out-of-catchment studies. For this study, only data from studies in south-eastern NSW and eastern Victoria were used to derive ECs. Export rates were scrutinised for accuracy and relevance. Particular attention was given to the rainfall and soil types recorded at the study sites during the monitoring period. NEXSYS was useful for reality checking ECs. NEXSYS does not give a single export rate, but instead provides a range in which actual exports are likely to fall. For the most part, the export rates used in this study are within the range estimated by NEXSYS (see Appendices 3 and 4).

Treatment of Grazing Land

Given the importance of grazing in the drinking water catchments, and the larger amount of data available for this land-use, special attention was paid to studies of exports reported from grazing land. For the purpose of attributing EC values, unimproved pasture was taken to mean pastures that were predominantly native or naturalised species that may be irregularly fertilised. Improved pasture was taken to mean pasture sown with introduced species, and regularly fertilised.

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4 The Reedy Creek monitored by Hollinger and Cornish (2001) is a tributary of the Kedamba River, situated in the Lower Cox’s River hydrological sub-catchment. It is not part of the Reedy Creek hydrological sub-catchment referenced throughout this report.
Much use was made of the Sustainable Grazing Systems (SGS) series of field experiments (see McCaskill et al., 2003; Ridley et al., 2003). The results from five sites in the high rainfall sheep zone of eastern Australia reported by McCaskill et al. (2003) were re-analysed to provide a set of ECs for P and N for extensive (unfertilised) and fertilised pasture. The analysis also included an early plot and catchment-scale study of improved pasture by Costin (1980) and more recent studies of dairies ranging from very intensive Baginska et al. (1998) to less intensive (Cornish et al., 2002). The dairy studies provided EC values at farm scale (0.4-2 km²). For calculations of EC using the data of McCaskill et al., (2003) the plot-scale values for nutrient concentrations in runoff were used together with gauging data from nearby streams, rather than plot runoff data. This procedure provided EC estimates that reflected hydrology of the local catchment, not plot runoff, which is always highly variable. The pastures in these studies were apparently well covered, with low erosion rates. Data were scrutinised to eliminate outliers, and a mean value adopted for each category of pasture. The combined dataset provided considerable insight into the processes that determine EC and its variation between sites and land-uses. Together, these sites are considered to best approximate the range of pasture situations in Sydney’s drinking water catchment area in terms of rainfall, soil type and intensity of use, except that no degraded pasture was included.

An EC for degraded pasture was determined by adjusting the values derived from the data of McCaskill et al., (2003) using results of Neil and Fogarty (1991) who examined sediment loads in farm dams in the Southern Tablelands of NSW. They estimated erosion rates from ‘degraded’ pasture to be approximately 7 times greater than from ‘native’ pastures. It was assumed that erosion was the main mechanism for mobilising P (attached to sediment) from degraded pasture and a 7-fold increase was applied to the EC for P from unimproved pasture to give an EC for degraded pasture. The N EC was also increased, but by a lower proportion, to reflect higher rates of runoff from sparsely vegetated ground. The use of this method to estimate an EC for degraded pasture means that the EC value is likely to factor in P from gully erosion. Experimental work is required to determine if the export rates used for degraded pasture are realistic.
Also, inspection of the catchment revealed significant areas of open forest that is grazed. It is likely that much of this area has been mapped as forest as there is no classification for ‘grazed forest’ in the land-use data. This being the case, nutrient exports from some forest areas will be underestimated.

*Export Coefficient Data Limitations*

The ability to develop a wide-range of ECs for many different land-uses was hampered by a number of practical limitations:

- There were few studies with which to derive ECs, although the availability and quality of data is steadily improving.
- There are significantly fewer published studies with locally relevant information available for N than for P.
- There was insufficient local data to differentiate ECs between plantation forests, native forests or grazed open forests. Existing land-use data does not allow identification of grazed forests areas.
- Based on south-eastern Australian studies, there was insufficient data to differentiate between sewered or unsewered and newly established or old urban areas. No attempt was made to differentiate these land-uses in the model.

### 6.1.5 Results

#### 6.1.5.1 Whole Catchment Results

The proportion of estimated P and N exports attributed to the most polluting land-use are shown in Table 6-3. The biggest estimated contributors of P were: Degraded pasture (38%); Unimproved pasture (32%); Forest (14%); Improved pasture (7%) and Dairies (4%). Degraded pasture and dairy pasture contribute to total P exports in much greater proportion than their area. The biggest estimated contributors of N were: Unimproved pasture (47%); Forest (32%); Degraded pasture (9%); Improved pasture (5%) and Urban Residential (3%).
Table 6-3. Modelled contribution to total potential P and N loads for major land-uses compared with area of catchment.

<table>
<thead>
<tr>
<th>Land-use Category</th>
<th>Percentage of:</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Total P Generated</td>
<td>Total N Generated</td>
</tr>
<tr>
<td>Forest</td>
<td>14</td>
<td>32</td>
</tr>
<tr>
<td>Unimproved pasture</td>
<td>32</td>
<td>47</td>
</tr>
<tr>
<td>Degraded pasture</td>
<td>38</td>
<td>9</td>
</tr>
<tr>
<td>Improved pasture</td>
<td>7</td>
<td>5</td>
</tr>
<tr>
<td>Urban Residential</td>
<td>1</td>
<td>3</td>
</tr>
<tr>
<td>Dairy pasture</td>
<td>4</td>
<td>&lt;1</td>
</tr>
</tbody>
</table>

6.1.5.2 Sub-catchment Results

The biggest estimated potential contributors of P on a unit area basis are Reedy Creek (0.7 kg/ha/yr); Mulwaree River (0.55); Boro Creek (0.48); Braidwood Creek (0.45) and Upper Wollondilly River (0.38) (Figure 6-2). In general, higher rates were attributed to extensive areas of degraded or unimproved pasture.

Half of the potential P exports in the drinking water catchments are estimated to originate from 5 of the 27 hydrological sub-catchments: Wollondilly (17%); Mulwaree River (12%); Reedy Creek (11%); Upper Wollondilly River (8%); and Wingecarribee River (7%) (Figure 6-3).

Sources of N are more evenly distributed than sources of P (Figure 6-4). Wollondilly River sub-catchment has much higher potential exports than other catchments (19% of total N generation), but this is mainly related to the larger size of the catchment (17% of the total catchment area) (Figure 6-5).
6.1.6 Comparison with Other Modelling

6.1.6.1 SedNet Modelling of Wingecarribee Catchment

The results from Wingecarribee River sub-catchment were compared with measured and modelled estimates of nutrient loads from a nutrient budgeting study undertaken by Olley and Deere (2003) using SedNet (Prosser et al., 2001). Comparing the approaches and outcomes used by Olley and Deere (2003) served to highlight commonalities and areas of difference between the two approaches.

Model One Results for Wingecarribee Catchment

Wingecarribee River sub-catchment covers approximately 76,000 ha. According to the EASI-Livestock dataset there are 14 dairy farms in the Wingecarribee sub-catchment. In model one, it was assumed that an average dairy farm was 100 ha, and that dairying occupied a total of 1,400 ha (~2% of the sub-catchment). The EASI-Horticulture dataset shows a total of 22 ha of vegetable farms (<0.1% of the sub-catchment). The “Broad Land Cover/Use 2002” dataset indicates that 44% of the sub-catchment is forest, 33% is unimproved pasture and 13% is improved pasture. Four percent of the sub-catchment is residential (Figure 6-6).

Eighty-five percent of potential diffuse P exports in Wingecarribee sub-catchment was estimated (using model one) to originate from grazing land. Dairy farms may produce up to one quarter of P that is generated, despite occupying only 2% of the sub-catchment (Figure 6-7). Improved pasture is relatively a much bigger contributor to potential P exports in Wingecarribee than in the drinking water catchment overall. A similar chart of N exports is given in Figure 6-8.

Figure 6-6. Proportion of Wingecarribee catchment occupied by each land-use category.
Comparison with Olley and Deere (2003)

Olley and Deere (2003) estimated sediment, nutrient and pathogen loads in Wingecarribee sub-catchment using SedNet and other methods. Estimates of P and N loadings were based on SedNet calculated sediment loads, P concentration in channel sediment (to estimate particulate P), and the ratio of particulate P to total P at a gauging station at Berrima Weir (to estimate dissolved P), together giving total P. Based on these calculations, Olley and Deere estimated the annual export of P from Wingecarribee sub-catchment to be 12 t/yr. They also estimated annual nutrient exports using only monitoring data of flow and nutrient concentration at Berrima Weir. Using the gauging station data, annual nutrient exports from Wingecarribee were estimated to be within the range of 6.7 and 66t/yr for P (depending upon flow), with a “best estimate” (p. 44) (based on long-term data) of 21t, and within a range of 108 and 300 t/yr, with a “best estimate” of 180 t for N.

Being based on in-stream data, Olley and Deere’s estimates also incorporate point sources of nutrients, which include 3 sewage treatment plants, a piggery and a stock sale yard. There were no data from the sale yard with which to estimate nutrient loads from that facility, but nutrient exports from the other point sources were estimated by the authors to be 2 t/yr for P and 21 t/yr for N.

After adding Olley and Deere’s point source data to the simple EC results, the estimated potential nutrient generation from the simple EC model for Wingecarribee River sub-catchment would be 29t/yr for P and 243 t/yr for N (Table 6-4).
Table 6-4. Comparison of estimates of P and N exported/generated in Wingecarribee River sub-catchment.

<table>
<thead>
<tr>
<th>Method and Source</th>
<th>Wingecarribee Nutrient Load (t/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Olley and Deere (2003) – Method 1*</td>
<td>P 12  N -</td>
</tr>
<tr>
<td>EC Modelling***</td>
<td>P (21)  N (180)</td>
</tr>
</tbody>
</table>

* Based on modelled sediment loads and P concentration in channel sediment to estimate particulate P, and the ratio of particulate P to total P at Berrima Weir to estimate dissolved P, together giving total P.
** P and N exports based on monitoring at Berrima Weir. The figure in parentheses is the ‘best estimate’.
*** Gross estimates of nutrients generated at each land-use.

The estimated generation of nutrients in Wingecarribee sub-catchment from the simple EC model were higher than the best estimates of monitoring-based load calculations of Olley and Deere (2003). The potential generation rate should be higher than in-stream measurements because of the influence of hill-slope and in-stream processes such as sedimentation along the flow path and in farm dams and streams, which are not explicitly accounted for in simple EC approaches. Also, the EC approach does not explicitly estimate nutrients derived from erosion of gullies and stream banks, although gully erosion may be implicitly included in some of the EC values.

6.1.6.2 Limitations of the Approach Used by Olley and Deere

The major weakness of the approach used by Olley and Deere (2003) is shared by all models that are based on empirical measurements. Routine measurements in streams of catchments with mixed land-use do not provide sufficient information to trace the nutrients back to their source. Whilst this can be achieved with reasonable success for sediment using geochemical tracing techniques, there is no viable approach for linking nutrients back to their source in mixed land-use catchments.

The strength of SedNet is sediment modelling. SedNet modelling and geochemical analysis by Olley and Deere (2003) found that 95% of sediment in Wingecarribee River was derived from gully and streambank erosion. An assumption that seems to have been drawn from this is that most of the P delivered to streams is also from gully and streambank erosion, or direct deposit of nutrients on or near the stream edge. This assumption is untested but had a major influence on their recommended
management response, which was that remedial action for diffuse nutrients should be focussed on establishing a buffer zone along the drainage network.

Olley and Deere (2003) used the ratio of estimated mean annual flows at the Berrima Weir gauging station (Berrima Weir) and the whole catchment; however, the land-use in the ungauged parts of the catchment is different to that of the gauged part of the catchment. Further, the flow data at the gauging station was not collected at the same time as the nutrient and sediment concentration data.

Another issue is temporal sensitivity. Although SedNet may appear more versatile than simpler approaches because of the model’s ability to disaggregate results to produce daily estimates (Wilkinson et al., 2004), the method used to achieve this, using the sediment rating curve and time-series flow data, is often inaccurate due to hysteresis effects associated with sediment supply and depletion (see Evens and Davies 1998; Asselman, 1999; Croke and Jakeman, 2001).

6.1.6.3 Similarities and Differences in the Two Methods

SedNet and EC models are both steady-state empirical models with algorithms and parameters based on observational data. The outputs from both models focus on the spatial pattern of pollution generation rather than temporal variability, with both models describing nutrient loading over a mean annual time-step, although SedNet has a dynamic component that can allow it to disaggregate its results to predict daily loads using sediment rating curves and flow time-series if desired (Wilkinson et al., 2004).

However, the SedNet approach is essentially an ‘inmission’ model, since empirical data of loads or concentrations in the stream and at the catchment outlet are used as a basis for determining characteristics higher up in the catchment. EC models are ‘emission’ models, because they use empirical emission data for different land classes in a catchment to describe the potential effect at the catchment outlet.

The most significant area of difference between the two approaches is implicit in the assumptions used by the modellers and relates to the part of the landscape in focus. The EC approach assumes that nutrient generation is uniform within individual land-use classifications, implicitly focussing on hillslope processes and implying that
management effort needs to be focussed across a wide area. By contrast, the approach used by Olley and Deere implies a strong link between erosion and P transport, focussing the user’s attention on management of gullies and streambanks.

In order for the SCA to focus management interventions on parts of the catchment that are most likely to produce the most efficient and cost-effective reductions in nutrient loadings, there is a need to identify which part of the landscape are generating the most nutrients. Both models place emphasis on the need to manage nutrient exports from grazing land, but whilst the work of Olley and Deere places the focus for P management on the main sources of sediment (and sediment-bound P), the EC approach implies that management attention should have a wider focus and include hillslopes.

Catchment monitoring reported by Baginska et al. (1998) and Cornish et al. (2002) suggests that nutrient delivery to streams can be high at scales up to 2 km² and there is growing evidence that P is mobilised from well-covered pasture mostly in the form of dissolved P (e.g., Haygarth et al., 1998; Gillingham and Thorrold, 1999; McGuickin et al., 1999; Nash and Hallwell, 1999, 2000; Cornish et al., 2002; McCaskill et al., 2003; Hart et al., 2004). It is likely that P that is later detected in particulate form (e.g., in channel sediments) after having sorbed to sediment during transport (Gillingham and Thorrold, 1999). If this were the case, nutrient delivery ratios from hillslopes would be higher than would be expected from sediment alone.

Further research is needed on the mobilisation processes of N and especially P from well-covered pastures, and the delivery of the nutrients to streams. Until then, models of nutrient transport will essentially only provide the SCA with a modeller’s opinion.

However, despite the different assumptions and methods used by Olley and Deere compared to that of the EC approach, the results of both models produced some similar results and conclusions. That is:

- the ratio between P and N concentrations estimated by the EC model are almost identical to those observed in the stream (1:8.5). This occurred despite the fact that the P to N ratio was not considered when choosing ECs. Only measurements reported in literature were used;
- pastures (i.e. cattle) are likely to be the major nutrient pollution source; and
over a year, diffuse sources outweigh point sources as the main source of nutrients in Wingecarribee River by about 10:1.

The SCA can be confident that these conclusions have a high likelihood of being correct and can tailor management priorities and interventions accordingly.

6.1.6.4 Comparison with Long (2003)

The most recent application of EC modelling of the drinking water catchments was undertaken by Long (2003), and a comparison of current work with that of Long was requested by the SCA. Long used a very similar EC approach to estimate nutrient source strengths, and attribute N and P loads to each major land-use and hydrological sub-catchment. Long initially used more land-use categories than are used in this research to estimate total loads of N and P, but these were combined in his final analysis to allow comparison of loads produced by eight land-use categories: agriculture, bushland, disturbed lands, forestry, mining, roads, sewage treatment plants and urban areas. Sewage treatment plants (STPs) were treated as point sources, and estimates of nutrient discharge from STPs were included in Long’s analysis. Long did not attempt to account for loads specifically from dairy or vegetable farms.

The focus in the simple EC approach that is the subject of this thesis has been on diffuse sources where locally relevant literature is available to support the allocation of ECs. Due to the absence of supporting data, for the current research, no attempt has been made to assess the contribution of forestry, mining or roads (except for informational purposes in Appendix 5). Forestry has been included with bushland and given a single EC. Mining and roads have only been included as part of the “Other” land-use category. All point sources, including sewage treatment plants, have also been ignored.

In order to compare, contrast and check the data analysis undertaken in both Long (2003) and the current analysis, the inputs and outputs of both models were closely examined. The outputs are not compared here, because fundamental differences in interpretation of the land-use data make any direct comparison meaningless. Of particular note, Long interpreted the “Agriculture - Intensive Pasture” in the data as “Cropping/Intensive plants?” (sic). Ground-truthing of the area showed no basis for this classification and, for the current work, “Agriculture - Intensive Pasture” was
reclassified as “Degraded pasture”. The subsequent ECs for this land area are different.

There is also some variation in EC values used by Long (2003) and the current research (Table 6-5). Variation in ECs highlights the subjectivity and uncertainty inherent in the allocation of ECs, and the importance of using robust and locally relevant EC values.

Table 6-5. Comparison of EC values used in the current study with EC values used by Long (2003).

<table>
<thead>
<tr>
<th>Land-use</th>
<th>This Study</th>
<th>Long (2003)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>P</td>
<td>N</td>
</tr>
<tr>
<td>Forest</td>
<td>0.05</td>
<td>1</td>
</tr>
<tr>
<td>Unimproved pasture</td>
<td>0.3</td>
<td>3.9</td>
</tr>
<tr>
<td>Degraded Pasture*</td>
<td>2</td>
<td>4</td>
</tr>
<tr>
<td>Improved pasture</td>
<td>0.7</td>
<td>4.2</td>
</tr>
<tr>
<td>Urban - residential</td>
<td>0.4</td>
<td>9</td>
</tr>
<tr>
<td>Urban - industrial</td>
<td>3</td>
<td>20</td>
</tr>
<tr>
<td>Rural residential</td>
<td>0.7</td>
<td>4.2</td>
</tr>
<tr>
<td>Dairy farms</td>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td>Horticulture-orchards</td>
<td>0.6</td>
<td>6</td>
</tr>
<tr>
<td>Vegetable farms</td>
<td>10</td>
<td>100</td>
</tr>
</tbody>
</table>

* Long (2003) interpreted this land area as “Cropping/Intensive plants?” [sic]

Although plantation forests and bushland were treated separately by Long (2003), the ECs applied to each were almost identical. Curiously, the EC used by Long (2003) for P is lower for plantation forest than for native forests. Long (2003) used ECs of 0.1 kg P/ha/yr and 1.5 kg N/ha/yr for native forest (“Bushland”) and 0.09 kg P/ha/yr and 1.5 kg N/ha/yr for plantation forest (“Forestry”). This is counter-intuitive, since plantation forests are likely to be fertilised and subject to more disturbance than unmanaged native forests, and therefore have higher nutrient export rates.

Another point of difference is that the area of Nattai River sub-catchment appears to have been doubled in the spreadsheet used by Long (2003). In this research, the area of Nattai River catchment was measured at 44,613 ha. Long (2003) records the area as 89,225 ha.

Consideration was given to the use of sewage treatment discharge data reported by Long in the current research, but this idea was rejected due to a lack of confidence in
the method used to calculate sewage treatment discharge loads. The estimates used by Long differ substantially to those of Olley and Deere (2003) for STPs at Berrima, Bowral and Moss Vale, and those reported by Sydney Water (2003) for STPs at Mount Victoria and Warragamba (Table 6-6). The discrepancy appears to have arisen because the calculations in Long (2003) are based on legal concentration limits, and not actual concentrations or loads.

Table 6-6. Comparison of STP discharge estimates.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>P</td>
<td>N</td>
<td>P</td>
</tr>
<tr>
<td>Berrima</td>
<td>219</td>
<td>438</td>
<td>10</td>
</tr>
<tr>
<td>Bowral</td>
<td>1,650</td>
<td>27,500</td>
<td>660</td>
</tr>
<tr>
<td>Moss vale</td>
<td>414</td>
<td>6,619</td>
<td>190</td>
</tr>
<tr>
<td>Mt Victoria</td>
<td>1,752</td>
<td>11,680</td>
<td>4*</td>
</tr>
<tr>
<td>Warragamba</td>
<td>1,606</td>
<td>12,410</td>
<td>953*</td>
</tr>
</tbody>
</table>

* Average of 3 years 00-01 to 02-03 inclusive.

6.1.7 Impact on Decision-making

Phosphorus

According to the simple EC model, pastures generate over 80% of the potential P load in the drinking water catchments and over 60% of the total N load while occupying less than 30% of the total catchment area.

Of the pastureland, degraded pastures appear to be of particular concern, generating 38% of potential P runoff from just 4% of the land area. Degraded pasture is most often found in the Reedy Creek, Mulwaree River, Boro Creek and Braidwood Creek sub-catchments. If the assumptions used in this model are reasonable, sub-catchments with large areas of degraded pasture should be priority catchments for SCA intervention.

However, based on the land-use map, sub-catchments with significant areas of degraded pasture are often situated farthest from reservoirs, so their impact on water quality may often be local. Due to the high rates of soil loss reported from degraded
pasture in these landscapes (Neil and Fogarty, 1991; Armstrong and Mackenzie, 2002), P loss from degraded pasture is likely to be predominantly by erosion and therefore, in terms of threat to reservoirs, accounting for in-stream processes including sedimentation (i.e. the delivery ratio from stream to reservoir) may be an important water quality indicator, which has not been considered in this study.

Parts of catchments with dairy and vegetable production have potentially high per unit area exports, but are not likely to be major contributors to total exports of N or P, contributing <1 and 4% to total potential pollutant loads respectively. The impact of dairies and vegetable farms in the catchments is more likely to be local than catchment-wide.

Wollondilly River sub-catchment has the greatest potential contribution of nutrients of all SCA hydrological sub-catchments, but exports per unit area are moderate, due to the presence of extensive areas of forest (57% of the sub-catchment) and unimproved pasture (36%). There may therefore be little scope to reduce P exports from this sub-catchment, except locally in areas of more intensive land-use or degraded pastures.

The model’s outputs were also sorted using a plot of the average potential P exports per hectare (i.e. the average export rate) against the total potential P export for each sub-catchment (Figure 6-9). Mulwaree River (43 tonnes P/yr, 0.55 kg/ha/yr) and Reedy Ck (40 tonnes P/yr, 0.7 kg/ha/yr) have high average ECs and high total potential P exports and would therefore rank high management intervention. Wollondilly River would rank lower because of low unit area loads (0.23 kg P/ha/yr), in spite of this sub-catchment delivering the highest estimated load overall (61 t/yr).

Consideration also needs to be given to the proximity of nutrient sources to ecologically sensitive areas and reservoirs. A map showing the total potential P exports and unit area P loads from each sub-catchment is given in Figure 6-10.
The SCA’s hydrological sub-catchment map units are too large to provide more than a rough guide to the proximity of nutrient sources to important water bodies. The EC approach allows finer scale analysis and more detailed outputs can be generated with the data that was used in the current study. However, this was not an objective of this study and was not undertaken.

Nitrogen

If the assumptions of the model are generally true, reducing diffuse N exports across the drinking water catchments will be difficult. Both this study, and the study by Olley and Deere (2003) found that sources of N are more evenly distributed than sources of P, with almost 80% of potential N exports estimated to originate from forest or unimproved pasture that are unfertilised (except for managed forests) and have low unit area exports. Nitrogen fertiliser use, and therefore N ‘hotspots’, will typically be restricted to dairy farms and vegetable farms from which exports may be locally high.
Figure 6-10. Potential annual exports and mean annual unit area exports of P for each sub-catchment.
Further, because of its chemistry, N is subject to more transformation processes and is more mobile in soil than P. Modellers (and catchment managers) therefore need to consider more fate and transport pathways, including lateral movement through soil to streams and vertical percolation through soil to groundwater and eventually to streams in base flow. Nitrogen is therefore considered to be more difficult to model and manage than P (McDowell et al., 2004). Identifying localised N export hotspots may not be a realistic proposition using readily available data at the catchment scale.

