USE OF THESES

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DYNAMIC TRADABLE DISCHARGE PERMITS FOR MANAGING RIVER WATER QUALITY: AN EVALUATION OF AUSTRALIA’S HUNTER RIVER SALINITY TRADING SCHEME

Qinghong Pu

A thesis submitted for the degree of Doctor of Philosophy of Australian National University

December 2008

The Fenner School of Environment and Society
Candidate's Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university. To the best of the author’s knowledge, it contains no material previously published or written by another person, except where due reference is made in the text.

Qinghong Pu

16 December 2008
Acknowledgements

My journey toward this PhD has been long and arduous. It would have been impossible for me to arrive at the destination without the support of so many great people. Here is a small tribute to all of them.

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Abstract

This study provides the first comprehensive quantitative assessment of Australia's pilot Hunter River Salinity Trading Scheme (PHRSTS). It casts new light on the relative merits of tradable permit system in terms of environmental and cost effectiveness under the PHRSTS, highlighting the potential benefits from an integrated regulatory instrument for management of natural resource and environmental quality.

The PHRSTS was introduced by the New South Wales Environmental Protection Authority (NSW EPA) in 1995 to regulate the discharge of saline water from the coal mines and electric power generators into the Hunter River, which was affecting other uses of the River. It was made permanent in 2002, becoming the formalised HRSTS (FHRSTS). Allowing for the total permitted salt discharge to vary dynamically from day to day subject to river flow conditions, and for salt permits be traded among the mines and power generators, the Scheme is widely known as Australia's first active water quality trading program, and still appears to be the only dynamic tradable permits scheme operating in the world. The NSW EPA has claimed that the PHRSTS achieved significant environmental and economic benefits, but until now there has been no rigorous examination of its operational performance. To help fill this gap, this study investigates the origins, evolution and institutional arrangements of the PHRSTS, examines the performance of the salt credits trading market of the PHRSTS and the two credit auctions of the FHRSTS, and evaluates the environmental and economic effectiveness of the PHRSTS.

In particular, this study finds that:

(1) The credits trading market of the PHRSTS was active in terms of both volume of trading and number of participants, in spite of the high proportion of intra-company trading. The successful bidders in the credit auctions of the FHRSTS were a mix of sellers and buyers on the credit trading market of the PHRSTS. The low, narrowly-spread auction prices suggest that the firms did not value the credits highly and that the differentials in marginal cost of salt control across the participants are not large enough to yield significant savings from the credit trading.

(2) The overall salinity objectives of the Hunter River were attained under the PHRSTS. However, the PHRSTS did not significantly improve the river salinity compared to the previous Trickle Discharge management system. The PHRSTS only generated trivial savings in social damage cost.
The PHRSTS generated measurable cost savings in the total control cost of saline water to its participants over its entire period. But this was minor in relation to the participants' sales revenues, and the tradability of the discharge permits accounted for only a very small proportion of the control cost savings. Dynamicism, instead of tradability, of the discharge permits was by far the main source of the cost savings.

This study therefore concludes that neither the environmental effectiveness nor the economic effectiveness of the PHRSTS is as impressive as that claimed by the NSW EPA. Nevertheless, the valuable experience drawn from the experimental design and operation of the PHRSTS should prove useful for broader water quality management strategies in Australia and elsewhere.
Acronyms

Please see Glossary for the full explanation of some key acronyms.

ABS Australian Bureau of Statistics
ACF Autocorrelation function
AIC Akaike information criterion
ARI Average Recurrence Interval
AWE Avoided withholdings event
CAC Command-and-control
COD Chemical oxygen demand
DEC Department of Environment and Conservation (NSW)
DECC Department of Environment and Climate Change (NSW)
DEP Department of Environment and Planning (NSW)
DIPNR Department of Infrastructure, Planning and Natural Resources (NSW)
DLWC Department of Land and Water Conservation (NSW)
DMR Department of Mineral Resources (NSW)
DN Dynamic non-tradable permits
DNR Department of Natural Resources (NSW)
DT Dynamic tradable permits
DWR Department of Water Resources (NSW)
EC Electrical conductivity
EPA Environmental Protection Authority (NSW)
FHRSTS Formalised Hunter River Salinity Trading Scheme (from 1/12/2002 to present)
GHG Greenhouse gases
HITS Hunter Integrated Telemetry Scheme
HVRF Hunter Valley Research Foundation (NSW)
ID Identification number
<table>
<thead>
<tr>
<th>Abbreviation</th>
<th>Description</th>
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<tbody>
<tr>
<td>IQQM</td>
<td>Integrated Quantity and Quality Model</td>
</tr>
<tr>
<td>MAC</td>
<td>Marginal abatement cost</td>
</tr>
<tr>
<td>MBI</td>
<td>Market-based instruments</td>
</tr>
<tr>
<td>MSD</td>
<td>Marginal social damage</td>
</tr>
<tr>
<td>NPS</td>
<td>Nonpoint sources</td>
</tr>
<tr>
<td>NSW</td>
<td>New South Wales</td>
</tr>
<tr>
<td>PACF</td>
<td>Partial autocorrelation function</td>
</tr>
<tr>
<td>PHRSTS</td>
<td>Pilot Hunter River Salinity Trading Scheme (from 1/1/1995 to 30/11/2002)</td>
</tr>
<tr>
<td>PS</td>
<td>Point sources</td>
</tr>
<tr>
<td>RO</td>
<td>Reverse osmosis</td>
</tr>
<tr>
<td>SN</td>
<td>Static non-tradable permits</td>
</tr>
<tr>
<td>SPA</td>
<td>State Planning Authority</td>
</tr>
<tr>
<td>SPCC</td>
<td>State Pollution Control Commission</td>
</tr>
<tr>
<td>ST</td>
<td>Static tradable permits</td>
</tr>
<tr>
<td>TAD</td>
<td>Total allowable discharge</td>
</tr>
<tr>
<td>TD</td>
<td>Trickle Discharge</td>
</tr>
<tr>
<td>TDS</td>
<td>Total dissolved salt</td>
</tr>
<tr>
<td>TSS</td>
<td>Total soluble salt</td>
</tr>
<tr>
<td>WAC</td>
<td>Waste assimilative capacity</td>
</tr>
<tr>
<td>WRC</td>
<td>Water Resource Commission (NSW)</td>
</tr>
</tbody>
</table>
### Glossary

Please see Acronyms for some of the terms listed below.

<table>
<thead>
<tr>
<th>Term</th>
<th>Description</th>
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<tbody>
<tr>
<td>A (single) trade of salt credits</td>
<td>One transfer of salt credits from one HRSTS participant to another.</td>
</tr>
<tr>
<td>Arm-relationship trades</td>
<td>The trades of salt credits under the HRSTS between the power stations and the coal mines that are the power stations' coal suppliers.</td>
</tr>
<tr>
<td>Autocorrelation effect (time series effect)</td>
<td>In a sequence of observations measured typically at successive times, spaced at (often uniform) time intervals, the observations are usually not independent of each other. Autocorrelation effect is the effect of the observation made at a particular time on those made at immediately preceding times.</td>
</tr>
<tr>
<td>Block (of river flow)</td>
<td>The amount of water that passes the Singleton gauging station during a 24-hour period. There are 365 blocks within a year defined under the HRSTS.</td>
</tr>
<tr>
<td>Conditional permit system</td>
<td>The quantity of permitted discharge of a pollutant at one time is a pre-specified function of instantaneous river conditions.</td>
</tr>
<tr>
<td>Conservative (dilutable) pollutants</td>
<td>Pollutants that do not reduce in mass on entering a river system and whose ambient concentration in the river is inversely related to the river flow, other things being equal.</td>
</tr>
<tr>
<td>Contribution fee</td>
<td>The fee that each PHRSTS participant was required to pay to share the operational cost of the PHRSTS.</td>
</tr>
<tr>
<td>Currency of a salt credit</td>
<td>The amount of salt load (in tonnes) that a salt credit allows to be discharged into a specific block of river flow.</td>
</tr>
<tr>
<td>Dilution Discharge</td>
<td>The strategy of increasing diluting effects of the river through artificially increasing the river flow.</td>
</tr>
<tr>
<td>Discharge permits</td>
<td>Permits that define the maximum amount of a specific pollutant allowed to be discharged from pollutant sources within a certain period of time, measured as kg/day or tonnes/day.</td>
</tr>
<tr>
<td>Term</td>
<td>Description</td>
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</tr>
<tr>
<td>Discriminatory pricing</td>
<td>An auction pricing rule that requires successful bidders to pay their own bid prices.</td>
</tr>
<tr>
<td>(pay-as-you-bid)</td>
<td></td>
</tr>
<tr>
<td>Dynamic permits</td>
<td>Regulatory changes in the total number of permits issued at any time in response to rapidly changing river conditions, such as flow and background concentrations.</td>
</tr>
<tr>
<td>Dynamic tradable permits</td>
<td>Discharge permits that vary in response to changing river conditions and are allowed to be traded between dischargers.</td>
</tr>
<tr>
<td>Dynamicism</td>
<td>Allowing the total permitted discharge to change over time in response to river conditions.</td>
</tr>
<tr>
<td>Economic effectiveness</td>
<td>The inverse of the control cost for achieving a given water quality standard.</td>
</tr>
<tr>
<td>(or cost effectiveness)</td>
<td></td>
</tr>
<tr>
<td>Economic performance</td>
<td>Measured by the control cost of achieving a given water quality standard.</td>
</tr>
<tr>
<td>Effluent standards</td>
<td>Dictate the amount of a pollutant that can be discharged from its source.</td>
</tr>
<tr>
<td>Electrical conductivity</td>
<td>Unit of ambient concentration of salinity. 1IEC = 1 micro Siemens per centimeter (µS/cm) at 25°C 1000 µS/cm = approximately 600–680 mg/L</td>
</tr>
<tr>
<td>Environmental effectiveness</td>
<td>Improvement in water quality, measured by a lower ambient concentration of a pollutant in a river in for example mg/L.</td>
</tr>
<tr>
<td>Environmental performance</td>
<td>Measured by achievement of a given water quality standard.</td>
</tr>
<tr>
<td>Flood flow period</td>
<td>(See Table 5.1 for Flood flow thresholds for Upper, Middle and Low river sectors of the Hunter River). In the Flood flow period, salt discharges are effectively unlimited, provided the maximum volumetric discharge rates set in discharge licences are met.</td>
</tr>
<tr>
<td>Flow augmentation</td>
<td>Alternative to achieving water quality standards for conservative pollutants by artificially increasing river flow to provide a waste assimilative capacity beyond that provided by natural river flow.</td>
</tr>
<tr>
<td><strong>Formalised Hunter River Salinity Trading Scheme</strong></td>
<td>Stage of the Hunter River Salinity Trading Scheme that became permanent on 1 December 2002, through the NSW <em>Protection of the Environmental Operation Regulation 2002</em>.</td>
</tr>
<tr>
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</tr>
<tr>
<td><strong>High flow period</strong></td>
<td>(See Table 5.1 for High flow thresholds for the Upper, Middle and Lower river sectors of the Hunter River). In the High flow period, the total quantity of salt discharge permits is dynamically determined by the river flow conditions and in-river salinity levels and thus varies on a daily basis. The discharge permits in the form of salt credits are tradable among industries, subject to river salinity objectives being met.</td>
</tr>
<tr>
<td><strong>Hot spot</strong></td>
<td>The phenomenon in which more discharges are concentrated spatially and in turn cause site-specific impacts on river quality as a result of permit trading.</td>
</tr>
<tr>
<td><strong>Hunter River Salinity Trading Scheme</strong></td>
<td>A dynamic tradable permit scheme introduced by the NSW EPA in 1995 to regulate the discharge of saline water from the coal mines and electric power generators into the Hunter River. It includes two phases: pilot HRSTS (PHRSTS) from 1 January 1995 to 30 November 2002, and formalised HRSTS (FHRSTS) from 1 December 2002 to the present.</td>
</tr>
<tr>
<td><strong>Inter-company trades</strong></td>
<td>The trades of salt credits under the HRSTS between the mines owned by different companies, and those between the power stations and mines that had no supply relationship with the power stations.</td>
</tr>
<tr>
<td><strong>Intra-company trades</strong></td>
<td>The trades of salt credits under the HRSTS between mines owned by the same parent company.</td>
</tr>
<tr>
<td><strong>Low flow period</strong></td>
<td>(See Table 5.1 for Low flow thresholds for the Upper, Middle and Lower river sectors of the Hunter River). In the Low flow period, the discharge of industrial saline water is prohibited.</td>
</tr>
<tr>
<td><strong>Lower river sector</strong></td>
<td>The river sector of the Hunter River between Glennies Creek/Hunter River junction and Singleton.</td>
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<tr>
<td>Term</td>
<td>Definition</td>
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<tr>
<td>Maximum (volumetric) discharge rate</td>
<td>A condition set in the discharge licences that specifies the maximum daily volume of saline water (measured in ML/day) that is allowed to be discharged by a licence holder from each licensed point, but does not limit the salt load in that volume.</td>
</tr>
<tr>
<td>Middle river sector</td>
<td>The river sector of the Hunter River between Denman and Glennis Creek/Hunter River junction.</td>
</tr>
<tr>
<td>Mine water</td>
<td>Water collected within mine workings as a result of groundwater seepage and rainfall accumulation.</td>
</tr>
<tr>
<td>Multiple-round auction</td>
<td>An open, iterative bidding procedure that allows bidders in each bidding round to make a series of bids, taking into account the information revealed by other earlier bids.</td>
</tr>
<tr>
<td>New development</td>
<td>Refers to Bengalla Coal Mine and Redbank Power Station, which participated in the PHRSTS in 2000 and 2001, respectively.</td>
</tr>
<tr>
<td>Non-conservative substances</td>
<td>Substances that are reduced in quantity by the physical, chemical and biological processes of receiving water</td>
</tr>
<tr>
<td>Periodic (dynamic) permit system</td>
<td>Wastewater discharge is regulated on the basis of calendar date.</td>
</tr>
<tr>
<td>PHRSTS participants</td>
<td>Refers to those coal mines and electric power generators that participated in the PHRSTS. At the start of the PHRSTS, there were 19 coal mines and 2 power stations participating in the Scheme. Two new developments, Bengalla Coal Mine and Redbank Power Station, then participated in the PHRSTS in 2000 and 2001, respectively.</td>
</tr>
<tr>
<td>Precautionary excess discharges (for coal mines)</td>
<td>The discharges that occurred whenever a mine, facing a high degree of uncertainty in the time and duration of the next discharge event, chooses to make an excess discharge one day even when it still has spare dam capacity.</td>
</tr>
<tr>
<td>Prioritised Discharge</td>
<td>A proposal for managing industrial wastewater discharged into the Hunter River based on the degree of reliability of use of the discharge permits.</td>
</tr>
<tr>
<td>Term</td>
<td>Definition</td>
</tr>
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</tr>
<tr>
<td>Quality margin</td>
<td>The difference between the concentration limit of a pollutant specified in water quality standards and the existing concentration of the pollutant in a river.</td>
</tr>
<tr>
<td>River salinity</td>
<td>Ambient concentration of salt in a river.</td>
</tr>
<tr>
<td>Salt credit</td>
<td>A salt credit entitles 0.1% of the total allowable salt load to be discharged into a specific block of river flow.</td>
</tr>
<tr>
<td>Salt load</td>
<td>The mass of salt (in tonnes/day or tonnes/year) passing a particular location within a given time period.</td>
</tr>
<tr>
<td>Salt spike</td>
<td>A sharp increase in the river’s salinity at the beginning of a storm event after a long drought period, caused by storm water flushing accumulated salt from the river streams.</td>
</tr>
<tr>
<td>Sealed-bid auction</td>
<td>One round of bidding in which the bidders simultaneously submit their demand quantities at one or more price levels without knowing the bids of other bidders. These bids are added to form the aggregated demand curve. The intersection between the demand curve and available supply determines the clearing price and allocation.</td>
</tr>
<tr>
<td>Staged Discharge</td>
<td>The strategy of allowing more industrial saline water to be discharged into the Hunter River when river flow is high and river salinity is low.</td>
</tr>
<tr>
<td>Static permits</td>
<td>The discharge permits of pollutants that are defined based on worst-case conditions (such as, river flow, flow velocity and water temperature) of the receiving water body, and remain unchanged at all times.</td>
</tr>
<tr>
<td>Total allowable discharge</td>
<td>The aggregated maximum amount of salt load (tonnes/day) that is allowed to be discharged into a High-flow river block without resulting in the river salinity exceeding 900 EC at Singleton.</td>
</tr>
<tr>
<td>Tradability</td>
<td>Defines whether individual permits are tradable or non-tradable among different dischargers.</td>
</tr>
<tr>
<td>Tradable permits</td>
<td>A total allowable discharge is established and allocated among firms in the form of permits, which can be bought and sold among firms, provided a firm’s discharge never exceeds the amount of permits it owns.</td>
</tr>
<tr>
<td><strong>Trading ratio</strong></td>
<td>Under a tradable permit scheme, the number of units of a pollutant that are required to be reduced for a source in order that a single unit of the pollutant can be credited for another source.</td>
</tr>
<tr>
<td>-------------------</td>
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</tr>
<tr>
<td><strong>Transaction costs</strong></td>
<td>The costs that are required to establish the pollution control program and those required to manage the program and keep in running.</td>
</tr>
<tr>
<td><strong>Trickle Discharge</strong></td>
<td>A licensing system that the NSW EPA adopted before 1995 to regulate the industrial wastewater discharge to the Hunter River. Initially it allowed industries to discharge wastewater into the nearest watercourse on a continuous, low-rate basis all year around, but with little regard for changing river flow and background salinity conditions.</td>
</tr>
<tr>
<td><strong>Trigger value approach</strong></td>
<td>Defines dynamic permits based on ranges of flows rather than individual instantaneous flows.</td>
</tr>
<tr>
<td><strong>Uniform pricing</strong></td>
<td>An auction pricing rule that requires all the winners to pay the same price (that is, the clearing price) for every item they acquire.</td>
</tr>
<tr>
<td><strong>Upper river sector</strong></td>
<td>The river sector of the Hunter River between Muswell Brook and Denman.</td>
</tr>
<tr>
<td><strong>Utilisation rate of credits</strong></td>
<td>The percentage of the actual aggregated salt load discharged on one day by the PHRSTS participants over the total allowable discharge on that day.</td>
</tr>
<tr>
<td><strong>Water quality standards</strong></td>
<td>Desired ambient concentrations of pollutants in rivers.</td>
</tr>
</tbody>
</table>
# Table of Contents

Candidate's Declaration ......................................................... i

Acknowledgements ................................................................... i

Abstract .................................................................................. iii

Acronyms .................................................................................. v

Glossary ..................................................................................... vii

Chapter 1  Introduction ............................................................. 1

1.1 Management of river quality .................................................. 1
1.2 Pilot Hunter River Salinity Trading Scheme ................................. 4
1.3 Research justification ............................................................ 6
1.4 Research questions ............................................................. 7
1.5 Research methodology ......................................................... 9
1.6 Structure and contributions of this thesis ................................. 11

Chapter 2  Overview of Strategies for Water Quality Management .... 14

2.1 Factors relevant to river pollution ............................................ 15
2.1.1 Sources ........................................................................ 15
2.1.2 Substances .................................................................... 15
2.1.3 Water quality management and water pollution control ......... 16
2.1.4 Enhancement of a river's wastewater assimilative capacity .... 20
2.2 Dynamic permits ............................................................... 21
2.2.1 Static permits versus dynamic permits ................................ 22
2.2.2 Environmental performance ............................................. 24
2.2.3 Economic performance .................................................. 25
2.2.4 Design and implementation issues ..................................... 25
2.3 Tradable permits ................................................................ 30
2.3.1 Tradable permits versus non-tradable permits ..................... 30
2.3.2 Economic performance .................................................. 31
2.3.3 Environmental benefits ................................................ 33
2.3.4 Tradable effluent permits versus effluent tax ....................... 33
2.3.5 Key design and implementation issues ............................... 35
2.3.6 Experience and lessons from effluent trading ...................... 41
2.4 Dynamic tradable permits – a theoretical perspective ............... 43
2.5 Summary ........................................................................... 51

Chapter 3  Hunter River Salinity and Industrial Saline Water Discharge 52

3.1 Catchment overview ............................................................ 52
3.1.1 Geology ........................................................................ 54
3.1.2 Rainfall and evaporation 54
3.1.3 Surface water 54
3.1.4 Groundwater 55
3.1.5 Social and economic activities 55
3.1.6 Principle water uses and discharges 56

3.2 Hunter River flow 57
3.3 Hunter River salinity 60

3.4 Mining and Hunter River salinity 63
3.4.1 Coal mines 63
3.4.2 Water demand and supply 68
3.4.3 Generation of mine water 70
3.4.4 Discharge of mine water 72
3.4.5 Impacts of mine water on river salinity 77

3.5 Power generation and river salinity 78
3.5.1 Macquarie Generation 78
3.5.2 Redbank power station – a new development under the PHRSTS 81
3.5.3 Relationship between coal mines and power stations 82
3.5.4 Agricultural effects on river salinity 83

3.6 Social impacts of the Hunter River salinity 84
3.7 Summary 86

Chapter 4  Historical Perspective of Water Quality Management in the Hunter River 87

4.1 Overview of strategies for water quality management in the Hunter River 87
4.1.1 Inclusion of quality management in quantity management 87
4.1.2 Domination of direct regulations 88
4.1.3 Exploration of flexible management strategies 89
4.1.4 Application of economic instruments 89

4.2 Trickle Discharge 91
4.2.1 Limits on discharge volume and concentration 92
4.2.2 Ambient and incremental limits 95

4.3 Staged Discharge 97

4.4 A trial of Dilution Discharge 99

4.5 Three-tier Scheme 101

4.6 Prioritised Discharge 103

4.7 Initiation of the pilot HRSTS 104

4.8 Summary 105

Chapter 5  Mechanisms and Operations of the PHRSTS 107

5.1 Mechanism of the PHRSTS 107
5.1.1 Main elements in the PHRSTS rules: sectors, flow periods and blocks 107
5.1.2 Why are some salinity objectives laxer under PHRSTS than TD? 110
5.1.3 Rules of discharge for Low, High and Flood flow periods 111
5.1.4 Initial allocation of salt credits 112
5.1.5 Rules of credit trading 115
5.1.6 Monitoring and reporting 116