6.1.8 Reflection

Key results of the current study were that:

- grazing accounts for most of the estimated N and P generated by diffuse sources in the SCA drinking water catchments;
- better management of degraded extensive pastures may provide an opportunity for significant improvement in water quality in some sub-catchments, however exports of nutrients from degraded land need to be experimentally verified;
- dairy farms occupy a small proportion of the total catchment area and are a relatively small generator of total nutrient loads, but they may be a significant source of nutrient exports in some sub-catchments. The model results suggests that dairy farms may contribute over one quarter of nutrients generated in Kangaroo River and Wingecarribee River sub-catchments, However verification of nutrient exports for dairy pasture is needed for these sub-catchments;
- vegetable farms occupy a very small proportion of the total catchment area and model results indicated that they are not a major contributor to total catchment loads of nutrients. However, vegetable farms (mostly market gardens) may generate almost one third of nutrients in the Werri Berri Creek sub-catchment. Where farm dams or other management features that mitigate these pollutants are not present, vegetable farms pose a local threat to water quality;
- nutrient pollution from urban sources is higher than from agricultural sources in the Blue Mountains sub-catchments and parts of the Lower Cox’s River sub-catchments; and
in Wingecarribee sub-catchment, the total estimated annual P and N generation of 29 t/yr and 243 t/yr was comparable with estimated loads of 21 t/yr and 180 t/yr, respectively calculated by Olley and Deere (2003) using existing water quality and flow data. Notwithstanding the limitations associated with measuring and modelling nutrients in the environment, the differences between predicted values in the current study and observed values may be explained as a result of the net attenuation in the flow path between the source and sub-catchment outlet, and mobilisation of other diffuse sources, such as channels and erosion gullies.

However, the certainty that can be attached to these conclusions is difficult to ascertain. The accuracy of an EC model is based on the reliability of the land-use map and the robustness and local applicability of the EC values. As is the case in all diffuse pollution modelling, the utility of model outputs is constrained by multiple levels of uncertainty. Several data issues and assumptions have had a significant bearing on the EC model’s results. Some assumptions and limitations highlight areas where further research is required if models are to deliver to the SCA more robust estimates of nutrient source strengths. The most fundamental issues that need to be addressed are listed below:

- Literature EC values such as those compiled in CECIL (AWT, 2001b) and other sources of consolidated literature values (e.g., Marston et al., 1995) were useful for developing comparisons of data availability and potential source strengths relating to different land-uses, but these tools appeared to have abandoned, rejected or forgotten by the SCA. The SCA should adopt the recommendation of the CECIL developers and continue to develop the CECIL database and make use of other tools, such as NEXSYS (Young et al., 1997);

- There are relatively few locally relevant research papers reporting N exports from any land-use with which to guide modelling, including the setting of N coefficients. If modelling N loads or concentrations in streams is deemed to be important, much further research into N mobilisation, transport and delivery to streams is needed;

- The land-use map was only marginally useful for coarsely defining areas of different nutrient source strengths or pollution potential. Although a new
land use map was in development at the time of the current study, SCA staff and the author felt that the new map was also a blunt tool for comparing nutrient exports from different land-use types. The grass-roots development of a land-use map that considers land-use in terms of soil and water pollution potential would greatly enhance the ability of any large-scale model to simulate potential nutrient fluxes;

- Given the potential impact of degraded lands on water quality the extent of degraded pasture needs to be better defined and quantified;
- In this study, the rate of N generation from degraded pasture was arbitrarily assumed to be 4 kg/ha/yr. The export of N from degraded pasture to receiving waters has not been substantively quantified in locally relevant catchments. A degraded pasture EC for N (or some other comparative quantitative measurement) needs to be experimentally established;
- A 7-fold increase in exports of P from degraded pasture (2 kg/ha/yr) compared with unimproved pasture (0.3 kg/ha/yr) was assumed. The high rate was based on a measured 7-fold increase in erosion reported in the Southern Tablelands (Neil and Fogarty, 1991). As for N, there are insufficient studies to provide a locally-relevant comparison of P export from degraded pasture as compared with any other pasture type;
- The size and intensiveness of dairy farms in the drinking water catchments was based on a simple set of assumptions that require verification;
- The estimated exports from some sub-catchments with dairy farms were very high. Verification of the EC values used in the current study, and on-site assessment of connectivity to receiving water should be used to confirm these estimates;
- According to the land-use map, only 2% of pasture is improved (i.e. sown to introduced species and regularly fertilised). Given the relatively high potential loads for this pasture type, it is important to verify the area of improved pasture in future land-use mapping;
- The area in the land-use map defined as ‘Rural Residential’ seems unrealistically small, and there are few relevant studies that can be used to confirm the EC values that have been used in the current research;
• For this study, native and plantation forests have been categorised as one unit, with the EC (0.05 kg P/ha/yr and 1 kg N/ha/yr) being mainly derived from studies of local native forests. There were insufficient locally relevant data to reliably estimate exports from local plantation forests. A report by Cullen et al. (1988) suggests that exports of P may be similar for pine forest and native forests, but literature on sediment exports indicates much higher rates of erosion from managed forests than from unmanaged, native forest (Neil and Fogarty, 1991; Wallbrink et al., 2002). This inconsistency requires further investigation; and

• The land-use map does not distinguish forest that has a grazed under-story. If grazed forest is categorised as ‘forest’, nutrient exports may be much higher than estimated for some forested areas. This is an issue for future land-use mapping, and, in order to estimate nutrient exports, for experimental examination;

In addition, the comparison between the assumptions used in the current study and the assumptions of Olley and Deere (2003) highlight the need for a better understanding of nutrient mobilisation processes, assimilation in the transport pathway and nutrient delivery to streams. Without a better understanding of these processes, the SCA will be unable to produce a robust nutrient budget, but may still.

Despite the major differences in assumptions, the current study produced a similar overall result and similar findings to that of Olley and Deere (2003), identifying cattle as the most significant source of nutrients across the SCA’s area of operations.

In order to develop a reliable nutrient budget, a better understanding of the processes of nutrient mobilisation, transport, assimilation and delivery from sources to streams is needed. This conclusion applies to the use of any modelling approach that seeks to produce a robust nutrient budget at scales applicable to the needs of catchment managers.

Other Issues Relating to Modelling for Decision-making

Although the EC approach described above was relatively simple, relating the land-use map and EASI data to the most recent locally relevant research allowed for
refinement of EC estimates. Basic EC models use simple GIS and spreadsheet functions and are easy to set up and manipulate. All input data are basic and can be modified iteratively. Outputs, including tables and charts can be produced effectively instantaneously.

Perhaps the most useful application of the approach and the modelling exercise itself was that in developing the model, important research needs were identified. In particular, it was noted that experiments indicating the relative importance of experimental studies of delivery from hillslopes to edge-of-stream should be undertaken, and that there was very little nutrient generation or delivery data for most land-uses.

6.2 Enhanced Export Coefficient Model

A major drawback of the basic EC approach is that it ignores spatial variations in the intensity of nutrient generation within land-use types, and therefore does not allow fine-scale targeting of management interventions or assessment of relative source strengths at high spatial and predictive resolutions.

Several workers have developed enhanced EC models that seek to overcome these and other drawbacks (Table 6-7). For example Johnes and Heathwaite (1997) and McNamara and Cornish (2004) weighted ECs based on proximity of some land-uses to streams; Atech Group (2000) weighted ECs based on annual rainfall (a surrogate for runoff) and geology. More sophisticated approaches have also been used; Endreny and Wood (2003) used a “contributing area and dispersal area” weighting function that adjusted ECs within the range reported in literature based on landscape position, run on from upslope contributing areas, and downslope nutrient trapping opportunities (buffers). In general however, there are not enough quantitative data to calibrate or verify these and most other models of nutrient fluxes in catchments.

Most SCA staff involved in this project expressed a desire to have an enhanced EC model. Enhancements seemed to be perceived as a way of generating predictions that had higher spatial and/or temporal resolution and greater accuracy, and made better use of available data and advances in knowledge and technology. However, the scientific basis for most feasible enhancements is limited and their utility in decision-making are sometimes questionable. As Donelly et al. (1994 p. 28) points out, even
the fact that algal blooms have increased since the arrival of Europeans is “an unproven assumption”, and algal blooms may be more closely linked to restriction of flow than P concentration or loads.

Table 6-7. Examples of enhancements to the EC approach.

<table>
<thead>
<tr>
<th>Reference</th>
<th>Enhancements</th>
</tr>
</thead>
<tbody>
<tr>
<td>CMSS (Davis et al., 1991 and Davis and Farley, 1997)</td>
<td>A software package that includes a flow routing/assimilation routine and error bars for generation rates and model outputs. Users can measure the potential impact of different management practices and/or land-use policy changes.</td>
</tr>
<tr>
<td>Johnes (1996)</td>
<td>EC values were adjusted based on stock and human population density and stream discharge.</td>
</tr>
<tr>
<td>Johnes and Heathwaite (1997)</td>
<td>EC values were adjusted based on stock and human population density and stream discharge, and on distance to stream, with the EC for land within 50m of a stream being given a higher value.</td>
</tr>
<tr>
<td>Atech (2000)</td>
<td>“Modification factors” were used to weight ECs based on geology and rainfall (as a surrogate for runoff). For land-uses that occurred on Tertiary basalt, the EC was increased by a factor of 1.5. Similarly, there were four different modification factors for rainfall (arid, 0.1; low, 0.5; medium, 1; high, 1.25).</td>
</tr>
<tr>
<td>Hanrahan et al. (2001)</td>
<td>An EC model was used to provide estimates of nutrient loads at monthly and annual time-steps.</td>
</tr>
<tr>
<td>Endreny and Wood (2003)</td>
<td>A “contributing area and dispersal area” weighting function was used to adjust generation rates, within the range reported in literature, based on landscape position, run on from upslope contributing areas, and downslope nutrient trapping opportunities (buffers)</td>
</tr>
<tr>
<td>Baginska et al. (2003)</td>
<td>A statistical technique (‘bootstrap’) was used to reduce subjectivity in the selection of ECs from a relatively small amount of available data and deliver improved estimates of confidence limits.</td>
</tr>
<tr>
<td>Dela-Cruz et al. (2003)</td>
<td>This paper describes a framework for EC modelling at different spatial resolutions. The approach relies on tiered sets of EC values that are sufficiently generalised to be representative at global, regional or local scales.</td>
</tr>
<tr>
<td>McNamara and Cornish (2004)</td>
<td>EC values were modified based on different landscape features. The modifications were based on NEXSYS outputs and included soil type, distance to stream, presence of gully erosion and connectivity to stream.</td>
</tr>
<tr>
<td>Khadam and Kaluarachchi (2006)</td>
<td>EC values were weighted using modelled estimates of sediment discharge as a surrogate for “hydrologic variability”.</td>
</tr>
</tbody>
</table>

One often-suggested enhancement of the EC model was that the catchments be divided into more land-use categories. Whilst this seems like a good use of more detailed land-use information, it ignores the fact that broad-scale land-use data is not
entirely ground-truthed. The most recent land-use data available to the SCA was found to contain many fundamental errors (M. Hart, pers. comm., Dec. 7, 2004). In any case, Baginska et al. (2003) found that only four or five broad land use categories were required to estimate the potential nutrient exports sufficiently well to identify the sub-catchments that were the largest nutrient contributors.

The scientific workshop and the literature on the subject highlight the fundamental lack of knowledge about the relative importance of processes that affect P mobilisation and transport at most scales. Much discussion at the scientific workshop revolved around debate about the importance of gullies and streambank erosion as sources of P in streams. Armstrong and Mackenzie (2002) argue that gullies are likely to be important source of sediment, and possibly P, even if concentrations of P in subsoils are low (see Donnelly et al., 1994). Other workers, such as Nash and Halliwell (1999, 2000), note that most P transported from well-managed pasture in southern Australia is in a dissolved form in surface runoff.

According to Donnelly et al. (1998), the relative importance of P releases from bottom sediments and delivery and in-stream transport of sediment-bound P is unknown, and in any case, the link between in-stream P concentrations and algal blooms is uncertain. Harris (1994) (in Donnelly et al., 1998 p. 23) argues, “in general it seems that, in Australia, phytoplankton dynamics are dominated by irregular and infrequent extreme events”, but he goes on to say that there is relatively little supporting evidence for this view.

Another issue raised by some SCA staff was whether it might be more worthwhile predicting concentrations rather than loads, since concentration data was seen to be more versatile. However, as pointed out by Harris (1994), although unproven, it is likely that algal blooms are linked to extreme climatic events and therefore load data may be more relevant than concentration data, because much exported P is trapped in sediment and becomes a source of bioavailable P.

At present, there is no reliable, generalisable catchment model of P generation or transport (Donelly et al., 1994).
6.2.1.1 Enhancing EC Modelling in Sydney’s Drinking Water Catchments

An assessment of the literature used to develop the generation rates for pasture used in the simple EC model seemed to show that variations in P generation between and within pasture types was predominantly a function of soil P and depth of runoff. When the data (Table 6-8) were analysed, a strong correlation between P exports, Soil P and runoff depth was observed (Equation 2).

<table>
<thead>
<tr>
<th>Site</th>
<th>Texture</th>
<th>Pasture Type</th>
<th>Soil P (Colwell) (mg/kg)</th>
<th>Runoff (mm/yr)</th>
<th>Conc. in runoff (mg/L)</th>
<th>Generation rate (kg/ha/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Carcoar¹</td>
<td>Sandy Loam</td>
<td>native</td>
<td>14</td>
<td>19</td>
<td>0.22</td>
<td>0.04</td>
</tr>
<tr>
<td>Vasey¹</td>
<td>Sandy Clay Loam</td>
<td>native</td>
<td>17</td>
<td>20</td>
<td>0.22</td>
<td>0.04</td>
</tr>
<tr>
<td>Vasey¹</td>
<td>Sandy Clay Loam</td>
<td>sown</td>
<td>43</td>
<td>20</td>
<td>0.38</td>
<td>0.07</td>
</tr>
<tr>
<td>Carcoar¹</td>
<td>Sandy Loam</td>
<td>sown</td>
<td>24</td>
<td>19</td>
<td>0.56</td>
<td>0.11</td>
</tr>
<tr>
<td>Ginninderra²</td>
<td>Sandy Loam</td>
<td>sown (est.)</td>
<td>20 (est.)</td>
<td>30</td>
<td>0.40</td>
<td>0.12</td>
</tr>
<tr>
<td>Wagga¹</td>
<td>Loam-Clay Loam</td>
<td>native</td>
<td>15</td>
<td>87</td>
<td>0.24</td>
<td>0.21</td>
</tr>
<tr>
<td>Maindample¹</td>
<td>Sandy Loam</td>
<td>native</td>
<td>14</td>
<td>105</td>
<td>0.32</td>
<td>0.31</td>
</tr>
<tr>
<td>Ruffy¹</td>
<td>Sandy Clay Loam</td>
<td>native</td>
<td>14</td>
<td>119</td>
<td>0.34</td>
<td>0.33</td>
</tr>
<tr>
<td>Ruffy¹</td>
<td>Sandy Clay Loam</td>
<td>sown</td>
<td>30</td>
<td>119</td>
<td>0.77</td>
<td>0.64</td>
</tr>
<tr>
<td>Wagga¹</td>
<td>Loam – Clay Loam</td>
<td>sown</td>
<td>41</td>
<td>87</td>
<td>1.00</td>
<td>0.87</td>
</tr>
<tr>
<td>Maindample¹</td>
<td>Sandy Loam</td>
<td>sown</td>
<td>38</td>
<td>105</td>
<td>0.76</td>
<td>1.22</td>
</tr>
<tr>
<td>Camden¹</td>
<td>Clay-Clay Loam</td>
<td>sown</td>
<td>60 (est.)³</td>
<td>158</td>
<td>1.40</td>
<td>1.90</td>
</tr>
<tr>
<td>Windsor²</td>
<td>Clay-Clay Loam</td>
<td>sown</td>
<td>&gt;300⁰</td>
<td>204</td>
<td>4.20</td>
<td>8.60</td>
</tr>
</tbody>
</table>

¹ From SGS Program experiments at several sites in southern NSW and VIC only: McCaskill (2003).
² Costin (1980).
³ Comish et al. (2002).
⁴ Baginska et al. (1998).
⁵ Measured soil P was ~40 (Bray).
⁶ This very high rate was attributed to P enrichment resulting from soil erosion.

Equation 2. Correlation between P exports, soil P and depth of runoff.

\[ PE = 0.002P_{(\text{colwell})} + 0.0064DR \]

\[ R^2 = 0.88 \quad (P< 0.001) \]

where: 
- \( PE \) is the amount of P exported;
- \( P_{(\text{colwell})} \) is the P concentration in soil (mm/kg); and
- \( DR \) is the depth of runoff (mm)

Nash and Halliwell (1999; 2000) have noted that, for well-managed pasture in southern Australia, most P is transported in a dissolved form in surface runoff, and that the most important factor determining the load of P lost in surface runoff is runoff volume. For well-managed pasture, runoff volume is generally found to be more important than soil type, management, including the history of fertiliser use, pasture
cover, soil erosion and other known factors that determine P exports (Nash et al., 1998).

6.2.1.2 The Selected Approach

A simple method was chosen to weight diffuse P generation rates in the SCA’s hydrological catchments according to relative variations in streamflow contribution across pastures (excluding dairy farms) and forests. The weighting was performed using standard GIS tools so that each location (pixel) in a GIS map of diffuse P generation in the hydrological catchments had an estimated generation rate that varies based on land-use and estimated streamflow contribution at that point.

The streamflow contribution estimates are based on empirical relationships between slope steepness, average annual rainfall, potential evaporation and average annual streamflow observed at gauged locations that are deemed to be representative of the SCA’s area of operations. Weighting generation rates allows users to identify areas that are likely to be more or less intense sources of diffuse P with greater resolution than the standard EC approach.

The variation in the weighted ECs reflects an assumption that the concentration of P in runoff is primarily related to land-use type, and that the total load of P from a given area is a function of both P concentration in surface runoff (*i.e.* quickflow) and the total volume of runoff. In effect, land-use type has been used as a surrogate for many natural and ‘anthropogenic’ factors than affect P mobilisation and transport, such as soil condition, vegetation cover and land management practice. Streamflow contribution has been used as a surrogate for surface runoff volume, because relative differences in surface runoff volume could not be predicted with confidence at the desired scale using available data.

The enhanced EC model highlights the importance of runoff since ECs for pastures and forests, which make up the bulk of the area of the drinking water catchments are weighted according to streamflow contribution. Ideally, only the surface runoff or ‘quickflow’ component of streamflow contribution should be used to weight generation rates, since it is likely that most P from hillslopes in south-eastern Australia is transported in surface runoff. Unfortunately, no method for reliably calculating surface runoff from readily available data was found during this study
(Croke, 2005). As a result, streamflow contribution was used as a surrogate for surface runoff.

Catchment-wide soil P data do not exist and are estimating soil P concentrations using surrogate data was deemed to be impractical, so this factor was not used in the enhanced approach, except implicitly, insofar as land-use influences typical soil P concentrations, which in turn are reflected in the EC values for each land-use.

The enhanced EC model was not used to assess N generation. The simple EC model showed that most diffuse N is generated by forests and unimproved pastures, despite relatively low unit area exports from these land-uses. There appears to be little or no scope to manage N exports from either forest or unimproved pasture, since, by definition, no N fertiliser is used and no legumes are sown or stimulated by P fertiliser in unimproved pasture and most (i.e. unmanaged) forests. Phosphorus is generally regarded as the principal nutrient driving algal blooms and accelerated eutrophication in freshwater systems (Ball et al., 2001), and is therefore the most significant pollutant in terms of nutrient management. Nitrogen may also be an important driver of eutrophication (Harris, 2001), but the chemistry of N means that modelling fluxes of N using available data at a whole-of-catchment scale using an enhanced EC approach is significantly more problematic than for P (Endreny and Wood, 2003).

When compared with the un-weighted generation rates from the simple EC model, hydrologic weighting increased estimated P export in humid sub-catchments such as Kangaroo River, Mid Cox’s River and Upper Nepean, and lowered P generation estimates in drier catchments such as Mulwaree River, Upper Wollondilly and Nerrimunga River. The weighted approach also highlights potential differences in P generation within sub-catchments. For example, weighted export rates at Brogers Creek, in one of the wettest parts of Kangaroo River sub-catchment are higher than those at Bundanoon Creek, in a generally drier part of Kangaroo River sub-catchment.

The primary use of this model would be as a screening or scoping tool that can be used to compare and contrast the relative source strengths of different land-uses, sub-catchments and regions within sub-catchments.
6.2.1.3 Assumptions and Limitations

The primary assumptions that support the enhanced EC model are that:

- the concentration of P in runoff is primarily related to land-use type;
- annual runoff loads of P for pasture-based land-uses and forest are largely a function of land-use type and depth of runoff;
- streamflow contribution from a given area of land is a function of slope steepness, annual rainfall, potential evaporation and grass/tree cover;
- relative variations in streamflow contribution can be quantified at relatively fine scales using these data; and
- variations in estimated streamflow contribution can be used as a surrogate measure of variations in runoff depth across a catchment.

The significance of land-use in determining variations in P exports is generally accepted (e.g., Johnes and O'Sullivan, 1989; Young et al., 1996; Worrall and Burt, 1999). Recent monitoring studies confirm the affect of runoff volumes in determining P exports, especially for well-managed pasture (see Nash and Halliwell 1999, 2000; Nash et al., 1998).

Most uncertainty in the enhanced EC approach described in this part relates to the reliability of the streamflow contribution estimates and, in the absence of a data-efficient method for filtering baseflow, the representativeness of streamflow contribution as a surrogate for surface runoff. Further discussion relating to these hydrological issues is contained in Appendix 6.

6.2.2 Method

The basic framework of the enhanced EC model was the same as for the simple EC model. For the enhanced model, potential P exports were estimated based on the same land-use classes used in the simple EC model (water; forest, degraded pasture, unimproved pasture, improved pasture, horticulture (orchards), urban residential; urban industrial, rural residential, dairy farms and vegetable farms). In the enhanced model, the potential export rates for pasture (including rural residential land and excluding dairy farms) and forest were weighted according to the estimated volume of streamflow that is generated at any given point in the catchment (i.e. streamflow
The weighting method ensured that the average generation rate for each land-use remained equal to the generation rate used in the simple EC model, since these were deemed to be typical P generation rates.

The method for calculating P generation from horticulture (orchards), urban residential, urban industrial, dairy farms and vegetable farms was identical to that used for the simple EC model. The method that was used for weighting the generation rates is not suited to urban or cropping situations, or where the boundary of a land-use cannot be accurately determined in relation to other landscape characteristics. Dairy farms and vegetable farms are only identifiable as point features in the existing land-use data (i.e. EASI data), and therefore their exact location and/or spatial extent is unknown. In any case, the extent of dairy and vegetable farms and other unweighted land-uses is small (5% of the SCA area of operations), and the P ECs that were used in the simple EC model should already reflect the hydrological conditions that are unique to those land-uses, such as the use of irrigation.

6.2.2.1 Streamflow Contribution Modelling

At the author’s request, Dr Barry Croke at the Australian National University provided streamflow contribution estimates for use in the enhanced EC model. The estimates of streamflow contribution were calculated in a GIS using a digital elevation model (to estimate slope steepness), average annual rainfall data supplied by the SCA and potential evaporation data obtained from the National Land and Water Resources Audit (NLWRA) database. The outputs of the streamflow modelling were two GIS data layers giving the streamflow contribution in mm/yr. One layer estimates streamflow contribution assuming 100% grass cover (Qg), and the other assumes 100% forest cover (Qf). Streamflow contribution is measured in millimetres (mm), but the estimates should only be regarded as an indication of the relative streamflow contribution (B. Croke, pers. comm., Feb. 11, 2005). For weighting ECs, relative estimates should be sufficient.