5.2 Operation and implementation of the PHRSTS 117
5.2.1 Occurrences of river flow periods 118
5.2.2 Salt load discharge 120
5.2.3 Transaction costs 122
Chapter 6 Examination of the Environmental Effectiveness of the PHRSTS

6.1 Introduction

6.2 Comparison of the monthly mean EC and flow before and after the PHRSTS
   6.2.1 Exploratory analysis: normality and autocorrelation of the data
   6.2.2 Graphical comparisons
   6.2.3 Exceedances

6.3 Regression models with autocorrelated errors
   6.3.1 Methodology
   6.3.2 Fitted models for monthly mean river salinity (EC)
   6.3.3 Fitted model for monthly mean river flow (Flow)
   6.3.4 Sensitivity analyses
   6.3.5 Comparison of the monthly mean river salinity in Low flow periods

6.4 Comparison of the environmental performance with versus without-PHRSTS using daily observed data
   6.4.1 Exploratory analysis
   6.4.2 Relationship between ec, ln(flow) and ln(d+1)

6.5 Estimation of environmental benefit

6.6 Summary

Chapter 7 Examination of the Economic Effectiveness of the PHRSTS

7.1 Theoretical model for estimating saline water control costs of four regulatory schemes
   7.1.1 Hypothetical regulatory schemes
   7.1.2 Storage costs for coal mines
   7.1.3 Treatment costs of power stations
   7.1.4 Minimisation of total control costs

7.2 An operational model for estimating the cost saving of the PHRSTS over the TD
   7.2.1 Cost savings of the PHRSTS for the coal mines
   7.2.2 Cost savings of the PHRSTS for the power stations
   7.2.3 Cost saving of the PHRSTS from the new development
   7.2.4 Total saving in control cost and its significance

7.3 Summary

Chapter 8 Conclusions

8.1 Main findings and contributions of this thesis

8.2 Recommendations for the FHRSTS

8.3 Prospects for future study
Appendices

Appendix 1 List of the HRSTS stakeholders consulted for this study 256
Appendix 2 Available data and their sources for this study 257
Appendix 3 System of coal-fired power stations 258
Appendix 4 Examination of incremental discharge 261
Appendix 5 Summary of credits held by the PHRSTS participants 263
Appendix 6 Example of discharge controlled by credits 264
Appendix 7 Simple statistical methods without autocorrelation 266
Appendix 8 Models considering foresight and transitional effects 269
Appendix 9 Parameters of theoretical models and their data availability 271
Appendix 10 Approaches to estimation of marginal abatement cost functions for individual firms 274

References 281

List of Tables

Table 1.1 Formats of discharge permits 3
Table 1.2 Notations and timeframes of systems for regulating Hunter River salinity 5
Table 2.1 Comparison of static and dynamic permits 30
Table 2.2 Constraints under the four different permit schemes 46
Table 3.1 Basic data on PHRSTS mines (the main mines of the Hunter coalfield) 65
Table 3.2 Mine water management strategies for the coal mines in the Hunter Valley 72
Table 4.1 Classification of the water quality management strategies for the Hunter River 91
Table 4.2 Evolution of license conditions under the Trickle Discharge system 92
Table 5.1 Summary of PHRSTS salinity targets and discharge rules 108
Table 5.2 Initial allocation of credits 114
Table 5.3 Comparison of predicted and actual daily river flows at Singleton gauging station 119
Table 5.4 Distribution of river flow periods against years 120
Table 5.5 Actual salt discharge versus allowable salt load discharge 121
Table 5.6 Sharing of transaction costs under the PHRSTS 123
Table 5.7 Trading records in 1995-2001 125
Table 5.8 Information about the PHRSTS credit trading participants 128
Table 5.9 Comparison of attributes of various auction approaches 134
Table 5.10 Environmental trading programs and their auction formats 135
Table 5.11 Summary of operational information about 2004 and 2006 auctions 138
List of Figures

Figure 2.1 Causal chain of discharge 17
Figure 2.2 Pollution control target 18
Figure 2.3 Dynamic pollution control target 19
Figure 2.4 Discharge permits under the static permits system and the dynamic permits system 22
Figure 2.5 Discharge permits and ambient water quality under the trigger value system 29
Figure 2.6 Economic and environmental performance of dynamic and tradable permits 47
Figure 3.1 The Hunter River Catchment 53
Figure 3.2 Distribution of gauging stations in the Hunter Valley 58
Figure 3.3 Daily river flow at the Hunter main gauging stations 59
Figure 3.4 Causes of river salinity 61
Figure 3.5 Comparison of open-cut and underground coal mining methods 64
Figure 3.6 Distribution of the coal mines in the Hunter Coalfield 66
Figure 3.7 Generation of open-cut mine water 70
Figure 3.8 Water use and wastewater discharge of an open-cut coal mine 75
Figure 5.1 Social costs of elevated salinity in the Hunter River 111
Figure 5.2 Lifespan of credits 136
Figure 5.3 Price signals of the 2004 FHRSTS auction 149
Figure 5.4 Price signals of the 2006 FHRSTS auction 149
Figure 6.1 Evidence for reduction of river salinity at Singleton cited by the NSW EPA 154
Figure 6.2 Monthly mean of river salinity at Singleton 1980-2002 158
Figure 6.3 Data distributions for EC and ln(EC), monthly mean, 1/1980-11/2002 159
Figure 6.4 Data distributions for Flow and ln(Flow), monthly mean, 1/1980-11/2002 160
Figure 6.5 ACF plots for ln(EC) and ln(Flow), monthly mean, 1/1980-11/2002 162
Figure 6.6 Box plots for ln(EC0) and ln(EC1), monthly mean, 1/1980-11/2002 163
Figure 6.7 Box plots for ln(Flow0) and ln(Flow1), monthly mean, 1/1980-11/2002 164
Figure 6.8 Comparison of the exceedances of river salinity 165
Figure 6.9 Comparison of the exceedances of river flow 167
Figure 6.10 ACF and PACF of ln(EC).i.m1 and ln(EC).ar1.m1 172
Figure 6.11 Cumulative periodograms of ln(EC).i.m1 and ln(EC).ar1.m1 173
Figure 6.12 Residual plots of ln(EC).ar1.m1 174
Figure 6.13 Fitted lines of AR(1) models of ln(EC) 175
Figure 6.14 ACF and PACF for ln(Flow).i.m1 and ln(Flow).ar1.m1 177
Figure 6.15 Cumulative periodograms of ln(Flow).i.m1 and ln(Flow).ar1.m1 178
Figure 6.16 Residual plots of ln(Flow).ar1.m1 178
Figure 6.17 Fitted lines for AR(1) models of ln(Flow) 179
Figure 6.18 Distribution of ec and ln(ec), daily river salinity, 1/1995-11/2002 184
Figure 6.19 Distribution of flow and ln(Flow), daily river flow, 1/1995-11/2002 185
Figure 6.20 Distribution of d and ln(d+1), daily industrial salt discharge, 1/1995-11/2002 185
Figure 6.21 ACF and PACF of ec and ln(ec), daily river salinity, 1/1995-11/2002 186
Figure 6.22 Relationship between ln(d+1) and ln(flow), daily data, 1/1995-11/2002 188
Figure 6.23 Relationship between ec and ln(d+1), daily data, 1/1995-11/2002 189
Figure 6.24 Flow and ec at Singleton, daily data, 1/1995-11/2002 190
Figure 6.25 Relationship between ec and ln(flow), daily data, 1/1995-11/2002 191
Figure 6.26 Cumulative periodograms of fitted models for ec 193
Figure 6.27 Observed ec of the PHRSTS versus stimulated ec of the TD 196
Figure 6.28 Exceedance curves for ec under the PHRSTS and under the TD 197
Figure 7.1 Data availability of the theoretical model for the coal mines 217
Figure 7.2 Assumed and actual total capital cost functions of storage construction 236

List of Boxes

Box 1.1 NSW EPA's claim on the success of the PHRSTS 6
Box 7.1 NSW EPA's claim on the PHRSTS contribution to the new development 241
Chapter 1  Introduction

1.1  Management of river quality

Management of water quality in rivers is recognised as an important and complex issue throughout the world, including Australia. A river's agricultural, industrial, ecological, aesthetic and recreational uses are all affected by the physical, chemical and biological aspects of river water quality (called “river quality” hereinafter). The strategies for managing river quality raise three main issues. First, how can a desired quality of water be determined? As a part of this problem, the degree of control of pollutant discharges needs to be decided. Second, how can a given level and pattern of water quality be achieved at a minimal cost? There is a wide variety of strategies potentially available to improve water quality. To find an optimal combination of strategies to achieve a given level and pattern of water quality requires comparison of the costs associated with impacts and control of pollutant discharges under various strategies. Third, what institutional capability and organisational arrangements for water quality management are needed? Defining the level of water quality to be achieved and the strategies to achieve it will be of no avail if there is no adequate institutional and organisational framework to implement the system and carry out the spectrum of activities involved in water quality management (Kneese and Bower 1968).

This study focuses on the second issue of water quality management, that is, to achieve a predetermined level of water quality at the minimal cost. It focuses particularly on discharges (effluents) of a conservative, dilutable pollutant, defined as a pollutant that does not reduce in mass on entering a river system and whose ambient concentration in the river is negatively related to the river flow, other things being equal. The study also focuses particularly on point sources of pollutant discharge, such as an industrial facility which may have a few outlets (“points”) which can be easily identified and monitored, rather than nonpoint sources associated with agriculture and other large-scale land uses. These issues are explored in both a theoretical and an empirical way for the case of industrial discharges of salt into the Hunter River, which flows to the central coast of New South Wales (NSW) in eastern Australia.

It is not possible to introduce even basic aspects of managing river water quality without briefly explaining some technical terms, even though full definitions must wait until Chapter 2. In the
Chapter 1 Introduction

literature on river quality management, many frequently-used terms are ambiguous. For example, “standards” could refer to ambient concentrations of pollutants present in receiving water bodies, or to pollutant concentrations in discharges. Moreover, many alternative phrases exist for the common terms in use. For instance, “standards” can also be “targets” or “limits” or “constraints” or “prescriptions” or “specifications”. These terms therefore need to be clarified. In addition, many frequently-used phrases like “pollution control target” and “environmental objective” are also ambiguous in the literature and thus need to be clearly defined as well.

In this study, water quality standards refer to desired ambient concentrations of pollutants in rivers, and effluent standards refer to desired concentrations of pollutants in effluent discharges. Correspondingly, environmental objectives are applied to river water quality (in a form of water quality standards). The term control is applied to pollutant discharges, in order to achieve a target defined in numerical terms, usually a discharge permit (in, for example, kilograms per day or tonnes per year), mostly to fulfil some prescription or specification usually expressed in the form of a statement.

How pollutant discharges affect ambient concentrations in a river is a complex question determined by not only the pollutant loads entering the river, but also the hydrological, chemical and biological conditions of the river. In theory, either controlling pollutant discharges to the river, or modifying river conditions, or both, can achieve water quality standards of a river. In practice, however, controlling pollution discharges into the river is far more feasible than modifying river conditions. For this reason, controlling pollutant discharge is the dominant strategy used for managing river water quality.

Control of pollutant discharge is often achieved using discharge permits, which set limits on pollutant load in discharges. Discharge permits can be defined in two dimensions. One is dynamicism, defining whether the total permitted discharge is changeable or unchangeable over time in response to river conditions (Kneese 1964; Reheis et al. 1982; Eheart et al. 1987; Noss and Gladstone 1987). The other is tradability, defining whether individual permits are tradable or non-tradable among different dischargers (Dales 1968; Montgomery 1972). Combining these two dimensions forms four schemes of discharge permits, shown in Table 1.1.
Table 1.1  Formats of discharge permits

<table>
<thead>
<tr>
<th>Scheme</th>
<th>Non-tradable permits</th>
<th>Tradable permits</th>
</tr>
</thead>
<tbody>
<tr>
<td>Static permits</td>
<td>Static non-tradable permits</td>
<td>Static tradable permits</td>
</tr>
<tr>
<td></td>
<td>(SN)</td>
<td>(ST)</td>
</tr>
<tr>
<td>Dynamic permits</td>
<td>Dynamic non-tradable permits</td>
<td>Dynamic tradable permits</td>
</tr>
<tr>
<td></td>
<td>(DN)</td>
<td>(DT)</td>
</tr>
</tbody>
</table>

The term "dynamic tradable permits" (DT) used in this study is therefore different from the idea of "dynamic tradable quotas" cited in the mainstream of literature on emission trading, such as Hagem and Westskog (1998), Ermoliev et al. (2000) and Armstrong (2008). The former means regulatory changes in the total number of permits issued at any time in response to rapidly changing environmental conditions, as often occur in rivers; an alternative name is flow-variable permits (Noss and Gladstone 1987). The latter, mainstream usage, also called intertemporal emission trading (Rubin 1996), means permit banking and borrowing for pollutants (for example, SO₂ and CO₂) which are long-lived in the environment, and hence diffuse very widely and accumulate steadily (discussed further in Section 2.3.5.3).

In comparing the above permits schemes, this study focuses on two kinds of performances: environmental performance, as measured by water quality standards attained; and economic performance, as measured by the total control cost. As will be elaborated in Chapter 2, any move from SN to one of ST, DN or DT is in theory a move to a control system either with a higher environmental performance, as measured by a fall in ambient concentration of the pollutant for example mg/L, called environmental effectiveness; or with a higher economic performance, as measured by a fall in control cost in for example $/year, called economic effectiveness or cost effectiveness. DT is expected to be, at least in theory, more cost-effective than either ST or DN alone.

Although conceptually appealing, DT has attracted little theoretical analysis compared to the prolific research on ST. Empirical studies of the DT system are also limited due to its few practical applications. As one of its research tasks, this thesis fills in this gap to some extent by constructing an analytical framework to compare various discharge permit schemes from both environmental and economic perspectives. This framework enriches the theoretical foundation of DT as the most cost-effective river pollution control scheme.
1.2 Pilot Hunter River Salinity Trading Scheme

This study sheds light on the topic of dynamic tradable permits (DT) through study of a real case, the pilot Hunter River Salinity Trading Scheme (PHRSTS). The PHRSTS was introduced by the NSW Environmental Protection Authority (EPA)\(^1\) in 1995 to manage the discharge of saline water from coal mining and power generating industries to the Hunter River. It was Australia’s first active water quality trading program and still appears to be the only DT scheme operating in the world. It makes an ideal setting for an illustrative case study of how various economic instruments can be integrated for management of the water quality of conservative pollutants from industrial point sources for a river system.

The Hunter River (see Figure 3.1) drains the largest coastal catchment in NSW and supports a diverse and productive industrial and agricultural economy and urban activities (HVRF 2003). Like most of Australia’s rivers, the Hunter River presents highly uncertain and variable river flow conditions (Day 1986). The salinity, the ambient concentration of salt, in the River is naturally high due to the catchment’s geological features. Salinity concentrations have a generally inverse relationship with river flows and progressively increase along the river downstream (NSW DLWC 2000).

Coal mining and coal-fired electricity generation are the two most important industries in the Hunter Region in terms of their contributions to the national and local economy and employment (HVRF 2003). They are also the primary point sources discharging saline water into the Hunter River. The coal mines accumulate mine water as they operate, and the power stations concentrate natural salts in their waste water in the production of steam for electricity generation (NSW EPA 1994a). Because discharges of the industrial saline water to the Hunter River at low river flows significantly elevate river salinity and in turn affect other water uses of the River, the key issue in managing industrial discharge is the timing of the discharge in relation to river flow, in addition to the amount of discharge (Croft & Associates 1983).

For about 20 years, starting in the 1970s, the NSW EPA used a traditional licencing system, known as Trickle Discharge (TD), to regulate industrial saline water discharge. Having evolved considerably over time, TD was not a particularly coherent system, but it was predominantly a

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\(^1\) Its name then changed to NSW Department of Environment and Conservation. It is now called NSW Department of Environment and Climate Change. To avoid confusion, NSW EPA will mostly be used throughout this thesis.
static non-tradable permit scheme (SN) in nature. Allowing industries to continuously discharge small quantities of saline water, but with little regard for river flow conditions and cumulative effects of salinity, the TD system resulted in long-standing conflicts over water use between the industries and the community because of the salinity cost to the community (NSW EPA 1994a; Smith 1995).

The NSW EPA introduced a pilot stage of the HRSTS in 1995, aiming to “manage saline water discharges so as to minimise impacts on irrigation, other water uses, and on the aquatic ecosystems of the Hunter River catchment, at least overall cost to the community; in an equitable and flexible manner; in a way that provides ongoing financial incentives to further reduce pollution” (NSW EPA 1995:1). It was developed on the basis of a number of historical strategies for management of water quality in the Hunter River.

The NSW EPA then formalised the Scheme, that is, made it permanent, on 1 December 2002 through the Protection of the Environmental Operation Regulation 2002 (the Regulation). In this thesis, FHRSTS denotes the formalised Hunter River Salinity Trading Scheme, as distinct from the pilot Scheme denoted PHRSTS. The following notations in Table 1.2 are used to indicate the key systems that have been applied in regulating industrial discharge of saline water into the Hunter River.

**Table 1.2 Notations and timeframes of systems for regulating Hunter River salinity**

<table>
<thead>
<tr>
<th>Notation</th>
<th>Timeframe</th>
</tr>
</thead>
<tbody>
<tr>
<td>TD (Trickle discharge)</td>
<td>1980-30/12/1994</td>
</tr>
<tr>
<td>HRSTS (Hunter River Salinity Trading Scheme)</td>
<td>1/1/1995-30/11/2002</td>
</tr>
<tr>
<td>PHRSTS (pilot HRSTS)</td>
<td>1/12/2002 – current</td>
</tr>
</tbody>
</table>

As explained in detail in Chapter 5.1, under both the PHRSTS and FHRSTS, discharges of industrial saline water to the Hunter River are prohibited when the river flow is in its Low period. Discharges are allowed at times of High and Flood flows when the salt dilution capacity of the river is greater and demands by and impacts on other uses are lower. In the High flow period, the total quantity of salt discharge permits is dynamically determined by the river flow conditions and in-river salinity levels, and thus varies on a daily basis. Furthermore, the discharge permits in the form of salt credits representing one-time right to discharge saline water into a certain water block of the Hunter River are tradable among industries, subject to
river salinity objectives being met. In the Flood flow period, salt discharges are effectively unlimited providing the maximum volumetric discharge rates set in discharge licences are met (NSW EPA 1995; 2002). Combining both dynamic and tradable dimensions of discharge permits, the Scheme is a unique DT program not found elsewhere in the world.

As one of the key changes accompanying the move to the FHRSTS in 2002, the salt credits that were initially allocated to the industries free of charge under the PHRSTS were re-distributed through periodic public auctions (NSW EPA 2002). Note that differing from the salt credits on the trading market, the salt credits put on auctions represent an ongoing right to discharge saline water into the Hunter River over a ten-year period. Three auctions were conducted in 2004, 2006 and 2008 respectively (NSW EPA 2004; NSW DEC 2006; NSW DECC 2008): this study examines the first two.

1.3 Research justification

The NSW EPA has claimed the success of the PHRSTS on a number of occasions, noticeable in its publications, such as NSW EPA (2001) and NSW EPA (2003a), and on its official website. One of the NSW EPA’s statements in regard to the success of the PHRSTS is provided in Box 1.1.

Box 1.1 NSW EPA’s claim on the success of the PHRSTS

“The Scheme has allowed major industries (coal mining and power generation) in the catchment to continue to discharge saline water to the river on a managed basis, and so has reduced the significant costs of water storage or treatment that would otherwise have been incurred by those industries under the previous discharge management system. It has also allowed new facilities to be established within the catchment, which may not have been possible under the previous management system.

At the same time, the Scheme has protected the environment. Since the Scheme’s commencement, the frequency with which the salinity target at Singleton was exceeded decreased from 33% before the Scheme to around 4% currently. None of the current exceedances are due to discharge by licensees.”

Source: NSW EPA (2001:v)
Although anecdotal evidence has been provided by the NSW EPA (2001), there has been no rigorous examination so far of the PHRSTS's performance to substantiate these statements. Because it is widely known as the first active trading program in Australia's history of water quality management, the PHRSTS has been frequently cited in the literature (for example, Brunton 1999; OECD 1999; Young and Hayes 2001; Action Salinity & Water Australia 2002; Whitten et al. 2003; Kraemer et al. 2004; US EPA 2004; Henderson and Norris 2008) as one of the few successful examples of effluent trading programs. Moreover, almost all of these reports cite the above-mentioned publicly available reports of NSW EPA and its website. However, no independent, critical attempt has yet been made to verify the NSW EPA's claim.

Evaluation of the actions and outcomes of public policy comes within a field of learning referred to as "evaluation research" (Robson 2002). It is a necessary exercise in policy analysis; evaluation is an important part of the policy cycle, the process where policy is created, implemented, evaluated and evolved (Robson 2002). An important consideration in any environmental policy mechanism is the ability to evaluate and monitor actions and outcomes, and to see how they achieve their objectives (Tietenberg and Johnstone 2004). The absence of ex-post measurement and accountability processes of the PHRSTS could limit policy learning and evolution.

This study attempts to fill in this gap through a comprehensive evaluation of the operational performance and actual experience of the PHRSTS. To this end, the study provides the interim feedback necessary to shape the evolution of the Scheme over time. In a broader sense, the study enriches our understanding of applications of water quality trading in a way that is relevant and would be beneficial to other potential tradable permit programs in the areas beyond water quality management. Moreover, the insights derived from this study on the relative merits of tradable permit system in terms of environmental effectiveness and cost effectiveness highlights the potential benefits of integrating various regulatory instruments for managing natural resources and environmental quality.

1.4 Research questions

Evaluating the operational performance of the PHRSTS should at least involve analysing how well the PHRSTS had achieved its objectives. However, while the objectives of the PHRSTS were wide and ambitious, they seem somewhat ambiguous and therefore hard to measure. For
example, how can one measure whether the PHRSTS achieved one of its principle objectives, “to minimise impacts on irrigations, other water uses and on the aquatic ecosystems of the Hunter River catchment” (NSW EPA 1995)? How can another principle objective of the PHRSTS, that is, minimising impacts “at least overall cost to the community” (NSW EPA 1995), be measured, while the baseline control system against which the PHRSTS is compared was not explicit in the statement of its objective?