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Although the grass and forest streamflow data were used separately in this analysis (i.e. pasture was assumed to be 100% grass covered, and forest 100% woody cover), where the percent of woody cover is known, the fraction of woody cover may be used to calculate streamflow contribution for a given area using a linear combination of these data layers, using the formula in Equation 3.

**Equation 3. Method to calculate streamflow contribution where the percent of woody cover is known (B. Croke, pers. comm., Feb. 11, 2005)**

\[ Q = f Qf + (1-f) Qg \]

where: \( Q \) is the streamflow; and
\( f \) is the woody cover (fraction of cell covered by trees/shrubs).

### 6.2.2.2 Data Preparation

Streamflow contribution data were supplied in a Geographic (lat/long) coordinate system. This was reprojected to a Projected (linear/metres) coordinate system that allowed mathematical computations in metric units to be performed using ArcView 3.3 and the Spatial Analyst extension. Reprojection was accomplished using a nearest-neighbour algorithm with the Grid and Theme Projector extension (Jenness, 2004). Projecting raster grids reduces the accuracy of data and is not desirable. Ideally, streamflow data should be produced in a projected coordinate system in the first instance.

The land-use map was converted from a vector format to a matching grid (raster) format (i.e. with same-sized grid squares that overlap the streamflow contribution data) using ArcView and Spatial Analyst. In a vector format, parcels of land are ‘discretised’ using polygons that represent an outline of the contiguous areas of each land-use. In the raster format, land used is discretised based on regularly shaped (usually square) cells (pixels) that combine to form a grid. The grid format allows the manipulation and analysis of data that is needed to weight ECs. For the purpose of this study, each pixel represents an area of approximately 24 m x 24m. This size was chosen to match the grid size used in the streamflow contribution layer.

### 6.2.2.3 Weighting of Pasture and Forest Export Coefficients

In the enhanced EC model, the generation rate at every pixel in each pasture-based land-use and forests was adjusted based on streamflow contribution estimates. The
weighting of generation rates was achieved by calculating the mean streamflow contribution for each land-use. If the value of streamflow contribution at a given location is equal to the mean streamflow contribution for the land-use that occurs at that location, then the generation rate for that location will be equal to the generation rate that was used in the simple EC model. The simple model’s generation rates were used as the basis for weighting generation rates in the enhanced model because they were deemed to be typical for each land-use across all of the SCA catchments (see Ch. 6.1). Weighting is used to identify areas that are likely to be higher or lower than the typical generation rate for a given land-use, and estimate the likely magnitude of variation. If the streamflow contribution estimate is 50% lower than the mean streamflow contribution, the generation rate will be half of that used in the simple EC model, and so on. This weighting method ensured that the mean generation rate for each land-use was equal to the generation rate used in the simple EC model, and that the total estimated loads were also equal.

The variation introduced by the weighting method changed the estimated loads from each sub-catchment. The amount of P generated at each pixel was calculated by multiplying the area of the pixel (a constant) by the weighted generation rate at each point in the grid. Summing all of the pixels for each land-use calculated the estimated total contribution for each land-use. Loads for remaining land-uses were calculated as per the simple EC model. Total loads for each sub-catchment were estimated by adding the loads from all land-uses.

6.2.3 Results

6.2.3.1 Range of Generation Rates
The resulting range of generation rates, and a comparison with generation rate given for similar land-use types using NEXSYS, is shown in Table 6-9. NEXSYS estimates a range in which P (or N) generation rates are likely to fall given land-use type, management practice, climate and/or landscape characteristics, based on expert opinion and published studies in Australia and overseas. With the exception of a small number of extreme outliers in degraded pasture and forest, export rates generated by the enhanced EC model fall within the range estimated by NEXSYS.
Table 6-9. Distribution of weighted generation rates (phosphorus) within each land-use category with reference to the range generated by NEXSYS.

<table>
<thead>
<tr>
<th>Land-use</th>
<th>NEXSYS Range</th>
<th>Mean (unweighted) EC*</th>
<th>Weighted Range**</th>
<th>Standard Deviation</th>
<th>Median</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>all values kg P/ha/yr</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Degraded pasture</td>
<td>0.5-3.0</td>
<td>2.0</td>
<td>1.36-12.17</td>
<td>0.57</td>
<td>1.82</td>
</tr>
<tr>
<td>Unimproved pasture</td>
<td>0.05-3.0</td>
<td>0.3</td>
<td>0.16-1.67</td>
<td>0.13</td>
<td>0.26</td>
</tr>
<tr>
<td>Improved pasture</td>
<td>0.2-3.0</td>
<td>0.7</td>
<td>0.2-2.19</td>
<td>0.36</td>
<td>0.63</td>
</tr>
<tr>
<td>Rural residential</td>
<td>N/A</td>
<td>0.7</td>
<td>0.37-1.99</td>
<td>0.40</td>
<td>0.44</td>
</tr>
<tr>
<td>Forest</td>
<td>0.01-0.20</td>
<td>0.05</td>
<td>0.004-0.270</td>
<td>0.03</td>
<td>0.05</td>
</tr>
</tbody>
</table>

* Originally developed for the simple EC model
** The result after applying the enhanced EC model weighting

Although the maximum estimated generation rate for degraded pasture (12 kg P/ha/yr) is higher than that reported in the limited number of relevant published studies, it might not be unrealistic. Very high generation rates might be expected from some degraded pasture sites. Neil and Fogarty (1991) reported erosion rates from degraded pasture in the southern tablelands that averaged 7 times that of unimproved pasture. In any case, only 5% of the area of degraded pasture has a generation rate higher than the maximum estimated by NEXSYS (3 kg P/ha/yr) and only 190 ha (0.3% of the total area of degraded pasture) has an estimated EC > 5 kg P/ha/yr. Less than 2 ha of degraded pastures have an estimated EC > 10 kg P/ha/yr.

6.2.3.2 Estimated Loads

In order to identify priority sub-catchments for management intervention, it is important to compare and contrast the total loads of P generated by each sub-catchment as well as the average unit area exports. Such a comparison allows the user to make judgements about the local and catchment-wide significance of diffuse P generated by each sub-catchment. The total potential P exports (t/yr) and average unit area P exports (kg/ha/yr) for each sub-catchment are mapped in Figure 6-11.

6.2.4 Comparison with Simple Model Results

To compare the effect of weighting generation rates on the ranking of sub-catchments, total potential P exports estimated using the simple EC model (i.e. unweighted) and the enhanced EC model (i.e. weighted) are graphed in Figure 6-12 and average potential P exports are graphed in Figure 6-13.
Figure 6-11. Potential mean annual exports and mean unit area exports of P for each sub-catchment.
EC model (ECM-A) and the enhanced EC model (ECM-B).

Figure 6.12. Potential P exports (tonnes/yr) for each sub-catchment calculated using the simple

EC model (ECM-A) and the enhanced EC model (ECM-B).
The magnitude and direction of the change in total potential P export and average potential P export induced by weighting generation rates is highlighted more graphically in Figure 6-14 and Figure 6-15 respectively.

The overall effect of weighting generation rates is that considerably higher estimates of total and average potential P exports are given for some catchments in the higher rainfall areas, notably Kangaroo River sub-catchment. A corresponding fall occurs in the drier areas, such as the Mulwaree River sub-catchment.

Within sub-catchment responses to weighting are illustrated by examples from Kangaroo River sub-catchment (Figure 6-16) and Mulwaree River sub-catchment (Figure 6-17). Weighting increased the estimated generation rates for pastures in the wetter, northern and eastern parts of Kangaroo River sub-catchment. In Mulwaree River sub-catchment, there is a reduction in estimated generation rates across most of the sub-catchment. In both catchments, only the weighted land-uses are represented.

The ability of the approach to identify potentially high source areas within land-use types is highlighted by examining the estimated P generation rates in the eastern half of Kangaroo River sub-catchment (Figure 6-16). The approach predicts high P generation rates in Brogers Creek and the upper Kangaroo River catchment. Water quality in these streams is described as poor and the area suffers from occasional outbreaks of blue-green algae (HRC, 1999; Sherman and Orr, 2003). Further, since only pastures and forests are shaded in Figure 6-16 and Figure 6-17, the figures illustrate the extent to which pastures and forests dominate the SCA’s area of operations, and the relevance of an approach that targets these land-uses.
Figure 6-14. Change in total estimated P export for each sub-catchment when generation rates are weighted. Values are tonnes per year. Green shades show a reduction and red shades show an increase in estimated potential P export. Yellow shades indicate little change. The intensity of the colour reflects the amount of change.
Figure 6-15. Change in estimated average P generation for each sub-catchment when generation rates are weighted. Values are kg/ha/yr. Green shades show a reduction and red shades show an increase in estimated potential P export. Yellow shades indicate little change. The intensity of the colour reflects the amount of change.
Figure 6-16. Maps of Kangaroo River sub-catchment showing generation rates for forest and pasture.

6.2.5 Reflection

The approach outlined in this part demonstrates a conceptually simple, spatially based approach for determining nutrient generation rates that is sensitive to landscape characteristics and could meet some SCA needs. The primary use of the enhanced EC model would be as a screening or scoping tool that can be used to compare and contrast the relative source strengths of different land-uses, sub-catchments and regions within sub-catchments for use in environmental reporting and prioritisation of management interventions for RAP.
Figure 6-17. Maps of Mulwaree River sub-catchment showing generation rates for forest and pasture.

Research to improve quantitative knowledge of the factors that affect nutrient mobilisation, transport and delivery to streams and inform future model development will be undertaken in the future as part of an Australian Research Council Linkage Project (LP0561858) entitled “Nutrient generation from rural land and delivery to streams in the Sydney drinking water catchments”.

When discussing the modelling needs with SCA managers and scientists at the second workshop, concern was expressed about the use of simple EC models, largely because they are “imprecise” and lack a hydrologic response component (See Chapter 5.2.6). The application of one nutrient EC for each nutrient and land-use type across all of the SCA area of operations was perceived as being too simplistic given the wide variation in climate, topography and soils across the area, and was perceived as being inadequate for use as a tool for assigning management intervention priorities and assessing land-use impacts. The enhanced EC model was therefore developed as a step towards meeting the SCA’s need for a conceptually simple, spatially based approach to the problem.

Weighting ECs based on estimates of streamflow contribution provides a statistical means for adjusting ECs based on rainfall/runoff variations that might otherwise be
achieved intuitively or subjectively. When compared with the range of ECs reported in literature, for experiments undertaken in the catchment and in similar catchments in south eastern Australia, the range produced by weighting produces a good match and seems to allow the user to identify areas within land-use categories that are likely to be relatively higher or lower sources of diffuse P.

Despite an attempt to derive EC values and land-use categories rigorously and as objectively as possible, the enhanced EC model remains a simple model of nutrient generation that, like all models, embody a number of subjective methodological choices and assumptions. Simplicity allows these choices and assumptions to be easily surfaced and evaluated. This is often not the case in more complex models that are portrayed as more scientifically rigorous, but usually also rely on equally contestable assumptions which are often hidden in the workings of the model. By surfacing assumptions, simple models such as the enhanced EC model may be better at highlighting knowledge gaps, stimulating enquiry and empowering managers to ask relevant questions that can give them an appropriate level of confidence in the model, the science that underpins it and/or the decisions that they make.

In developing the enhanced EC model, several other refinements were considered. Most importantly, a method for reliably estimating the quickflow component of streamflow contribution could be developed. Further improvement might also be possible by taking account of other hydrological and topographic factors that are easily mapped and known to influence the mobilisation and delivery of P to receiving streams and reservoirs. For example, ECs could be weighted based on proximity to stream, soil moisture (e.g., topographic wetness index) and/or soil erosion potential (e.g., using the universal soil loss equation). An assimilation component could be added to simulate the attenuation of P across hillslopes and/or in streams and the sensitivity of model results to uncertain generation rates and/or land-use data could be assessed using a Monte Carlo method, or another statistical method. As noted previously however, adding enhancements to the model that involve the use of contestable assumptions may not improve the utility of the model as a decision-support tool.

Throughout the development of these models, it was noted that many in the SCA desired further enhancement of the EC approach so that more factors that affect
variations in nutrient fluxes were accounted for, or more land-use types were considered. Whilst this may appear worthwhile and might increase the spatial resolution of the model predictions, the algorithms used to incorporate any enhancement will rely on additional untested and contestable assumptions and introduce new sources of uncertainty. Yet in developing the simplistic enhancements used here, there was observed to be a fundamental lack of quantitative knowledge to verify the assumptions that were used. The same limitation would necessarily apply to any catchment models of relative nutrient source strengths at scales that are appropriate to management. As with all models, the contestability of the methodological choices and assumptions used in the enhanced EC model will limit the ability of managers to use the model as a regulatory support tool.

However, these recommendations need to be tempered by the understanding that it is currently impossible to directly measure the emissions of nutrients from soil to surface water and that verification of any model that describes the emissions of nutrients from soil to surface water will be problematic for the foreseeable future.
7 FOCUS GROUP DISCUSSION

7.1 Introduction
Chapter 6 described two nutrient export models that were developed to meet SCA needs that were identified at a workshop involving SCA managers the year before (Chapter 5.1). Not all of the needs could be met due to process knowledge and data constraints that were identified in literature and at a meeting of modellers and water quality scientists conducted following the SCA workshop (Chapter 5.2).

Two models were developed. The first model was a simple EC model based on an extensive review of nutrient export studies. The second model was an enhanced EC model in which the coefficients used in the simple model were weighted on a pixel-by-pixel basis to account for estimated variations in streamflow contribution across the SCA’s area of operations.

Following completion of the modelling, an information session and focus group was organised by the author. This chapter reports on the proceedings and analyses the outcomes with respect to the objectives and broader aims of this thesis, in particular with reference to the affect of model sophistication on confidence in the models and with an expectation that engaging the SCA in the process of model development should lead to higher confidence levels.

In this chapter, the main issues arising from the session are discussed. The purpose of the meeting, the method used to organise and run the focus group and analyse the proceedings are also described.

7.1.1 Purpose
The purpose of the information session and focus group session was three-fold. Firstly, the session provided an opportunity to engage SCA staff in an activity to understand the models and evaluate them against their needs. Secondly, the focus group provided an opportunity to explore the impact of participation over the course of this research, and the impact of model sophistication, on participant’s perceptions and confidence in the models. Thirdly, the session allowed the author to more deeply understand the motivations and value judgments that these catchment managers apply
when selecting or rejecting models as decision-support tools, and in doing so, elucidate the criteria that they used to evaluate them.

7.2 Logistics and Method

7.2.1 Identifying Participants

Participants were selected using a purposive, maximum variation sampling method (Richie et al., 2003 p. 78). Purposive sampling ensures that all participants are relevant to the research. All selected persons were current or potential users of models or model outputs and were chosen on the basis of their ability to provide insights that informed the research questions. Maximum-variation samples are a type of purposive sample that aims to be more representative than an ordinary purposive sample by ensuring that selected participants include diverse or divergent viewpoints that might not be captured if a random sample was taken, or a homogeneous purposive sample representing only the most commonly-held views (Richie et al., 2003 p. 79). In this case, participants were chosen from different sections in the SCA and, based on ad hoc interactions at various seminars and meetings, were thought to have different views about models and modelling.

Nine participants were selected for the focus group. Although participation was voluntary, only one participant could not attend the session, due to a prior commitment. Five participants worked in the Catchment Operations and Major Projects section (COMPs), two from Environment and Planning (E&P) and one from Policy and Governance Directorate (P&G). Seven of the eight participants were also present at the first workshop. In order to protect the privacy of participants, direct reference to the participant or the section that a participant works in has been avoided here.

7.2.2 Venue and Outline of Proceedings

The proceedings of the session were divided into two parts. In the first part, the author gave a PowerPoint presentation describing the models and model outputs. The second part of the proceedings was the focus group discussion itself.

The session was held in the morning in November 2005 at the SCA’s offices in Penrith, western Sydney. All participants were sent information about the two models
one week before the focus group commenced. The initial presentation was 1.5 hours in duration and the focus group session ran for 2 hours, following a brief intermission. Lunch was provided after the meeting to encourage attendance and to thank participants for their contribution.

Questions

Six questions were asked of participants. To prevent the formation of preconceived opinions, the questions were not revealed until after the PowerPoint presentations. The questions were:

1. How does your work unit make catchment management decisions?
2. What prompts you to use water quality models as decision-support tools?
3. How well do the alternate approaches used by UWS demonstrate variations in nutrient source strengths across the SCA catchments?
4. How useful are the alternate approaches for prioritising rectification actions? Why? *
5. How useful are the alternate EC approaches for evaluating DAs for the NorBE test? Why? *
6. Would you use any of the EC approaches for prioritising rectification actions or assessing development applications for NorBE? Why/Why not?

* In the course of the discussion, significant overlap existed between the responses to these questions and they were, for the most part, treated as one question.

Recording and Analysis

Typical methods for recording and analysing focus group discussions are described by Krueger (1988) and Knodel (1993). In this research, the focus group session was recorded using a digital audio recorder. Notes were taken throughout the session on a whiteboard to ensure that the topics that the group felt were important were recognised. An additional SCA staff member was present who acted as note-taker, in case of failure of the audio equipment. He did not contribute to the session.

After the discussion, important topic areas were identified by quickly reviewing the recording and looking for commonalities and divergent views amongst participants. Discussion points relating to issues that seemed relevant to the research topic were
transcribed and sorted into different ‘themes’ that reflected the focus group questions that were asked and the issues that participants found important. A report was sent to each participant outlining the major themes. The next section contains a synthesis of the results for each question.

7.3 Focus Group Results

Question 1  How does your work unit make catchment management decisions?

For field-workers, decisions about “where to go next” (i.e. deciding which parts of the SCA’s area of operations to target for management intervention) are often based on expert opinion, field observations and context-specific considerations. Air photos and GIS are sometimes used to get a “feel” for what is going on. Modelling was described as something that is “going on in the background”. At the planning level, decisions are often based on modelling, but also on political, economic and “equity considerations”. For the NorBE test, the MUSC model (Wong et al., 2001) is often used. Simple EC models are sometimes used, but these are too crude for many applications. NorBE disputes are often resolved through discussion and negotiation rather than model results. More generally, models are one input into the decision-making process – and not necessarily the most important input.

Question 2  What prompts you to use water quality models?

Amongst the group in general, model use was associated with scientific credibility, logic and repeatability. That is, models are a tool for codifying scientific knowledge to facilitate objective and consistent decision-making. Models are also used to get a “feel” for what is going on in the catchments. There also seems to be pressure to meet the expectations of “the Executive” and catchment auditors, and impressive model outputs (e.g., colourful maps) and more sophisticated modelling approaches seem to be used to meet this need.

Question 3:  How well do the alternate approaches used by UWS demonstrate variations in nutrient source strengths?

For this question, discussion largely focussed on the limited range of land-use and land-management types and landscape characteristics that were used to differentiate
areas of differing nutrient source strength in the models. The second model, which adjusts ECs on a pixel-by-pixel basis depending on estimates of streamflow contribution, was seen as a “step in the right direction”.

There was general support for the method of weighting ECs according to hydrological considerations in order to represent areas of varying source strength. A participant advised that the SCA has been developing a similar approach using L-THIA, with estimates of nutrient concentrations rather than loads. The same participant suggested that concentration-based models (e.g., EMC) might be more versatile than load-based models.

Three participants considered that, to be useful, a model needed to include other factors, including land management, land capability, proximity to stream, stream density, soil characteristics and take account of temporal variations (e.g., change in land-use, land management or ground cover, response to rainfall events of different magnitudes, seasonal variations, climate, etc). The desirability of adding an economic component was also mentioned by a participant.

It appeared that the desire for more spatial and temporal detail was driven in part by a need for guidance about which parts of the catchments have the “greatest scope for improvement”, and by a need to ensure that catchment health was not impaired in specific conditions, rather than simply focusing on annualised catchment loads.

*Questions 4 and 5:* How useful are the alternate approaches for prioritising rectification actions? and; How useful are the alternate export coefficient approaches for evaluating DAs for the NorBE test?

These two questions were discussed simultaneously because issues affecting RAP and NorBE seemed to be viewed similarly by the participants. The primary area of concern was that the models did not go far enough to account for differences in nutrient export that were the result of variations in landscape characteristics and land management. Likewise, some considered it necessary to include factors that allow modelling of temporal variations in nutrient fluxes across events, seasons and years, and to include a land capability component that could be compared with land-use to highlight inconsistencies. Some discussion also focussed on the need to account for differences in nutrient generation or export that related to features such as gully
erosion, streambank erosion and streambank vegetation, and that EC models may not be the best tool if these “linear” features need to be incorporated.

A more fundamental concern that was raised by a participant was subjectivity inherent in the selection of EC values in the simple EC model, and that the subjectivity therefore permeated into the other model, even though the intention of the author was to be objective and rely on locally relevant literature.

A statement was made by one participant suggesting that any model outputs should not focus on large sub-catchments, since comparisons between these were “useless” from a management perspective. This viewpoint led to some debate. Most participants said that they could see the benefit of sub-catchment scale comparisons from a strategic and planning perspective.

For three people, the EC models were seen to match the available data and scientific knowledge and therefore provided a potentially useful tool for analysing nutrient relative nutrient source strengths at a macro scale as an input into the planning process. To this subset, the models were seen as a conservative approach that could be used in the absence of more data and more sophisticated and data hungry modelling approaches which are in development or unavailable.

**Question 6:** Would you use any of the export coefficient approaches for prioritising rectification actions or assessing development applications for NorBE?

The view of those familiar with the MUSIC model considered MUSIC to be more suitable for NorBE tests in urban and peri-urban areas. Participants directly involved in the use of models reported that, in rural areas, the simplest EC model was already being used to provide “crude” first-cut estimates of nutrient exports for NorBE tests, although they considered this as only a stop-gap measure, until a more robust method is available. A future product of the eWater CRC (i.e. WATERCast⁶) was mentioned as a potentially viable replacement. One participant stated that the second model was

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seen as “the way to go”, but the approach needed to include more landscape and land-use elements.

For RAPs, other purpose-built DSS-style modelling tools are being developed in-house for each of the SCA’s major sub-catchments, and a view was expressed that these are more suitable than the ECs described here. The primary problem with the approach conveyed at the meeting was the limited number of variables, the perceived quality of the hydrological data used to weight the ECs and the potential for subjectivity in the selection of ECs.

Although the focus of the meeting was on RAP and NorBE, there seemed to be some scope for a simple, low-input and broad-scale model such as the EC approaches in priority-setting and problem identification in some other areas of the SCA’s scope of operations.

7.3.1 Reflection

The consensus of those present at the focus group was that good modelling can add scientific credibility and process to planning and decision-making that might otherwise be perceived as subjective and ad hoc, however, virtually any model input can be perceived as being subjective. For example, although the subjectivity of the ECs used in this study were questioned, they were the result of an extensive literature review using a large suite of published research results for similar catchments in south-eastern Australia. It is unlikely that the selection process for inputs of ECs (or EMCs) could draw upon a more complete set of locally-relevant data or be less subjective than the values used in this study.

The focus group reinforced the idea garnered from workshops and other formal and informal meetings that participants have relatively little confidence in the quality or utility of results produced by any existing catchment-scale nutrient model. The lack of confidence may also be reflected in the large range of modelling approaches previously used by the SCA and by the SCA’s perceived need to develop new modelling tools in-house. The lack of confidence in models appears to be evidenced by the numerous and arguably excessive variety of catchment models that have been produced by researchers over the past 25 years. Researchers and model reviewers often comment on the “plethora” of water quality and quantity models and modelling
techniques that are available to natural resource managers and researchers (e.g., OTA, 1982; Simonovic and Bender, 1996; Letcher et al., 1999; Refsgaard and Henriksen, 2002; Chiew et al., 2002; Caminiti, 2004), and the new models that are continually being developed to take their place.

Although diffuse nutrient models are considered to be one of the most essential tools in the planning and decision-making toolkit, the focus group comments gave the impression that models may hold little weight when firm catchment management decisions need to be made. As a result, the link between modelling and the implementation of catchment management actions seemed unclear.

Three often competing themes were conveyed by the group; i) an aversion to the use of uncertain data, such as the “subjective” assignment of ECs, and uncertainty in the hydrological data; ii) in spite of what was said at the first workshop, a desire for increased sophistication, to maximise the number of scenarios that can be compared and accuracy and spatial resolution for targeting sites for rectification action; and iii) reluctant acceptance of the limitations in scientific knowledge and data and the opportunity to make use of simple tools until better knowledge and data are available. It is difficult to resolve which of these competing themes has the most bearing on the choice or appropriateness of any model for any given task. The precedence seems to be based on personal preference as much or more so than it is based on context.

Although uncertainty in model outputs was raised as a concern by some participants, issues relating to the measurement and management of uncertainty in models were barely mentioned. Uncertainty appeared to be seen by most participants as something that could be managed by using more or better data or better models, particularly models that used more input data and therefore accounted better for natural variation. However, for catchment-scale modelling of nutrients, both the literature (Young et al., 1996; Letcher et al., 1999) and SCA experience seems to suggest that more complex models offer little improvement over simpler models.

The desire for increased sophistication became particularly evident when the second model was introduced to the focus group. Early comments from about half of the participants were questions asking if the model accounted for other factors that they perceived to be important, including estimates of daily nutrient loads, soil
characteristics, streambank condition and other trapping mechanisms such as farm
dams, drainage density and proximity to streams, dairy farms and N. Later in the
discussion, participants also said that it was important that a model should account for
land management, land capability, rainfall events and economic considerations. The
utility of the model was also questioned because it is likely to be inaccurate at finer
scales.

Whilst this part of the discussion possibly constitutes an interminable ‘wish list’ rather
than a considered opinion about what should realistically be included in a model, the
discussion appeared to highlight a preference amongst some for all-inclusive models.

The combined expectation of the group is beyond the scope of what existing models
are able to achieve with available data and process knowledge. The second workshop
and literature review revealed considerable gaps in quantitative knowledge of
processes at scales relevant to SCA management. Any model that purports to include
most if not all of these features must rely substantially on subjective opinions and
intuition. Examples of fundamental areas of scientific uncertainty identified over the
course of this research included the following issues:

- Nutrient delivery ratios;
- The importance of ‘first flush’;
- Importance of ‘variable source areas’;
- The utility of ‘curve number’ as a method for determining runoff rates;
- The use loads or concentrations as the preferred method for determining
  relative source strengths;
- The reliability and utility of existing land-use and soil data; and
- Methods used to discretise land-use types.

Although confidence in all of the model outputs was low, and there was at least a
methodological preference for the enhanced EC model (i.e. it was seen as a “step in
the right direction” and was similar to a subsequent SCA modelling approach based
on L-THIA), the enhanced model generated the most discussion and critical debate.

It was obvious from the focus group, first workshop and other interactions with the
SCA that many SCA managers have good scientific knowledge of nutrient
mobilisation, transportation and delivery processes and seem to be well connected to
scientific research through present and former professional roles or associations. Although this suggests that there could be a healthy conduit for the transfer of scientific information to management action, there may be a divide within the organisation. One participant referred to modelling as an activity that “goes on in the background”, but seemed to have little relevance to day to day decision-making. Some concern was also expressed regarding the lack of connection between managers, modellers and “the Executive”.

The wide range of models that are available to researchers that wish to quantify nutrient fluxes is testament to the fact that there is no generally agreed upon methodology for analysing or predicting sources of diffuse nutrient pollution. This is compounded by the fact that it is difficult to advocate a single perspective that encompasses everything in a system in the face of the multiplicity of spatial and temporal scales, and therefore important processes. The consideration of multiple scales in time and space means that advocating a single perspective becomes increasingly difficult and less effective (Poch et al., 2004)

In isolation, any model provides only an opinion. Different models provide different opinions. Lack of consensus about the probable source and extent of land-use effects on nutrient mobilisation, transport and delivery have the potential to stymie regulatory and planning decisions. And, in any case, whether nutrient exports will be higher or lower from a given area of land in future depends on the spatial and temporal concatenation of an unpredictable and unknowable range of factors that include, but are not limited to, land-use, land management, vegetation cover, geomorphology, weather and climate.

In spite of uncertainty, models can be used to build consensus and confidence for evidence-based policy development and decision-making, but this may require a more participatory process to ensure that the necessary subjectivity of model inputs are accepted and acknowledged. A primary theoretical challenge for the SCA and similar organisations is to move beyond methodological individualism and generate theory that gives sufficient weight to competing conceptualisations and theories about how natural systems function and how that functioning should be represented in models.
8 REVIEW OF MODELLING PRACTICE GUIDELINES

At the first workshop, participants identified a desire for ‘standard operating procedures’ that could be used to guide model selection and use in the SCA. In response to the outcomes of the workshop, a review of existing modelling ‘guidelines’ was undertaken. This chapter presents the results of the review and discusses the findings in relation to the methodological lessons learned throughout the research. In chapter 9, new criteria for better modelling practice are developed and discussed.

The review and related analyses contained in chapters 8 and 9 informed the second research aim, i.e., to identify key criteria that catchment managers can apply to choose and use models more effectively; and met an objective of the research; to develop evaluation criteria or guidelines to assess the utility of catchment models in different decision-making and policy development situations.

8.1 Introduction

There has been recognition of the need for tools to support the selection of water quality models and guidelines for using models to support decision-making for many years (e.g., Grimsrud et al., 1976; Renard et al., 1982; Anderson and Woessner, 1992). Thirty years ago, Grimsrud et al. (1976) wrote a handbook entitled “Evaluation of water quality models: a management guide for planners”. The handbook catalogued the characteristics of most models available at the time and included guidance advice for selecting appropriate models and using models more effectively. The aspects of modelling that Grimsrud et al. (1976) used to evaluate models, including data requirements, cost, accuracy and ease of use, continue to be used today. With few exceptions, the practice of modelling for decision-making has not evolved substantially, and no standard methodology for selecting and using environmental models for decision support has been widely accepted by modellers or decision-makers.

With respect to surface water quality modelling, Refsgaard et al. (2005a) argue that quality assurance (QA) guidelines for the selection and use of models are not well developed. According to Refsgaard et al. (2005a), the key elements in the technical guides include:
• definition of the purpose of the modelling study;
• collection and processing of data;
• establishment of a conceptual model;
• selection of code or alternatively programming and verification of code;
• model set-up;
• establishment of performance criteria;
• model calibration;
• model validation;
• uncertainty assessments;
• simulation with model application for a specific purpose; and
• reporting.

Many guidelines reviewed later in this chapter also include suggestions for regulating the interaction between the manager and the modeller, including the following (Refsgaard et al., 2005a p. 1208):

• “Definition of the purpose of the modelling study, including translation of the end-users needs to preliminary performance criteria;
• Establishment of performance criteria. The accuracy of the model predictions has to be established via a trade off between the benefits of improving the accuracy in terms of less uncertainty on the management decisions and the costs of improving the accuracy through additional model studies and/or collection of additional field data; [and]
• Reviews with subsequent consultation between the modeller and the end-user at different phases of the modelling project”.

The successful use of models as decision-support tools depends on a number of factors that include the features of the model (Benbasat and Nault, 1990), the quality of the modelling process (Anderson and Woessner, 1992; Rykiel Jr, 1996), the culture and structure of the decision-making organisation (Russell and O’Grady, 1996; Pielke Jr, 2003) and the decision-space - or freedom that an organisation has to make decisions given the socio-political context in which the organisation operates (Jacobs, 2005). In order to evaluate the utility of a model in a given context, one should therefore evaluate models with reference to the way the model will be used and the context in which modelling is being undertaken. Whilst the mathematical techniques
used to simulate natural processes and measure model accuracy and uncertainty are of particular interest to scientists who seek to better understand natural systems, the immediate beneficial impact of such inquiry in catchment management, although widely assumed, is questionable (Loucks, 1992; Oreskes et al., 1994; Oreskes, 1998 Pielke Jr et al., 2000; Pielke Jr, 2003), because of the inherence of uncertainty and the range of competing values that are present for most decision-making problems.

The following sections of this chapter contain a review of some of the published guidance material, undertaken to better understand the extent to which other workers offer guidance for decision-makers that seek to negotiate and manage the abovementioned issues.

8.2 Review of Published Guidance Material

Several efforts have been undertaken to evaluate water quality models and/or offer guidance on the role and practice of water quality modelling in decision-making (Table 8-1). Some of these efforts have focused on direct comparison of different models or modelling approaches, whilst others focus on the evaluation of modelling as a ‘process’, and include focus on the wider decision-space. Other approaches look at both models and the modelling process.

8.2.1 US Guidelines

The earliest published guidance material was developed in the USA in the 1970s and 1980s and most work has been undertaken by the US EPA. Older guidelines typically have much in common with newer guidelines. For example, Grimsrud et al. (1976) catalogued the models that were available at the time and reviewed them against criteria such as applicability, data requirements and cost. Gass and Thompson (1980) explained a process for evaluating models against five criteria: 1) documentation; 2) theoretical, data and operational validity; 3) computer verification; 4) maintainability; and 5) usability. Each of these criteria form the basis of most of today’s model evaluation efforts.
Table 8-1. Some guidelines for the use of science or modelling to support policy development and decision-making (sorted by date).

<table>
<thead>
<tr>
<th>Title</th>
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<tbody>
<tr>
<td>(USA)</td>
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<tr>
<td>US GAO Guidelines for Model Evaluation (USA)</td>
<td>Gass and Thompson, 1980</td>
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<tr>
<td>(USA)</td>
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<tr>
<td>Currently Available Models (USA)</td>
<td>Renard et al., 1982</td>
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<tr>
<td>CAMASE Guidelines (Europe)</td>
<td>Plentinger and Penning de Vries, 1996</td>
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<tr>
<td>Evaluation Criteria for Water Quality Models (USA)</td>
<td>Parsons et al., 1998</td>
</tr>
<tr>
<td>Dutch Good Modelling Practice Handbook (Europe)</td>
<td>Van Waveren, et al., 1999</td>
</tr>
<tr>
<td>Review of Techniques to Estimate Catchment Exports (Australia)</td>
<td>Letcher et al., 1999</td>
</tr>
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<td>Bay-Delta Modeling Forum Protocols (USA)</td>
<td>BDMF, 2000</td>
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<tr>
<td>Checklist for Quality Assistance in Environmental Modelling (Europe)</td>
<td>Risbey et al, 2001</td>
</tr>
<tr>
<td>American Society of Agricultural Engineering. Southern Cooperative</td>
<td></td>
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<tr>
<td>Series Bulletin #598 (USA)</td>
<td></td>
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<tr>
<td>Model evaluation and performance (International)</td>
<td>Beck, 2002</td>
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<tr>
<td>Handbook in Groundwater Modelling (Europe)</td>
<td>Henriksen, 2001 [in Danish]; see Henriksen, 2002</td>
</tr>
<tr>
<td>Guide to Good Practice for the Development of Conceptual Models and</td>
<td>McMahon et al., 2001</td>
</tr>
<tr>
<td>the Selection and Application of Mathematical Models of Contaminant</td>
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<tr>
<td>Transport Processes in the Subsurface (USA)</td>
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<tr>
<td>Guidance on the Development, Evaluation, and Application of</td>
<td>Pascual et al., 2003</td>
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<tr>
<td>Regulatory Environmental Models (USA)</td>
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<tr>
<td>Handbook for Developing Watershed Plans to Restore and Protect Our</td>
<td>USEPA, 2005</td>
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<td>Waters (USA)</td>
<td></td>
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<tr>
<td>The Catchmod Cluster of Projects (Europe)</td>
<td>see Blind et al., 2005</td>
</tr>
<tr>
<td>CRCCH Series on Model Choice (Australia)</td>
<td>CRCH 2005a, 2005b</td>
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<tr>
<td>NOAA Connecting Science, Policy and Decision-making Handbook (USA)</td>
<td>Jacobs, 2005</td>
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<tr>
<td>Ten iterative steps in development and evaluation of environmental</td>
<td>Jakeman et al., 2006</td>
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<td>models (Australia)</td>
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Much of the effort to develop modelling guidance material in the US is a response to s. 303(d) of the *Clean Water Act (1972)*\(^7\) requiring that US States, Territories and Tribes set water quality standards for waterbodies. Total Maximum Daily Loads (TMDLs) are used to determine how much pollution from point sources and diffuse sources must be reduced to avoid exceeding the maximum pollutant load for a waterbody.

In 1991, the US EPA published *Guidance for Water Quality-based Decisions: The TMDL Process* (USEPA, 1991). The purpose of this document was to “help [US] State water quality program managers understand the application of total maximum daily loads ... to establish pollution control limits for waters not meeting water quality standards”. Appendices D and E contain some guidance material for the selection and use of computer-based mathematical models. In appendix D, a brief rationale for model selection is provided (p. 48). Firstly, the authors argue that it is important to characterise models based on temporal and spatial scale and processes and pollutants modelled. Model selection, it is argued, should be a four-step process that included each of the following steps (p. 49):

1. *Identifying models applicable to the situation;*
2. *Defining the appropriate level of analysis;*
3. *Incorporating practical constraints into the selection criteria;* [and]
4. *Selecting a specific model.*

In discussion of each step, only technical issues are considered. Interestingly, the guidance material implies that participation of the decision-maker is merely optional, a view which has become somewhat outdated (see for example Krauss *et al.*, 2005):

“...the selection of an appropriate model should be made by a water quality analyst [and] it is useful for program managers to be familiar with the decisions which must be made” (USEPA, 1991 p. 48).

In USEPA (1998), the EPA Scientific Advisory Panel described various surface water quality models and evaluated them against a range of criteria including model operation, model components, inputs and outputs (what is modelled and how); model

\(^7\) See: http://www.epa.gov/owow/tmdl/policy.html
application (examples of why the models have been used); model validation (using lists of publications that compare modelled vs. observed pollution); model availability and support; and strengths and weaknesses of the models that were evaluated. The criteria used at this time were not substantively different to those used by Grimsrud et al. (1976) and Thompson (1980).

In 2000, the Bay-Delta Modeling Forum (BDMF) published a generic set of principles and guidelines that it called the Bay-Delta Modeling Forum Protocols (BDMF, 2000). The authors contend that in order for models to see widespread and practical application, model developers need to address four specific questions (p. 1):

1. what is the original purpose of the model?
2. under what conditions will the model perform correctly?
3. what accuracy can be expected under the best conditions? and
4. what are the limitations?

As with most guidelines discussed in this chapter, the protocols focused on the need to define modelling objectives, manage relationships between modellers and other stakeholders and use appropriate technical methods for calibration and validation of model outputs. An aspect of this material that sets it apart from some other guidance material is a section containing guidelines specifically targeting the use of models in planning studies.

The authors contend that adherence to these protocols “…should enhance the credibility and effectiveness of modeling work and reduce the effort needed to respond to technical controversies”. Based on my experience in this research, I question the accuracy of the latter part of this statement, at least in the context of modelling for decision-support. As has been observed, the relative importance of the processes that drive nutrient transport at scales appropriate to management are unknown and debated. A modelling study focussed on catchment planning and decision-making or regulatory support is an unlikely tool for addressing a source of fundamental scientific debate. Such tools models should not be expected to advance scientific knowledge or minimise scientific debate and controversy.

In general, the protocols assume that a process whereby model use incorporates stakeholder involvement and/or peer review, controversy will become resolvable, or
at least more manageable. Whilst this can be the case for very simple scenarios (e.g., McNamara and Cornish, 2004), it may also serve to highlight and accentuate methodological differences of opinion amongst stakeholders and within the scientific community and subsequently increase the need to respond to controversy (see Chapter 7).

The ASAE Southern Cooperative Series Bulletin #398 (Parsons et al., 2001a) comprises almost 200 pages related to the selection and use of models. Although most of the report contains evaluations of individual models, some generic guidance material is provided in the opening chapters. The editors consider that the provision of up-to-date model evaluation resources for users of models is essential for meeting the needs of their “clientele”. The document includes:

“an overall description of model evaluation efforts, the model evaluation criteria used for each model in this document, a matrix describing many of the different models and their general characteristics, individual/extensive model evaluations for at least 14 widely used models, and at least three unique model evaluation/use papers” (abstract).

In “Introduction and Scope”, Thomas et al. (2001b) argue that there is an inherent unwillingness on the part of model developers or users to “relearn new technologies or models” (p. 1), and modellers will therefore tend to create new models based on older models. To a large extent, the decision to use a simple EC model in this page was based on successful use of this simple technology in previous applications. Support for this approach tended to come from those that had used the methodology previously. Further, the authors argue that “model evaluations need to be undertaken periodically, to encompass new developments and components which were not available in previous versions” (p. 4).

In the chapter entitled “Summary of Model Characteristics”, Thomas et al. (2001a) imply that it is difficult to identify model characteristics based on literature searches and discussions with model experts and users. The authors created several matrices that are intended to help identify a number of important model attributes, including:

- model characteristics: temporal and spatial scale, computational time step, and target audience (research or management);
• validation: an approximate guide to the physiographic regions in the USA in which the model has been tested;
• documentation: the availability of supporting documentation (e.g., manuals);
• user interface: the presence of ‘input helpers’ and GIS integration;
• processes and pollutants simulated: describes if the model simulates nutrients and/or pesticides, if the model simulates surface and/or subsurface flow, precipitation and other basic natural processes; and
• model contact information: a list of contact details for each of the models that were evaluated.

In the chapter entitled “Evaluation Criteria for Water Quality Models”, Parsons et al. (2001b) discuss “a set of evaluation criteria [intended] primarily to assist users in selecting a water quality model for their application”. The criteria discussed here are essentially similar to those described in the previous chapter and are mostly of a technical nature. The appropriate target audience for the model and the expertise required and opportunity for training are also mentioned as requirements for consideration, but are not discussed in any detail.

In 2002, the USEPA published a comprehensive document entitled “Guidance for Quality Assurance Project Plans for Modeling” (USEPA, 2002). The report is intended as a tool for modelers, project managers, planners and QA personnel who wish to document the type and quality of data and information needed for making environmental decisions, and contains advice and recommendations on how users of predictive models can develop a QA Project Plan for projects that involve model development or application.

The guidance document largely focuses on technical aspects of model use, but is more thorough than most guidelines and represents an improvement over earlier US EPA guidance material. The authors highlight the need to use models that are “scientifically sound, robust, and defensible” (p. 2), and argue that confidence in model outputs can result if the modelling activity is thoroughly planned. Technical issues and data needs are discussed in relative detail. However, the document does not discuss what to do when only uncertain input data are available. In fact, the authors imply that “input data and parameters that are accurate and appropriate for
the problem” are essential to ensure that model outputs are sound, robust and defensible (p. 2). However, the authors concede that the level of QA required for non-regulatory priorities and “ballpark” estimates are lower than for those used to set regulatory requirements.

An extensive model selection/development and application process is described in this document. Part of this process includes a ‘needs analysis stage’ in which the need for modelling is questioned at the outset, a rare feature of most guidelines.

Another step-wise improvement in guidance material is the Handbook for Developing Watershed Plans to Restore and Protect Our Waters (USEPA, 2005). Chapter 8 of this publication contains a relatively comprehensive and ‘plain-English’ discussion on techniques for estimating and modelling pollution loads. The document as a whole covers most aspects of catchment planning and is aimed at US government and non-government agencies and organisations that develop watershed (i.e. catchment) plans.

Not directly related to modelling and water quality, but relevant nonetheless is a handbook, entitled, Connecting Science, Policy, and Decision-making: A Handbook for Researchers and Science Agencies (Jacobs, 2005). The handbook was published by The National Oceanic and Atmospheric Administration (NOAA) Office of Global Programs, an agency of the US Department of Commerce. The book can be used by researchers and government agencies to better connect science with policy and decision-making. The content focuses on many of the so-called softer issues of using science to inform management, including the importance of: understanding the context (“Decision-space”); the actors involved in an issue, communication and collaboration between stakeholders; the incentives and motives that drive participants involved; and other issues that affect the successful use of science in policy and management. The guidance contained in this handbook is no less relevant to the decision-makers that use models than it is to decision-makers that use any other output of science and research.

8.2.2 European Guidelines

European workers generally approach the problem of model use more systemically than similar guidelines in the US. Some of the most mature guidance material and
research originates from The Netherlands and much of work is focused on providing guidance and support for government agencies responding to Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy, more commonly known as the Water Framework Directive (WFD)\(^8\).

The Good Modelling Practice (GMP) Handbook (van Waveren et al., 1999) was developed by Dutch environmental authorities as part of their Generic Framework Water (GFW) program. The aim of the handbook was to improve the effectiveness of modelling by addressing misuse of models that resulted from problems such as careless handling of input data, insufficient calibration and validation, irreproducibility of modelling studies and using models outside of their intended scope (Scholten et al, 2000).

The handbook was designed as a technical, QA tool to guide model users through the major steps in the process of using models for decision support. A seven-step modelling procedure is advocated in the handbook. The seven steps are:

1. Starting a model journal;
2. Setting up [and defining] the modelling project;
3. Building the model;
4. Analysing the model;
5. Using the model;
6. Interpretation of model results; and
7. Reporting the modelling outcomes and ‘filing’ so that the study is reproducible

Much of the content is common to many modelling guidelines. However, two areas where this guideline goes beyond most others are in steps two and six. For step two, the handbook outlines an activity in which the objective of modelling is described:

“The objective must be described in terms of:

- the domain and the problem area;
- the reason for solution of the problem by means of a model;
- the questions to be answered by the model;[and]

\(^8\) See: http://ec.europa.eu/environment/water/water-framework/index_en.html
• *the scenarios to be calculated*” (p. 2-2).

The second bullet point (above) is unusual and is not described in most other guidance materials, but is an important first step in the analysis of a decision problem for which modelling is proposed (Pielke Jr *et al.*, 2000). For step six, the guidelines are also notable because they call on users to analyse the outcomes of modelling for the research question:

“Unfortunately, the [modelling] procedure followed will often produce an unsatisfactory solution, a compromise between feasibility and affordability. This may have various consequences:

• *the response to the modelling project is negative (particularly if the modeller keeps too many options open);*

• *the modelling project exposes gaps in the domain knowledge, thus generating new research questions;*

• *the modelling project requires more field observations/measurements;*

• *a follow-up modelling project has to be initiated to thoroughly investigate all matters involved;*[and]

• *the client is dissatisfied, or, quite the contrary, satisfied”* (p. 6-2).

Although there is no substantive guidance on how these should be achieved, most guidelines are virtually silent in relation to such matters.

The guidelines also include a discussion of pitfalls and sensitivities associated with the use of different types of models used by water managers to help users avoid adverse outcomes. As with USEPA (2002) (above) and RISBey *et al.* (2001) (below), the authors have included a checklist.

The Netherlands Environmental Assessment Agency (MNP), an agency of the Netherlands National Institute for Public Health and the Environment (RIVM), developed a suite of tools called the *RIVM/MNP Guidance for Uncertainty Assessment and Communication* (Janssen *et al.*, 2004). The related *Tool Catalogue for Uncertainty Assessment* (van der Sluijs *et al.*, 2004) provides an overview of a selection of quantitative and qualitative methods relevant to the identification and assessment of uncertainty associated with environmental problems and modelling. The tools mentioned in the Catalogue are listed below, with a brief description:
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- Sensitivity Analysis: The use of a quantitative or qualitative method to determine how the variation in the output of a model can be apportioned to different sources of variation.

- Error Propagation Equations: A very simple method for assessing how quantified, normally distributed uncertainties in model input parameters propagate in model calculations and affect model outputs.

- Monte Carlo Analysis: A method for statistically assessing how quantified uncertainties in model input parameters propagate in model calculations using random or pseudo-random samples selected from virtually any probability distribution.

- Scenario Analysis: The consideration of possible alternative outcomes (scenarios). A user may gain insights into the potential impact of uncertainty on decisions by adjusting model parameters or algorithms to reflect different assumptions about uncertain inputs or unknown future events.

- Expert Elicitation: A structured process that can be used to obtain subjective probabilistic distributions of risk or uncertainty for model parameters when scientific data is absent, limited or conflicting.