An important and complex initial issue for this thesis was thus choosing the criteria against which to evaluate the PHRSTS. A number of criteria can be used to evaluate an environmental program. These criteria include, but are not limited to, environmental effectiveness, cost effectiveness, dynamic incentives, enforceability, equity, flexibility and long-term effects (Environment Australia 1997; Dietz and Vollebergh 1999; Perman et al. 2003; Tietenberg and Johnstone 2004). This study focuses on both the environmental effectiveness and economic effectiveness of the PHRSTS. The indicator for evaluating the environmental effectiveness is improvement in the river salinity level in the Hunter River. The indicator for evaluating the economic effectiveness is the saving in total control cost of saline water to the PHRSTS participants. Accordingly, two research questions of this study were developed:

(1) Did the PHRSTS significantly improve the river salinity of the Hunter River?

(2) Did the PHRSTS generate substantial saving to its participants in control costs?

In order to address these research questions, the following specific research activities were conducted:

For the conceptual analysis, this study

- reviewed two basic dimensions of discharge permits for rivers – dynamicism and tradability;

- conceptualised DT building on these two basic dimensions; and

- constructed an analytic framework comparing both environmental and economic performances of various discharge permits schemes.
For the empirical study, this study

- investigated the origins, history and evolution of the PHRSTS;
- reviewed the design features of the PHRSTS;
- examined the operation and implementation of the PHRSTS;
- examined the credit trading market under the PHRSTS and its implications for the two credit auctions in 2004 and 2006 under the FHRSTS;
- compared the river salinity at Singleton gauging station under the PHRSTS to that under the TD; and
- estimated actual savings in control cost generated by the PHRSTS over the TD system.

### 1.5 Research methodology

This study adopted the research philosophy and methodology of a “real world enquiry” espoused by Colin Robson (2002). It seeks to say something sensible about a complex and interactive situation in the real world. This research cannot be conducted in a laboratory, but a scientific attitude is adopted through conducting the research systematically, skeptically and ethically for the purpose of seeking the “truth” about the subject. Particular attention is given to avoiding prejudice and reaching conclusions that are objectively justified. The methodological approaches used in this study are wide-ranging, being a mix of qualitative and quantitative approaches, as follows.

Critical review and analysis of literature (as detailed in Chapter 2) has been used to construct the theoretical framework of this study. Both algebraic and diagrammatic models are developed to examine the cost effectiveness of DT in controlling pollution.

A systematic search for relevant information and a synthesis of this information formed part of this study to investigate the evolution of strategies for management of river salinity in the Hunter River and the origins of the PHRSTS. As part of information gathering, this study involved site visits and extensive consultations with the HRSTS stakeholders (see Appendix 1) in the form of interviews, target group meetings and discussions and electronic communications. This study uses both primary and secondary information, sourced from
publicly available documents (for example, published academic papers, government reports and industrial reports) and confidential documents (for example, consultant reports, firms’ records and government financial reports).

Quantitative analysis is the dominant approach in this study in examining the environmental and economic performances of the PHRSTS. Both the methodologies used in the qualitative and quantitative analyses were largely determined by the availability of data and the accuracy of the available data. Lack of data is a key, uncontrollable problem in this study. The main points concerning methods and data in this study are summarised here. The data used in quantitative analysis, including their availability and sources are detailed in Appendix 2. The availability of the data and their implications for each quantitative approach are discussed fully where appropriate in Chapters 5, 6 and 7.

The daily flow and salinity monitoring data are available for the PHRSTS period (from 1/1995 to 11/2002), but only monthly average data (with a number of missing data) are available for the TD period (from 1/1980 to 12/1994). These data problems could affect the accuracy of the comparisons of the river flow and salinity between the PHRSTS and the TD, and as well as the accuracy of modelling of the relationship of flow, discharge and river salinity.

Lack of data was also the most difficult technical barrier to the application of the purpose-built theoretical model to compare the total control costs under various real and hypothetical discharge permit schemes. Data on industrial discharges at either aggregate or individual level for the TD period are not available. Information on the industrial control costs for either the TD or the PHRSTS period is limited. The trading prices of salt credits under the PHRSTS are also not available. Due to these and other data problems, even the operational model, which has fewer data requirements, can only make a rough estimate of the cost saving generated by the PHRSTS. Therefore, the findings of this analysis should be treated as indicative but still capable of yielding justifiable conclusions, rather than as being quantitatively accurate.
1.6 Structure and contributions of this thesis

The thesis consists of eight chapters including this Introduction chapter, followed by ten appendices (see Table of Contents).

Chapter 2 first reviews the literature on theories, designs and practices of discharge permits in two separate domains: static versus dynamic permits and tradable versus non-tradable permits. This review facilitates the positioning of DT within established disciplines and theoretical discussions. Following the overview, I construct a theoretical framework for comparing various discharge permits schemes from both environmental and economic perspectives. The main contributions of this chapter are building bridges between two basic dimensions of discharge permits, dynamicism and tradability, and enriching the theoretical foundation of DT as the most cost-effective river pollution control strategy available at present. The analytical framework will inform other empirical studies of strategies for river quality management where data are available.

Chapter 3 provides a general background of the Hunter River, including the main features of its river flow and salinity. The focus is on the factors affecting the generation of industrial saline water and those determining the impacts of industrial discharges on river salinity. This helps provide a better understanding of the nature and scale of industrial saline water discharges and their implications for the development of the strategies for managing the river salinity in the Hunter.

Chapter 4 provides an overview of the historical strategies for the management of salinity in the Hunter River before the inception of the PHRSTS. A number of important strategies and events are described in detail. This chapter presents the findings from the collation of unpublished literature and several field trips. These trips involved consulting widely with people from a number of agencies, organisations, institutions and local industries who participated in the initiation, development and implementation of the PHRSTS. This chapter provides the basis for a thorough understanding the origin and institutional arrangement of the PHRSTS. It also identifies alternative management strategies against which the PHRSTS can be compared.

Chapters 5, 6 and 7 form the core of this thesis. Together they provide the first comprehensive quantitative assessment of many aspects of the PHRSTS, in particular, its environmental and
economic performances. Quantitative measurements are therefore the main outputs of these chapters. The findings of these chapters not only enable an overall conclusion on the PHRSTS performance to be reached, but also contribute to ongoing improvement of the FHRSTS operation. In addition, the methodologies described in these chapters offer a useful reference for other research on river quality management, in particular, the methodology used to estimate the cost saving for the coal mines can easily be extended to water quality research on storable pollutants.

Chapter 5 starts with a description and interpretation of the main mechanisms of the PHRSTS. Examination of the operation and implementation of the PHRSTS then follows. This chapter also undertakes a preliminary analysis of the credit trading markets under the PHRSTS and the two credit auctions held in 2004 and 2006 under the FHRSTS. Findings from this examination and analysis are then used to proceed to and interpret the environmental and economic analyses in Chapters 6 and 7.

Chapter 6 examines the environmental performance of the PHRSTS. It addresses this first research question - Did the PHRSTS significantly improve the river salinity of the Hunter River? A hydrologic and water quality analysis is conducted from both the before-vs-after PHRSTS perspective and the with-vs-without PHRSTS perspective. From the before-vs-after PHRSTS perspective, the long-series (from 1980 to 2002) monthly river salinity and flow between the TD periods (from 1/1980 to 12/1994) and the PHRSTS periods (from 1/1995 to 11/2002) are compared using advanced regression models that account for the time series effect in the data. From the with-vs-without perspective, a relationship of river salinity, river flow and industrial salt discharge is developed using available daily observations during the PHRSTS period (from 1/1995 to 11/2002). Based on this relationship, the river salinity of the Hunter in a without-PHRSTS scenario is simulated and then compared with the observed salinity with the PHRSTS. Following this simulation, the social benefits from the improvement of the river salinity under the PHRSTS are also quantified. In addition to yielding the first set of quantitative measurements for the environmental performance of the PHRSTS, the key contribution of this chapter is the new approach to accounting for the time series effect in Hunter River quality analysis. The flow-discharge-salinity model developed in this analysis will contribute to the improvement of the ongoing FHRSTS operation by providing a more advanced model to predict the discharge events and calculate the salt discharge permits.
Chapter 7 presents an economic analysis to estimate actual cost-savings of the PHRSTS over the previous TD system. It addresses the second research question of this study – *Did the PHRSTS generate substantial saving to its participants in control costs?* This chapter starts with an innovative, purpose-built theoretical model to estimate the total control cost under various real and hypothetical discharge permits schemes, based on the theoretical framework constructed in Chapter 2. However, the absence of the required information at the individual firm level makes it impossible to proceed further. As an alternative, a simple operational model is developed to estimate the cost saving of the PHRSTS over the TD derived from its dynamicism and tradability, which are certainly the two radical elements of the PHRSTS, separately. Significance of the total cost saving to the industry is then measured. In addition to delivery of the first set of quantitative measurements of the economic performance of the PHRSTS, the key contribution of this chapter is that the methodology used in the operational model provides cost estimation for storable river pollutants, a procedure which is lacking in the literature.

Chapter 8 summarises the research findings, recommends some improvements on the FHRSTS and reviews prospects for further research.
Chapter 2  Overview of Strategies for Water Quality Management

Water pollutants from various sources have different features. The impact of wastewater discharge on river quality depends not only on the nature of the pollutant and the pollutant load that enters into the river, but also on the conditions of the receiving water. Water quality objectives for the management of a river system can be achieved by controlling pollutant discharge, or by modifying the receiving water, or by some combination of the two. The focus of this study is on river water quality and optimum measures for its management. This chapter provides a broad overview of the strategies for water quality management of river systems. It briefly summarises a voluminous theoretical and applied literature on water quality management. This review aims to be comprehensive and relevant rather than definitive and exhaustive. The purpose of this chapter is to provide a base for understanding the evolution of the strategies for the Hunter River quality management and for assessing the environmental and economic effectiveness of the pilot Hunter River Salinity Trading Scheme (PHRSTS).

The chapter starts with a short presentation of the factors relevant to river pollution (Section 2.1). These characteristics crucially influence the adoption of strategies for managing river quality. The aim of this section is to provide background to the environmental and economic aspects of river quality management. The chapter then focuses on various types of discharge permits put in place for the control of pollution sources. Section 2.2 compares static and dynamic permits, while Section 2.3 compares non-tradable and tradable permits. Section 2.4 introduces the concept of dynamic tradable permits (DT), building on dynamic permits and tradable permits. The primary purpose of this section is to introduce DT as the most cost-effective river pollution control system via both algebraic and diagrammatic models. This section also provides an analytical framework for the examination of environmental and economic performances of the PHRSTS in Chapters 6 and 7.
2.1 Factors relevant to river pollution

2.1.1 Sources

Water pollutants result from a great variety of processes. They have markedly different characteristics, produce complex changes in receiving waters and affect subsequent water uses in numerous ways. Based on the generation sources, water pollutants can be broadly classified into point-source pollutants and nonpoint-source pollutants.

*Point-source* (PS) pollutants enter into receiving waters at discrete and identifiable locations. They usually can be directly measured or otherwise quantified and their impact directly evaluated. Major point sources include effluents from industrial facilities, municipal sewage treatment plants and effluents from farm buildings or solid waste deposition sites (Krenkel and Novotny 1980). Unlike PS pollutants, *nonpoint-source* (NPS) pollutants are not generated from a discrete conveyance, but come from many different sources and locations, usually associated with rainfall or snowmelt runoff moving over the ground or groundwater discharge carrying natural and human-made pollutants into the receiving water body (Jarvie and Solomon 1998). As the NPS pollutants enter water bodies over a large area and the load is often highly variable, NPS pollutants are more difficult to detect, model and manage than PS pollutants.

2.1.2 Substances

Water can be polluted by many substances with many very distinctly different effects on aquatic ecosystems. Based on their behaviour in the waters that receive them, pollutants can be grouped into conservative and non-conservative substances.

*Conservative* substances do not degrade or change in character. Once they enter the receiving waters, they are usually diluted and may be changed in form, but they are not appreciably reduced in total mass. Therefore, given the mass of a pollutant in effluent and the volume of the effluent, the resultant concentration of a conservative pollutant over time in receiving waters is relatively easy to predict because dilution is the dominant process that occurs when the substance is dissolved. Major conservative pollutants include inorganic chemicals, such as chlorides (for example, salt), some synthetic organic chemicals and inorganic suspended solids (Kneese and Bower 1968). Unlike conservative pollutants, *non-conservative* pollutants degrade...
or change in character. They are reduced in quantity by the physical, chemical and biological
processes of the receiving water (Kneese and Bower 1968). For this reason, predicting the
temporal and spatial patterns of concentration of non-conservative pollutants is technically more
difficult than for conservative pollutants. The emphasis in this research is on a point-source
conservative pollutant – salt.

2.1.3 Water quality management and water pollution control

This review and the whole study focus on managing the water quality of conservative pollutants
from the point sources in a river system. Clarification of some of the basic terminology is a
useful starting point.

One clarification is to distinguish between water quality management and water pollution
control. Water quality is not synonymous with water pollution, and similarly, water quality
management should not be equated only with water pollution control. The following definitions
are derived from Krenkel and Novotny (1980:5): “Water quality management deals with all
aspects of water quality problems relating to the many beneficial uses of water, while water
pollution control usually connotates adequate treatment and disposal of wastewater.”

The distinction between water quality standards and effluent standards should also be made.
Water quality standards define water quality goals of a water body by designating uses of the
water and by setting the criteria necessary to protect uses of the water (for example, drinking
water, fisheries, irrigation, aquatic life and recreation). The criteria usually specify the threshold
values of pollutants allowed in the receiving water (Krenkel and Novotny 1980). Effluent
standards dictate the amount of a pollutant that can be discharged from its source. While water
quality standards are often defined as ambient concentration, effluent standards can be based on
either concentration and/or total mass discharged or on the degree of treatment required in order
to maintain a specific waste load discharge. When effluent standards are based on total mass
discharged, they are often called effluent limits, discharge permits or load targets (Krenkel and
Novotny 1980). The term discharge permits is used throughout this thesis.

The impact of an effluent on the quality of a river depends on the load of pollutants entering the
river and the conditions of the river at the time. In river quality management, waste assimilative
capacity (WAC) is a term widely used to connote the capability of a receiving water body to
assimilate a certain quantity of a pollutant under certain conditions. WAC depends upon processes within the water body that can reduce the concentration of pollutants. These processes include dilution, biological degradation, adsorption, sedimentation and volatilisation (Kerenkel and Novotny 1980). The WAC of a water body can thus be influenced by a number of factors, including the runoff or flow rate in the water, the background water quality and temperature of the water, as well as the flora and fauna in the area.

Figure 2.1 illustrates the causal chain of effluent discharge, water quality, social damage and costs. Despite information gaps and measurement errors in each step, and the level of ignorance increasing with each step, the pollution control targets may be positioned conceptually at any point in the chain. The most fundamental question is whether, in setting control targets, only damage or both damage and abatement costs should be considered.

Economists advocate that a discharge target should be set at the efficiency level of discharges, where both damages and abatement costs are considered. The efficient level is the one at which marginal social damage (MSD, cost of damage to the society caused by the incremental units of discharge) is equal to marginal abatement cost (MAC, cost incurred by an additional unit of discharge reduction), shown as D* in Figure 2.2. At this level, the sum of total abatement cost and social damage cost should be the least (Perman et al. 2003). In practice, however, more often a discharge target is set at the acceptable level based on the damage threshold alone, to assure human health and reasonable quality of environment (shown as D in Figure 2.2). It is expected that the MSD curve would rise sharply if the discharge exceeds the acceptable level. In
this case, ambient concentration standard of a pollutant, as proxy of social damages of this pollutant, is used to set control target of this pollutant.

Figure 2.2 Pollution control target

MAC = marginal abatement cost
MSD = marginal social damage

The above discussion is implicitly framed in the context of these pollutants whose damage of discharge depends directly on the level of discharge. For river pollutants, including salt in this study, the damage of discharge is a function of the ambient concentrations of the pollutants, which are jointly determined by both the level of discharge and the river conditions such as flows. While the damage is associated with the ambient concentrations, what environmental regulators typically control or regulate are effluent discharges (Perman et al. 2003: 117). Therefore, it is necessary to understand the linkage between the discharges, ambient concentrations and damage when setting control targets for river pollutants.

For a river pollutant whose social damage varies considerably with river flows, a related issue is whether or not pollution targets should be the same over time. As shown in Figure 2.3 and as will be further discussed in Section 2.2 and demonstrated by the case of the PHRSTS (see
Chapter 1), allowing the discharge targets to vary in response to river flow will ensure the same level of social damage (or ambient concentration) but at lower total abatement cost.

**Figure 2.3 Dynamic pollution control target**

![Graph showing dynamic pollution control target]

- MSD<sub>L</sub>, MSD<sub>H</sub> – marginal social cost of pollutant discharge at low and high river flows respectively
- D<sub>L</sub>, D<sub>H</sub> – acceptable levels of pollutant discharge at low and high river flows respectively
- C<sub>L,H</sub> – ambient concentration of a pollutant at low and high river flows

The process of setting of water quality standards for a river system is a task that involves several disciplines. It involves, or should involve, epidemiologists, toxicologists, chemists, hydrologists, water quality modellers, and other disciplines as required. The difficulties and the many areas of uncertainty in setting water quality standards are recognised, the resolution of which may require additional extensive data. It is not the purpose of this study to determine levels of ambient standards of pollutants for river systems. Therefore, this study will not provide in-depth background information, nor a detailed discussion of the many areas of debate associated with water quality standard setting. Nor will it address the numerous bifurcations in the process where a decision needs to be made, nor details of technical nuances for all situations that may arise when setting a standard. Rather, the study will focus on the strategies that can achieve pre-determined water quality standards at a reduced cost. This gives a simple and realistic setting in which various pollution schemes can be compared.
2.1.4 Enhancement of a river’s wastewater assimilative capacity

The load of pollutants discharged from various sources to a river and the waste assimilative capacity (WAC) of the river are the two major factors determining ambient concentrations of the pollutants. Correspondingly, a pre-determined water quality standard can be achieved by either controlling the discharge of pollutants, or modifying the WAC of the river, or by doing both simultaneously. For a river system where there is a monotonic, inverse relationship between river flow and water quality, enhancing natural WAC by flow augmentation was often recommended in some early literature (Kneese 1964; Kneese and Bower 1968; Institute of Water Resources 1973; Mills 1978) as an engineering alternative to achieving water quality standards for conservative pollutants. With this alternative, river flow is artificially increased during critical low-flow periods, which often coincide with a heavy background concentration of pollutants and/or high water temperature, to provide a WAC beyond that normally provided by natural river flow. An array of options is available to augment flow, including controlled release from reservoir storage, withdrawal of water from groundwater sources and releasing it into a surface watercourse, and transfer of water from other water sources, such as rivers and lakes (Kneese 1964; Kneese and Bower 1968; Institute of Water Resources 1973; Mills 1978).

While conceptually appealing, the operational feasibility of flow augmentation depends on a number of factors. The feasibility of flow augmentation depends critically on the availability of water, particularly during low-flow periods. Water taken from an upstream impoundment for flow augmentation incurs an opportunity cost. This opportunity cost varies between basins, and may not be easily quantified when it is derived from flood control, water supply and recreational considerations (Institute of Water Resources 1973). If opportunity costs are low, the likelihood of providing flow augmentation is enhanced. Otherwise, the option of flow augmentation is not favoured. Even so, philosophically this means that society pays the cost of controlling the pollution compared with the polluters.

The effectiveness of flow augmentation depends on both the type of waste and the type of receiving water involved. For example, with respect to organic wastes, increasing WAC by flow augmentation is effective for streams, but not for lakes, and only to a limited extent for estuaries. With respect to conservative wastes such as chlorides, however, increasing the flow into streams, lakes and even estuaries can result in a significant improvement in water quality (Kneese and Bower 1968). In cases where augmented flow is provided by controlled release
from reservoir storage, the effects of reservoir storage itself on the quality of water later released have both positive and negative aspects. While bacteriological quality tends to be stabilised in reservoir storage and summertime releases from reservoirs tend to be cooler than normal stream flows, harmful impacts of impoundment on the dissolved oxygen content of releases have been found (Kneese and Bower 1968). For these and other reasons, the application of flow augmentation in practice is limited to dilution of conservative pollutants to maintain or improve the water quality, or using the release as cooling water to maintain a desired stream temperature. A trial flow augmentation for managing salinity for the Hunter River, as well as its environmental effectiveness and operational feasibility will be discussed in Section 4.4.

While flow augmentation offers an alternative for maintaining river quality, controlling pollutants entering rivers, that is, water pollution control, is the dominant strategy for river quality management and involves actions being taken by polluters. Coincident with this strategy, management of point sources is usually achieved using discharge permits, which define the maximum amount of a specific pollutant that is allowed to be discharged from pollutant sources within a certain period of time, measured as kg/day or tonnes/day. As will be presented in the following subsections, discharge permits take various forms, but are generally defined in two basic dimensions: dynamicism, defining whether the permits are changeable or unchangeable over time, and tradability, defining whether the permits are tradable or non-tradable among different dischargers.

### 2.2 Dynamic permits

In river quality management, water quality standards serve a dual role: first, they establish water quality benchmarks that specify the limits on the effects of pollution; and second, they provide a regulatory basis for establishing water quality-based pollution control programs. For any water pollution control program, the challenge is to logically translate the water quality standards into load targets – discharge permits. Discharge permits are set by back-calculating from ambient concentration to ensure that discharges will not result in exceedance of a water quality standard at a location of reference point. Noss and Gladstone (1987) introduced a precise term - quality margin (measured by mg/L), to represent the difference between the concentration limit of a pollutant specified in water quality standards and the existing concentration of the pollutant in a river, for implementing discharge permits. The quality margin is virtually a proxy of WAC (measured in tonnes/day) and is generally an increasing function of river flows. It is used here to
help convey the distinction between static permits and dynamic permit in terms of the degree of use of WAC.

2.2.1 Static permits versus dynamic permits

Under a static permits system, the effluent limits for a particular pollutant are determined by a steady state analysis of worst-case conditions of the receiving water body. These worst-case conditions are usually specified in terms of a critical volumetric flow, velocity of flow, water temperature or/and concentration background of pollutants. Furthermore, once this single basic permit has been determined, it is enforced at all times without taking account of the temporal variations in the quality margin of the river (shown as Figure 2.4). The underlying philosophy of this system is that the discharge permits based on the worst conditions ensure that the ambient concentration of a pollutant is always below the threshold of water quality standards or exceeds it only over a certain limited, acceptable percentage of the time.