- NUSAP (Numeral Unit Spread Assessment Pedigree): A notational system devised by Funtowicz and Ravetz (1993) that provides for an analysis and diagnosis of uncertainty in science for policy. The approach complements conventional statistical approaches to assessments of uncertainty (inexactness) by adding methodological (unreliability) and epistemological (ignorance) dimensions, and examining uncertainty in terms of spread (the amount of inexactness) and strength (the quality of the underlying knowledge base) (see also van der Sluijs et al., 2004).

- PRIMA (Pluralistic jFramework of Integrated uncertainty Management and risk Analysis): PRIMA is described as “a meta approach (organising framework) to structure the process of uncertainty management” (p. 47). The approach allows the user to attain insight into the impact of uncertainty in decision-making by systematically accommodating the views of people or groups with different perspectives on the issue being modelled. This can be achieved, for example, by having multiple perspective-based model runs
whereby a model’s controversial algorithms or input parameters are adjusted to accommodate differing viewpoints.

- **Checklist for Model Quality Assistance**: [treated separately. See below]
- **A Method for the Critical Review of Assumptions in Models**: A methodological framework, related to NUSAP, for the critical appraisal of value-laden choices and assumptions. The method is designed to be used for environmental assessments where the outputs from multiple models are used to make a final assessment (i.e. a “calculation chain”).

The “Checklist for Quality Assistance in Environmental Modelling” (Risbey et al., 2001) seeks to help users of complex models ensure that they are following good modelling practice, and ensure that model results are sufficiently useful for their intended purpose. The Checklist focuses on technical aspects of models and the modelling process, and pays particular attention to the impact of different kinds of uncertainty. The Checklist is designed for users of “relatively complex models where validation of model outputs is not possible or is at best partial” (p. 4). The authors state that the checklist is only useful if quality is at stake, and included in the checklist are questions aimed at helping the user determine whether or not the checklist can be of assistance.

The types of questions raised in this checklist are relevant to the problems identified over the course of this research. As well as asking questions that are relevant to the scientific community and analysts, several questions are asked that are directly relevant to management and the use of models by scientists, managers and other stakeholders, including the public. For example, the checklist (p. 5) asks:

- “Is the model well accepted for use on the desired application by: peers; users; and/or stakeholders?
- Is the model application salient to stakeholders and the public agenda?
- Is the legitimacy of the model community an issue among stakeholders?
- Is public accountability of science important to the policy process?”

The checklist, which is more of a questionnaire, asks users to consider technical and non-technical issues, and many of the questions ask for qualitative assessments as answers. For example, users are asked: “For this problem, what are the key value issues?” (p. 8). In this example, users list the key value issues and categorise them
according to how central they are to the given problem: either “peripheral”, “relevant” or “central”. Most questions require qualitative answers similar to these. Users are also asked to think about other issues relevant to the decision-making, including facts that are in dispute, groups with vested interests and the role that models should play in setting policy on the issue.

Part 4.4 of the checklist (p. 12) asks questions relevant to “Robustness”. One such question is “How vulnerable is the model to “hack and crack”? (i.e. Is it possible to produce an arbitrarily chosen output by tweaking the system?)”. In order to answer the question, users must respond to a subsequent sub-question concerning model sensitivity: “If you were asked to change the main result of the model for a factor of 2, how much would you need to ‘tweak’ the most sensitive parameter values?”. Answers which the user may choose are “barely – well inside range of expert opinion”; “moderately – moving to tails of expert distributions”; or “radically – outside expert distributions”.

Users of the checklist are also asked to question issues such as the relevance of the model to the policy process, the investments that modellers and other stakeholders might have in model results and evidence of modellers being influenced by funders or other stakeholders. They also ask a rhetorical question (p. 22): “Who will quality control the quality controllers?”

CatchMod Cluster Projects

A variety of inter-related projects have been funded by the European Commission as part of the EU’s 5th Framework Programme to support the Union’s Water Framework Directive (WFD) legislation. Of these, the Catchment Modelling Cluster (CatchMod) projects deal with issues including international co-operation and collaborative planning, models and other Information and Communication Technology tools (Table 8-2).

Aspects of the projects relevant to modelling for decision support are described below:
Table 8-2. CatchMod cluster projects.

<table>
<thead>
<tr>
<th>Project Acronym and Title</th>
<th>URL</th>
<th>Focus</th>
</tr>
</thead>
<tbody>
<tr>
<td>Harmoni-CA</td>
<td><a href="http://www.harmoni-ca.info">www.harmoni-ca.info</a></td>
<td>Synthesis and Co-ordination</td>
</tr>
<tr>
<td>BMW</td>
<td><a href="http://www.environment.fi/syke/bmw">www.environment.fi/syke/bmw</a></td>
<td>Model Selection</td>
</tr>
<tr>
<td>Euroharp</td>
<td><a href="http://www.euroharp.org">www.euroharp.org</a></td>
<td></td>
</tr>
<tr>
<td>Cline</td>
<td><a href="http://www.water.hut.fi/clime">www.water.hut.fi/clime</a></td>
<td></td>
</tr>
<tr>
<td>TransCat</td>
<td><a href="http://www.harmoniqua.org">www.harmoniqua.org</a></td>
<td>DSS Development</td>
</tr>
<tr>
<td>TempQSim</td>
<td><a href="http://www.tempqsim.net">www.tempqsim.net</a></td>
<td></td>
</tr>
<tr>
<td>Tisza River</td>
<td><a href="http://www.tiszariver.com">www.tiszariver.com</a></td>
<td></td>
</tr>
<tr>
<td>HarmoniCoP</td>
<td><a href="http://www.harmonicop.info">www.harmonicop.info</a></td>
<td>Stakeholder Participation</td>
</tr>
<tr>
<td>HarmoniQuA</td>
<td>harmoniqua.wau.nl</td>
<td>Quality Assurance</td>
</tr>
<tr>
<td>HarmoniRiB</td>
<td><a href="http://www.harmonirib.com">www.harmonirib.com</a></td>
<td>Data-model and Uncertainty</td>
</tr>
<tr>
<td>HarmonIT</td>
<td><a href="http://www.harmonit.com">www.harmonit.com</a></td>
<td>Open modelling interfaces and architectures</td>
</tr>
</tbody>
</table>
Harmoni-CA

Harmoni-CA is a key component of CatchMod that is intended to:

“create a forum for unambiguous communication, information exchange and harmonisation of the use and development of ICT-tools relevant to integrated river basin management, and the implementation of the WFD” (Refsgaard et al., 2005b).

The project is oriented towards the coordination of science support to implementation of the WFD through the development of guidance and methodologies that can be applied by stakeholders involved in catchment planning, policy development, management and modelling.

Two significant outputs of Harmoni-CA so far have been “Guidance Document on Uncertainty Analysis” (Refsgaard et al., 2005b) and “Model-supported Participatory Planning for Integrated River Basin Management” (Becker, 2005).

HarmoniQuA Toolbox

HarmoniQuA aims to provide guidance for multi-disciplinary teams working in model-based integrated water management. A software tool named MoST (Modelling Support Tool) and a “Knowledge Base” have been developed under the HarmoniQuA project. According to (Middlemis, 2004), the current version of the guidance content of MoST is largely based on the (Dutch) Good Modelling Practice Handbook (van Waveren et al., 1999) (above). Additional content has reportedly been drawn from selected information from the Bay-Delta Modeling Forum protocol (BDMF, 2000), and from a Murray Darling Basin Corporation guide (Middlemis, 2001).

MoST decomposes the modelling process into five major steps, which are primarily focussed on common technical steps in model use:

1. Model Study Plan;
2. Data and Conceptualisation;
3. Model Set-up;
4. Calibration and Validation; and
5. Simulation and Evaluation
BMW

The principle objective of the BMW project is the establishment of a set of benchmark criteria to assess appropriateness of models for use in implementation of the WFD (see Dilks et al., 2003). A hierarchical series of benchmark criteria, to be applied on a case specific basis, have been developed for selection of diffuse pollution models. The benchmarking process, as developed within BMW, consists of two main stages, evaluation of model codes in relation to qualitative benchmark criteria and assessment of model performance relative to other models.

Additionally, two layers of benchmarking criteria have been developed: Generic criteria can be applied to all models; and domain specific criteria for application within individual modelling domain, including diffuse pollution modelling. For diffuse pollution modelling (p. 207), criteria include:

- **model suitability for use**;
- **data availability**;
- **modelling objectives and requirements of the WFD**;
- **spatial and temporal scale resolution**;
- **transformation and transportation processes**;
- **data processing**; [and]
- **model output and model integration**.

For each criterion, a series of questions are asked of the modeller. The questions are answered by selecting the most appropriate statement from three options. This should be done with the modelling objective in mind. The statements classify the model application as good, ok or poor with regard to a particular criterion. Selection of certain statements results in the model code being *not recommended* for use. The number of criteria falling into each class provides the assessor with an indication of model suitability for use.
Euroharp

Euroharp has two objectives: 1) to provide European environmental policy-makers with a thorough scientific evaluation of nine “quantification tools”\(^9\) (models) and their ability to estimate diffuse N and P losses to fresh surface water systems and coastal waters to facilitate the implementation of the WFD; and 2) develop a decision support system (“toolbox”) for the identification of benchmarking methodologies with respect to both costs and benefits, for the quantification of diffuse nutrient losses under different environmental conditions across Europe (Arheimer and Olsson, 2003).

With the exception of SWAT (Arnold et al., 1998), the models in the toolbox are relatively unknown in Australia.

8.2.3 Australian Guidelines

As in Europe and the US, Australia is an active developer of many water quality models (Irvine et al., 2002). However, Australian guidelines are typically less well-developed than their European and US counterparts. Whereas in the US and Europe, guideline development is often focussed on meeting the requirements of TMDL limits (USA) and the WFD (Europe), in Australia, guidelines typically support regional catchment management, ‘State of the Environment Reporting’ and National Pollutant Inventory (NPI)\(^10\) reporting.

In 1999, the NSW EPA published a *Review of Techniques to Estimate Catchment Exports*. The report reviewed and compared the relative merits of a comprehensive range of models and methods for estimating nutrient exports from diffuse sources (Letcher et al., 1999). The review includes information that can assist intending model users to assess the relative merits of each model or method and was primarily designed to help readers identify the most suitable approaches for assessments of aggregated catchment nutrient emissions to water for the National Pollutant Inventory (NPI).

\(^9\) See: http://euroharp.org/pd/pd/index.htm#5

In relation to model ‘implementation’, the review discusses issues of goodness of fit (“calibration acceptance criteria”), model complexity, ease of use, hardware requirements, model validation, the importance of identifying and modelling key hydrological and erosional processes, spatial and temporal variability, model structure, and the objectives of the model user. The authors also discuss the accuracy of modelled estimates or predictions when compared with observational data (p. 69) and note that it may be difficult or impossible to conclude which models are most accurate because of a lack of observed water quality data. The authors also discuss the different needs of researchers versus managers. In this respect, a conclusion of the authors was that for catchment management planning in Australia, where model input data is usually sparse and models need to be flexible, simple empirical models are usually most useful.

In 2002, the Cooperative Research Centre for Catchment Hydrology (CRCCH) began to develop a Toolkit Assistant (Podger, 2002) that was to help intending model users select appropriate models for a given task from a CRCCH software repository called the “Modelling Toolkit”\(^\text{11}\). However, although the range of models in the Toolkit represents only a small subset of the models available to users from all sources, development of the Toolkit Assistant was halted when researchers found that it was prohibitively difficult to attempt to prescribe which toolkit model to use in any given situation. Instead, the CRCCH focussed resources towards the publication of a series of booklets on model choice (CRCCH, 2005a, 2005b) designed to assist model users to better understand issues in catchment modelling and model selection (Argent and Podger, 2005).

The intent of the series is to highlight the major issues that should be considered when selecting appropriate models for particular applications. The first two booklets have been published and both focus on data-related scientific and technical issues that limit the effective use of models, in particular, the booklets examine links between model complexity, data limitations, and predictive accuracy. The importance of issues relating to uncertainty and temporal and spatial scale are also discussed. The first booklet in the series (CRCCH, 2005a) presents issues that apply to many types of

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\(^{11}\) See: http://www.toolkit.net.au/
catchment models, while the second booklet (CRCCH 2005b) focuses on models that estimate or predict nutrient fluxes, and includes a comparative evaluation of relevant CRCCH Toolkit models. In general, the series emphasises a need for a ‘horses for courses’ approach to model selection and use.

There is limited discussion of cultural issues that inhibit effective model use. In the first booklet, it is stated that a basic consideration for a potential model user is to understand the objectives of the modelling exercise, so that both “clients” and “consultants” (p. 17) have a reasonable understanding of a potential model and its limitations and shared expectations about what a model can achieve.

Key questions that the author considers important include the following (p 15):

- *Is the broader context of the modelling clear?*
- *How are the results of the modelling going to be used?*
- *What specific output is needed?*
- *Where will the model be applied?*
- *What are the proposed actions that need to be represented?*
- *Who will be interpreting the results and what decisions will they be making?*

The author notes that these types of considerations are rarely explicitly or adequately addressed and as a result, modellers and/or model users “are often polarised in their approach to output from models – either blind-faith or total disbelief” (p. 17). The need for timely and salient results, especially when models are used for practical applications is also mentioned, emphasising the need for modelling to meet the client’s practical needs.

### 8.3 Discussion and Reflection

Although many guidelines have been developed over the last 30 years, Refsgaard and Henriksen (2002) note that there has been relatively little advancement in the ideas and methods used to assure the quality of the modelling process. They argue that most of the model guidelines described in this chapter have been derived from a modelling protocol advocated 15 years ago by Anderson and Woessner (1992) (Figure 8-1 and Figure 8-2).
Figure 8-1. ‘A modelling protocol’: The steps in model use identified by Anderson and Woessner (1992 p. 168) that now form the basis of many guidelines to good modelling practice (see also Figure 8-2).
Figure 8-2. Variations on a theme: Most modelling protocols are similar in form and function.
However, whilst the steps in the flow chart by Anderson and Woessner (1992) have proved to be a popular foundation prescribing a basis for good modelling practice, Dahl and Wilson (2001) argue that this method of conceptualising modelling problems is of limited use because it does not take into account the complexity involved in setting appropriate modelling objectives. They consider that it is necessary to consider “an outer level of iteration encircling the setting of modelling goals” (p. 7), that is, to consider modelling more broadly and with greater reference to the decision problem. Similarly, from my experience, I argue that it is necessary to consider more acutely how models and model outputs help managers negotiate their ‘decision-space’ (Jacobs, 2005). In terms of assessing whether or not a model is fit for a given purpose, model evaluation is essentially a qualitative process that seeks to determine if a model is “more useful” or “less useful” for a given problem (Risbey et al., 2001).

The overwhelming issue that recurs in every guideline is that of uncertainty. Some workers (e.g., van Asselt and Rotmans, 2002) recognise that uncertainty in data and information used in decision-making cannot be adequately addressed with conventional methods and tools (e.g., sensitivity analysis and Monte Carlo analysis). According to van Asselt and Rotmans (2002):

“[this] especially holds for uncertainty in model structure and uncertainty due to behavioural and societal variability, value diversity, technological surprise, ignorance and indeterminacy.” (p. 82)

They further argue:

“Uncertainty is usually treated as a marginal issue, as an additional physical variable, as a mathematical artefact” (p. 82).

Tools such as NUSAP (Funtowicz and Ravetz, 1993) and the Checklist for Model Quality Assistance ” (Risbey et al., 2001) seem to offer new ways of thinking about uncertainty that, especially in the sphere of policy development and management can complement and sometimes replace conventional statistical methods.

Uncertainty is especially problematic for decision-makers because, as van Asselt (2004) points out, “uncertainty legitimates different perspectives on policy issues” (p. 47), and can therefore encourage disagreement and debate. Most guides reviewed in
this chapter support the view of Middlemis (2001) and imply that decision-makers lack confidence in models because of a “…lack of consistency in approaches, communication and understanding between modellers and managers” (page S2). However, based on the experience of this research, these issues are problems within and not just between the science disciplines and management disciplines. Lack of consistency in particular is at least partly due to both imperfect knowledge and simply different ways of seeing the world.

Likewise, two reasons proposed by BDMF (2000) for decision-maker’s lack of confidence in models are inadequate understanding of modelling principles and inconsistencies in the way models are developed. Although these issues are important, the experience of this research was that agency staff lacked confidence in models precisely because they did understand modelling principles, and that, amongst both the decision-makers and the scientists, there were different views about what approach should be taken to model nutrient fluxes. Settling on an approach to the modelling process does not resolve this fundamental problem.

Refsgaard and Henriksen (2004) argue that the poor quality of modelling work is the result of two key factors: 1) inadequate use of guidelines and inadequate role play between managers and analysts; and 2) a lack of data and methodology in hydrological science. The authors point out that issues relating to the first factor are almost exclusively addressed by modelling guides (such as those discussed in this chapter) and that the second factor is almost exclusively addressed in the scientific literature. In order to improve the modelling process, the authors argue that it is crucial that the two key issues be combined.

In discussing the problem of determining what regulatory interventions are justifiable when scientific certainty cannot be provided, Krauss et al. (2005) discuss the importance of collective judgments of quality based on deliberation about the characteristics of information at hand. The authors argue that model quality assessment should follow a relatively simple rationale in which a model should be chosen that “is suited to the purpose and yet bears some reasonable resemblance to the real phenomena” (p. 1). The authors point out that “purpose” is not a technical issue, and refers to “the policy and societal contexts in which the assessment results are used” (p. 6). In this approach to modelling, Krauss et al. (2005) stress that non-
scientists and non-modellers must be involved in the assessment process. The “challenging” question of “how can decision makers use uncertainty information and quality assessments in the policy context?” (p. 7) is left for the reader to ponder at the end of the paper.

My experience in developing and running a simple model with potential to be used as a regulatory support tool and tool to support the allocation and prioritisation of catchment management resources seemed to show that exposing uncertainty might in fact paralyse decision-making, regardless of how well it is communicated to the decision-maker, and that, in any case, different problems may negatively impact upon the utility of models as decision support tools. In general, there was little interest shown in the potential application of methods for estimating the impact of uncertainty on model outputs that, in any case, are difficult or impossible to adequately determine due to data deficiencies. The fact that there was uncertainty at all seemed to be a very fundamental problem (see Chapter 7).

Many of the managers in the SCA with responsibility for catchment planning and management, and the regulation of new developments under NorBE, have a background as scientists and are familiar with water quality models and, albeit to a lesser extent, tools available to scientists to quantify uncertainty. As scientists, these managers are well-equipped to identify the substantial ways in which a very complex biophysical system needs to be simplified to allow for mathematical estimation of chemical loads in streams across a very large catchment. Disagreements amongst the manager groups and the science groups at the workshops and the focus group reflect normal scientific debate and the inadequate quantitative knowledge of the processes that affect nutrient mobilisation and transport.

Differences of opinion with respect to modelling methodology, which were often based on uncertainty about the processes that affect nutrient fluxes within catchments, produced debate and were not conducive to building consensus that afforded model users with confidence in model outputs. One response to this might be to advocate the importance of data collection that is linked to the needs of the modelling community. However, my experience was that there were very limited funds available for long-term routine monitoring to support model development. Instead, funds for monitoring were essentially short term and project-based (an institutional
barrier recognised by Bosch et al., 2003). In any case, the logistical problems associated with monitoring at scales appropriate to management mean that such an exercise would be ambitious and would not correspond with either the need to act on a serious environmental problem or the timeframes allocated to managing the problems as laid out in relevant policy documents, such as the Drinking Water Catchments Regional Environmental Plan No 1 (2006-289 NSW).

Beven (2001) (in Perrin et al., 2002) proposed the following steps that can be used for model ‘rejection’:

1. Prepare a list of models under consideration;
2. Prepare a list of variables predicted by each model and those required, including considerations on spatial and temporal scales;
3. Prepare a list of assumptions made by the model;
4. Make a list of inputs required;
5. Determine whether you have any models left. If not go back to 2 and relax criteria.

The implication of these steps is that modelling is essential. There is no opportunity in any of the steps for the user to question or re-evaluate the need for modelling or for the user to modify their expectations of models, which may be unrealistic. Prospective model users need to think deeply about the relevance and impact of models to specific decision problems given the limitations of data and knowledge, and most existing guidelines, with the notable exception of some of the innovative guidance material summarised in the Tool Catalogue for Uncertainty Assessment (van der Sluijs et al., 2004), do not encourage this. Most guidelines intended to help individuals or agencies use models as decision-support tools generally take a narrow view of how scientific knowledge is transferred into policy and decision-making, focussing almost entirely on science-based methods (e.g., Figure 8-3). The implication seems to be that if a scientifically rigorous process is used in modelling, science and modelling will become more usable in policy and decision-making.
However, decisions and decision-makers can be affected more by exogenous factors, including influences that are uninteresting to biophysical scientists, and these need to be borne in mind when considering the quality of models as decision-making tools. For example, Land and Water Australia (Schofield, 2005) found that the successful use of the CMSS model was affected by changing political priorities and institutional restructuring. The author states:

“...the impact of CMSS [modelling] was reduced partly due to a general shift in the topical issues regarding water quality from algal blooms in the 1990s to environmental flows and salinity and then to water sharing and native vegetation and land management that affect water quality. This changing situation, together with catchment management structural changes, meant that the implementation of plans developed from the use of CMSS were significantly delayed.” (p. 3)
Virtually all guidelines begin with agreeing on the problem definition (or purpose of the modelling study), but different scientists can have qualitatively different mental models and disagree about the “primacy of causal factors” of the issue being modelled. Consensus amongst scientists can also be difficult because of different modes of cognition. Some decision-makers are more analytical and others are more intuitive (Mumpower and Stewart, 1996 p. 200). In order to resolve disagreements that can be traced back to different modes of cognition, Mumpower and Stewart (1996) believe that experts would have to agree to use a common mode of thought - analytical thinkers would need to think intuitively, or vice versa. According to Mumpower and Stewart (1996), when a group of individual’s organising principles are different, “methods for diagnosing and treating disagreement are poorly understood” (p. 191).

8.3.1 Form and Function of a Better Guide

Many workers express the belief that model selection for decision-support is a matter of “horses for courses” (e.g., Dent, 2000; Grayson and Blöschl, 2001a; CRCCH, 2005a, 2005b; Van Dijk and Podger, 2005), and that view was expressed by some participants at the second workshop (Chapter 5.2.7) and in passing conversation numerous other times. In this metaphor, the horse is the model, and the course is typically the issue that needs to be managed (Dent, 2000). Based on this research, what appears to be missing from this cliché is the organisational context. The metaphor might therefore be expanded to provide a more complete systemic representation of modelling for decision-making by including the organisation as the jockey and the decision-space as the state of the track.

At the commencement of this project, it was assumed that dialogue between modeller and manager was important for choosing and using the right model for the decision-making problem. It was also proposed that the success of a modelling project was likely to be influenced by the sophistication of the model, which was likely to have a bearing on the confidence that managers placed on model results. This theme is emphasised in much of the scientific literature discussing the role of science in environmental management and discussion of the issue pervades many of the guidelines to the use of models in decision support. However, in this research, whilst
participation and uncertainty and model complexity are seen to be important issues, they are issues that cannot be resolved without addressing other issues relating to how individuals and organisations conceptualise and contextualise problems and prospective solutions (see Mumpower and Stewart, 1996).

At several meetings, SCA staff expressed a desire for a SOP for the use of models as decision-support tools. Standard operating procedures are written instructions for tasks that outline the preferred, safest, recommended or mandated method for undertaking a specified task and can be used for Quality Assurance and to ensure that best practice is followed. According to Refsgaard et al. (2005a), poor modelling outcomes are often the result of “inadequate use of guidelines and quality assurance procedures, and improper interaction between the manager (client) and modeller (consultant)” (p. 1202).