Figure 2.4 Discharge permits under the static permits system and the dynamic permits system

For most rivers, river flow is the most important physical parameter affecting the capacity of a river to assimilate conservative pollutants. Poor water quality usually occurs in drought periods due to insufficient flow to dilute the pollutants. Droughts are thus critical for defining the waste assimilative capacity (WAC) of streams for various pollutants. In pollution control programs, the lowest seven-day average flow with a recurrence interval of ten years, denoted as $7Q_{10}$, is used as a prevailing design flow for total allowable discharge permits (Krenkel and Novotny 1980), and $7Q_{10}$, which is often exceeded about 97-99% of the time on unregulated rivers, is
regarded a fairly conservative design parameter. Studies (Reheis et al. 1982; Noss and Gladstone 1987) have shown that discharge permits based on 7Q10 flow could ensure water quality remains above specified levels almost all of the time.

Dynamic permits, as the name suggests, allow a variable quantity of pollutant to be discharged at any point in time depending on the river conditions. (As discussed after Table 1.1, there is an alternative, intertemporal meaning of "dynamic permit" which we do not use here.) The concept behind this method is to induce maximum use of quality margin by allowing more discharges during times when WAC of a river is relatively large. The idea of dynamic permits seems to have been originated by Kneese (1964) as an engineering approach instead of a cost-effective pollution control instrument. Nevertheless, it did not receive adequate academic attention until the 1980s. A search of the literature does not reveal any further studies of dynamic discharge permits in recent years. Thus, the literature reviewed in this section is mainly the work done in the 70s and 80s. While a limited number of early studies focused on simulation and design aspects (Reheis et al. 1982; O’Neil 1983a; Eheart et al. 1987; Noss and Gladstone 1987), few provided rigorous theoretical justification for the cost saving potential of the dynamic permits.

Eheart et al. (1987) grouped dynamic permits into two forms: periodic permits and conditional permits. Periodic (dynamic) permits are also called seasonal permits, timing discharge, phasing discharge or staged discharge in the early literature (Kneese 1964; Eheart et al. 1987; Croft & Associates 1983). They regulate wastewater discharge on the basis of calendar date. In such systems, the year is divided into several periods based on a statistical analysis of the historical record of river parameters, usually flow and temperature, each exhibiting relatively constant conditions. For each period, different discharge permits are specified so that water quality standards can be met with reliable probabilities. The periodic permit is more applicable to cases where river variations are quite reliable and recur regularly over the course of an annual cycle and therefore are predictable. In its simplest form, only two periods are set: a critical period in which only low or no pollutant discharge is allowed; and a non-critical period in which higher or unlimited pollutant discharge is permitted.

Conditional (dynamic) permits are discussed broadly in the literature under the terms variable flow permits, variable effluent limits or time-varying discharge permits (Reheis et al. 1982; Eheart et al. 1987; Noss and Gladstone 1987). They differ from periodic permits in that the
quantity of discharge permitted at a time under the conditional permit system is a pre-specified function of instantaneous stream conditions. For example, for the pollutants whose concentrations highly correlate with river flows, the total discharge permits might be proportional to the instantaneous river flow.

Kneese (1964) offered a number of options for industrial plants and municipalities in response to the mechanism of dynamic discharge. For instance, firms can reschedule production in accordance with pre-programmed dynamic permits. For some firms the lowest cost solution might be to cut back production over short periods of time, perhaps by planning staff vacation schedules to coincide with typical periods of critical river conditions or even temporarily shutting down. Also, treatment processes can be designed and operated to yield varying levels of treatment of pollutants. In addition, the wastewater can be withheld in storage for some periods of time, providing the wastewater is storable without causing secondary pollution (as appears to be the case of the HRSTS. For details of the mechanisms of the PHRSTS, please see Section 5.1). Nevertheless, this option seems more applicable in non-urban areas where detention ponds could be built without great costs due to land being cheaper.

2.2.2 Environmental performance

While both well-designed static and dynamic permits can achieve water quality standards with an acceptably low possibility of non-compliance of water quality, most of the time the river water quality under static permits is superior to that under dynamic permits, and this superiority shows a positive correlative with river flows.

 Appropriately designed and effectively implemented dynamic permits can protect the river quality to the required standards all the time. However, Reheis et al. (1982) were concerned about the long-term effect of dynamic permits on water quality. While dynamic permits can reduce treatment effort without necessarily increasing the frequency of water quality violations, the cumulative mass of pollutants discharged to the receiving water is greater than for constant year-round, static permits. This may cause a negative effect on water quality from a long-term perspective.
2.2.3 Economic performance

A dynamic approach can potentially yield protective but less restrictive permit limits if a positive correlation exists between the point sources and instream dilution capability. Dynamic permits could represent significant cost savings to dischargers without compromising the attainment of water quality standards. Such cost-savings result from the less stringent limits on pollutant discharge and in turn, less pollution abatement is required during high flow conditions.

Although examples of the dynamic permit programs are rare in practice, the cost-saving potential of dynamic permits has been supported by several simulation studies, such as Yaron (1979), Reheis et al. (1982), Eheart et al. (1987), and Noss and Gladstone (1987). The general conclusion of these studies was that compared to static permits, dynamic permit programs have the potential for meeting water quality standards at a lower total control cost. The savings of dynamic permits appear to mainly come from operational and maintenance cost savings, since the capital plant installed must still be able to meet the more stringent discharge limits during some critical river conditions.

Noss and Gladstone (1987) also raised concerns about the impact of the use of quality margins for discharge on maximisation of social welfare. While a case can clearly be made for dischargers to increase discharge during periods of high flow, where excess quality margin in a river exists, this may not always maximise social welfare. Noss and Gladstone suggested that this surplus quality may be of value to other water users and the benefits to fishing, for example, may outweigh the benefits of using the excess quality for increased waste assimilation. This concern is more related to the efficiency level of discharge mentioned in Section 2.1.3. However, further discussion is outside the scope of this study.

2.2.4 Design and implementation issues

While the concept of dynamic permits is appealing, its application poses a range of design and implementation challenges, as discussed below.

2.2.4.1 Suitability

The nature of the effluent pollutant affects the application of dynamic permits. The dynamic permit approach appears to be more suitable for the management of conservative pollutants than
for non-conservative pollutants. A low flow is often selected as a critical design river flow because it is commonly thought to represent a minimally acceptable condition for the assimilative capacity of a river. The assumption that the worst water quality occurs at the lowest river flow may not always hold in instances involving multiple discharges and non-conservative pollutants. The additional dilution resulting from higher flow may be offset by adverse changes in the parameters that govern water quality and in decreased residence time, which allows the river less time to recover from the effect of one discharge before receiving another. Eheart (1988) demonstrated that for Chemical Oxygen Demand (COD) the critical pollutant impact on a river may increase through increasing the river flow. This suggests dynamic permits may have limited application in the management of river quality for non-conservative pollutants.

The characteristics of the effluent load also affect the application of static permits and dynamic permits, as well as their effect on river water quality (Butcher and Diamond 2003). The static permit approach works well when the effluent load is constant in time and independent of the dilution capacity (and other factors affecting impacts) of the receiving water. In this situation, given a constant load, the maximum impacts occur at low flows. The probability of the exceedance of water quality standards is thus a direct function of the distribution of river flows. The static permit approach can provide a simple but effective means of evaluating the total allowable permits associated with a desired frequency and duration of water quality standard exceedance. Even if the load has a high degree of variability, this can be addressed by statistical considerations in setting the permit limit.

However, when the loads are not constant, and particularly when the loads and critical conditions are not independent, but correlated, such as precipitation-driven pollution sources where the effluent load is positively correlated with dilution capacity or background concentration, the analysis becomes more complex. It may not be easy to identify the most appropriate critical conditions. Use of the simple steady-state analysis approach can result in permit limits that are either under-protective or over protective. An over-protective permit imposes unnecessary costs on the dischargers and potential economic costs on the community, but incurs no direct environmental costs. An under-protective decision reduces immediate costs to the dischargers, but can result in environmental impacts with a cost which may be difficult to express in monetary terms.
2.2.4.2 Equity issue

Dynamic permits allow discharge rates to increase and decrease according to changes in river quality associated with different flow rates and background concentrations over time. Another issue associated with dynamic permits that is rarely mentioned in the literature is equity of discharges among the sources along a river. River flow progressively increases downstream as more tributaries join the mainstream along a river. Dynamic permits tend to be biased to upstream sources on discharge rates. As the background concentration of a conservative pollutant is often higher downstream than upstream, however, there is the potential that dynamic permits are biased against downstream sources. As these two effects may often interact, the sequencing effect on the magnitudes of the biases against discharge sources can only be determined on a case by case basis for a whole river. This issue needs to be considered in the design of any dynamic permits program. As will be described at length in Section 5.1, PHRSTTS addressed this issue through using different salinity objectives and flow thresholds for different sectors of the river.

2.2.4.3 Risk

The risks associated with decisions made about abatement costs that the dischargers would bear are different under static and dynamic permits (Eheart et al. 1987). In water pollution control programs, each discharger has to face two decisions regarding abatement costs: the abatement capacity that needs to be constructed; and the intensity of operating the abatement facilities within each time period. Under static permits, because a set of implicit permits has been specified in advance, dischargers know exactly what abatement level is required for them at any time. So they have a relatively high degree of certainty when undertaking cost-efficient investment in waste treatment facilities to meet these abatement levels. Furthermore, the dischargers assume no risks are associated with unexpected fluctuations in the natural river conditions, in which water quality may become unacceptable even though the pollutant discharge below the permits limits, because their discharge rates are fixed. Instead, environmental regulators, and hence society, bear the risk (Eheart et al. 1987).

Under dynamic permits (in particular, conditional permits), however, dischargers' decisions on abatement investments have to account for uncertainties in the frequency and duration of critical river conditions. Stringent controls must also be placed on pollutant discharges under these
conditions. Dischargers have to make the trade-off between the expected abatement costs and the penalty for violating the regulations. Thence, the dischargers have to bear the risk when the water quality objectives are not met. Without a high degree of certainty in undertaking cost-efficient investments in pollution control, the dischargers may plan to respond to the more severe reduction in discharge permits by attempting to avoid detection or simply by paying the penalty, rather than by treating their wastewater at a higher level (Eheart et al. 1987).

2.2.4.4 Design and implementation

The efforts required for design and implementation are also different between static permits and dynamic permits. A significant barrier to the use of dynamic modelling techniques is the additional level of effort and expertise required. In practice, a compromise must be made between the technical accuracy and high level of effort needed for dynamic modelling approaches and the ease of application and relatively small level of effort needed for static modelling approaches. The need for short-term advance notice, sophisticated water quality modelling, and great enforcement costs may limit practical applications of the conditional permits (Eheart et al. 1987). In many cases, steady-state, design condition analyses are preferred by environmental regulators because they are simpler and quicker to implement and manage.

Noss and Gladstone (1987) proposed a trigger value approach to provide a middle ground between the efficient use of quality margins and the simplicity of implementation. In this approach, trigger values are used to define flow variable permits based on the ranges of flows rather than on individual instantaneous flows. In this case, the discharge permit would be a step function and the ambient water quality would have a saw-tooth appearance as illustrated in Figure 2.5. As will be elaborated in Section 7.2.1.1, both periodic (dynamic) permits and conditional (dynamic) permits are adopted by HRSTS for regulation of saline water discharge from industrial sources. This feature makes HRSTS a unique discharge permits program throughout the world.
Figure 2.5  Discharge permits and ambient water quality under the trigger value system

Source: Adapted from Noss and Gladston (1987)

I end this section by summarising difference between static permits and dynamic permits in Table 2.1.
Table 2.1  
Comparison of static and dynamic permits

<table>
<thead>
<tr>
<th>Characteristic</th>
<th>Static permits</th>
<th>Dynamic permits</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dynamicism of discharge permits</td>
<td>Remain unchanging regardless of changes in river conditions</td>
<td>Vary in response to changes in river conditions</td>
</tr>
<tr>
<td>Environmental performance</td>
<td>Tend to be over-protective and then create superior quality</td>
<td>With the risk of being under-protective</td>
</tr>
<tr>
<td>Economic performance</td>
<td>Unnecessary extra costs due to over-protection of water quality</td>
<td>Potential cost savings from the less operational and maintenance costs</td>
</tr>
<tr>
<td>Application conditions</td>
<td>Effluent loads are constant, and independent from dilution capacity of the river.</td>
<td>Effluent loads are not constant and correlate with dilution capacity of the river; conservative pollutants</td>
</tr>
<tr>
<td>Design condition</td>
<td>Worst-case conditions of receiving water</td>
<td>Real conditions of receiving water</td>
</tr>
<tr>
<td>Effort of design</td>
<td>Least</td>
<td>Greatest</td>
</tr>
<tr>
<td>Effort of implementation</td>
<td>Least</td>
<td>Greatest</td>
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<td>Effort of regulation</td>
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<td>Greatest</td>
</tr>
<tr>
<td>Risk bearers</td>
<td>Regulators</td>
<td>Dischargers</td>
</tr>
</tbody>
</table>

Sources: Kneese (1964); Kneese and Bower (1968); Reheis et al. (1982); Eheart et al. (1987); and Noss and Gladstone (1987).

2.3  Tradable permits

2.3.1  Tradable permits versus non-tradable permits

Traditionally, environmental regulators have relied almost exclusively on command-and-control (CAC) approaches to regulate water pollution. One of the most common and straightforward CACs to control effluent discharge from industrial point sources is non-tradable permits. Non-tradable permits are uniformly applied to all firms and are not allowed to be exchanged between them. The firms are penalised and fined for non-compliance with their individual discharge permits.
In contrast to CACs, market-based instruments (MBIs) operate through market processes or other financial incentives to make polluters voluntarily change their environmental behaviours. The most common forms of MBI for water pollution control are pollution taxes (Pigou 1920; Baumol and Oates 1971) and tradable discharge permits (Dales 1968; Montgomery 1972). Pollution taxes allow the market to determine the total quantity of pollution by means of establishing the price (via a tax, fee, charge or levy) to be paid for polluting, while tradable permits allow the market to determine the price by means of rationing total quantity of government-created tradable (transferable, marketable) discharge (emission, effluent) permits (allowance, licences, rights).

Under a tradable permits system, a total allowable discharge is established and allocated among firms in the form of permits. The firms are then allowed to use the permits to discharge or to increase effluent in exchange for equivalent or larger discharge reductions at the same or other facilities. Tradable permits have a number of economic, environmental and social advantages over the non-tradable permits. Theoretical studies (Montgomery 1972; Eheart 1980; Baumol and Oates 1988; Xepapadeas 1997) as well as practical experience (Tietenberg 1985; Hahn and Hester 1989; Tietenberg 1990; Stavins 1998; Carlson et al. 2000; Kerr et al. 2000; Sterner 2002) from tradable permit programs have proven their potential value. Tradable permits can not only achieve economically ambitious environmental goals, but also safeguard environmental quality in the face of industrial growth, as well as offering incentives for technological innovation.

2.3.2 Economic performance

The main reason underlying the cost saving potential of tradable permits over non-tradable permits is the Equimarginal Principle. According to the Equimarginal Principle, the prerequisite of attaining cost-effectiveness is to equalise MACs over all firms undertaking pollution control in a region. Theoretically, non-tradable permits could satisfy the Equimarginal Principle if all firms have the same MAC or as long as permits are chosen for each firm so that the MACs are equal over all firms (Baumol and Oates 1988).

However, in reality, the MACs of individual firms do differ, and enormous efforts are needed for collecting sufficient information on the MACs of individual firms and keeping it up-to-date. Therefore, rather than setting effluent standards for firms individually, environmental regulators
design and apply uniform effluent standards for discharge sources in an industrial sector or a region. The option of non-tradable uniform permits, which ignores differences in MACs across sources, violates the Equimarginal Principle. Therefore, while non-tradable discharge is capable of effectively achieving control targets, it cannot provide a cost-effective allocation of the pollution control burden among sources (Baumol and Oates 1988).

In contrast, a tradable permits scheme allows sources facing higher pollution control costs to meet their regulatory obligations by purchasing environmentally equivalent pollutant reductions from those sources with lower costs. To this end, tradable permits allocate pollution abatement responsibility in an efficient way: firms with relative low MACs undertake most but not all of the total abatement efforts, and firms with high MACs undertake less, as long as they collectively meet the required pollution control targets. The greater the MACs differ between sources, the more cost-saving tradable permits would achieve relative to non-tradable permits (Baumol and Oates 1988).

The cost saving potential of tradable permits has been well demonstrated by its application in air pollution control (Tietenberg 1985; Stavins 1998, 2002, 2003, 2007). The cost saving potential is also generally supported by a number of simulation studies for management of water quality (O'Neil 1983a; Lence et al. 1988; Letson 1992; Hoag and Hughes-Popp 1997; Zhang and Wang 2002; Hung and Shaw 2005), although the empirical cost saving from real application is still anecdotal.

Another economic advantage of tradable permits is their ability to manage new developments while protecting the environment, as stressed by Jarvie and Solomon (1998) and by Bjornlund (2003) in the case of irrigators. Tradable permits could provide a practical solution to the tensions caused by the need to achieve water quality objectives and the pressures for increased discharge from new developments. Under a tradable permits system new developments that wish to either create new loads or increase existing loads are allowed to purchase permits from the existing sources instead of undertaking load abatement on site. This not only ensures environmental objectives can be achieved because of no net increase in discharge of pollutant loads, but also may generate substantial cost savings for the new developments compared to having to fully abate the discharge on site. However, empirical demonstrations of cost savings for new development from permits trading are scarce.
2.3.3 Environmental benefits

As the aggregated load is limited by the total number of permits available to trade, permit trading under a cap-and-trade policy scenario can achieve the environmental performance approximating to that which can be achieved through a non-tradable permit control system. Moreover, a tradable permit system has the potential to achieve a higher environmental performance than non-tradable permits system. Once the tradable permit system is installed, firms have incentive to sell their permits and gain some revenue. This encourages industries to further adopt pollution reduction and take up innovative technologies to reduce pollution beyond current limits (Kneese and Bower 1968; Jaffe and Stavins 1995; Jung et al. 1996; Jaffe et al. 2002). In addition, tradable permits can engage more NPS in solving water quality problems, meaning tradable permits can achieve a higher environmental performance more quickly than non-tradable permits (Jarvie and Solomon 1998).

While the potential environmental benefits of tradable permits are acknowledged, it is worthwhile to note that permit trading could also result in less effective environmental performance than non-tradable permits (Ng and Eheart 2005) due to its higher technical complexity (for example, trading ratios) and managerial uncertainties (for example, implementation and monitoring). The key elements that need to be considered in the design of tradable permit programs will be presented in Section 2.3.5.

2.3.4 Tradable effluent permits versus effluent tax

In addition to being compared with non-tradable permits, tradable permits are often compared with their counterpart, effluent or pollution taxes. The comparison made between effluent taxes and tradable permits in the literature generally falls into the discussion of “control by price” and “control by quantity” (Weitzman 1974). Because pollution tax is not the purpose of this study, only a brief comparison between tradable permits and pollution taxes is outlined here for completeness.

In the theoretical setting where (1) the same amount of effluent from different sources has equal external costs; (2) there is no uncertainty about the costs and benefits of pollution control; and (3) a perfectly competitive market prevails, pollution taxes and tradable permits are equivalent. That is, they can both achieve the same level of pollution reduction with minimum levels of
aggregate abatement costs (Tietenberg 1985; Baumol and Oates 1988) and provide dynamic incentives for the innovation and diffusion of cheaper and better pollution control (Wenders 1975; Downing and White 1986; Milliman and Prince 1989; Jaffe and Stavins 1995; Jung et al. 1996; Montero 2002). These characteristics allow tradable permits and pollution taxes to achieve larger reductions in pollution than would result from CACs and provide an opportunity to address sources of pollution (for example, NPS) that are not easily controlled with traditional CACs (Hahn 1989).

Under some strong simplifying assumptions, these two kinds of instruments are symmetrical and hence economically equivalent (Pezzey 1992; Farrow 1995). However, this symmetry begins to break down when there is asymmetric information about environmental damage and abatement costs (Weitzman 1974; Adar and Griffin 1976; Fishelson 1976; Roberts and Spence 1976; Baumol and Oates 1988), in the presence of high transaction costs (Stavins 1995; Cason and Gangadharan 2003) and/or under other conditions (Watson and Ridker 1984; Moledina et al. 2003). These factors offer basic grounds for the choice between taxes and tradable permits for pollution control.

Nevertheless, there are several factors that make tradable permits superior to pollution taxes for the control of water pollution. The factors include (1) the risk of missing reduction targets appears lower for tradable permits than for pollution taxes because, by design, the tradable permits hold environmental quality constant. This is particularly important when environmental damage exhibits a threshold effect or irreversibility; (2) tradable permits have an information advantage over pollution taxes because the environmental regulators can achieve a given aggregate pollution control target cost effectively without knowing anything about an individual firm's cost of pollution control; (3) tradable permits can automatically accommodate themselves to inflation and economic growth with an increase in pollution, and in turn avoid adjustment costs arising from the iteration in charge fees; (4) tradable permits tend to address spatial and temporal dimensions of a pollution problem in a flexible way; (5) although both taxes and permit trading systems increase costs to industry and consumers, permit trading tends to make those costs less obvious to both groups and thereby may incur less political resistance; (6) tradable discharge permits allow industrial growth and economic development without compromising the protection of water quality; (7) tradable permits allow the public and environmental groups to actively participate in order to reduce pollution; and (8) effluent trading may bring otherwise unregulated pollution sources, such as NPS, under control.
Tradable permits are preferred to pollution taxes in some circumstances. In reality, the choice that ultimately determines which instrument is used to regulate pollution is influenced by many other factors. These factors include, but are not limited to, the nature of a pollutant and its geographical setting, the pre-existing regulatory context, market development, political acceptability, institutional capacity, and social equity. (Tietenberg 1985; Baumol and Oates 1988; Woodward and Kaiser 2002).

**2.3.5 Key design and implementation issues**

 Tradable permits are among the most challenging MBIs in terms of both their design and implementation. In the setting of a permits trading program, an array of elements needs to be considered, including the selection of pollutants, structure of trade, units of trade and minimum quantities, eligible traders, size of the trading area, trading ratio, monitoring requirements, banking and inter-pollutant trades, calculating loads, assuring real trades, trade approvals, enforcement and responsibility, registration system, verification of credits, and public information (National Wildlife Federation 1999; Tietenberg 2002; Ning and Chang 2007). In the remainder of this section, I will discuss in detail some key design and implementation issues associated with development of an effluent trading program, including the initial allocation of permits; environmental equivalence, such as geographic or temporal flexibility or restrictions; and transaction costs. I discuss these issues based on theoretical guidelines and empirical experience on effluent trading programs and draw conclusions regarding these issues.

**2.3.5.1 Initial allocation of permits**

Initial allocation of permits is the most important and controversial aspect in the design of effluent trading programs. Closely related to this issue is the question of whether the initial permits should be given away or sold. Under theoretically ideal conditions, the initial distribution of permits is regarded as having little effect on the equilibrium allocation of control and hence the aggregate costs of control (Montgomery 1972) as long as the permits are distributed fairly widely. However, this does not hold when there are transaction costs,
uncertainty and market power (Montero 2002). The initial allocation of permits can affect both long-run efficiency and equity of final equilibrium, as well as the speed of introduction of new production technologies and products (Stavins 1995; Kling and Zhao 2000).

Generally, the allocation of initial permits may be based on two different basic approaches: grandfathering or auction. Under a grandfathered scheme, initial permits are allocated to firms without charge, usually based on their historical pollution. Another version of the grandfathered scheme is connecting all existing discharge permits to tradable permits when the tradable discharge permits program is an “add-on” mechanism to a pre-existing non-tradable permit system. The grandfathering approach has a certain initial appeal because it appears to offer a great degree of political control over the distributional effects. This approach does not incur political resistance from the existing sources as it avoids additional financial burden on firms (Tietenberg 2002). The grandfathering approach can also provide incentives for polluters to identify themselves and report their discharges (Field 1994).

A grandfathering approach, however, may have several drawbacks. Among these is the drawback that receives most serious criticism: grandfathering creates political and economic bias against new sources entering the product market, since existing firms do not pay for their permits while new firms have to buy them (Howe 1994; Stavins 1998). One way suggested to temper the cost advantage created by grandfathering is to hold back a certain proportion of initial permits for allocation to any new firm (Hinchy et al. 1998), as it was in the case for the PHRSTS to be presented in Section 5.1.3. Grandfathering may also provide a perverse incentive to elevate current discharges or over-report discharge records if the allocation is based on historical records (Field 1994). For these reasons, in theory auctioning of permits is usually preferred to grandfathering for allocation of initial permits.

Allocation of the permits by means of a regular public auction has broadly similar economic effects to a pollution tax in the case of a competitive market (Pezzey 1992). The auction approach has a list of attractive features compared with the grandfathering approach. These include (1) delivering a signal of the scarcity of resources and embodying the polluter pays principle, in turn reducing the need for politically contentious arguments over the allocation of valuable rights; (2) helping to establish a market price for permits and therefore removing any uncertainty about the value of permits allocated free of charge; (3) eliminating bias against new sources bias; (4) providing greater incentives for innovation; and (5) providing flexibility in
distribution of costs (Hinchy et al. 1998; NSW EPA 2001; Tietenberg 2002). However, the auction approach is not perfect: it also has disadvantages. One of its significant disadvantages is that auctioned permits may add a financial burden to pollution sources and are thus prone to resistance from sources (Tietenberg 2002). This may partially explain why almost all tradable permit programs in place, including the PHRSTS, have started with the grandfathering approach (OECD 1999).

A hybrid system, which is a mix of the grandfathering and auction approaches, is often suggested for allocating initial permits. Grandfathering and auctioning can be mixed in at least two dimensions. Within a given timeframe, a certain number of permits can be distributed free of charge, and additional permits could be auctioned. Along a timeline, permits can be given away at the initial stage of a trading scheme, and then the proportion of permits auctioned can be gradually increased over time, as in the case of the PHRSTS to be presented in Section 5.1.3. A hybrid system is regarded as being superior to either the grandfathering approach or the auction approach individually. A hybrid system will weaken any cost advantages created for existing firms over potential entrants by grandfathering approach (Hinchy et al. 1998), and have a less severe cost impact on firms than the auction approach because firms will not be forced to face suddenly increasing costs in adjusting to an abatement regime (NSW EPA 2001).

In addition to the methods of allocating initial permits, the lifespan of the initial permits and how to change the number of permits as pollution policies change are other issues that need to be clearly defined in the initial allocation of permits. This has been less well researched in the literature. Permits can be issued with long or short duration. Permits with a short duration would allow the regulators more flexibility in adapting the program to new information on abatement technologies or water quality, but may generate uncertain expectations of the firms regarding the decision to invest in permits and long-lived capital facilities. Long-term permits allow firms to plan capital investments with less uncertainty and more confidence and might allow improved cost efficiency in water management (Center for Public Leadership Studies 1999). Milliman (1982) suggested that permits be issued with varying maturities when information is imperfect, particular when commencing a trading program. The number or lifetime of permits can be changed progressively once more information is available.
2.3.5.2 Spatial dimension

Compared to air emissions trading, effluent trading is significantly more complicated due to the inherent spatial issue. The environmental impact of a unit of discharge entering the river at one location is different to the impact of an equivalent unit discharged at another point along the river. Discharges further upstream may have more impact on river quality than discharges further downstream where flow is greater. Effluent trading, even though it does not result in an increase in total pollutant discharges, may affect the water quality reliability of a whole river system (Ng and Eheart 2005) or induce *hot spots* (Tietenberg 1985) where larger discharges are concentrated and in turn cause site-specific impacts. This emphasizes the need for discounting the value of the permits as a function of the location of the pollution sources.

One approach in this regard is the zonal approach (Tietenberg 1985; Hung and Shaw 2005). Under this approach, the designed trading area is divided into a number of zones. An effluent cap can be set for individual zones. *Trading ratios* are employed to indicate the number of units of a pollutant that are required to be reduced for a source in order that a single unit of the pollutant can be credited for another source. While the sources within the same zone are regarded as having a similar environmental impact on a control point and are allowed to trade on a one-to-one basis, trades between the sources of different zones are only allowed using predefined trading ratios.

Determining accurate trading ratios is not easy. It requires a solid understanding of the effects of pollutant reduction by sources at different points in relation to the river quality. This needs substantial study and could be very costly in terms of time, money and effort. Inappropriate trading ratios not only influence the environmental effectiveness of the trading programs, but also affect the economics of the trades (Jarvie and Solomon 1998). As the trade ratio increases, the lower-cost sources’ MAC rises, because more water pollution reduction is required. Trading programs are designed to bring together high and low cost sources. If a trading ratio is too high, it will lessen or eliminate the cost advantages of trading and consequently could prevent effluent trades from occurring (Jarvie and Solomon 1998).

In addition to the challenge of determining the correct scientifically-based trading ratio, the zonal approach faces another challenge of increasing transaction costs. The issue of transaction costs will be discussed at length in Section 2.3.5.4.
2.3.5.3 Temporal dimension

Permit trading between different periods of time can occur through banking or borrowing. *Banking* allows sources to save permits for future use or sale, and *borrowing* allows sources to use more permits than allowed in one period and pay them back in the future.

Kling and Rubin (1997) investigated the incentives of individual sources for emission banking and borrowing theoretically and showed that a source's choice of banking or borrowing is contingent on the conditions of the marginal abatement costs (MACs), production costs, output prices, environmental standards and the effect of discounting. In general, when MACs, output prices and allowable discharge of a source are stationary over time, the source prefers borrowing to banking as a result of the discounting effect. The higher the discount rate, the more permits the source should borrow. Nevertheless, in the situation where MACs or output prices are rising, marginal production costs are falling or the ceiling of allowable discharge is being lowered over time, sources may be prone to banking permits.

Kling and Rubin's analysis is, however, framed in the context of certainty. In reality, without information on costs, a source may bank its permits, perhaps because of the limited spatial dimension of the trading market caused by environmental constraints or slow market development (Ellerman 2002). Or a firm might use banked permits to cover a period of particularly high emission levels because of a sudden increase in the demand for output. In addition, the reasons for borrowing can include anticipated changes in production (Ellerman 2002). There is an increasing body of literature (Rubin 1996; Kling and Rubin 1997; Yates and Cronshaw 2001; Phaneuf and Requate 2002) examining the effects of inter-temporal trading from different angles. Thus, the conclusions from these studies on the desirability of allowing permit banking and borrowing are mixed.

Borrowing and banking schemes are easier to implement for air quality management as emissions (such as, SO₂ and Greenhouse Gases (GHG)) are "global" in that they disperse uniformly across large areas and there is little difference in damage per unit emission across locations and time. There is an increasing argument that in water volume trading markets, inter-temporal trading might well be more valuable than intra-season trading in terms of potential economic gains (Zekri and Easter 2005). In a regime of water quality management, however, the nature of water pollutants raises several concerns about the application of banking and
borrowing. One of the concerns is that banking and borrowing would increase the possibility of temporal uncertainty in achieving water quality objectives. Banking or borrowing can lead to some variations in effluent load over time, while the water quality standards are required to be maintained at all times. This implies the possibility that the water quality standards are overachieved or underachieved in one period and underachieved or overachieved in another. Another concern is about the implementation of sanction (Kling and Rubin 1997). This explains why few effluent trading programs have opened banking and borrowing options to firms.

2.3.5.4 Transaction costs

Transaction costs of an effluent trading program refer to those costs required to establish the program and those required to manage the program and keep it running. These costs generally fall into three main categories: searching for trading partners and acquiring information; bargaining and deciding on alternatives; and monitoring and enforcement costs (Stavins 1995). In practice, transactions costs may constitute a substantial proportion of the total costs of pollution control and become a significant barrier to trading. In the extreme situation where the transaction costs faced by the trading sources exceed the abatement-cost differential between those sources, that is, the economic benefits of trading, trades will not take place (Hahn 1989; Hahn and Hester 1989; Jarvie and Solomon 1998; Gangadharan 2000; Woodward 2003; Morgan and Wolverton 2008). Therefore, in the development of an effluent trading program, every possible effort should be made to minimise transaction costs.

One clear indication that the transaction costs are low would be that the market was quite active. Stavins (1995) noted that search and information transaction costs would be reduced in markets with a relatively large number of potential trading sources because more frequent transactions would generate more trading information. There are a number of actions that environmental regulators can take to help reduce transaction costs, such as providing an easy means for buyers and sellers to transact; providing systematic public information on trading prices; acting as a broker; and/or paying for monitoring and enforcement activities (Hoag and Hughes-Popp 1997; Jarvie and Solomon 1998; Woodward and Kaiser 2002; Morgan and Wolverton 2008).

In addition to the size of transaction costs, the spread of the transaction costs burden is another important concern, which may affect trade effectiveness. Such issues as who should bear the costs and by how much should be fully considered in the design of a trading program (Center
for Public Leadership Studies 1999). Transaction costs do not need to be entirely borne by the firms or environmental regulators. It is conceptually possible to design programs where all parties involved in trading and trading process share the costs, with the primary beneficiary sharing a substantial portion (Center for Public Leadership Studies 1999). The method of transaction cost sharing for the PHRSTS will be discussed in Section 5.2.3.

2.3.6 Experience and lessons from effluent trading

Although the theoretical merits of tradable permits have long been understood, in practice, tradable permit systems were implemented only in a few counties and with limited scope (Sterner 2002). In the US the tradable permits have received the earliest, dating back to early 1970s, and most widespread application in its environmental protection programs. These built on its pre-existing detailed systems of source-by-source regulations and individual pollution permits or licences (Stavins 1998, 2002, 2007).

Tradable permits in the US are mainly applied to air pollution control, with a clearly increasing application in water pollution control since the 1980s. Hahn (1989) and Woodward and Kaiser (2002) attributed the growing interest in the inclusion of the tradable permit concept in water pollution control to a variety of reasons. The main reasons are (1) the US EPA’s emphasis on the Total Maximum Daily Load (TMDL) programs to achieve water quality goals, which shifted the center of water pollution control from regulating pollutant loads of individual discharge sources to limiting local total pollution loads and allocating the reduction responsibilities among the dischargers; (2) the prevalence of NPS problems and the widely accepted perception that effluent trading may bring unregulated NPS under control in cost-effective ways; and (3) being promoted by the success of the Acid Rain program and the potential to substantially reduce abatement costs while still meeting air quality goals (Hahn 1989; Woodward and Kaiser 2002). In 1996 the US EPA released a guideline (that is, Draft Framework for Watershed-Based Trading) to the states to assist in evaluating and designing effluent trading programs (US EPA 1996). This not only represents a national effort to encourage and facilitate the development of watershed-based effluent trading programs, but also fosters the popularity of tradable permits in water pollution control.

By the end of last century, the US EPA had some 37 water quality trading programs that were in various stages of initiation, development, pilot and implementation stages (Environomics 1999;
Rousseau 2005). Morgan and Wolverton (2008) provided a systematic overview of water quality trading in the US based on information from a recently compiled database and updated the number of water quality programs to 58 (27 programs in operation whilst 31 in proposal). Water quality trading programs have been expanding not only in number, but also in diversity. The trading programs have expanded from individual facilities, a group of local dischargers affecting the same water body to an entire watershed or basin (Environomics 1999; Rousseau 2005; Morgan and Wolverton 2008). Participants have covered private firms, public owned treatment works, farmers, non-point sources (NPS) of cities and countries, government and regulatory agencies and non-for-profit organisations (Environomics 1999; Rousseau 2005; Morgan and Wolverton 2008). Trades are also various, not restricted to homogeneous pollutants, but include cross-pollutant trading, different pollutants which have the same environmental impacts (Lenee et al. 1988; Lenee 1991), two pollutant and bi-seasonal pollution offsets (Letson 1992) and pre-treatment trading (Environomics 1999).

Despite the various forms, water quality trading programs in the US generally fall into two distinct categories: cap and trade for existing discharges to meet limits and offset for new or expanding discharges. The market structures in pollution trading fall into four main categories: exchange, bilateral negotiations, clearinghouses and sole-source offsets (Woodward and Kaiser 2002; Woodward et al. 2002). The most common type of trading program in the US occurs between point and non-point sources, and most commonly traded pollutants are nutrients such as phosphorus and nitrogen (Morgan and Wolverton 2008).

In spite of the fact that effluent trading programs have been burgeoning in water quality management, it appears that such markets have not been as active as anticipated. According to Morgan and Wolverton (2008), by 2005 while trades had occurred in 11 out of 19 water quality trading programs in US, only four had experienced a large number of trades. The other programs had only one or two trades during more than a decade. Although the reasons for the inactivity of effluent trading in the US differ between situations, the following factors have been identified as the main causes: (1) low environmental targets and lack of trade demand; (2) government interference; (3) difficulties in setting proper trading ratios between PS and NPS and defining who is responsible for the risk of the NPS not delivering the credits that the PS purchased; (4) high transaction costs; (5) the small number of participants; and (6) uncertainties in trading policy and rules (Hahn 1989; Jarvie and Solomon 1998; Woodward 2003; Morgan and Wolverton 2008).
Chapter 2 Overview of Strategies for Water Quality Management

While there is little doubt that well designed and implemented tradable permits can provide the opportunity to achieve any given level of water pollution control with substantial cost savings, the theoretical cost saving potential of effluent trading has not materialised to the extent expected. Some of the programs provided the estimated savings of the trading programs (O'Neil 1983a, 1983b; Letson 1992; Kramer 1999; Kerr et al. 2000; Keplinger et al., 2004). These estimated savings are difficult to compare but appear to be quite substantial. However, there have been no insights into the benefits that were truly generated by the programs, nor their environmental impacts. Therefore, it is not easy to conclude whether the trading programs have been successful or not. This provides the motivation for this study to investigate the PHRSTS, in particular to provide comprehensive assessment of it from both environmental and economic perspectives.

A number of factors that may increase the likelihood of successful implementation of effluent trading have been identified from the previous research on US water quality trading program (Hahn 1989; Hoag and Hughes-Popp 1997; Jarvie and Solomon 1998; Kraemer 2003; Kraemer et al. 2004; Morgan and Wolverton 2008). Those include, but are not limited to (1) legislative or regulatory underpinning and backing; (2) clearly defined environmental targets and a need for discharge reduction; (3) accurate and sufficient information on stream characteristics, discharge sources and trading prices; (4) an adequate number of participants with diverse MACs; (5) low transaction costs; and (6) support from the local community. These factors have laid the ground rules for development of a smoothly functioning effluent trading program. Some of these factors in implementation of the PHRSTS will be discussed in Chapters 5, 6 and 7.

2.4 Dynamic tradable permits – a theoretical perspective

The combination of dynamic permits and tradable permits forms a system of dynamic tradable permits (DT). In such a system, dynamic permits provide a platform into which tradable permits may be incorporated. Because it builds on the dynamic permits and tradable permits, DT has a lot of intuitive appeal. Nevertheless, compared to the large body of literature on tradable permits and the considerable amount of literature on dynamic permits, the research on DT is limited.

Among the limited literature on DT, O'Neil (1983b) simulated the total abatement cost associated with tradable permits under varying stream flow and temperature conditions for the Fox River, Wisconsin, US. The simulation arrived at the general conclusion that worst-case-
based permit trading program is more costly than a time-variant permit trading program and indicated that the magnitude of the cost saving of the dynamic permits depends on the threshold of river flow and the temperature on which the permits are based. In addition, Dudley et al. (1993) conducted an exploratory analysis to raise awareness of potential benefits of applying DT to the management of water quality for Australian rivers, given that most of them are characterised by pervasive, highly uncertain and variable river flow conditions.

So far a limited number of studies have focused on the feasibility of applying DT in the management of river quality, but neither the theory nor practice of DT appears to have been formally researched to date. Thus, it is appropriate to consider DT theoretically before moving on to examine its application in the PHRSTS. The remainder of this subsection will provide a theoretical comparison of the environmental and economic performances of various permits schemes with the aid of an approximate algebraic model and a simple diagrammatic analysis. This comparison not only enriches the theoretical foundation of the cost effectiveness of DT, but also builds an analytical framework for examining the environmental and economic effectiveness of the PHRSTS in the later Chapters 6 and 7.

The principle assumption underlying this analysis is that for a conservative pollutant, a mixed system involving both dynamic permits and tradable permits is preferable to either dynamic permits or tradable permits separately.

Let

\[ i \]  Any firm in a region.
\[ n \]  The number of the regulated firms in the region, \( i = 1, \ldots, n \).
\[ t \]  Any day over a period of days.
\[ T \]  Number of days in the period, \( t = 1, 2, \ldots, T \).
\[ g_{it} \]  Firm \( i \)'s pollutant load that needs to be discharged on day \( t \) (tonnes).

Then, the whole industry (consisting of \( i \) firms)'s aggregated pollutant load that needs to be discharged on day \( t \) is

\[ G_t = \sum_{i=1}^{n} g_{it} \]  (tonnes)

\[ r_{it} \]  Firm \( i \)'s actual discharge reduction on day \( t \) (tonnes).
Then, the whole industry's aggregated actual discharge reduction on day $t$ is

$$R_t = \sum_{i=1}^{n} r_i \text{ (tonnes)}$$

$d_i$ Firm $i$'s actual discharge of pollutant load on the day $t$ (tonnes).

Then, the whole industry's aggregated discharge of pollutant load on day $t$ is

$$D_t = \sum_{i=1}^{n} d_i \text{ (tonnes)}$$

$p_i$ Discharge permits for firm $i$ on day $t$ (tonnes) under dynamic permits system, which vary from time to time.

The total discharge permits for the whole industry on day $t$ (tonnes)

$$P_t = \sum_{i=1}^{n} p_i \text{ (tonnes)}$$

$P_i$ Discharge permits for firm $i$ on any day under static permits system, which remain constant over time.

Then, the total discharge permits for the whole industry on any day is

$$P = \sum_{i=1}^{n} P_i \text{ (tonnes)}$$

$c_i(r_i)$ Firm $i$'s cost function of abatement activities ($$/\text{tonnes.day}).$

The total abatement cost of the whole industry under static non-tradable permits (SN), dynamic non-tradable permits (DN), static tradable permits (ST), dynamic tradable permits (DT), denoted $TC_{SN}, TC_{DN}, TC_{ST}, TC_{DT}$ respectively, is the resolution to the following equation subject to their corresponding constraints as listed in Table 2.2.

$$TC = \min \sum_{t=1}^{T} \sum_{i=1}^{n} c_i(r_{it}) \quad [2.1]$$
### Table 2.2  Constraints under the four different permit schemes

<table>
<thead>
<tr>
<th>Non-tradable</th>
<th>Static</th>
<th>Dynamic</th>
</tr>
</thead>
<tbody>
<tr>
<td>(SN)</td>
<td>$d_{it} = g_{it} - r_{it} \leq p_i \quad \forall i, t$</td>
<td>(DN) $d_{it} = g_{it} - r_{it} \leq p_i \quad \forall i, t$</td>
</tr>
<tr>
<td>(SN)</td>
<td>$d_{it} (SN) \equiv g_{it} - r_{it} \leq p_i \quad \forall i, t$</td>
<td>(DN) $d_{it} (DN) \equiv g_{it} - r_{it} \leq p_i \quad \forall i, t$</td>
</tr>
<tr>
<td>Tradable</td>
<td>(ST)</td>
<td>(DT)</td>
</tr>
<tr>
<td>(ST)</td>
<td>$D_t = G_t - R_t = \sum_{i=1}^{n} g_{it} - \sum_{i=1}^{n} r_{it} \leq P \quad \forall t$</td>
<td>(DT) $D_t (ST) = G_t - R_t = \sum_{i=1}^{n} g_{it} - \sum_{i=1}^{n} r_{it} \leq P_t \quad \forall t$</td>
</tr>
<tr>
<td>(ST)</td>
<td>$D_t (ST) = G_t - R_t = \sum_{i=1}^{n} g_{it} - \sum_{i=1}^{n} r_{it} \leq P_t \quad \forall t$</td>
<td>(DT) $D_t (DT) = G_t - R_t = \sum_{i=1}^{n} g_{it} - \sum_{i=1}^{n} r_{it} \leq P_t \quad \forall t$</td>
</tr>
</tbody>
</table>

Constraint [2.2] is set to assure that the discharge of each firm ($d_{it}$) on any day does not exceed its corresponding static permits ($p_i$), which is defined based on the worst river conditions and remains constant throughout the year. The purpose of Constraint [2.3] is the same as that of Constraint [2.2], except that the permits for a specific firm ($p_{it}$) may vary with the daily local river conditions. Constraint [2.4] and [2.5] are set to make sure that the aggregated discharge from the firms ($D_t$) does not exceed the total discharge permits ($P$ or $P_t$) on any day, although the permits under the static and dynamic permit systems are different. The relaxation of constraint that occurs moving from [2.2] and [2.4] to [2.3] and [2.5], respectively, reflects the dynamicism of permits. The relaxation of constraints that occur moving from [2.2] and [2.3] to [2.4] and [2.5], respectively, reflects the tradability of permits. $T_{CDT}$, as the solution to the equation subject to the loosest constraint, will be the least among $T_{C_{SN}}$, $T_{C_{DN}}$, $T_{C_{ST}}$, $T_{C_{DT}}$.