The collaborative research projects, of which this research was a part, should have overcome some of these problems, by moving away from the traditional funder/provider model of interacting with the consultant researcher to a collaborative framework, however, the authenticity of the collaborative arrangement was questionable. For example, as a collaborative researcher stated at a workshop in October 2005 (not formally part of this research), “The nature of the collaboration has never been clear to me”. Another researcher stated in a frustrated tone, “Knowledge of SCA’s goals and expectations, strategic plans, guidelines, targets, capabilities for impacting on catchment management is important – What are they and what can SCA do?” Given that these comments highlight communication problems that probably impacted on the quality of research outcomes, the issue of analyst/manager communication is pertinent, however, based on experience in this research, this may not be the main challenge facing organisations that wish to use models to support decision-making.

The utility of a model in management and decision-making depends on a range of factors, and accuracy may not always be important. In the context of decision-making, policy development and regulatory support, the evaluation criteria that are important are those that relate to a given model’s fitness for purpose. The evaluation criteria also relate to the quality and utility of the modelling process, since effective use of a model depends not only upon the selection of an appropriate tool, but an
appropriate method for using the model to support decision-making. What appeared to be missing in the SCA approach and in much of the literature on modelling for decision-making was due consideration of organisational and socio-political issues.

Whilst some workers look to technology to resolve the problems faced by individuals or institutions that wish to use models to support decision-making, for example through new software or changes to model structures (e.g., Argent and Grayson, 2001; Kassahun et al., 2004), the experience of the author in this research is that considerable advances can be made by re-conceptualising environmental decision problems and thinking more broadly about how and why models are used.

A concept developed by computer scientists in the 1990s that holds relevance to the problem of environmental modelling for decision-making is that of ‘patterns’ (Gamma et al., 1994) and ‘anti-patterns’ (Brown et al., 1998). These terms have been used to categorise good and bad ‘solutions’ to software engineering problems. Essentially, ‘patterns’ are simple, succinct, repeatable solutions to common software engineering problems that occur in particular contexts, whilst ‘anti-patterns’ are commonly reinvented problem solutions that are ineffective or result in perverse outcomes and software design failures. A similar conceptualisation may be useful when developing criteria that are intended to ensure efficiency and efficacy in modelling for decision-making. For example, a term that Brown et al. (1998) labelled as an anti-pattern that already receives some usage in environmental modelling and management (e.g., Barry, 1999 p.11; Kaufman et al., 2003; Newham et al., 2004 p.18) is that of ‘analysis paralysis’. ‘Analysis paralysis’ refers to situations where further decision analysis occurs even though the cost of further analysis is higher than the benefits that can be gained from further analysis. I speculate that the continual search for new and better modelling tools, in the SCA (Chapter 1.1.4) and in the field of environmental and catchment management in general (as evidenced by the plethora of alternative models and guidance materials), may constitute evidence that this is occurring.

Another anti-pattern used by Brown et al. (1998) is ‘golden hammer’, which in effect is an antonym of the metaphor ‘horses for courses’. A ‘golden hammer’ is a tool or idea that is widely lauded for solving one or more problems even if the tool or idea is inappropriate for solving the problem. For many decision-makers, in the absence of
obvious alternatives, and in the presence of abundant models and scientists pushing modelling tools, models themselves may have become a ‘golden hammer’. As Maslow (1966) argued in his book, *The Psychology of Science*, “…it is tempting, if the only tool you have is a hammer, to treat everything as if it were a nail”. The influential systems practitioner, Sir Geoffrey Vickers foresaw this phenomenon in relation to computer technology in the late 1970s, warning that computer-based technology had the potential to create a narrow-focussed world view that merely reflected an understanding that could be represented using the technology (Vickers, 1978; in Stansfield, 2001). For problems such as NorBE and RAP, quantitative simulation modelling may be useful, but it may not be more useful than qualitative risk-based assessments, adaptive management strategies that incorporate management experiments and monitoring, using expert systems and/or participative or judgment-based decision-making using site-assessments and local knowledge and expertise.

By looking at the problem of modelling for decision-making in terms of patterns and anti-patterns, I have teased out six issues that I observed over the course of this research that receive scant or no attention in existing guidance documents, but, if not managed, have the potential to render models ineffective as decision-support tools. These are:

1. *The rightful role of models and modelling in decision-making*: When selecting or evaluating models, the models should be assessed against criteria of *relevance* to the decision-making problem, and *impact* on the decision-making process. In some cases this may demonstrate the futility rather than utility of model use.

2. *Methodological tension*: Most guidelines to the selection and application of models offer no substantive support for organisations confronting challenges related to different ways that individuals and organisations perceive problems and potential solutions. Epistemological and methodological tension amongst individuals *within* organisations, departments and disciplines may inhibit effective model use more than the problem of managing divides between researchers and managers.

3. *Treatment of uncertainty*: In general, analysts that run models should not be expected to go beyond the available data to make inferences and predictions. Scientists are subject to the same problems as lay people when they draw inferences.
4. *Information generation*: Information gathering is a relatively small part of the decision process and catchment managers should avoid undue time generating information. Given uncertainty in data as well as knowledge of the fundamental processes that result in nutrient runoff, information gathering may amount to little more than ‘opinion gathering’.

5. *Information transformation*: Before embarking on a modelling project, decision-makers need to determine what constitutes ‘evidence’ for given decision-making problems.

6. *The roles and responsibilities of actors*: Politics and the vested interests of different actors frequently interfere with decision-making. The catchment manager should actively manage a quality assurance process for a modelling activity that is cognisant of vested interest and focused on problem resolution.
9 IMPROVING GUIDELINES FOR MODELLING PRACTICE

9.1 Emergent Issues

Through both the experiential (modelling, ‘workshopping’, active participation) and theoretical (literature reviews) aspects of this research, two overarching factors that negatively affect the quality of modelling for decision-making in the SCA became apparent: 1) inadequate, inconsistent or unused methodological procedures that define good practice in modelling for better management and decision-making; and 2) inadequate scientific understanding and quantitative knowledge of the processes that affect nutrient movement in catchments.

Both of these factors imply a need for catchment management organisations such as the SCA to consider issues in addition to the model itself to ensure more effective use of models in decision-making. In particular, organisational and epistemological factors need to be considered. The first factor can be addressed through the development or use of appropriate guidelines (see Refsgaard et al., 2005a). The second factor is an issue of uncertainty and, to some extent, might be addressed through further research and monitoring or, in a more pragmatic and timely way, through the use of management tools that are tolerant of uncertainty, such as adaptive management. However, there also appears to be a need to better understand how institutions and individuals perceive and respond to uncertainty.

Good Modelling Practice

The need for guidance for the use of models and their application to decision-making problems was expressed at the first workshop. Participants generally agreed that the SCA could benefit from the use of a SOP, which could take the form of a decision-tree or checklist (Chapter 5.1.5). To this and similar ends, many guidelines covering ‘good modelling practice’ in water resource management and other spheres of natural resource management have been produced by government agencies, research organisations and others in the USA, Europe and Australia (for an overview, see Refsgaard and Henriksen, 2002; Refsgaard and Henriksen, 2004; Jakeman et al., 2006). Over twenty guidelines were reviewed for this research and many of these are discussed in the previous chapter. Although the guidelines are consistently strong
with respect to the technical issues relating to model selection and/or application, most are weak with respect to guidance for managing the cultural issues that sometimes negatively impact upon effective model use.

Data and Knowledge Deficiencies and Uncertainty

Since its inception, the SCA has been tasked with moving the locus of catchment management and responsibility towards the ‘outer catchment areas’ and away from the so-called ‘special areas’ adjoining the water supply reservoirs (CSIRO, 1999). The objective has been to abandon “all vestiges of the management regime that distinguishes between inner and outer catchments... [and replace it with] ...an integrated total catchment management strategy” (CSIRO, 1999 p. 8). A very significant effect of this change in focus has been to expose uncertainty about pollution sources and downstream effects at scales appropriate to management and mitigation. Previously the uncertainty was effectively ‘black boxed’.

In the context of modelling for decision-making, uncertainty appears to be viewed as a two-dimensional problem. Firstly, there is the problem of minimising and quantifying uncertainty (e.g., Reckhow, 1994; Reckhow and Chapra, 1999), and secondly there is the problem of communicating uncertainty to managers and other stakeholders to allow them to appropriately assess the quality and utility of model outputs and make planning, management or regulatory decisions based on them (e.g., Loucks, 1992; Mackenzie, 1999; Burgman, 2001). Most existing guidelines for assuring quality in modelling for decision-making devote much discourse towards addressing these two dimensions using methods that have changed little in the last few decades. Methods for dealing with uncertainty have not advanced to meet the increasing needs of decision-makers and there is currently no generally accepted approach to communicating or managing uncertainty in model-based decision-support (Walker et al., 2003).

Throughout this research it appeared that exposing uncertainty in model inputs and outputs in ways traditionally advocated in the literature (through transparency, communication and interaction between modeller and manager) did not necessarily improve the use of model outputs, but rather tended to inflame latent methodological tensions and, potentially, paralyse decision-making (See Question 5 in Chapter 5.2.6).
The fact that there was uncertainty at all seemed to be a fundamental problem, which was usually seen to be caused primarily by flawed models, even when the lack of quantitative scientific knowledge would affect any replacement model in the same way (Chapter 7.3.1). In addition, there was generally little interest shown by SCA managers in the potential application of methods for estimating the impact of uncertainty on model outputs that, in any case, are difficult or impossible to adequately determine due to gross data deficiencies (Chapter 7.3.1).

A predictable organisational response to environmental problems that are characterised by uncertainty is to establish a research program to identify and fill knowledge gaps, and thereby provide the proof needed to make rational policy decisions and take justifiable regulatory or management action (Wynne, 1992; Lövbrand and Öberg, 2005). Wynne (1992) refers to this as the ‘institutionalisation of science’. Others refer to related phenomena as the ‘scientisation of environmental policy’ (Lövbrand and Öberg, 2005) or the ‘politicisation of science’ (Pielke Jr, 2002). The institutionalisation of scientific research in the SCA can be traced through each SCA audit report (CSIRO, 1999, 2002; DEC, 2003, 2005). A large part of the research effort has been expended attempting to identify the sources of fluxes of nutrients in the drinking water catchments. However, despite these efforts, both scientists and SCA managers agree that there remain substantial and fundamental gaps in their quantitative knowledge of the biophysical processes, and they continue to call for more and better research and more and better modelling to provide robust science to resolve the uncertainty.

Wynne (1992 p. 112) writes, “the uncertainties which pervade attribution of environmental effects to specific environmental discharges are often large enough to sustain chronic conflict and indecision”. According to Wynne, an important step in the process of managing or avoiding potentially debilitating uncertainty and indecision is to seek to understand the “social character” of the uncertainties, including within the domain of “scientific knowledge”. Some characteristics of scientific knowledge that Wynne (1992) identified as important are listed below:

- When scientific knowledge is institutionalised, the scope and power of that knowledge can be exaggerated.
• The normal scientific approach to uncertainty is weakened because it tends to ignore inherent uncertainties, indeterminacy and ignorance (see also van Asselt and Rotmans, 2002). Some forms of “not knowing” cannot be quantified using normal (scientific) methods because it is impossible to completely describe or differentiate ‘what is’ or predict ‘what will be’ (see also O’Connor, 1994). Protagonists can use this characteristic to highlight the concealed uncertainties and undermine the case for action (see also Wynne and Mayer, 1993; O’Connor, 1994).

• For decision-making, the importance of uncertainty is related only to the extent to which policy or management plans are reliant on the relevant bodies of knowledge. Scientific uncertainty is not important in and of itself.

Aspects of these three factors may be relevant to the SCA, and may create conditions favourable to conflict and indecision. Each is discussed further below:

1) The institutionalisation of science in the SCA is apparent in the content of SCA audit reports and this view was re-enforced by most SCA staff observed throughout this research, who saw science-based decision-making as key to successful catchment management. For example, at an August 2004 meeting of SCA managers and collaborative researchers, in response to a question about the relevance of science-based management, one senior participant stated, “Of all the factors that influence our management... science, economics, politics, etc... science is the critical element”.

2 and 3) Policies such as the Neutral or Beneficial Effect Test (SCA, 2006) have been structured such that they rely strongly on quantitative assessments using models and scientific knowledge that are based on imported data, generalisations and assumptions that cannot be tested (validated) in identical locations, times and circumstances to that in which they were developed, and therefore, it can conceivably be argued, ignore inherent uncertainties and indeterminacies. At the same time, there has been an apparent preference amongst key decision-makers for deterministic model outputs and an expectation that models should be able to quantitatively differentiate nutrient exports based on subtle differences in topography, hydrogeology, soil type, land-use, and land management. These expectations go substantially beyond what any catchment-scale model is currently
able to reliably achieve and, if the model outputs are to be robust and without subjective assumptions, increases data requirements well beyond that which might be achievable in the foreseeable future.

These examples go some way to revealing how some SCA decision-makers cope with uncertainty. van der Sluijs (2005) draws an analogy whereby scientific uncertainty is viewed as a “monster”. He goes on to describe four “coping strategies”, originally described by Smits (2004), that individuals or organisations might use to deal with the “monster”. These strategies are termed “monster-exorcism”, “monster-adaptation”; “monster-embracement”; and “monster-assimilation”.

**Monster-exorcism:** A coping strategy that commonly appeared to be used in the SCA was a “monster-exorcism”:

“Monster-exorcists want to expel the monster. Uncertainty simply does not fit within symbolical order where science is seen as the producer of authoritative objective knowledge. They call for more objective research that should aim at reducing uncertainties. The borders between facts and values, knowledge and ignorance, science and policy are seen as real and inflexible and often the categories are also seen as norms (as in the notion that it is a good thing to keep science and policy, facts and values, objective and subjective separated)”.

A major problem with this coping strategy is that for each uncertainty that research attempts to resolve, more ‘monsters’ tend to appear as new-found knowledge also unearth other otherwise unforeseen or ‘black boxed’ complexity. An alternative strategy that monster-exorcists may adopt is to conceal uncertainty to avoid controversy or avoid a perception that uncertainty is evidence of poor-quality science.

**Monster-adapters:** “Monster-adapters” were also observed in the SCA. Monster-adapters seek to ‘tame’ the monster by quantifying uncertainty using statistical methods. When there is no objective ground for quantification, “monster-adapters may use subjective probability and Bayesian approaches to quantify uncertainties in terms of the degrees of belief that experts assign to their knowledge claims”. However, monster-adaptation tends to ignore methodological
uncertainties, in model structure for example, and the value-laden assumptions in most models.

**Monster-embracers:** Some ‘monster-embracers’ are fascinated by the “unfathomable complexity” of things and tend to highlight the limitations of positivist, reductionist schools of thought and offer philosophical alternatives. Other monster-embracers are reactionists or sceptics that tend to highlight and exaggerate the impact of uncertainty for personal or political gain.

**Monster-assimilators:** Monster-assimilators view all knowledge as value-laden and socially constructed. They seek to manage uncertainty and pluralism through transparency and participation of all affected parties rather than seek to identify a single ‘truth’. An example of a methodology used by monster-assimilators is ‘post-normal science’, an approach to decision-making that utilises an extended peer community of all affected stakeholders to review facts and help develop a consensus or a way forward that is more palatable and potentially more accurate and robust than ‘normal’ scientific decision-making (see Funtowicz and Ravetz, 1993).

### 9.1.1 Other issues

Many science and management staff in the SCA have a scientific background and at least a general familiarity with a variety of approaches to water quality modelling and the methods available to scientists to quantify uncertainty. As scientists, these managers are also well-equipped to understand the ways in which a very complex biophysical system needs to be simplified to allow for mathematical estimation of chemical loads in streams at multiple scales across a very large catchment. When discussing the most appropriate way to model nutrient fluxes, disagreements between and amongst catchment managers and scientists about the relative importance of the different biophysical processes were frequently observed, with individuals tending to advocate the use of methodologies and modelling approaches that reflected their personal expert opinion (*i.e.* favoured assumptions – see Question 5, Chapter 5.2.5 and Chapter 7.3.1). Often this resulted in groups dismissing existing modelling approaches and calling for the development of new and better models or the collection of more data because the existing crop of models and existing data repeatedly failed to meet group expectations.
Reflecting upon this research, it appears that the absence of an agreed conceptual model of nutrient fluxes severely limited the utility of mathematical models as prediction-based decision-support tools. Whilst most employees in the SCA understand the range of biophysical processes that affect the mobilisation, transport and delivery of nutrients from hillslopes and streambanks to streams and reservoirs, there is no consensus amongst scientists or within the SCA about the relative importance of each factor at scales applicable to management. Modelling efforts aimed at improving understanding and consensus about the relative importance of each factor, or at least the areas of highest uncertainty and significance to management, may therefore prove more useful than modelling aimed at delivering predictions based on an expert’s favoured assumptions.

9.2 Improving Guidelines for Model Selection and Use

The research identified six issues that were important in the context of this research. For the most part, published guidelines were silent with regard to these. The six issues were:

1. The rightful role of models and modelling in decision-making
2. Methodological tension
3. Treatment of uncertainty
4. Knowledge and information management: Information generation
5. Knowledge and information management: Information transformation
6. The roles and responsibilities of actors.

Each issue is presented below with some comment and some bulleted suggestions for managing each issue. Each bullet point could be regarded as a checklist item or guideline that needs to be addressed by all departments engaged in resolving a decision-making problem and, where applicable, implementing a subsequent management intervention. These guidelines are supplemental to the technical guidelines such as those provided in most existing guidance materials (Chapter 8).
9.2.1 The rightful role of models and modelling

Models are imperfect and incomplete decision-making tools. The limitations associated with predictive modelling have caused some investigators (e.g., Loucks 1992; Pielke Jr. et al., 2000; Reichert and Borsuk, 2005) to question the extent to which models and the science of simulation and prediction can benefit decision-making. Model use (or misuse) may sometimes have the effect of making the complexities and uncertainties associated with an environmental issue more conspicuous, increasing political controversy and producing a false perception of a need for more research rather than management intervention.

Where models are useful in decision-making, modelling is likely to be just one input in the decision process, and not necessarily the most important input. Further, the precision or otherwise of model predictions is not necessarily of prime importance.

Most guidelines for the use of models offer little support to users who wish to evaluate the utility of modelling for decision-making. Instead, guidelines usually assume that models will be useful and focus on aspects of modelling that emphasise adherence to procedures that maximise repeatability and model precision, effectively assuming that precision and scientific rigour are the most important criteria for model selection and use.

Similarly, decision-makers should have realistic expectations about how modelling can contribute to decision-making given the uncertainty in input data and process knowledge and the wide range of social, political and economic factors that decision-makers must consider in the face of competing needs and wants. According to Refsgaard and Henriksen (2002 p. 2-12):

“Translation of the manager's objectives/needs to performance criteria is a very difficult issue, but also very important, but is hardly considered in any of the existing [modelling] guidelines. It must be realised that this involves socio-economic considerations...”

It follows then that decision-makers should assess the performance of a given model against a measure of how well the modelling activity and/or modelling outputs assist decision-making. Evaluation criteria that focus only on mathematical precision and
scientific rigour, although these are the focus of most critical assessments of water quality models, are inadequate for decision-making. It is proposed here that more useful evaluation criteria are relevance to the decision-making problem and impact on the decision-making process.

**Managing the ‘Role of Models’ Problem:**

- Do not assume that models are the most effective decision-making tools.
- Consider alternative or complementary tools before embarking on a modelling exercise. These might include adaptive management experiments or tools that make qualitative estimates of risk rather than quantitative predictions of loads or concentrations.
- Be realistic about how much a model can contribute to decision-making, given uncertainty in data and scientific knowledge, competing socio-political and economic values and other cultural and organisational factors.
- Consider each modelling exercise in terms of the decision that needs to be made. The academic task of advancing the relevant science should not usually be a coincidental modelling objective.
- Assess the quality of a model and the modelling process against criteria of ‘relevance’ to the decision-making problem, and ‘impact’ on the decision-making process. The quality of the science, whilst important, may not be the area that needs improvement to achieve greater relevance and impact with the model in decision-making.

### 9.2.2 Managing methodological tension

Researchers concerned with the utilisation of science in policy development and environmental management often report on miscommunications and methodological tensions between researchers (scientists/analysts) and the users of research (policy-makers/managers). Much utilisation research has therefore focussed on identifying tools that bridge the gaps between the cultures, needs and beliefs of these “two-communities” (Caplan et al., 1975; Wingens, 1990; Cash et al., 2003). Solutions are often centred on methodologies that aid the ‘transfer’ of information and knowledge across the disciplinary divide. However, this research showed that epistemological
and methodological tensions within each community may constrain the use of models as effective decision-making tools more than barriers across disciplinary divides.

In the context of this research, methodological tension is the result of uncertainty in quantitative knowledge of the relative importance of the different processes that affect nutrient mobilisation, transport and delivery. Researchers had different and sometimes conflicting sets of assumptions and methods for conceptualising environmental processes and representing those problems in mathematical models. For example, there was much unresolved debate amongst scientists about the relative importance of hillslopes, streambanks and VSAs as contributors to diffuse nutrient pollution in streams and reservoirs. It is important to note that neither the current knowledge of the processes involved nor local data are sufficient to resolve this debate. Significantly, the SedNet approach of Olley and Deere (2003) in the Wingecarribee River Catchment and application of the EC models used here (Chapter 6) make very different implicit assumptions about the importance of hillslope nutrient sources versus streambank erosion, yet arrive at much the same conclusions in relation to management objectives to reduce nutrients loads. Likewise, there was ongoing discussion about the best method for modelling nutrient fluxes, with some participants preferring so-called ‘top-down’ modelling approaches that are relatively simple and based on field data; whilst, for the same problem, others had an obvious methodological preference for ‘bottom-up’ approaches that seek to represent all of the relevant biophysical processes.

Uncertainty in the science means that researchers are unable to draw definitive and unambiguous conclusions, and the range of competing conceptual models and methodologies means that individual approaches can lack authority and credibility. Models are more likely to be effective decision-making tools if the theory that backs them is authoritative and unambiguous (i.e. robust).

The methodological tension in the scientific community was replicated in SCA management. In his seminal work on the sociology of groups, Homans (1950) observed that complexity and conflict in the external environment tends to be replicated within a group when information from the environment is imported into the group. Throughout this research it was evident that uncertainty and methodological tension in the wider scientific community was also present amongst SCA staff.
charged with designing, managing and implementing regulations and management interventions.

Whilst unceasing enquiry, debate and challenging of opinions are valued and important features of the scientific process, each is typically seen as an obstruction to the decision-making process. At some timely point in the process of making a decision, decision-makers must, unlike scientists, withdraw (at least temporarily) from the task of intelligence gathering, or ‘scouting’ for new or better technologies or understanding, and focus on applying their understanding towards making and implementing a decision (Ancona and Caldwell, 1992). It follows that the SCA and other organisations should ensure that model selection, modelling or other research activities are not undertaken at the expense of intuitive and applied parts of the decision-process (i.e. ‘analysis paralysis’).

Whereas much literature on research utilisation is focussed on the problem of knowledge or information transfer (from researchers to managers), a bigger issue encountered during this research was the problem of ‘information transformation’. Most SCA managers and field staff are well informed about the relevant science, but the issue lay in confidently transforming this knowledge into rectification action plans and management interventions. For the outcomes of research and modelling to be useful in decision-making, the information obtained by decision-makers needs to be transformed into ‘evidence’.

The belief that decision-making should be ‘evidence-based’ (or science/research-based) is widely held by employees in the SCA, who wish to use research findings to inform decision-making in order to minimise inefficient use of resources or wrong decisions and reduce the possibility that decision-making is perceived as being subjectively biased. The weight given to research findings as evidence for management action is not only based on the quality and quantity of the science that was invested in a particular problem, but also on organisational and socio-political factors. According to Gibson (2003), evidence is strongest when research is invested with meaning and power through institutional, social and political processes. In this respect, Gibson (2003), drawing from Dery (1984), highlights the importance of understanding ‘organisational epistemology’ - which is an organisation’s rules of inquiry, observation and inference, and the basis upon which an organisation decides
what data or information is relevant to it’s operation. Adopting a frame of reference focussed on organisational epistemology can lead to more effective use of research tools such as models because:

“…it locates the drivers of data selection [and therefore inputs to decision-making] in the dynamic interaction between the structure of the organisation, the characteristics of the policy problem, and the political risks and opportunities created by these” (Gibson, 2003 p. 231).

Existing guidelines for good modelling practice generally offer no substantive support for decision-making organisations confronting challenges related to diverse individual epistemologies, dysfunctional or unique organisational epistemologies, and methodological tension amongst individuals and groups (Chapter 8). Much more attention is given to the seemingly lesser problem of negotiating the perceived divide between researchers and managers.

Managing the ‘Methodological tension’ Problem:

- Consider epistemological diversity and sources of methodological tension within and across different groups (executives, managers, analysts, field workers) and develop strategies for managing divergent expectations and conflicting standpoints.
- Consider the impact of organisational epistemology on the modelling process and how this affects the utility of models and modelling.

9.2.3 Managing uncertainty

Water quality models suffer from higher levels of uncertainty than some other water resource models due to the range of climatic, chemical, hydrologic, edaphic, topographic and hydrodynamic factors that affect pollutant mobilisation, transport and delivery. In relation to managing uncertainty in modelling projects, guidelines for the selection and use of models often highlight important and under-utilised statistical techniques for quantifying uncertainty in data and model predictions, such as sensitivity analysis, error propagation equations and Monte Carlo simulation (see van der Sluijs et al., 2004 for an overview of each technique).
However, in many modelling situations, desired levels of certainty cannot be achieved or uncertainty may be difficult or impossible to estimate due to gross data or knowledge deficiencies. Most guidelines are silent with respect to how uncertainty affects user confidence and impacts on decision-making. Inappropriate treatment of uncertainty may cause a kind of ‘analysis paralysis’ that hinders decision-making in much the same way as methodological tensions.

Although he did not intend to focus on the issue of model uncertainty, Walters (1997) offers three inter-related explanations as to why decision-makers may respond inappropriately to uncertainty. These include: 1) a misguided belief that certainty is necessary to maintain agency credibility; 2) the promotion of process research by scientists; and 3) inaction as a rational choice by bureaucratic decision makers.

*Uncertainty and Credibility:* Some SCA staff seemed to view unavoidable levels of uncertainty in data and model outputs as critical weaknesses that must ultimately lead to inaction or ineffective compromises, regardless of the application for which the modelling was likely to be used. Walters (1997) notes that some agency staff feel that they must “…present options with confidence and certitude to maintain credibility with political decision makers and players from other agencies”. The stakes are presumably higher for regulatory and other decisions that might ultimately be challenged in courts. In seeking to avoid perceptions of pretence and maintain credibility, some SCA staff seemed to dwell on uncertainty and push the need for new research and/or the use of more or ‘better’ *(i.e. usually more complex)* models unrealistically aimed at virtually eliminating uncertainty.

Also, at times it seemed that some SCA participants wanted modellers to produce or use models that went beyond the data and make statements about pollution sources and pollution loads that could not be justified given the limits of process knowledge and lack of input data and calibration data at relevant scales (see Appendix 1 and Chapter 5.2.5). Participants sometimes seemed to see models as a temporary alternative to the acquisition of data from scientific experiments and monitoring, without recognising that in the absence of reliable data and knowledge, modellers must use the same subjective judgment processes that lay people use and therefore model outputs lose the objectivity and credibility that decision-makers seek (Hammond *et al.*, 1975; Stewart, 1991).
In addition, more complex models generally rely on larger input datasets, employ more complex algorithms and rely on more assumptions than simpler models. Aside from the often expressed benefits of simpler models in decision-making (e.g., flexibility, ease of use, lower cost, realistic input data requirements and lower time and expertise requirements), relatively simple empirical and conceptual models may be more useful in decision-making because they make fewer implicit claims to precise knowledge of processes that protagonists can use to demonstrate the failure of the model to make accurate predictions.

Promotion of Process Research by Scientists: Walters (1997) laments that:

“It is depressingly easy for scientists to convince themselves, and bureaucratic funding agencies, that ‘fundamental understanding’ of the process or mechanism that they study is somehow important to predictions about impacts of ecosystem management policies”.

Given that researchers are in direct competition with each other for limited funding, it is unsurprising that at every opportunity throughout this research most lead-scientists involved in collaborative projects with the SCA were observed peddling the importance of their research and seeking new funding opportunities, usually on the basis that existing scientific knowledge was insufficient to draw the firm conclusions that managers wanted. One such example was provided by a physical geographer who, at a workshop in October 2005, raised the possibility that an earthquake might spark a landslide that could cause a tsunami-like pressure wave that might threaten dam walls. The discussion was followed by the geographer asking, “Who do I contact about getting funding for this kind of research?” In this competitive environment there may be a tendency for researchers to oversell the value of their research (or the ability of models and scientists themselves to provide managers with certitude) when applying for funding.

Inaction as a rational choice: Just as researchers have a vested interest in selling the need for research, decision-makers may consciously or subconsciously feel that they must avoid action when faced with uncertainty or unclear choices.

The difficulty that SCA managers might face in this respect is evidenced in the lengthy delays in finalising the Regional Environment Plan that sets out, amongst
other things, the rules for NorBE and RAP. In a 2003 audit report (HRC, 2003), the Healthy Rivers Commission of NSW was critical of the Department of Infrastructure Planning and Natural Resources and the SCA because the Regional Environment Plan, which was launched in July 1999 and due for completion in early 2001 was “bound up in process and complexity” and still in draft form (p. 8). This delay caused setbacks with respect to Rectification Action Planning and other catchment management initiatives. The plan was not finalised until June 2006 and did not come into effect until January 2007. Although this research was expected to produce outcomes that could guide modelling for the NorBE test and the RAP process, definitions were vague and operational issues were unclear. Following the release of the Neutral or Beneficial Effect Assessment Guidelines in December 2006 (SCA, 2006), it remains unclear how the SCA plans to use or assess any development using any model of nutrient fluxes, except the models sanctioned in the Guidelines, MUSIC (Wong et al., 2001) and DAM (McGuinness and Martens, 2003).

Decisive action by managers might produce outcry from stakeholders that are disaffected by decisions or cause professional embarrassment if outcomes differ from those expected (Walters, 1997). Walters (1997) points out that it is comparatively easy for decision-makers to justify inaction using the very same arguments that scientists use when promoting the need for further research – that there is simply insufficient evidence to justify intervention.

Where present, individual or organisational aversion or antipathy towards uncertainty needs to be replaced with an attitude that leads to more productive management of uncertainty. Although most guidelines offer only technical guidance for the most common statistical methods quantifying uncertainty, the Dutch National Institute for Public Health and the Environment has produced comprehensive guidance on the assessment and communication of different types of uncertainty using both quantitative and qualitative methods (Van der Sluijs et al., 2004).

Alternative responses to the dilemma of uncertainty versus the need for action, such as adaptive management experiments or decisions based on qualitative estimates of risk rather than quantitative predictions, did not seem to be considered seriously by participants in the current research.

*Managing the ‘Uncertainty’ Problem:*
There are multiple sources and types of uncertainty. Sources of uncertainty should be identified early so that users are able to identify areas requiring further research and have reasonable expectations of modelling. Useful guidance for assessing and communicating uncertainty are contained in van der Sluijs et al. (2004).

Some aspects of the science of diffuse nutrient management are actively debated in the scientific community and warrant further research aimed at achieving a degree of confidence and informed consensus. For the foreseeable future, a degree of uncertainty is unavoidable and should be expected and accepted.

In the absence of adequate input, calibration and validation data at appropriate temporal and spatial scales, models should not be viewed as a viable alternative to the acquisition of data from experiments and monitoring.

Do not ask modellers to go beyond the available data to make inferences and predictions unless you expect a subjective opinion. In drawing inferences, scientists use the same judgment processes that lay people use, and they are subject to the same problems. The use of models, although often perceived as an objective scientific method, is often a subjective process.

9.2.4 Information generation

On the surface, SCA managers seek to focus on the outcomes of research rather than purely academic, information-gathering pursuits. In a meeting of all collaborative researchers in August 2004, one researcher commented:

“It is good to learn more about different research topics even if there isn’t a direct benefit to the SCA. Scientific knowledge has intrinsic value”

An SCA manager replied:

“I don’t agree, and neither would some hard-nosed managers…”

A short time later, a different researcher stated:

“We need to work at convincing politicians to give us money to do our research”

To which, the same SCA manager rebuked:
“No. They will give you money to solve SCA problems!”

Also, in October, 2005, a senior manager said:

“What I want to see from the collaborative projects is an appropriate level of quality. Close enough is good enough”

The manager followed up defensively after that remark by stating:

“saying that makes me sound like a bureaucrat”.

The comments above reflect fundamental and stereo-typical differences between the mental models of scientists, who traditionally focus on more academic pursuits, and managers, who focus on solving management problems. However, the experience of the author is that significant parts of the SCA appear to think and act more like ‘scientists’ than these statements suggest. A driving factor for much of the SCA’s nutrient modelling effort is a strong desire to reduce or remove quantitative uncertainty in nutrient load predictions, even though this may be occurring at the expense of efficient and effective problem-solving that might occur in spite of inherent uncertainties.

One outcome of this information-driven paradigm seems to be that nutrient modelling and related research have produced several orphan datasets and reports and relatively untested and mistrusted hypotheses that are contained in those reports, for example earlier model reviews and modelling produced by Australian Water Technologies (AWT 2001a, 2001b, 2003), including the CECIL database (a product of AWT, 2001b). The work of AWT shared many of the objectives and some of the outcomes that have been identified in the current research.

Decision-makers need effective strategies for utilising uncertain model predictions so that they are comfortable promoting alternative strategies and prescribing solutions in a timely manner. Inappropriate treatment of uncertainty may make it difficult to move through the successive phases of the decision process and induce ‘analysis paralysis’. One way of achieving this may be for the manager to be mindful of the different phases of the decision-process. Lasswell (1971) divides the decision-making process into seven ‘phases’: 1) Gathering intelligence; 2) Promoting alternatives; 3) Prescribing the solution; 4) Implementation; 5) application; 6) Terminating the decision; and 7) Evaluating the decision.
Managing the ‘Information Generation’ Problem:

- Consider the impact that uncertainty is likely to have on all phases of the decision process.
- Evaluate the magnitude of the information gathering task and, if it should be a relatively small part of the decision process, avoid undue time generating information. Given uncertainty in data and knowledge, information gathering may amount to little more than ‘opinion gathering’.

9.2.5 Information transformation

Information transformation is vital for effective evidence-based decision-making.

In literature on the use of science in decision-making, there is a strong focus on the transfer of knowledge and information from the domain of science to the domain of management (e.g., Somlyódy et al., 1998; Kininmonth et al., 2003; McIntosh et al., 2004). This typical view was exemplified by an SCA manager at a collaborators meeting in December 2004:

“Scientific knowledge is only useful if it is transferred”

In relation to this comment, one collaborative researcher with considerable experience dealing with government agencies stated:

“The SCA is one of the worst organisations I have seen for knowledge transfer.”

Although the transfer of information and knowledge from scientists to managers is obviously important, the author’s opinion is that information transfer is not a significant issue for the SCA. This opinion is based on observation that the SCA has an abundance of science-trained and science-literate staff working in planning and management roles. Although less discussed in literature, a bigger issue for the SCA appears to be the problem of transforming scientific opinion into evidence that can support the SCA’s regulatory decisions (i.e. NorBE) and management objectives (i.e. RAP).

For the outcomes of scientific research and modelling to be useful in decision-making, the information needs to be transformed into something akin to ‘evidence’. Where knowledge transformation is absent or inadequate, the information from
research or modelling may be perceived as irrelevant or merely personal opinion. However, the science is inherently uncertain and there is no universally accepted law of nutrient flux at scales appropriate to management. Decision-making must therefore rely substantially on judgments, hypotheses, assumptions or ‘rules of thumb’. To some extent, evidence might be created by overcoming uncertainty through further research, but in the absence of conclusive science, knowledge transformation might also achieved by blending information from disparate sources using participatory and consensus-building activities.

Collating information from many different modelling, monitoring and research efforts undertaken at different times and by different staff or departments should be an early step towards transforming disparate datasets into some form of evidence. However, finding out what information was available and where it was held proved difficult for most collaborative researchers. For example, at a meeting in August 2004, one researcher commented:

“They [in the SCA] who should know about data and information held by the SCA don’t know where stuff is. The information needs to be stored and retrievable”.

The collation of both SCA data and relevant research undertaken by others will allow the SCA to build a compelling body of data and information that may help transform these disparate research efforts into a compelling body of knowledge that can support decision-making. Other researchers have highlighted this need, for example the development of the CECIL database of nutrient generation rates (AWT, 2001b), but these efforts appear to have been shelved and virtually forgotten.

Also, to target research or modelling outcomes so that they provide the type of results that the SCA can use to build a case that supports its decisions, researchers need to have a clear idea of why the research is being undertaken and what kind of results are required. Several collaborative researchers were frustrated by the lack of direction from the SCA. As one researcher commented:

“To effectively use science, the SCA needs an appropriate body to identify issues and [research] questions. The researchers need to know what the questions are! ... The SCA needs to take ownership of this problem”.
Managing the ‘Information Transformation’ Problem:

- Decision-makers need to determine what constitutes ‘evidence’ in their organisation for given decision-making problems.
- Consider how research and modelling can help produce useful ‘evidence’. This might involve assessing the ‘pedigree’ of the model, the expertise of the modellers, the use of a prescribed protocol for quality assurance, a peer review process or participatory activities that involve stakeholders or experts in the selection of model inputs or review of outputs.
- Ensure that SCA staff and researchers have access to the results of previous research, modelling and monitoring activities to avoid duplication and build a knowledge-base that assists in building a case for action.

9.2.6 Roles and responsibilities of actors

When using science to help make decisions, the role of the different actors involved in the decision process can become blurred. Lach (2000) reported on a survey of American scientists and natural resource managers and found that scientists tended to assess their credibility on the basis of the quality of methods used, data generated, and the hypotheses and theories used. These are the tools of “the scientific method”. Managers, on the other hand, judged the credibility of scientists by their ability to communicate with managers and translate results into usable information.

In literature, there are some arguments that scientists should play a more significant role in decision-making through advocacy (e.g., Lubchenco, 1997; Cortner, 2000) and/or by working with managers to help frame appropriate questions (e.g., Cullen et al., 2001), and that stakeholders and managers should play a greater role in model development and use. However, many scientists are more comfortable investigating “what is” and see the question of “what should be” as one that should not be their purview (Findlay, 1992; Lach, 2000). Similarly, managers that seek to use science to legitimise their management activities may be uncomfortable using non-technical inputs that contaminate what they perceive to be an objective and quantitative scientific approach.

The perception that governments fund scientific research in an effort to “…solve our problems” misplaces the role of science in decision-making. In reality, disciplinary
scientists cannot be expected to solve difficult problems that have a wide range of socio-political and economic components that impact upon the efficacy and effectiveness of decisions and management interventions.

In any decision process, nutrient modelling is likely to occur as part of Lasswell’s (1971) pre-decision phase (*i.e.* intelligence gathering), sharing this part of the decision process with other important information gathering activities, including information gathering within domains that include the socio-economic and political aspects that influences decisions.

*Managing the ‘Roles and Responsibilities’ Problem:*

- In selecting researchers for a particular modelling assignment, consider their ability to communicate and translate their research into *usable* forms.
- Modelling should not be seen as an opportunity for scientists to “Tell us [the decision-maker] what to do”. The expert role that scientists fulfil is not that of either advocate or decision-maker. The role of science and modelling is to inform and/or provide an input to decision-making.
- When negotiating modelling objectives with researchers, be aware of the problems caused by the competitiveness of the research funding process and the vested self-interest of researchers.
- The manager/decision-maker should actively manage a quality assurance process. Assessments of the quality of the modelling process and model outputs can only be expected in practice if the manager/decision-maker prescribes their use. This is especially the case when modelling is subject to tendering and cost plays a role in selection of the best bid.
- Maintaining the quality of the relationship between scientist and decision-maker should be the responsibility of relationship managers (aka boundary managers) on the side of the decision-maker (the funder). This role is similar to that of a ‘champion’, put forward by participants at a workshop with SCA managers, or an ‘ambassador’ (Ancona and Caldwell, 1992).
- Consider if decision-makers have a vested interest in avoiding decision-making when faced with uncertainty or unclear choices; *e.g.*, if decisions may be challenged by disaffected stakeholders; or if decision-makers might
suffer professional embarrassment when outcomes differ from those expected.

9.3 Tools for Improved Use of Models

To more usefully apply models to decision-making problems, it is necessary to address technical problems and uncertainty in the context of organisational and other cultural factors that affect the decision-maker’s ability to act with confidence. Two ways of achieving this may be through the use of Standard Operating Procedures and effective ‘boundary management’. Each is discussed below.

Standard Operating Procedures

An appropriate response to many of the issues identified in this part of the project could be the use a SOP. The SOP ‘solution’ was proposed by participants at a workshop of SCA management that formed part of this research. Guidelines for the analytical components of the modelling process, although under-utilised, are already fairly well developed (Chapter 8) and could be adapted for use by the SCA, in the form of an SOP, relatively easily.

Any SOP should focus on the over-riding problem of decision-making under uncertainty, and it should be recognised that, in relation to policy-development and catchment management, this is not primarily a model-related problem, but rather a culture- and process-related problem. By way of example, a useful starting point for a more complete guideline might be the NOAA Connecting Science, Policy and Decision-making Handbook, a guide that provides practical suggestions for researchers and agencies about ways to improve the focus of research intended to be useful to decision-makers (Jacobs, 2005). Guidance material should also address the six issues raised in this chapter.

Boundary Management

Any SOP should include a prominent role for staff to function as ‘boundary managers’ (Cash et al., 2003; McNie, 2007), and could benefit from the appointment of a person whose role is that of a ‘champion’ (as advocated by SCA staff at the first workshop, Chapter 5.1.5) or ‘ambassador’ (Ancona and Caldwell, 1992). In this
context, ‘boundary management’ refers to the process by which organisations or groups within organisations (i.e. departments) manage the flow of information to and from other entities. The flow of information may be vertical (e.g., to and from executive-level management) or horizontal (e.g., to and from departments within an organisation or to and from analysts and consultants commissioned by an organisation). ‘Ambassadors’ or ‘champions’ are persuasive people with good communication skills that support the interests of the modelling ‘team’ through negotiation and networking, including liaison with senior management, lobbying for resources, and undertaking other supportive ‘boundary management’ activities.
10 CONCLUSION

The aim of this research was to develop two models differing in sophistication and use them to prioritise nutrient pollution sources and calculate nutrient loads in the drinking water catchments of Sydney; and, in doing so, to learn methodological lessons that catchment managers can apply to choose and use models more effectively.

Additionally, barriers that inhibit the effective use of models in decision-making, and in particular, the transfer of scientific knowledge into management action were identified. Criteria were developed that catchment managers can use to assess the utility of catchment models in different decision-making and policy development situations. The impact of model sophistication and catchment manager’s participation in model development and use were also qualitatively assessed.

The research was undertaken with the SCA using a systemic, qualitative approach. Systemic approaches to research are commonly used where the problem is ill-defined, as it is here. Action research (Carr and Kemmis, 1983) is one qualitative, systemic methodology, and was used here as the guiding methodology to explore the model-related problems faced by catchment managers.

Activities undertaken within this action research framework were:

- a review of relevant literature from diverse disciplines;
- a workshop involving SCA catchment managers;
- a workshop involving water quality scientists and modellers;
- the selection of nutrient ECs and their application in a simple EC model;
- the enhancement of an EC model by weighting ECs using estimates of streamflow contribution;
- a focus group discussion to review the models involving SCA catchment managers; and
- a follow-up review of over 20 papers containing guidelines for good modelling practice in environmental management.
10.1 Prioritising Nutrient Pollution Sources

Two export coefficient models (Chapter 6) of varying complexity were used to identify nutrient pollution sources and strengths in the drinking water catchments. Phosphorus (P) and nitrogen (N) exports were estimated using the first EC model (Chapter 6.1). The results for a single sub-catchment, Wingecarribee River, were compared to calculations based on water quality and flow data (Olley and Deere, 2003). The EC model estimated annual P and N loads of 29 tonnes per year (t/yr) and 243 t/yr respectively compared to calculated loads in the river of 21 t/yr and 180 t/yr (Chapter 6.1.6).

For the Sydney drinking water catchments, modelled estimates showed that:

- Most grazing land is a weak source of N and P per unit area, but by virtue of its extent accounts for most of the N and P generated in the catchment;
- Better management of degraded pastures may provide an opportunity for significant improvement in water quality in some sub-catchments, but further studies are needed to confirm the ECs used in this study;
- Dairy farms occupy a small proportion of the catchment area and their pastures are a relatively small generator of total nutrient loads, but they may be a significant source of nutrients in the Kangaroo River and Wingecarribee River sub-catchments if there are inadequate means of attenuating concentrations or flows between the farm boundary and streams;
- Vegetable farms occupy a very small proportion of the total catchment area and modelling indicates that they are not a major contributor to total catchment nutrients loads, except for the Werri Berri Creek sub-catchment, where they may generate one third of the nutrients. They pose a local threat to water quality if farm dams or other management features are not present to reduce delivery to streams; and
- Modelled nutrient losses from urban areas are higher than from agricultural areas in the Blue Mountains and Lower Cox’s River sub-catchments.

Sub-catchments were prioritised for attention through the combined assessment of total modelled loads of P (t P/yr) and load per unit area (kg P/ha/yr). On this basis, Mulwaree River (43 t P/yr, 0.55 kg P/ha/yr) and Reedy Creek (40 t P/yr, 0.7 kg
P/ha/yr) sub-catchments rank high for attention, whereas Wollondilly River ranks low because of low per unit area loads (0.22 kg P/ha/yr) despite this sub-catchment delivering the highest load overall (60 t P/yr).

In the literature, variation of runoff depth was shown to be a major contributor to variation in the loads of nutrients from pastures (Nash and Halliwell, 1999, 2000). Therefore, it was considered that a significant improvement to the EC modelling approach would be to ‘weight’ EC values based on estimated variation in runoff using a simple approach requiring only commonly available data. In the second stage of modelling (the ‘enhanced’ EC model, Chapter 6.2), estimates of stream flow contribution (Croke, 2005) were used as a predictor (surrogate) of runoff depth and used to weight the EC values used in the simple model (for P only) on a ‘pixel-by-pixel’ basis. When applied across the entire catchment, the hydrologically-enhanced model increased estimated P exports in humid sub-catchments (e.g., Kangaroo River, Mid Cox’s River and Upper Nepean River) and lowered P exports in drier catchments (e.g., Mulwaree River, Upper Wollondilly River and Nerrimunga River). The approach also highlighted potential differences in P generation within land-use types and sub-catchments.

10.2 Choosing and Using Models

In general, scholarly discourse regarding the evaluation of environmental models has focused on describing the mathematical techniques used to represent the biophysical processes simulated by the models (e.g., Letcher et al., 1999). Similarly, much attention has been given over to quantifying model reliability by comparing model estimates and predictions against real-world observations (e.g., Young et al., 1996; Letcher et al., 2002). On the role of models in environmental management, consideration of the mathematical techniques and accuracy of models is widely perceived to be worthwhile to decision-makers because, by reducing uncertainty in models used for decision-support, decision-makers can consequently reduce the uncertainty associated with competing management options and more confidently arrive at the optimal decision (van der Sluijs, 2005).