Any move from SN to one of ST, DN or DT is a move to a permit control system with a higher economic performance but lower or more uncertain environmental performance. Although regulators may not always admit it, there is almost always — at least in the long term, once laws, institutions and capital equipment can be adjusted — a trade-off between environmental and economic performance. Within any permit control system, without reforming the overall system itself, the trade-off can be made between tightening or lowering discharge permits, which will improve the water quality but drive up the cost, and relaxing or increasing discharge permits, which will lower the water quality but reduce the cost.
Figure 2.6 shows these trades-off for all four schemes as curves of falling ambient concentration as cost rises, or looked at it in another way, falling environmental performance as economic performance rises. Curve \( sn \) shows the long-term trade-off that was available in the original, SN system, and curves \( st, dn \) and \( dt \) show the trades-offs available under the ST, DN and DT systems respectively. The actual distances between these curves are purely arbitrary in the Figure, being drawn for diagrammatic convenience.

**Figure 2.6 Economic and environmental performance of dynamic and tradable permits**

This study evaluates the performance of the PHRSTS, a dynamic tradable (DT) control system (that is, PHRSTS, see Section 5.1) against the Trickle Discharge (TD), an approximate static non-tradable (SN) control system (that is, TD, see section 4.2). Because of trade-offs between two dimensions of performance, and because the PHRSTS contains two new elements, dynamicism and tradability, just defining the conceptual framework for assessing the PHRSTS's performance is quite complex and rather arbitrary. Even if all data before and after the
PHRSTS was introduced were available — which they were not — costs and (ambient) concentration data would be only about the points labelled in bold on the Figure:

- the initial, before-PHRSTS point $SN$, with cost $C_0$ and concentration $A_0$; and

- the final, after-PHRSTS point $DT$, with cost $C_{DT}$ and concentration $A_{DT}$ (which may or may not be at society's optimal point *, where the opportunity set $dt$ touches the most desirable indifference curve $PP''$, representing society's trade-off of preferences between cost and concentration, touches, that can be reached — a theoretical ideal that we are unable to pay any further attention to).

This immediately raises the question of how we can define the (relative) performance of the DT system compared to the initial, SN system. There is an almost irresistible temptation to seek a single number to measure performance. The single number could be

$$\sqrt{\left(\frac{C_0 - C_{DT}}{C_{DT}}\right)^2 + \left(\frac{A_0 - A_{DT}}{A_{DT}}\right)^2} \quad \text{or} \quad \frac{C_0 - C_{DT}}{C_{DT}} \frac{A_0 - A_{DT}}{A_{DT}} \quad \text{[2.6]}$$

However, probably the best way to find the performance of the DT would be always to give two numbers together, by saying:

- the overall performance of DT over SN is \(\{C_0 - C_{DT}, A_0 - A_{DT}\}\), a combined improvement of cost and concentration.

Two conceptual interests would then be what I called in this study:

- the equivalent economic performance of DT over SN: $C_0 - C_{0DT}$, or

- the equivalent environmental performance of DT over SN: $A_0 - A_{0DT}$.

As shown in Figure 2.6, $C_{0DT}$ is the hypothetical cost — using a convention throughout the Figure that hypothetical rather than actual numbers are shown in ordinary rather than bold type — that could in principle have been achieved if the DT scheme had been introduced without any overall fall in ambient concentration, by setting numerically higher (that is, laxer) effluent
standards under DT (but in different times and places than under SN, thanks to dynamic tradability). So $C_0 - C_{0DT}$ is the fall in cost that notionally could have been achieved by DT compared to SN. Likewise, $A_{0DT}$ is the hypothetical (ambient) concentration that could have been achieved if DT had been introduced without any overall fall in cost, by setting numerically lower (that is, more stringent) effluent standards under DT than under SN; so $A_0 - A_{0DT}$ is the fall in concentration achieved.

Given the unique DT nature of the PHRSTS, it is of considerable interest to decompose its overall performance into that achieved by dynamicism and that achieved by tradability. But as can be seen from the Figure 2.6, there are several possible decompositions of performance. If points ST and DN would have been optimally chosen under ST and DN control system respectively, then we could define five pairs of performance measures for tradability:

- the **overall performance of tradability** := \{ $C_0 - C_{0ST}$, $A_0 - A_{0ST}$\} or \{ $C_{DN} - C_{DT}$, $A_{DN} - A_{DT}$\} (actual falls in cost and concentration that would *optimally* be achieved by tradability, before or after dynamicism is introduced); or separating economic from environmental performance:

- the **actual economic performance of tradability** := $C_0 - C_{0ST}$ or $C_{DN} - C_{DT}$ (actual falls in cost that would *optimally* be achieved by tradability, before or after dynamicism is introduced, both of which allow for simultaneous falls in concentration);

- the **actual environmental performance of tradability** := $A_0 - A_{0ST}$ or $A_{DN} - A_{DT}$ (actual falls in concentration that would *optimally* be achieved by tradability, before or after dynamicism is introduced, both of which allow for simultaneous falls in cost);

- the **equivalent economic performance of tradability** := $C_0 - C_{0ST}$ or $C_{DN} - C_{DT}$ (falls in cost before or after dynamicism is introduced, holding concentration constant)

- the **equivalent environmental performance of tradability** := $A_0 - A_{0ST}$ or $A_{DN} - A_{DT}$ (falls in concentration before or after dynamicism is introduced, holding cost constant)

and an analogous set of five pairs of performance measures for dynamicism:
Chapter 2 Overview of Strategies for Water Quality Management

- the overall performance of dynamicism \( := \{C_{\theta} - C_{DN}, A_{\theta} - A_{DN}\} \) or \( \{C_{ST} - C_{DT}, A_{ST} - A_{DT}\} \) (actual falls in cost and concentration that would optimally be achieved by dynamicism, before or after tradability is introduced); or separating economic from environmental performance:

- the actual economic performance of dynamicism \( := C_{\theta} - C_{DN} \) or \( C_{ST} - C_{DT} \) (actual falls in cost that would optimally be achieved by dynamicism, before or after tradability is introduced, both of which allow for simultaneous falls in concentration);

- the actual environmental performance of dynamicism \( := A_{\theta} - A_{DN} \) or \( A_{ST} - A_{DT} \) (actual fall in concentration that would optimally be achieved by dynamicism, before or after tradability is introduced, both of which allow for simultaneous falls in cost);

- the equivalent economic performance of dynamicism \( := C_{\theta} - C_{DT} \) or \( C_{ST} - C_{DT} \) (falls in cost before or after tradability is introduced, holding concentration constant)

- the equivalent environmental performance of dynamicism \( := A_{\theta} - A_{DT} \) or \( A_{ST} - A_{DT} \) (falls in concentration before or after tradability is introduced, holding cost constant)

With so many different performance measures available, it would be easy to lose sight of the key conclusion that is immediately obvious from Figure 2.6, which is that DT gives a greatest improvement in economic performance. However, which out of DN and ST gives a greater economic performance is less obvious, being a case-by-case answer.

As stated in Section 1.4, this study poses two primary research questions. Those are: (1) Did the PHRSTS significantly improve the river salinity of the Hunter River? (2) Did the PHRSTS generate substantial saving to its participants in control costs of saline water? Further, can much of the actual cost saving generated by the PHRSTS be attributed to the dynamicism or tradability of the discharge permits? To address these questions, based on the available operational data of the PHRSTS, the study measures (1) actual environmental performance represented by \( A_{\theta} - A_{DT} \), (2) actual economic performance \( C_{\theta} - C_{DT} \), and (3) actual cost saving due to tradability represented by \( C_{\theta} - C_{DT} \). A conclusion on the overall performance of the PHRSTS will then emerge based on these measures.
2.5 Summary

Control of pollution sources is a dominant strategy for management of river quality. Management of point sources is usually achieved using discharge permits, which can be defined in two dimensions: dynamicism and tradability. Combinations of these two dimensions form static non-tradable permits (SN), static tradable permits (ST), dynamic non-tradable permits (DN) and dynamic tradable permits (DT). Any move from SN to one of ST, DN or DT is a move to a control system with higher economic performance, but also with less reliable environmental performance. DT, a mixed system involving both dynamic permits and tradable permits, is more cost-effective than either dynamic permits or tradable permits separately. The theoretical foundation developed in this chapter provides an analytical framework for examining the environmental and economic effectiveness of the PHRSTS.
Chapter 3 Hunter River Salinity and Industrial Saline Water Discharge

This chapter describes the characteristics of the Hunter River based on a review of the available literature on salinity, both general and specific, as it applies to the Hunter Catchment. It presents the main features of river flow and salinity in the Hunter River, factors affecting the generation of industrial saline water, and factors determining the effects of industrial discharge on river salinity. This is to aid understanding of the nature and scale of the industrial saline water discharge and their implications for saline water management strategies for the Hunter River.

3.1 Catchment overview

The Hunter River (the River) is one of the most important river systems in Australia from geographic, resource, economic and political viewpoints. Situated on the east coast of Australia, the River is located in the State of New South Wales (NSW), between 31.5° and 33° south, and 150° and 152° east. Figure 3.1 shows the Hunter catchment. The River, which originates in the Barrington Tops, north of Muswellbrook, flows south-west to where it meets the Goulburn River. It then flows east to enter the Pacific Ocean at Newcastle, with a total length of 467 kilometres (km). The floodplain gradually expands as the river flows to the sea. The floodplain in the upper part of the catchment is around 3 km wide, expanding to almost 24 km in width by Maitland, and reaching widths of up to 40 km in some places in the lower reaches (NSW DLWC 2000; HVRF 2003).
While most of the stream flow comes from the north-eastern part of the catchment, the Hunter River is fed by a number of tributaries along its course. The major tributaries, from upstream to downstream, are the Pages River, Isis River, Rouchel Brook, Goulburn River, Glennies Creek, Wollombi Creek, and the Paterson, Allyn, Chichester and Williams Rivers, together with a number of other minor tributaries draining the surrounding sub-catchments. Among them, the Goulburn River in the Weston part of the catchment contributes 23% of the river flow while draining almost half of the total catchment. In total, the rivers and creeks of the Hunter drain a total catchment of 22,000 km² (NSW DLWC 2000; HVRF 2003). This study focuses on the segment of the Hunter River mainstream between Denman and Singleton, to which the PHRSTS applies (See Section 5.1).
3.1.1 Geology

A significant feature of the Hunter Valley geology is a major fault line which separates the carboniferous rocks exposed along the northern sides of the valley from the coal measures sequences of Permian age in the central and south-eastern areas and the Triassic sandstone in the south (NSW DLWC 2000). The Permian rock underlying most of the Hunter Valley is one of the main factors governing the salinity of both groundwater and surface water. The Permian rock consists of conglomerate, sandstone, shale and coal; it derives from ancient marine sediments and therefore contains salt. As a consequence, many of the streams that occupy the central valley floor are naturally saline. Many groundwater areas are also saline and the land is prone to dryland salinity when the salt is brought to the soil surface by rising water tables (NSW DLWC 2000).

3.1.2 Rainfall and evaporation

Maritime influences play an important role in all aspects of the climate in the Hunter catchment. Rainfall in the Hunter Valley varies considerably in both temporal and spatial dimensions. Average annual rainfall ranges from approximately 1350 mm along the coast margins to about 550 mm in the upper Hunter Valley (AGC Woodward-Clyde 1992). A seasonal rainfall pattern is also observed. Summer months are generally wetter than winter months except in the coastal area (NSW DLWC 2000). The average annual evaporation increases from east to west across the Hunter Valley, ranging from 1180 mm to 1323 mm (AGC Woodward-Clyde 1992).

3.1.3 Surface water

Within the Hunter catchment, approximately 80% of total water flows are unregulated. The remainder of water flows is regulated by three major storage dams: the Glenbawn Dam on the upper Hunter mainstream, Glennies Creek Dam on Glennies Creek and Lostock Dam on Paterson River. These dams regulate 320 gigalitres/year (GL/year) stream flow (NSW DLWC 2000), accounting for 17.8% of the average catchment flow yield (see Section 3.2). These dams are owned and operated by the NSW Department of Land and Water Conservation (DLWC)².

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² DLWC has changed its names from time to time. Its previous names were NSW Water Resource Commission (WRC); Department of Water Resources (DWR). It was then renamed as the Department of Infrastructure, Planning and Natural Resources (DIPNR) and the Department of Natural Resources, in that order. On 27 April 2007 a new Department of Water and Energy (DWE) was created, combining divisions of the former DLWC and the former Department of Energy, Utilities and Sustainability (DEUS) http://www.naturalresources.nsw.gov.au/aboutus/index.shtml. The acronym DLWC is used in this thesis to reflect the fact that it was used for the entire PHRSTS period.
the Hunter Water Corporation and the Macquarie Generation to provide a large volume of water during low flow periods to meet the requirements of irrigation, industry and town water supply. In general, the flow regulation has reduced high flows in the Hunter mainstream due to the storage of water in the dams (NSW DLWC 2000).

Salinity, algal blooms and bacteriological contamination are the main issues of concern in relation to water quality in the Hunter River (HVRF 2003). The high salinity in the River can be attributed to its prevailing geological conditions, but this has been exacerbated by human activities, such as discharge of industrial saline water and agricultural irrigation. The impacts of industrial discharges and agricultural irrigation on the Hunter river salinity will be described in Sections 3.4, 3.5 and 3.5.4. Algal blooms result occasionally from excess nutrients in rivers and water storage areas in the Hunter. Bacteriological contamination arises from runoff from livestock and septic systems (HVRF 2003).

3.1.4 Groundwater

Groundwater resources in the Hunter Valley generally fall into one of two categories - alluvial aquifers in the Hunter River plains, and hardrock aquifers. These aquifers maintain base flow in many streams during dry times (Mackie Martin-PPK 1994).

Alluvial aquifers adjacent to the Hunter River are the principle groundwater storages in the Hunter Valley. These aquifers are hydraulically connected to the river and are exploited for irrigation water supply in the region, from Glenbawn dam to the confluence of the Hunter and Goulburn Rivers. The alluvial deposits act as local river water storage, exchanging waters with the river as river levels rise and fall (Mackie Martin-PPK 1994). Hardrock aquifers offer only limited groundwater resources. In the northern parts of the valley, Carboniferous rocks provide water mostly suitable for stock use. In the central areas, Permian rocks and coal measures yield mostly brackish water (Mackie Martin-PPK 1994).

3.1.5 Social and economic activities

The Hunter Valley is one of the most highly developed areas in Australia in terms of the extent and diversity of the agricultural activities and the significance of industrial activities. The main agricultural activities undertaken are cropping, grape cultivation and wine making, sheep
grazing and the wool industry, house studs and fishing. These activities directly and indirectly support some 5900 people and generated production with a total value of approximately $393.5 million in 1999-2000 (HVRF 2003).

Besides farming, the industries of the Hunter region include coal mining, power generation, heavy metal smelting and various manufacturing. Among them, coal mining and power generation are the most important industries in the region. The coal mining industry employs around 8000 people and supplies the export and domestic markets. Black coal mining accounts for more than 1% of Australia's GDP (NSW DMR 2002). The Macquarie Generation operates two coal fired power stations in the mid Hunter catchment, contributing more than 15% of the electricity in the National Electricity Market (Macquarie Generation 2003). Detailed descriptions of the coal mining and power generation in the Hunter Region will be provided in Sections 3.4.1 and 3.5.1.

3.1.6 Principle water uses and discharges

The coal mines and power generation, along with agricultural activities, smaller industries and urban areas, rely on the Hunter River and its tributaries for water supply, wastewater discharge or both (HVRF 2003). DLWC supplies water to each of these groups of users on the basis of a security rating — high security and low security through extraction licences that specify the allocated volume of water. High security supply ensures some level of guaranteed supply throughout the worst droughts while low security supply is subject to water availability during very dry periods. Users of the water for irrigated agriculture are generally consumers of low security water, while many industrial users and town water suppliers receive high security water (NSW DLWC 2000).

Point source discharge of pollutants including from coal mining and power generating industries can potentially occur in all weather and river flow conditions, and non-point source pollutant runoff from groundwater discharge and agricultural activities occurs in wet weather when river flow is generally higher. While the former is regulated by the discharge licenses and various NSW EPA regulations, the latter is not regulated (NSW EPA 2001).
3.2 Hunter River flow

River flows within the Hunter catchment result from a number of sources, including rainfall runoff, groundwater seepage, regulated discharge from Glenbawn and Glennies Creek dam, mine water releases and other sources (NSW DLWC 2000). While rainfall runoff and groundwater seepage are natural processes, regulated dam releases and mine water releases are not.

Average runoff from the Hunter catchment is 1800 GL/year or about 12.5% of the total catchment rainfall. Of the total runoff, 42% actually comes from the Paterson, Allyn and Williams Rivers where the annual rainfall is more than double the average rainfall over the whole of the Hunter (NSW DLWC 2000). Discharge of shallow groundwater, primarily found in the river alluvia along the length of the Hunter, is the key contributor to river flows during dry periods (NSW DLWC 2000).

The monitoring of the Hunter River mainstream and its tributaries has a long history. The NSW DLWC maintains an extensive gauging network throughout the Hunter Valley. Figure 3.2 indicates the locations of major gauging stations in the study area. The oldest gauging at Singleton (Point 15) was established in 1891, but most gauges were installed in the late 1950s or 1960s. Early flow information was collected by visual observation of staff gauges with low frequency (for example, monthly) but suffered from considerable discontinuity. Later records were obtained by continuous recording with float gauges or pressure gauges with an increasing frequency (M. Simons [Resource analyst, NSW DNR] 2003, pers. comm., 7 July). Recent records have been collected through newly developed radio telemetry with high frequency down to 5 to 10 minute intervals since 1993 (NSW DLWC 2000). As will be described in Section 5.1, Denman (shown as Point 2 in Figure 3.2), confluence of the Glennies Creek and Hunter mainstream (denoted Glennies/Hunter, shown as Point 14 in the Figure) and Singleton (shown as Point 15 in the Figure) are the control stations for defining river flow periods (such as Low, High and Flood) and salinity standards for the river sectors under the HRSTS.
Chapter 3 Hunter River Salinity and Industrial Saline Water Discharge

Figure 3.2 Distribution of gauging stations in the Hunter Valley

Source: Hunter Water Quality Task Group (1994: 9); point 14 = "Glennies/Hunter"

Stream flows within the Hunter Valley exhibit a high degree of both inter-year and intra-year variability. Flows are also seasonally variable with highest flows occurring during summer months when highest rainfall receipts and more intense storms tend to occur. This characteristic has been well documented in the literature, such as, Croft & Associates 1983; Day 1986; AGC Woodward-Clyde 1992; NSW DLWC 2000.

Examination of the available daily flow records at Singleton for the period 1993 to 2002 in this study has confirmed this characteristic. The annual average daily river flow for this period varied by a factor of 7 for Denman, 12 for Glennies/Hunter and 19 for Singleton. The extreme daily flows of the River also present a huge range of dispersion. In the same period, the maximum daily flows varied from the minimum daily flows by the factors of 32~1860 for Denman, 32~1489 for Glennies/Hunter and 60~1416 for Singleton. Figure 3.3 presents the daily river flow at Denman, Glennies/Hunter and Singleton gauging stations. It illustrates considerable variations in river flow in the Hunter mainstream.
Chapter 3 Hunter River Salinity and Industrial Saline Water Discharge

Figure 3.3  Daily river flow at the Hunter main gauging stations

Source: Complied from data provided by the NSW DLWC in 2003

Day (1986) attributed the large variability of river flows to a number of natural factors, including considerable spatial variations in altitude, geology, landform, soils, rainfall, runoff and land use in the Hunter catchment, and to the great differences in periodicity of flow and runoff contributions per unit of catchment area of the major tributaries to the Hunter mainstream. In addition to these natural factors, human activities, such as direct extraction from and discharges into the river systems, as well as flow regulation through the construction of dams, may also have had effects on river flows through changing frequency of large events, volume and seasonality.

Another important characteristic of the river flow is that while the flow is higher in the middle and lower river sectors than in the upper sector, the flow in the middle sector is higher than in the lower sector, as the major tributaries join the mainstream in the middle sector. For example, based on the river flow data provided by the NSW DLWC, during the period 1993 to 2002, the average daily flow at Denman, Glennies/Hunter and Singleton gauging stations were 843 ML/day, 1427 ML/day and 1297 ML/day, respectively.
3.3 Hunter River salinity

Salinity refers to ambient concentrations of salts, such as sodium chloride, magnesium and calcium sulphates and bicarbonates, present in soil and water. Salinity can be measured in milligrams per litre (mg/L) of Total Soluble Salt (TSS) or Total Dissolved Salts (TDS). A common way to find the salinity level of water is by measuring its surrogate, electrical conductivity (EC), given that the concentration of salt in water affects the electrical resistance of the solution, and 1 EC = 1 micro Siemens per centimetre (µs/cm) at 25° C; typically a conversion factor of 0.6–0.68 is applied to convert EC (in µs/cm) to TDS (in mg/L) for the calculation of salt load. That is, 1000 EC = approximately 600–680 mg/L of TDS. Salt load, the mass of salt passing a particular location within a given time frame, is generally measured in tonnes per day or tonnes per year.

River salinity can develop naturally. Rainwater that falls into a catchment, surface runoff and seepage of groundwater all contribute salt to river streams (see Figure 3.4), though the contributions of each are disproportionate. Among the contributions, surface flows from salt-affected land and saline groundwater discharges to streams are the two main factors resulting in high salinity concentrations in river systems.

The water quality of the Hunter River is naturally saline due to its geological features described in Section 3.1.1. The NSW EPA (1994a) estimated that approximately 80% of the salt load in the River comes from natural sources. High salinity levels in the River are mainly attributed to natural seepage of saline groundwater to the surface. AGC Woodward-Clyde (1992) estimated that the direct groundwater accession of salt to the Hunter streams over the whole catchment above Singleton is 16,300 tonnes per year, based on an average concentration of groundwater of 4650 mg/L.
The history of observation of river salinity in the Hunter River is not as long as that of river flows. Although the NSW DLWC started to record river salinity at some key gauging stations in the 1970s, the sampling methods and frequency of records have not been constant over time. Early sampling was done via a hand-held probe or by a water sample being sent to laboratories for analysis. Only sporadic samples were taken at a selected number of stations when salinity became a concern. In the early 1980s, the DLWC installed four automatic salinity recorders at Muswellbrook, Liddell, Singleton and Greta (see Figure 3.1 for locations) to continuously monitor salinity levels in the River. Monthly readings have since then been supplemented by weekly or daily observations (M. Simons [Resource analyst, the former NSW Department of Natural Resources] 2003, pers. comm., 7 July). Since 1993, the monitoring coverage increased and the frequency and quality of river salinity observations dramatically improved with the installation of radio telemetry (NSW DLWC 2000). Monthly salinity observations of 1980-2002 and daily salinity observations of 1995-2002 at Singleton are used in this study to examine the environmental performance of the PHRSTS (Chapter 6).

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Recharge refers to the process of water entering the groundwater system from the surface. Discharge refers to the process of groundwater rising to the soil surface in low-lying areas or on the break of a slope, or flowing directly into stream (NSW DLWC 2001).
With the increasingly frequent observations and wide spatial monitoring network, a better understanding of the salinity in the Hunter River was also developed. Two main characteristics of the Hunter salinity became better understood, as discussed below.

The first is that salinity concentrations have a general inverse relationship with river flows. A number of factors account for the high salinity level during low flow from dry weather conditions. Those are: (1) the highly saline groundwater, which is the main source of the River’s base flow; (2) limited or non-existent surface runoff to dilute groundwater salinity; and (3) high evaporation rates, which concentrate salt in surface water (NSW EPA 1994a). It has been reported that regulation of the River (through water release from the dams) has ensured reasonable flows in all years. This in turn has removed the threat of extremely high salinities in periods of drought (NSW DLWC 2000b). During rainfall runoff events, salinity in the River varies with time. After initial flushing of runoff when a salt spike may occur, the river salinity decreases gradually with fresh rainfall runoff. This complexity makes it difficult to establish a precise quantitative relationship between river salinity and flow (see Section 6.2.2 for further discussion of this phenomenon).

Another important characteristic regarding the river salinity is that salinity progressively increases downstream, because water draining from the upper part of the catchment is of relatively low salinity. For example, the water in the Glenbawn Dam located in the upper Hunter River (Figure 3.2) is within the range of 350–400 EC. Upstream of Muswellbrook, salinity remains below 700 EC most of the time and only exceeds this level under extremely low flow conditions (NSW DLWC 2000). Further downstream, salinity steadily increases. Salinity at Glennies/Hunter is considerably higher than that at Denman (Figure 3.1), with few exceptions. However, on most occasions salinity at Glennies/Hunter tends to be slightly more saline than that at Singleton. This situation is attributed to high salinity water inputs from the tributaries, notably from the Goulbourn River and lower Wollombi Brook, which merge with the Hunter mainstream between Denman and Glennies/Hunter (Croft & Associates 1983). The Goulbourn River and lower Wollombi Brook, which are incised into saline geologies and flow through the saline rocks of the valley floor, are regarded as major sources of riverine salinity in the Hunter River (Croft & Associates 1983).

4 A salt spike is a sharp increase in the river’s salinity at the beginning of a storm event after a long drought period, caused by storm water flushing accumulated salt from the river streams.
Although most of the salt in the Hunter River comes from natural sources, discharges from human activities can periodically cause significant short-term increases in river salinity. The principal human activities that release salt into the Hunter River are coal mining, power generation and agricultural practices. The following sections describe the characteristics of the wastewater discharged from each of these activities and discuss their impacts on the water quality of the Hunter River.

3.4 Mining and Hunter River salinity

3.4.1 Coal mines

As noted Section 3.1.1, the Permian system dominates the geologic structure of the Hunter Valley. This system contains marine strata, which includes rich coal measures. As a result, underlying most of the Hunter region are numerous seams of black coal of various thicknesses and at various depths (Creelman 1994). It was estimated that recoverable coal reserves in the Hunter Valley account for approximately 63% (6,011 million tonnes) of the total coal in NSW (9500 million tonnes) (HVRF 2003).

The Hunter Valley has had a long history of coal mining, over 200 years. Early mining activities in the region can be traced back to 1799, begun under the auspices of the colonial government (HVRF 2003). The importance of the coal mining in the region is shown by the original name of the Hunter River, “Coal River” (Marsden 1999). Coal mining is also the reason for the settlement and development of Newcastle, which was called “Coalopolis” (meaning Australia’s only coal port) in the late 19th century (Marsden 1999). Though having a very long history, only in the 1980s did the mining industry start to experience its rapid growth. This was driven by two factors: increase in the demand for coal from overseas markets; and increase in domestic demands as a result of construction and development of coal-fired thermal power stations in the Hunter Valley, notably Liddell power station (NSW Minerals Council 2001).

Nowadays, the Hunter Valley possesses one of the best-developed coal mining industries in the world, and coal mining has become one of the key industries in the region. The following statistics provided by the NSW Department of Mineral Resources (NSW DMR 1991; 1992; 1993; 1994; 1995; 1996; 1997; 1998; 1999; 2000; 2001; 2002; 2003) indicate the industry’s importance in the region’s development and economy. During the period 1991 to 2003, the coal
production of the Hunter area increased steadily, at an average annual growth rate of 7%. As a result, the share of NSW production from the Hunter coalfield increased from 50.7% in 1992/1993 to 94.4% in 2001/2002. Approximately 75% of coal produced in the Hunter Regions supplied the export markets, and the rest was consumed by domestic users, primarily by local electricity generators.

Open-cut (or surface) and underground (or deep) mining are the two main methods used for coal mining. The choice of mining method is largely determined by the geology and depth of the coal seam below the surface. Where the coal seam is relatively close to the surface, open-cut mining is economical. Where the coal seam is too deep for open-cut methods, underground mining is the only suitable method. An illustration of open-cut and underground coal mining methods is in Figure 3.5.

**Figure 3.5 Comparison of open-cut and underground coal mining methods**

![Comparison of open-cut and underground coal mining methods](source: University of Kentucky (2004))

In the Hunter coalfield, vast coal reserves lie at comparatively shallow depths. Open-cut is therefore the dominant mining method employed in this region. Of the total saleable coal in 1992-2002, 87% was produced by open-cut mines, and only 13% was from underground mines (NSW DMR 1992; 1993; 1994; 1995; 1996; 1997; 1998; 1999; 2000; 2001; 2002).
Chapter 3 Hunter River Salinity and Industrial Saline Water Discharge

Up to 2002, 22 coal mines operating in the study area (Table 3.1) participated in the PHRSTS. In terms of their geographic distribution (Figure 3.6), ten mines were concentrated in the middle segment of the Hunter River, and the rest were equally distributed across the upper and lower Hunter. Among these 22 coal mines, 15 were open-cut, five were underground, and two were combined open-cut and underground operations.

Table 3.1 Basic data on PHRSTS mines (the main mines of the Hunter coalfield)

<table>
<thead>
<tr>
<th>Coal mines</th>
<th>Ownership</th>
<th>Location</th>
<th>Mining methods</th>
</tr>
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<tbody>
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<td>Bengalla*</td>
<td>Coal &amp; Allied</td>
<td>U</td>
<td>O</td>
</tr>
<tr>
<td>Bulga/South Bulga</td>
<td>Xstrata Coal</td>
<td>U</td>
<td>O/U</td>
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<td>Camberwell Coal</td>
<td>M</td>
<td>O</td>
</tr>
<tr>
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<td>Xstrata Coal</td>
<td>M</td>
<td>U</td>
</tr>
<tr>
<td>Dartbrook</td>
<td>Anglo Coal</td>
<td>U</td>
<td>U</td>
</tr>
<tr>
<td>Drayton</td>
<td>Anglo Coal</td>
<td>U</td>
<td>O</td>
</tr>
<tr>
<td>Glennies Creek</td>
<td>Glennies Creek</td>
<td>M</td>
<td>U</td>
</tr>
<tr>
<td>Hunter Valley Operations north</td>
<td>Coal &amp; Allied</td>
<td>M</td>
<td>O</td>
</tr>
<tr>
<td>Hunter Valley Operations west</td>
<td>Coal &amp; Allied</td>
<td>M</td>
<td>O</td>
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<td>(Howick)</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Lemington</td>
<td>Coal &amp; Allied</td>
<td>L</td>
<td>O</td>
</tr>
<tr>
<td>Liddell</td>
<td>Xstrata Coal</td>
<td>M</td>
<td>O</td>
</tr>
<tr>
<td>Mount Owen</td>
<td>Xstrata Coal</td>
<td>M</td>
<td>O</td>
</tr>
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<td>Coal &amp; Allied</td>
<td>L</td>
<td>O</td>
</tr>
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<td>O</td>
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<tr>
<td>Ravensworth-east</td>
<td>Xstrata Coal</td>
<td>M</td>
<td>O</td>
</tr>
<tr>
<td>Ravensworth-Narama</td>
<td>Xstrata Coal</td>
<td>M</td>
<td>O</td>
</tr>
<tr>
<td>Rixs Creek</td>
<td>Rixs Creek</td>
<td>L</td>
<td>O</td>
</tr>
<tr>
<td>United</td>
<td>Xstrata Coal</td>
<td>L</td>
<td>U</td>
</tr>
<tr>
<td>Wambo</td>
<td>Wambo Coal</td>
<td>L</td>
<td>O/U</td>
</tr>
<tr>
<td>Warksworth No.1</td>
<td>Coal &amp; Allied</td>
<td>L</td>
<td>O</td>
</tr>
</tbody>
</table>

*U = upper sector of the Hunter River.
*M = middle sector of the Hunter River.
*L = lower sector of the Hunter River.
*O = open-cut mines.
*U = underground mines.
* = new development during PHRSTS.

Sources: NSW DMR (2002) and P. Smith ([Assistant Director, Environment, NSW Minerals Council] 2003, pers. email. 16 September)
Figure 3.6 Distribution of the coal mines in the Hunter Coalfield

In 2002, the total raw production and saleable production of coal amounted to 93.62 Mt and 69.55 Mt respectively, almost double that for 1992/1993 financial year (NSW DNR 1993; 2002). The growth of production can not only be attributed to the increase in demands from overseas, but also to the significant improvements made in mining technology, which have enabled large tonnages of coal to be exploited cheaply and thus competitively in the world markets (NSW Minerals Council 2001).

One noticeable feature of the Hunter coal mining industry is its high number of cross-company ownerships, though the ownership of some mines has changed over the past two decades. As shown in Table 3.1, by 2002, 14 out of the 22 mines actually belonged to two big companies: Xstrata Coal Pty (eight mines) and Coal & Allied (six mines). This has implications for the FHRSTS credit trading (discussed in Chapter 5).

The contribution of these mines to regional social development is illustrated by the following statistics. In 2002, the 22 mines employed a total of 5213 people (compared to 5487 people employed in agriculture in the whole Hunter Region), and generated some $430 million in local wages income (NSW DMR 2002). While 84% of these people were employed by the open-cut mines, the remainder worked for the underground mines. The percentages of employees working in the mines in the upper, middle and low Hunter region were 33%, 42% and 25%, respectively (NSW DMR 2002).

Open-cut mining tends to be capital rather than labour-intensive. Statistical data for 1999-2002 (NSW DMR 1999; 2000; 2001; 2002) show that the average annual saleable output per employee for open-cut mines (14340 tonnes) was 1.5 times that of underground mines (8244 tonnes). Although underground mining is usually more expensive and labour-intensive than open-cut mining, the geology of coal deposits is likely to result in a mine’s continuation long after open-cut seams have been exhausted (NSW Minerals Council 2001). This may partly explain the fact that in recent years the annual production growth rate from underground has been twice that of the production of open-cut mines.

As noted in Table 3.1, Bengalla Coal Mine is one of the two new developments (another new development is Redbank Power Station) under the PHRSTS. Detailed information about this mine is not readily available, but some general relevant information is obtained from its publicly accessible website (Rio Tinto 2008). For example, Bengalla Coal Mine is located four
kilometres west of Muswellbrook in the upper Hunter region. Although its proposal as a project goes back to 1990, it did not start its operation until 1999. In 2007, Bengalla produced a total 5.2 million tonnes of saleable production and had a total of 233 employees. The freshwater use rate was 102 litres per tonne of coal produced.

3.4.2 Water demand and supply

Despite the significance of the coal mining industry in terms of its contribution to the region's economic and social development, it is relatively insignificant in terms of its consumption of river water. The industry uses a small proportion (some 5%) of the regulated river water allocation while generating large financial returns on this water usage (NSW Minerals Council 2001). The figures for economic returns of per unit water usage for the Hunter coal industry are not available, but the State's average can be used as an indication. Based on the statistics provided by NSW Minerals Council (2001), the economic return on the same level of water usage for the mining industry in the State is one thousand times that of agriculture. The modest consumption of river water by the coal mining industries is attributable to two related factors. On one hand, the coal industry, by its nature, is able to use poor quality water that may not be suitable for most other purposes. On the other hand, considerable improvements have been made in the efficiency of water recycling and reuse in the industry (NSW Minerals Council 2001).

The main demands for water on a mine site are for coal washing, dust suppression on haul roads and coal stockpile areas (NSW Minerals Council 2001). The demand for coal washing varies with the amount of coal production. The demand for dust suppression, however, depends not only with the length of haul roads and area of hardstand that need to be watered, but also varies with regional prevailing climatic conditions. For example, dust suppression on haul roads and other areas was estimated to have risen from 1.2 ML/day to more than 1.6 ML/day for the extension of Warkworth Coal Mine (Mackie Environmental Research 2002). Other uses of water on a mine site include wash-down of industrial facilities and ablution, irrigation on rehabilitated land and fire fighting, but these activities only account for a small volume (NSW Minerals Council 2001).

The water used on a mine site can originate from a range of sources, divided into four categories based upon the source, quality and management requirements: potable water, clean water, dirty water and mine water (NSW Minerals Council 2001). Potable water, which is used for drinking,
administration office supplies and bathhouses, is the highest category and quality. It is supplied from off-site sources, such as pumping water from the Hunter River under extraction licences, connection to a town water supply, or periodic delivery by tanker. Potable water is tested regularly for occupational health reasons (NSW Minerals Council 2001).

*Clean water* comes from the rainfall runoff from undisturbed, vegetated parts of a site. It is diverted around or away from the areas affected by mining by means of drains and diversion banks. While this water can be discharged off-site subject to no controls, the prevailing practice is withholding the water in dams to augment the mine supply (NSW Minerals Council 2001). In contrast to clean water is *dirty water*, which is the runoff from so-called disturbed areas, the areas affected by mining activities. The water of this category usually contains suspended silt material and is drained to holding dams and sediment ponds for settling. The runoff from disturbed and undisturbed areas has little impact on river salinity because of the general low level of soluble salts. Discharging of this water is subject to the limits set down in the *Australian Water Quality Guidelines for Fresh and Marine Waters* (NSW Minerals Council 2001).

The last category is *mine water*, which is the water collected within the mine workings as a result of groundwater seepage, dewatering and rainfall accumulation (NSW Minerals Council 2001). This water is usually saline and can be discharged only under the discharge licenses and in compliance with NSW EPA regulations (NSW EPA 1994a). Mine water is the focus of this study. The ways in which coal mines generate mine water and the impacts of mine water discharge on river quality will be discussed in the next section.

The amount of water used varies among mine sites and depends on their on-site operations and environmental conditions. In general, the water demand of underground operations is less than that of open-cut operations, as the latter need more water for dust suppression (NSW Minerals Council 2001). Over the last ten years, improvement in self-sufficiency has become a central aim of water management systems for the coal mining industry. The overall objectives of a water management system are to reduce the demand on external sources and reduce the need for disposal of any excess through maximizing on-site water recycling and reuse. To achieve these objectives, the dirty water and mine water stored in on-site dams are used as priority sources for coal washing and dust suppression. Wastewater from coal washing is, in turn, stored and recycled to limit the demand on external water supplies (NSW Minerals Council 2001).
Information from the NSW Minerals Council (2001) shows that considerable improvement has been made in the efficiency of water use in the coal mines of the Hunter region. The average amount of water consumed dropped from 283 litres per tonne of coal in 1978 to 193 litres per tonne in 1999-2000. Consequently, despite the increase in coal production over that time of almost 800%, the proportion of licensed entitlements of water use held by the mines has only risen from 1% to 5.5% of total entitlements across the whole Hunter Valley (NSW Minerals Council 2001).

### 3.4.3 Generation of mine water

Groundwater seepage dewatering of mine workings and rainfall accumulation are the two primary sources of mine water. The ways in which coal mines generate mine water is depicted in Figure 3.7, which uses an open-cut mine as an example.

**Figure 3.7 Generation of open-cut mine water**

As noted in Section 3.1.1, marine sediment deposits dominate the geology of the Hunter Valley. While they contain substantial coal reserves, they also contain salts, which render the groundwater very saline. Moreover, for geological reasons, the best coals are usually found in rocks with the highest solute contents (Creelman 1994). Pit groundwater inflow/seepage is a constant product for both open-cut and underground coal mines, and its volume progressively
increases over the life of the mine as the mine excavation increases in depth and in area and after regular dewatering. For example, URS Australia (2000) predicted that the total groundwater seepage at the Mount Arthur North Coal Mine would rise from zero to a rate of 3 ML/day over the first 10 years and increase further to 3.6 ML/day over the next 10 years. Also, Mackie Environmental Research (2002) projected that the groundwater seepage, with a salinity concentration ranging from 2900-18400 EC, would rise from an initial rate of 0.64 ML/day to a rate of 4.2 ML/day at the Warkworth Coal Mine after 18 years of operation.

While both open-cut and underground mines experience groundwater seepage, open-cut mines may accumulate rainfall from the floor and side slopes of the workings in the pits. As coal seams alternate with a conformable series of marine sandstones, siltstones and mudstones, open-cut mine works expose substantial areas of salt-bearing rocks and liberate the saline groundwater at much higher rates than would occur naturally. As a result, the accumulated water in the mining pits becomes very saline. Though part of this water can be lost by evaporation, salt cannot. The residual water, whose saline content is highly concentrated and gravitates to the low points of the pits, has to be pumped out of the pits so that mining activities can continue (Croft & Associates 1983).

Mine water in the Hunter coal mines therefore exhibits two special characteristics. First, although the groundwater is a constant product of mine operations, the overall amount of mine water generated on a mine site depends not only on rainfall features (that is, duration and intensity), but also on the catchment area of the site, which varies over the mine’s life. In general, open-cut mines tend to accumulate more mine water than underground mines, and tend to accumulate mine water at higher rates in wet years and at lower rates in dry years. Because of this characteristic, unlike other industrial or municipal wastewater treatment plants whose effluent flows are continuous and relatively predictable, the generation of mine water is unpredictable and the discharge of mine water tends to be discontinuous.

The second characteristic of mine water, which is closely related to the first, is that the salinity of mine water, even for the same mine, may vary significantly over time. In general, compared to the average rainfall condition, the salinity of mine water in a dry period is higher, as it mainly consists of the highly saline groundwater seepage. Conversely, the salinity of mine water in a wet period is lower because of the dilution from rainfall runoff. AGC Woodward-Clyde (1992)
estimated that the mean salinity of mine water discharged from all operating mines in the Hunter was about 5000 EC.

### 3.4.4 Discharge of mine water

The requirements to maximise the reuse of mine waters for mine water supply, minimise the external mine supply demand, and minimise the discharge of mine water from the site are the three key considerations in development of appropriate on-site water management strategies in the mining industry (NSW Minerals Council 2001). The prevailing water management strategies of the coal mines in the Hunter region are summarised in Table 3.2.

**Table 3.2 Mine water management strategies for the coal mines in the Hunter Valley**

<table>
<thead>
<tr>
<th>Type of water</th>
<th>Characteristics</th>
<th>Management strategies</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pit water from open-cut mines</td>
<td>Sourced from groundwater access</td>
<td>Stored in on-site dams</td>
</tr>
<tr>
<td></td>
<td>Dependent on the depth of mining pits</td>
<td>Used as mine water supply source</td>
</tr>
<tr>
<td></td>
<td>Too saline to allow uncontrolled discharge to the Hunter River</td>
<td>Controlled discharge of any excess to the Hunter River in accordance with NSW EPA regulations</td>
</tr>
<tr>
<td>Pit runoff from open-cut mines</td>
<td>Dependent on pit catchment area and rainfall conditions</td>
<td>Stored in dams on site</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Used as mine water supply</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Controlled discharge of any excess to the Hunter River in accordance with NSW EPA regulations</td>
</tr>
<tr>
<td>Water from underground mines</td>
<td>Predominantly groundwater</td>
<td>Stored in dams on site</td>
</tr>
<tr>
<td></td>
<td>High salinity concentration</td>
<td>Used as mine water supply</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Controlled discharge of any excess to the Hunter River in accordance with NSW EPA regulations</td>
</tr>
<tr>
<td>Surface runoff from undisturbed areas</td>
<td>Natural runoff water</td>
<td>Diverted around areas disturbed by mining activities and allowed to discharge from the site without control</td>
</tr>
<tr>
<td>Active mining overburden emplacement and establishing rehabilitation areas</td>
<td>Likely to contain elevated levels of suspended sediment</td>
<td>Runoff collected in catch drains and directed through sediment traps and settling dams</td>
</tr>
<tr>
<td></td>
<td></td>
<td>After removal of suspended sediment, allowed to drain from the site</td>
</tr>
</tbody>
</table>
### Chapter 3 Hunter River Salinity and Industrial Saline Water Discharge

<table>
<thead>
<tr>
<th>Mine infrastructure areas</th>
<th>Likely to contain elevated levels of suspended sediments</th>
<th>Isolation of mine infrastructure area catchments with diversion drains</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Potential for contamination from hydrocarbon and other chemical spills</td>
<td>Use of sediment traps and oil separators to remove suspended sediments, and oil and grease, prior to collection in dams</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Collection of runoff in catch dams located immediately downstream of infrastructure area</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Transfer of the water in catch dam to other dams for use as mine water supply</td>
</tr>
</tbody>
</table>

| Mine infrastructure construction areas | Likely to contain elevated levels of suspended sediments | Development and implementation of an erosion and sediment control plan in accordance with the DLWC requirements |

<table>
<thead>
<tr>
<th>Sewage effluent</th>
<th>Likely to have elevated levels of suspended solids, nutrients and BOD</th>
<th>Treated on site prior to discharge from the site</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Not suitable for discharge from the site without treatment</td>
<td></td>
</tr>
</tbody>
</table>

Sources: Adapted from URS Australia (2000); Gilbert & Associates (2003)

For most coal mines, on-site water management operates predominantly as a closed system, shown schematically in Figure 3.8. In this system, mine water is not discharged from the wastewater system off-site. Instead, mine water is pumped from the mine pits to on-site storage dams and used as one of the priority water sources for coal washing and dust suppression.