Much previous scholarship has also been devoted to exploring ‘barriers’ to the effective use of science and models in decision-making (Cullen, 1990; NRC, 1990;
Loucks, 1992; Roots, 1992; Barnes, 1995; Somlyódy, 1997; Bernknopf and Herman, 1998; Berry et al., 1998; Cortner et al., 1998; Bouyssou et al., 2000; Dent, 2000; Argent and Mitchell, 2003; Bosch et al., 2003; Brezonik and Renwick, 2003; Cash et al., 2003; Landry et al., 2003; Caminiti, 2004; McIntosh et al., 2004; van Kerkhoff, 2005; Saloranta, In Press). To consolidate this diverse literature, barriers were divided into two fundamental types: 1) technical barriers; and 2) cultural barriers. Technical barriers are essentially problems of ‘hard’ systems. They are usually well-defined problems and relate to the use of technology and the acquisition and manipulation of data and information. Cultural barriers are typically viewed as ‘soft’ systems problems. Cultural barriers are those that relate to socio-cultural problems (i.e. problems relating to people, groups and organisations and the way they think and interact) (Gunderson et al., 1995; Cortner et al. 1998; Bosch et al. 2003). These barriers are usually ill-defined and difficult to quantify (see Checkland 1994).

Just as the barriers to the effective use of models in decision-making can be divided into two fundamental types, so can the approaches to evaluating models as decision-support tools. In general, two broad areas are discussed in the literature: 1) models and modelling products; and 2) the modelling process (Pielke Jr et al., 2000). The product produced by a model is usually a prediction or estimation. Much attention has been given to evaluating models and modelling products, and this has led to domination in the literature of ‘hard science’ papers focussed on how well models overcome the scientific and technical aspects of modelling to produce predictions. The modelling process refers to the way in which the model is used. Evaluating the modelling process demands more focus on the socio-cultural aspects of modelling, including the decision-space (the freedom that an organisation has to make decisions given the socio-political context in which the organisation operates: Jacobs, 2005), and is usually explored using ‘soft systems’ methodologies. This field of research has become more prevalent in scholarly publications in recent times.

A thorough review of discourse reveals that to fully evaluate the potential for success of a model in the context of decision-making, there is a need to consider the product, the process and the decision-space. From the entirety of the research including the review of literature, and the experiential components of the research (workshops, focus group, interactions with modellers and managers), six issues critical to the
effective implementation of modelling for decision-making were identified. Each is
critical to the effective implementation of models and yet all receive scant treatment
in most existing guidelines of good modelling practice:

1. Consideration of the role of models and modelling in decision-making
2. Managing methodological tensions
3. Qualitative treatment of uncertainty
4. Avoiding excessive information gathering and generation
5. Transformation of information into evidence
6. The roles and responsibilities of actors

Discovery and exploration of these issues was partly effected by viewing the
modelling process in terms of patterns (Gamma et al., 1994) and anti-patterns (Brown
et al., 1998), a conceptualisation normally used in the field of software engineering,
but equally relevant in the context of this research.

*Model Transparency, Model Complexity and Participation*

Haag and Kaupenjohann (2001) reflect the views of most workers interested in the
practice of using science and modelling in environmental decision-making. They
believe that models should be transparent, and that framing of models and model
choice and the evaluation of models should involve stakeholders and “local actors”.
In this research, model development occurred interactively and openly, with input and
opinion sought from SCA managers and scientists following an extensive review of
literature.

The results from this process did not appear to have a beneficial effect on SCA
manager’s confidence in the models, partly because the expectations of many
participants appeared to be incongruent with the constraints imposed on modelling by
data availability and quantitative knowledge of environmental processes and partly
because scientists were unable to agree on the primacy and relative importance of the
factors that affect nutrient export (see Mumpower and Stewart, 1996).

Nevertheless, attention to the six critical issues listed above can provide decision-
makers facing similar problems with methodological insights that can help them
negotiate their decision-space and apply models to decision problems with greater efficacy.
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APPENDIX 1  SCA’S MODELLING NEED

“The SCA needs a conceptually simple, spatially based approach to determining nutrient export rates that will enable the Authority to:

1. Reference the typical nutrient export rates (kg/ha) for different land uses;
2. Understand how the export rates for a land use vary with geographic location (i.e. different landscape/soil/hydrologic combinations);
3. Produce a catchment-wide map of nutrient export rates under current land use;
4. Inform rectification priorities and the NorBE (Neutral or Beneficial Effects Test) by estimating nutrient export loads (i.e. total load delivered to edge of stream) for areas of different sizes and complexity across the catchments including:
   • Drainage units typically 1000 – 10,000 hectares in area;
   • Rural lots typically 1-100 hectares in area;
5. Inform rectification actions by identifying & ranking the significance of different nutrient sources within a drainage unit;
6. Inform rectification planning and best practice guidelines by understanding the sensitivity of the export coefficients to different land use management practices; and
7. Understand how nutrient export rates vary with rainfall event size, and with spatial differences in annual average rainfall and rainfall variability”.

Written by an SCA manager for Workshop Two
## APPENDIX 2  RECLASSIFICATION KEY

Land use reclassification key for Models One and Two:

<table>
<thead>
<tr>
<th>SCA land-use/cover class</th>
<th>New Land-use category</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO DATA</td>
<td>NO DATA</td>
</tr>
<tr>
<td>Urban - Built up Areas</td>
<td>Urban - Residential</td>
</tr>
<tr>
<td>Urban - Residential</td>
<td>Urban - Residential</td>
</tr>
<tr>
<td>Urban - Rural Residential</td>
<td>Rural residential</td>
</tr>
<tr>
<td>Urban - Environmental Protection</td>
<td>Urban - Residential</td>
</tr>
<tr>
<td>Urban - Open Space Recreation</td>
<td>Urban - Residential</td>
</tr>
<tr>
<td>Heritage - Cemetery</td>
<td>Urban - Residential</td>
</tr>
<tr>
<td>Urban - Commercial</td>
<td>Urban - Industrial</td>
</tr>
<tr>
<td>Urban - Industrial</td>
<td>Urban - Industrial</td>
</tr>
<tr>
<td>Pollution - Waste Disposal</td>
<td>Urban - Industrial</td>
</tr>
<tr>
<td>Pollution - Biosolids Disposal</td>
<td>Urban - Industrial</td>
</tr>
<tr>
<td>Pollution - STP</td>
<td>Urban - Industrial</td>
</tr>
<tr>
<td>Pollution - Water Filtration</td>
<td>Urban - Industrial</td>
</tr>
<tr>
<td>Mining - Mines</td>
<td>OTHER</td>
</tr>
<tr>
<td>Mining - Quarries</td>
<td>OTHER</td>
</tr>
<tr>
<td>Mining - Tailings</td>
<td>OTHER</td>
</tr>
<tr>
<td>Transport - Airfield</td>
<td>OTHER</td>
</tr>
<tr>
<td>Transport - Minor Road</td>
<td>OTHER</td>
</tr>
<tr>
<td>Transport - Main Road</td>
<td>OTHER</td>
</tr>
<tr>
<td>Transport - Highway</td>
<td>OTHER</td>
</tr>
<tr>
<td>Transport - Railway</td>
<td>OTHER</td>
</tr>
<tr>
<td>Recreation - Golf Course</td>
<td>OTHER</td>
</tr>
<tr>
<td>Agriculture - Intensive Use</td>
<td>OTHER</td>
</tr>
<tr>
<td>Agriculture - Improved Pasture</td>
<td>Improved pasture</td>
</tr>
<tr>
<td>Agriculture - Unimproved Pasture</td>
<td>Unimproved pasture</td>
</tr>
<tr>
<td>Vegetation - Sparse</td>
<td>Unimproved pasture</td>
</tr>
<tr>
<td>Agriculture - Intensive Pasture</td>
<td>Degraded pasture</td>
</tr>
<tr>
<td>Lands - Degraded</td>
<td>Degraded Pasture</td>
</tr>
<tr>
<td>Horticulture - Orchard</td>
<td>Horticulture - Orchards</td>
</tr>
<tr>
<td>Plantation - Old Softwood Growth</td>
<td>Forest</td>
</tr>
<tr>
<td>Plantation - New Softwood Growth</td>
<td>Forest</td>
</tr>
<tr>
<td>Plantation - Native Vegetation</td>
<td>Forest</td>
</tr>
<tr>
<td>Plantation - Cleared</td>
<td>Forest</td>
</tr>
<tr>
<td>Vegetation - Forest or Woodland</td>
<td>Forest</td>
</tr>
<tr>
<td>Vegetation - Heath</td>
<td>Forest</td>
</tr>
<tr>
<td>Vegetation - Wetland</td>
<td>Forest</td>
</tr>
<tr>
<td>Vegetation - Rainforest</td>
<td>Forest</td>
</tr>
<tr>
<td>Water</td>
<td>Water</td>
</tr>
</tbody>
</table>
## APPENDIX 3  NEXSYS COMPARISON AND KEY DATA SOURCES (P)

Data sources used to assign phosphorus export coefficients for Model One and base EC values for Model Two.

<table>
<thead>
<tr>
<th>Land-use Category</th>
<th>NEXSYS Range (kg P/ha/yr)</th>
<th>Chosen EC (kg P/ha/yr)</th>
<th>Key References/Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest</td>
<td>0.01 – 0.20</td>
<td>0.05</td>
<td>Cullen et al. (1988) (plantation pine forest); Hollinger and Cornish (2001) measured nutrient exports from two areas within the drinking water catchments (unmanaged native forest).</td>
</tr>
<tr>
<td>Unimproved pasture</td>
<td>0.05 – 3.00</td>
<td>0.30</td>
<td>Cullen (1991); McCaskill et al. (2003); Ridley et al. (2003).</td>
</tr>
<tr>
<td>Degraded pasture</td>
<td>0.50 – 3.00</td>
<td>2.00</td>
<td>Neil and Fogarty (1991).</td>
</tr>
<tr>
<td>Improved pasture</td>
<td>0.20 – 3.00</td>
<td>0.70</td>
<td>Baginska et al. (1998); McCaskill et al. (2003); Ridley et al. (2003); Costin (1980); Cornish et al. (2002).</td>
</tr>
<tr>
<td>Urban - residential</td>
<td>0.40 – 4.00</td>
<td>0.40</td>
<td>Measured values are consistently at the lower end of the range estimated by NEXSYS. Campbell (1978); Smalls (1986); GHD Pty Ltd and EPA VIC (1987).</td>
</tr>
<tr>
<td>Urban - industrial</td>
<td>2.00 – 4.00</td>
<td>3.00</td>
<td>Campbell (1978); Smalls (1986); GHD Pty Ltd (1987).</td>
</tr>
<tr>
<td>Rural residential</td>
<td>N/A</td>
<td>0.70</td>
<td>Very little supporting data. Baginska et al. (1998). See also Improved pasture.</td>
</tr>
<tr>
<td>Dairy farms</td>
<td>0.20 – 3.00</td>
<td>5.00</td>
<td>Nash et al. (2000); Cornish et al. (2002); Baginska et al. (1998).</td>
</tr>
<tr>
<td>Horticulture (orchards)</td>
<td>0.10 – 3.00</td>
<td>0.60</td>
<td>Very little supporting data. Cuddy et al. (1994) used 0.3 kg P/ha/yr. Not a major land-use.</td>
</tr>
<tr>
<td>Vegetable farms</td>
<td>1.00 – 5.00</td>
<td>10</td>
<td>Little supporting data, except for market gardens. Where soil cover is low and fertiliser inputs are high (e.g., potatoes, market gardens) export rates may be very high. Baginska et al. (1998); Hollinger and Cornish (2001).</td>
</tr>
</tbody>
</table>
APPENDIX 4  NEXSYS COMPARISON AND KEY DATA SOURCES (N)

Data sources used to assign nitrogen export coefficients for Model One and base EC values for Model Two.

<table>
<thead>
<tr>
<th>Land-use Category</th>
<th>NEXSYS Range (kg N/ha/yr)</th>
<th>Chosen EC (kg N/ha/yr)</th>
<th>Comments/Key References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest</td>
<td>0.5 – 10.0</td>
<td>1</td>
<td>Hollinger and Cornish (2001) measured nutrient exports from two areas within the drinking water catchments.</td>
</tr>
<tr>
<td>Unimproved pasture</td>
<td>0.5 – 10.0</td>
<td>3.9</td>
<td>McCaskill et al. (2003).</td>
</tr>
<tr>
<td>Degraded pasture</td>
<td>0.5 – 10.0</td>
<td>4</td>
<td>Very little supporting data.</td>
</tr>
<tr>
<td>Improved pasture</td>
<td>0.5 – 15.0</td>
<td>4.2</td>
<td>Baginska et al. (1998); McCaskill et al. (2003); Costin (1980).</td>
</tr>
<tr>
<td>Urban - residential</td>
<td>5.0 – 20.0</td>
<td>9</td>
<td>Smalls (1986); GHD Pty Ltd (1987).</td>
</tr>
<tr>
<td>Rural residential</td>
<td>N/A</td>
<td>4.2</td>
<td>Very little supporting data. Baginska et al. (1998). See also Improved pasture.</td>
</tr>
<tr>
<td>Dairy farms</td>
<td>2.0 – 15.0</td>
<td>5</td>
<td>Baginska et al. (1998).</td>
</tr>
<tr>
<td>Horticulture (orchards)</td>
<td>2.0 – 15.0</td>
<td>6</td>
<td>Very little supporting data. Cuddy et al. (1994) used 4.7 kg N/ha/yr, but not a major land-use.</td>
</tr>
<tr>
<td>Vegetable farms</td>
<td>6.0 – 30.0</td>
<td>100</td>
<td>Little supporting data, except for market gardens. Where soil cover is low and fertiliser inputs are high (e.g., potatoes, market gardens) export rates may be very high. Baginska et al. (1998); Hollinger et al. (2001).</td>
</tr>
</tbody>
</table>
APPENDIX 5 PLANTATION FOREST

Published data permitting estimation of unit area exports of nutrients from plantation forests in south-eastern Australia are limited. Of the literature that is available, there appears to be incongruity in the data reported for sediment exports compared to that for nutrients. For example, Cullen reported phosphorus (P) exports from radiata pine forest in the ACT of between 0.05 and 0.07 kg/ha/yr. This is similar to data reported for relatively undisturbed native forests in the Blue Mountains, part of Sydney’s drinking water catchments (Hollinger and Cornish 2001). However, researchers studying sediment exports have reported soil erosion rates well over 30 times higher from pine forest than from unmanaged native forest (Neil and Fogarty, 1991; Wallbrink et al., 2002). If most P leaving plantation forests is associated with eroded sediment, then P exported from pine plantations would be expected to be much higher than reported in the available literature.

Based on data contained in the SCA’s Broad Land Cover/Use 2002” GIS dataset, plantation forest occupies 1.6% of the total area of the Sydney drinking water catchments. Kowmung River sub-catchment contains the largest proportion of plantation forests, with coverage of 8.3%. If we assume that most P exported from managed forests is associated with erosion and apply a 30-fold increase in P generation from pine forests compared to native forests (i.e. applying the same methodology to the estimation of P generation from pine forests to that which was used in the current report for degraded pasture), the P export coefficient for plantation forest would be 1.5kg/ha/yr.

For the purposes of performing a comparative analysis, the model used in the current report was modified to include plantation forest and an EC of 0.05 kg/ha/yr (i.e. the same rate as applied to all forest areas in the current report) and then changed to an EC of 1.5 kg/ha/yr (i.e. 30 times higher than remaining, unmanaged forest areas). The results are given in Table A.

If P export rates from plantation forests are similar to that for unmanaged forests (i.e. 0.05 kg/ha/yr), the contribution of plantation forests to the total P load is 0.5%. In Kowmung River, P exports from plantation forest would comprise approx. 5% of total
sub-catchment loads. If the higher P EC of 1.5kg/ha/yr is applied, the estimate for P generation from plantation forest increases 30-fold, from 1,200 kg to 38,000 kg or 9% of total catchment P generation. The overall contribution of P from plantations to total P generated in each sub-catchment also increases. In Kowmung River sub-catchment for example, the estimated contribution of P to total sub-catchment loads increased from 5% to approx. 60%.

Table A. Modelled P exports from plantation forest in the Sydney drinking water catchments assuming EC values of 0.05 kg/ha/yr and 1.5 kg/ha/yr.

<table>
<thead>
<tr>
<th>Sub-catchment</th>
<th>Area of plantation forest (ha)</th>
<th>Percent of sub-catchment</th>
<th>Plantation Forest EC 0.05 kg/ha/yr</th>
<th>Contribution to total P exports</th>
<th>Plantation Forest EC 1.5 kg/ha/yr</th>
<th>Contribution to total P exports</th>
</tr>
</thead>
<tbody>
<tr>
<td>Back &amp; Round Mountain Creek</td>
<td>1,434</td>
<td>4.2%</td>
<td>72</td>
<td>0.8%</td>
<td>2,151</td>
<td>18.5%</td>
</tr>
<tr>
<td>Boro Creek</td>
<td>1,159</td>
<td>3.2%</td>
<td>58</td>
<td>0.3%</td>
<td>1,739</td>
<td>9.3%</td>
</tr>
<tr>
<td>Braidwood Creek</td>
<td>390</td>
<td>1.0%</td>
<td>19</td>
<td>0.1%</td>
<td>584</td>
<td>3.4%</td>
</tr>
<tr>
<td>Bungonia Creek</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
</tr>
<tr>
<td>Endrick River</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
</tr>
<tr>
<td>Grose River - Blue Mts</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
</tr>
<tr>
<td>Jerrabattagulla Creek</td>
<td>684</td>
<td>1.9%</td>
<td>34</td>
<td>0.5%</td>
<td>1,026</td>
<td>13.8%</td>
</tr>
<tr>
<td>Kangaroo River</td>
<td>2,293</td>
<td>2.7%</td>
<td>115</td>
<td>0.6%</td>
<td>3,439</td>
<td>14.9%</td>
</tr>
<tr>
<td>Kowmung River</td>
<td>6,368</td>
<td>8.3%</td>
<td>318</td>
<td>4.9%</td>
<td>9,351</td>
<td>60.4%</td>
</tr>
<tr>
<td>Lake Barragorang</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
</tr>
<tr>
<td>Little River</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
</tr>
<tr>
<td>Lower Cox's River</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
</tr>
<tr>
<td>Mid Cox's River</td>
<td>1,920</td>
<td>1.8%</td>
<td>96</td>
<td>0.8%</td>
<td>2,880</td>
<td>18.5%</td>
</tr>
<tr>
<td>Mid Shoalhaven River</td>
<td>3,264</td>
<td>6.6%</td>
<td>163</td>
<td>3.3%</td>
<td>4,896</td>
<td>50.2%</td>
</tr>
<tr>
<td>Mongarlowe River</td>
<td>2,139</td>
<td>5.0%</td>
<td>107</td>
<td>1.6%</td>
<td>3,208</td>
<td>33.0%</td>
</tr>
<tr>
<td>Mulwaree River</td>
<td>176</td>
<td>0.2%</td>
<td>9</td>
<td>&lt; 0.1%</td>
<td>265</td>
<td>0.6%</td>
</tr>
<tr>
<td>Nattai River</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
</tr>
<tr>
<td>Nerrimanga River</td>
<td>1,463</td>
<td>3.0%</td>
<td>73</td>
<td>0.5%</td>
<td>2,194</td>
<td>13.4%</td>
</tr>
<tr>
<td>Reedy Creek</td>
<td>623</td>
<td>1.1%</td>
<td>31</td>
<td>0.1%</td>
<td>935</td>
<td>2.3%</td>
</tr>
<tr>
<td>Upper Cox's River</td>
<td>824</td>
<td>2.2%</td>
<td>41</td>
<td>0.4%</td>
<td>1,236</td>
<td>11.4%</td>
</tr>
<tr>
<td>Upper Nepean River</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
</tr>
<tr>
<td>Upper Shoalhaven River</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
</tr>
<tr>
<td>Upper Wollondilly River</td>
<td>39</td>
<td>0.1%</td>
<td>2</td>
<td>0.0%</td>
<td>58</td>
<td>0.2%</td>
</tr>
<tr>
<td>Werri Berri Creek</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
<td>0</td>
<td>0.0%</td>
</tr>
<tr>
<td>Wingoera River</td>
<td>800</td>
<td>1.0%</td>
<td>40</td>
<td>0.1%</td>
<td>1,200</td>
<td>4.2%</td>
</tr>
<tr>
<td>Wollondilly River</td>
<td>1,713</td>
<td>0.6%</td>
<td>86</td>
<td>0.1%</td>
<td>2,569</td>
<td>4.0%</td>
</tr>
<tr>
<td>Total</td>
<td>25,288</td>
<td>1.6%</td>
<td>1,264</td>
<td>0.5%</td>
<td>37,933</td>
<td>9.4%</td>
</tr>
</tbody>
</table>

The analysis presented here highlights the impact of uncertainty in EC values on efforts to prioritise land-uses and parts of the catchment for rectification action planning and assessment of development applications.
APPENDIX 6  OVERLAND FLOW

Both the simple export coefficient (EC) model and the enhanced EC model assume that P generation can occur at any part of the catchment. Annual streamflow contribution estimates were not adjusted based on groundwater and soil moisture conditions, which are important in identifying so-called ‘variable source areas’ (VSAs) that, in some hydrological systems may account for almost all run-off. This implies that the dominant mechanism for runoff generation is infiltration-excess overland flow.

In humid climates with regular rainfall, such as north-eastern USA, Europe and Japan, VSA hydrology is often used to explain variations in runoff (Ward, 1984) and to identify areas of differing P source strength within catchments (Gburek and Sharpley, 1998). VSA hydrology usually assumes that runoff in a catchment is dominated by saturation excess overland flow resulting from high water tables or high soil moisture in discharge zones near streams. Saturation excess overland flow predominates in cool, humid climates and in catchments with dense vegetation and water tables that are close to the surface. In VSA catchments, saturated areas contribute significantly to overland flow, and since most P is transported in surface runoff, the contributing areas are much stronger sources of diffuse P delivered to streams than non-contributing areas. The size and shape of a contributing area can change rapidly, and is a function of precipitation, topography, soil type, geology, groundwater levels and catchment moisture status (Gburek and Sharpley, 1998).

However, research into the relationship between exports of P from pasture and VSA hydrology has been limited (Dougherty et al., 2004). In Australia, VSA hydrology has been used to explain the runoff response to rainfall in some agricultural catchments in southern Victoria (Lyon et al., 2004; Wasimi, 2001) and in forested catchments of central Victoria (Burch et al., 1987), ACT (O’Loughlin 1981) and southern NSW (Mackay and Cornish, 1982). Cornish et al. (2002) found evidence suggesting that temporal variations in P exports from a dairy farm at Camden, south-west of Sydney, was the result of runoff from a VSA. The VSA concept was used by Endreny and Wood (2003) to weight generation rates for a catchment in New York, USA.
In general, Australian catchments differ from those found in areas where VSA hydrology has been extensively used to explain runoff response to rainfall. Flows in Australian catchments are storm-driven, and are “typically peakier, base flows are of lower proportion, runoff coefficients are smaller, and dry periods are longer and more variable, than in European and North American catchments” (Croke and Jakeman, 2001, p. 52).

The approach used for the enhanced EC model in the Sydney drinking water catchments assumes that surface runoff that reaches streams is generated across most of the catchment, and that infiltration excess overland flow is an important runoff generating process. Infiltration excess overland flow tends to predominate in hotter, drier climates, agricultural catchments with relatively low vegetation density (such as pastures), and/or catchments where rainfall intensity regularly exceeds soil permeability. In these catchments, runoff and erosion occurs over a much larger area of the catchment, and most areas of the catchment contribute to streamflow.

Nevertheless, VSA hydrology and saturation excess overland flow may explain runoff response in some parts of the catchment. VSA hydrology may be predominant for short duration rainfall events or during unusually wet seasons. In the more humid parts of the catchment and in forested areas, VSA hydrology may be particularly important. Studies conducted in forests of the ACT and central Victoria have highlighted the potential importance of VSA hydrology in some forested parts of the SCA’s area of operations (see O’Loughlin, 1981; Burch et al., 1987).

The type of overland flow that is dominant in a catchment has important implications for management. In VSA catchments (saturation excess dominant), nutrient export abatement actions need only be focussed on relatively small and predictable ‘critical source areas’ near streams. In non-VSA catchments, abatement actions need to be focussed on managing concentrated sources across the whole catchment (Pionke et al., 2000; Schnabel, 2000).