For individual mines, a water balance that captures all inflows and outflows on site needs to be developed to provide the basis for the on-site water management system. The water balance includes water demand and discharge in response to the level of coal production, size of the pits and rainfall, and in turn determines the capacity of mine water storage required to meet most mine site operational conditions. Detailed explanation and discussion of this will be provided in Section 7.2.1.1. As the on-site water balance fluctuates with climatic conditions, and as the extent and status of the mining operation evolve over time, there may be periods when the availability of water on site is insufficient to meet the demand. Under these circumstances, water would be imported from external sources, either opportunistically from adjacent mining operations, or from the Hunter River mainstream and its tributaries in accordance with the extraction licenses (URS Australia 2000).
Under "normal" weather conditions mines are net water users, supplementing on-site water needs with the water from clean water diversion systems. During prolonged wet periods, however, there may be times when the quantity of mine water exceeds that required on site and the excess cannot to be stored indefinitely on site due to the limitations of storage capacity (NSW EPA 1994a). This situation would be exacerbated by mining to greater depths (where groundwater inflow rates increase) and the development of additional and larger mines (where more rainwater accumulated in the pits and voids). In these circumstances, excess mine water has to be removed off site.

There are several options open to a mine with excess mine water, such as transferring it to adjacent mines that could benefit from additional water or temporarily storing it in the lower levels of open-cut pits contingent upon the mining operations (Gilbert & Associates 2003; Mackie Environmental Research 2002). When these options are not available, excess mine water has to be discharged to the Hunter River. Therefore, the need for a mine to discharge mine water to the Hunter River not only depends on the amount of mine water generated, but also relates to the efficiency of on-site water use and the storage capacity of on-site dams.

AGC Woodward-Clyde (1992) estimated the excess water from the 14 mines operating in 1991 under different climate conditions over the period 1990-1995. It was estimated that the total amount of excess water required to be discharged would be in the range of 8000-12000 ML/year in 1:20 wet year conditions, with an average of 2800-6500ML/year. These figures were later updated by NSW EPA (1994a), which estimated that 6280 ML of saline water would be generated in 1994, of which 3302 ML saline water containing 12,872 tonnes of salt would have to be removed so that mining could proceed. This is a substantial amount of saline water to be disposed of in terms of both volume and cost.
Figure 3.8  Water use and wastewater discharge of an open-cut coal mine

P = precipitation; E = evaporation

Sources: Adapted from AGC Woodward-Clyde (1992); URS Australia (2000); NSW Mineral Council (2001); Gilbert & Associates (2003); NSW EPA (1994a).
3.4.5 Impacts of mine water on river salinity

The impacts of mine water on salinity in the Hunter River depend on several factors. Generally, discharge of saline water from coal mines can be easily accommodated when the river flows are high. This is because high river flows are usually associated with periods of above-average rainfall; when in-river salinity is relatively low, coal mines need to dispose of the large quantities of mine water accumulated, and irrigators have low demands for river water. However, the typical rainfall pattern in the Hunter Valley is a succession of wet years followed by one or two dry years. Even within one year, dry periods and wet periods occur alternately (NSW EPA 1994a). Thus, there may be excess mine water carried over from a series of wet periods into a dry period. In such a situation, discharge may be required even though the excess water is not generated then.

However, during dry periods the average flow in the Hunter River falls considerably. The background or natural salinity then rises because river flow is sustained by groundwater inflows that commonly have conductivities of 1000-3000 EC (AGC Woodward-Clyde 1992). As a result, the river salinity levels are more likely to be affected by mine discharges in dry periods than in wet periods. However, demands on river water for irrigation and other uses are at their highest during dry periods. Although the volume of excess water requiring disposal in dry periods may be substantially lower than in wet periods, the impacts of one unit of mine discharge on river salinity in dry periods would be expected to be greater than in wet periods.

In addition to discharge of mine water, mining activities potentially contribute to river salinity in some other ways. Surface water runoff from mine landforms and disturbed areas may contain soluble salts and discharge them via overland flow to the streams. Also, when coal mines use mine water for spray irrigation and road dust suppression, the salt can eventually find its way back to the river. As well, the wastewater storage dams on coal mine sites, which are used to trap particulate matter and store excess mine water before it enters streams, may overflow in wet weather. While the salt load entering the Hunter River in these ways is analogous to non-point sources (NPS), the contribution to and impacts on river salinity by overflows are not significant in comparison with those generated by mine water.

Other than the above impacts, mining activities could have long-term profound effects on the quality of the Hunter River. As mining operations permanently remove coal and change the
structure and morphology of the coal removed, they make permanent alterations to the natural soil, rock strata and the groundwater regime. These alterations create new connections between separate aquifers and change flow paths and permeability, which may have far-reaching effects on the river system and groundwater salinity for a century or more after mining ceases (Healthy Rivers Commission of NSW 2002). So far, the understanding of long-term consequences of mining operations is extremely limited due to the lack of research on these issues.

3.5 Power generation and river salinity

Hydroelectric power production in Australia is limited to mainly the Snowy Mountains and Tasmania due to the dry nature of most of the continent. Despite continuing development in the wind and solar power generation sectors, over 85% of electricity produced in Australia today is still from coal-fired thermal power stations (Australian Coal Association 2008). There are three main power generation stations in the study area: Liddell, Bayswater and Redbank. Liddell and Bayswater power stations have had a long history in the area and had been PHRSTS participants since 1995. Redbank was established only in 2000 and is the other new development under the PHRSTS (the first being Bengalla coalmine). The remainder of this section will provide background to these power stations and their relationships with the coal mining industry in the Hunter Valley. The focus will be on the saline water generated by the power stations and its impact on the Hunter River salinity.

3.5.1 Macquarie Generation

3.5.1.1 General background

Macquarie Generation owns and operates the Liddell and Bayswater power stations which are located close together in the upper Hunter Valley (see Figure 3.6). They have been in service since the early 1970s and in the middle of 1980, respectively. In 1991 the two power stations formed a company called “Pacific Power”, which was taken over by Macquarie Generation, a state-owned corporation, in 1996 in line with reforms in the NSW electricity industry. With a combined generating capacity of 4,640 MW\(^5\) (2,640 MW from the Bayswater Power Station and 2000 MW from the Liddell Power Station), Macquarie Generation has become the largest power

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5 For comparison, the electricity generating capacity of the Snowy Mountains Scheme is 3756 MW (Snowy Hydro Limited 2007).
Macquarie Generation is the largest single user of the flow of the Hunter River, at a rate of some 60,000 ML per year (approximately 25,000 ML per year for Liddell’s operations, and approximately 36,000 ML per year for Bayswater’s operations), for steaming, cooling and other purposes (Macquarie Generation 2006). Macquarie Generation is restricted to extracting water from the Hunter River corresponding to the flood pulses in the River, or through special releases from Glenbawn dam in drought times. Although the Macquarie Generation’s water needs represent just over 5% of the total average annual flow available (100 year average) of the Hunter River, it is comparable to the total abstraction by all irrigators from the River (NSW EPA 1994a).

In addition to extracting water from the Hunter River, Macquarie Generation has the option of taking water from adjacent coal mines and pre-treating it or pumping it into Lake Liddell. However, at an operational cost of $100 per ML (excluding capital investment in pumps and
pipes for the coal mines) this option is regarded as too expensive to be economically viable (NSW EPA 1994a).

Salinity in the Hunter River has implications for the power stations’ water treatment practices and therefore the overall cost of using water withdrawn from the Hunter River and discharging wastewater to the Hunter River. By using a large volume of water withdrawn from the Hunter River for cooling and for producing steam for electricity generation, the power stations bring large quantity of salt into their systems. As described in Appendix 3, Macquarie’s two power stations use a desalinisation process to remove the salt from the water. The products of this process are treated water and extracted salt. While the treated water returns to the cooling tower for re-circulation and becomes more saline again, the salt accumulates on site. Creelman (1994) estimated that each day Macquarie Generation’s two stations have to store approximately 60 tonnes of salt, equivalent to around 20,000 tonnes of salt annually. Thus, if the power stations could uptake water with less salt, they would accumulate less salt. Conversely, if salinity increases in their uptake water from the Hunter River, they will be forced to handle greater volumes of salt. The NSW DLWC (2003: 29) estimated that the cost to Macquarie Generation of an increase of 10 EC in the salinity of the water extracted from the Hunter River would be in the order of $180,000 per annum.

Based on the above discussion, it may not be fair to regard the Macquarie Generation as a polluter, as the power stations actually reduce the net salt in the Hunter River through their water treatment process, and because the nature of the salt discharged from the power stations is exactly the same as that occurring naturally in the Hunter River. For example, in the 1999/2000 financial year Macquarie Generation reduced the net salt in the Hunter River by more than 5700 tonnes through its water treatment process (Macquarie Generation 2000). While it is unclear what happened to the solid salt produced by desalinisation, the prevailing methods for desalination plants dealing with the extracted salt include producing salt products or filling in land after it has dried out (see Appendix 3). Therefore, the main issue confronting the power stations is the salt load, that is, disposal of the extracted salt and discharge of the concentrated saline water off site. This is in contrast with the coal mines, where the main concern is the disposal of excess water with little concern about the salt load.
3.5.1.3 Discharge of saline water

Discharge of saline water from Macquarie Generation’s two power generations to the Hunter River (via Bayswater Creek) is currently regulated by the NSW EPA discharge licencing system and the HRSTS. In general, controlling the saline water from Lake Liddell is much easier than controlling the mine water of coal mines because of its 152 GL of storage capacity. As a comparison, Macquarie Generation discharged a total of approximately 27 GL of saline water into the Hunter River during the whole PHRSTS period from 1995 to 2002 (NSW DLWC 1995; 1996; 1997; 1998; 1999; 2000a; 2001a; 2002), which is far less than the storage capacity of Lake Liddell. The large capacity of Lake Liddell makes the withholding of saline water for Macquarie Generation flexible.

Also, in regard to the concentrated saline water, Macquarie Generation has the technical capacity to operate without discharging saline water off site. For example, based on the HRSTS Annual Report (NSW DLWC 2002), no salt was discharged by Macquarie Generation to the Hunter River in 2002. Even so, discharge of saline water could bring Macquarie Generation a significant benefit. The NSW EPA (2001: 6) estimated that the discharge of saline water could save Macquarie Generation between $100/ML and $400/ML in treatment costs.

3.5.2 Redbank power station – a new development under the PHRSTS

Redbank Power Station is a new development that has occurred since the inception of the PHRSTS. Joining the power generating industry in 2000, Redbank Power Station is the youngest coal-fired power station in the Hunter region. The normal capacity of electricity generation of Redbank Power Station is 130 MW (Redbank Power Station 2007). Redbank Power is also important to local development, producing over $10 million a year to the local economy and providing more than 250 employment opportunities (Redbank Power Station 2007).

The water needed for the operation of Redbank Power Station is sourced from controlled release of the Glenbawn and Glennies Creek dams, and this water is purchased from the DLWC. Similar to the Liddell and Bayswater power stations, Redbank also relies on the Hunter River

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5 This is estimated based on a total of 43154 tonnes salt into the Hunter River during the period of 1995 to 2002 (NSW DLWC 1995; 1996; 1997; 1998; 1999; 2000a; 2001a; 2002) and average discharge concentration of 3833 mg/L reported in Macquarie Generation’s Annual Return during the same time.
for water supply and wastewater discharge. The river water is pre-treated to produce clean water for Redbank’s operational uses. This water is recycled many times before it is discharged into the Hunter River. While the total treatment capacity is not known, it is known that cooling water is cycled throughout Redbank power generator at 400 ML/day, and the water is recycled as many as 14 times (Redbank Power Station 2007). Like the two Macquarie power stations, the salt that Redbank discharges to the Hunter River is the same salt that it draws from the Hunter River; nevertheless, the quantity of the salt discharged is less than that received from the River due to the salt being removed by the water pre-treatment. According to the HRSTS Annual Report (NSW DLWC 2001a; 2002), Redbank Power Station discharged a salt load of 227.8 tonnes in 2001 and 354 tonnes in 2002, accounting for 1% and 62%, respectively, of the total salt load discharged by the Hunter industries in the corresponding years.\(^7\)

Because it is a new development, information about the operation of Redbank Power Station is sparse. Despite some innovative features in this Station, it seems reasonable to assume that the basic operation system used by the Macquarie power stations also applies to Redbank Power Station, and therefore that the new Redbank Power Station will also receive significant benefits if it discharges the saline water to the Hunter River instead of treating it on site.

### 3.5.3 Relationship between coal mines and power stations

The power generating and coal mining industries in the Hunter Valley have a long-term, close relationship. This relationship can be viewed from the following two perspectives.

Firstly, the coal needed for the power stations is all sourced from the mines in the Hunter region. A demand-supply relationship between the power stations and the local mines has been maintained for many years. Macquarie Generation is the biggest coal consumer in the Hunter region. As a general rule for black coal generators, dividing GWh of electricity produced by two will give an approximate number of millions of tonnes of coal consumption (J. Devine [Public Affairs Manager, Macquarie Generation] 2003, email comm., 15 Dec). For example, 20 GWh is produced from approximately 10 million tonnes of coal. In 2006, 13 million tonnes of coal were consumed for the generation of electricity at its Liddell and Bayswater power stations (Macquarie Generation 2006). Statistical information from NSW DMR (1991; 1992; 1993; 1994; 1995; 1996; 1997; 1998; 1999; 2000; 2001; 2002; 2003) shows that the coal used to

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\(^7\) The total salt discharged by the PHRSTS participants in 2001 and 2002 amounted to 22337 tonnes and 571 tonnes, respectively.
produce electricity by the Liddell and Bayswater power stations is sourced from nearby coal mines. Five coal mines, Bayswater, Cumnock No. 1, Drayton, Hunter Valley Operations and Ravensworth-Narama, were the major coal suppliers during 1993 to 2002. Several other coal mines, such as Nardell, Ravensworth-east, Muswellbrook No.2 and Mt Arthur Coal, have entered the supply chain in recent years.

Secondly, the power stations and coal mines have combined to make a joint effort in water management. One example is that Macquarie Generation takes excess mine water from adjacent coal mines and treats the water to supplement their own water intake from the Hunter River. In this way, the coal mines reduce the need (and maybe the cost) for discharging saline water into the Hunter River, and the power stations obtain extra water for their operations when the river flow is too low to be extracted (Macquarie Generation 2003). Another example is Redbank Power Station delivers its treated water to the Warkworth Coal Preparation Plant for Warkworth’s operations (Redbank Power Station 2007). This relationship between the power stations and coal mines has had some effects on the PHRSTS credit trading, which will be discussed in detail in Section 5.3.

3.5.4 Agricultural effects on river salinity

In addition to industrial sources, agricultural activities can also result in saline water intrusion into river systems. In the case of the Hunter catchment, due to its specific soil and hydraulic conditions as noted in Section 3.1.1, this is likely to occur through dryland salinity and irrigation salinity. Dryland salinity occurs in non-irrigated areas where native vegetation has been replaced with smaller leaf canopy and shallow-rooted annual crops and pastures, or has been lost due to clearing, overgrazing or erosion. Irrigation salinity is usually caused by excess irrigation, inefficient water use and poor drainage in agricultural practices. In spite of the different causes, both dryland salinity and irrigation salinity have the same direct result of raising water tables and in turn have implications for river salinity. As water tables rise, more salt will be discharged into river systems and then carried downstream. Both dryland and irrigation salinity could contribute to increases in salinity in the Hunter River (NSW DLWC 2001b).

Irrigation practices in the Hunter Valley mostly rely on the main surface water, but much alluvial groundwater is also used to supplement surface water in dry periods (Creelman 1994).
This groundwater will enter into the river system with rainfall runoff, and in turn increase the likelihood of river salinity problems developing. This discharge, along with discharge from natural causes and dryland and irrigation salinity described, are the diffuse discharges of salt in the Hunter region. As these diffuse discharges are not currently regulated and not included in the PHRSTS (NSW DLWC 2001b), they are beyond the scope of this study.

3.6 Social impacts of the Hunter River salinity

Salinity in the Hunter River has negative impacts on agricultural production, urban and other industries' water uses and aquatic ecosystems.

Natural river salinity has few effects on aquatic ecosystems and human health. However, in the presence of elevated salinity, some negative impacts appear. Although understanding of the effects of salt on natural systems is generally poor, it has been recognised that adverse impacts, particularly on irrigated agriculture, are likely to occur when salinity concentrations reach 1500 EC over certain periods of time (NSW DLWC 2003). For some sensitive plants, such as rice and horticultural crops, the yield is affected by saline water at levels as low as 700 EC (Croft & Associates 1983). This may explain why 700 EC was used as the threshold of river salinity standard under the TD system (NSW EPA 1994a). Higher salinity will also restrict the potential uses of river water for livestock watering and human consumption. As well, higher salinity will incur additional water treatment costs for municipal and industrial water users. The costs of infrastructure operation and maintenance will rise due to increased corrosion and wear on facilities caused by high salinity (AGC Woodward-Clyde 1992).

Prior to the late 1970s, little was known about the impacts of mining activities on the water quality of the Hunter River. Though some knowledge was acquired in the course of preparing the Environmental Impact Statements for the mines (for example, Coal & Allied Operation 1980; Croft & Associates 1984) and from sporadic government surveys, it was only in the early 1980s that substantive studies (for example, Croft & Associates 1983; AGC Woodward-Clyde 1992; NSW EPA 1994a; Hunter Water Quality Task Group 1994) were undertaken. From these studies, it was recognised that industrial saline water discharged to the Hunter River without restriction could result in elevated salinity concentrations in downstream reaches and in turn cause negative impacts on agricultural production, urban and other industries, as well as aquatic ecosystems. It is difficult to separate the impacts of discharges from individual sources, but two studies have estimated the overall social cost of higher salinity.
The earliest attempt at estimating the cost relating to the elevated salinity in the Hunter River was that undertaken by Croft & Associates (1983). However, the focus of the study was solely on the impacts of salinity on agricultural outputs. It concluded that although the threshold tolerance level, the maximum level of salt that plants can tolerate without causing a reduction in yield, varied among different crops and growth stages, water with conductivity between nil and 700 EC can be used for irrigation of most crops on most soils with little probability of damage. The study also found that increases of 10% to 20% above these values may have little effect if accompanied by favourable weather conditions and appropriate irrigation management. These findings formed the basis for setting up salinity objectives for the Hunter River before 1995, as will be described in Chapter 4.

AGC Woodward-Clyde’s study (1992) was a subsequent attempt to estimate the damage costs of incremental salinity for the Hunter River. It used the damage parameters from the Murray-Darling Basin Commission’s SALTCOST model and proportioned them by the number of urban, non-urban and irrigation users of the Hunter River (AGC Woodward-Clyde’s study 1992). The study estimated that the annual cost of an increase of 1 EC, starting from a conservative threshold of 600 EC, was $10,000 in 1992 prices for the Hunter River, of which $7000 was from urban users, $1200 from non-urban users and $1800 from irrigation (AGC Woodward-Clyde’s study 1992).

While identification of the salinity impacts reported in the AGC Woodward-Clyde’s study (1992) seems more complete than that of Croft & Associates’ (1983), the evaluation of these impacts appears oversimplified in the AGC Woodward-Clyde’s study. For example, AGC Woodward-Clyde assumed a linear marginal social damage (MSD) function for the impact of river salinity. While this linear MSD function is appropriate when the river salinity is lower than a threshold of 1500 EC, beyond this threshold the MSD function may increase significantly due to catastrophic failure. Also, according to AGC Woodward-Clyde’s estimation, the cost of higher salinity to urban users is far more than that to non-urban users and irrigation. However, only the conflict between industries and agricultural irrigations has been widely documented in the literature (see Section 4.2). The possible explanation for this could be that the effect of salinity represents a higher cost per person for irrigators than for urban users.

There is still a substantial gap in the knowledge of the costs associated with elevated salinity in the Hunter River, and more research in this area is needed. Estimation of social cost associated
with Hunter River salinity will require substantial studies involving identification of damages and estimation of their market and non-market values. These studies will require extensive data and information and thus are well beyond the scope of this thesis. This study uses AGC Woodward-Clyde’s estimation to calculate the social benefits derived from the improvement of river salinity under the PHRSTS, as will be discussed in Section 6.5.

3.7 Summary

The Hunter River exhibits a high degree of variability in its stream flows. The River also displays a high level of salinity due to the catchment’s geological features. Whilst most of the salt in the River comes from natural sources, the discharge of saline water from mining and power generating industries contributes to the river salinity periodically. Salinity and volume of industrial effluent from coal mines in particular, flow and background salinity of the Hunter River and the impact of industrial effluent on other water uses of the River are all direct or indirect functions of rainfall. As such, they have an interactive and complex relationship. This has implications for management of industrial discharge, that is, both load of discharge and time of discharge need to be considered.
Chapter 4  Historical Perspective of Water Quality Management in the Hunter River

The last chapter described the features of saline water discharged from the coal mines and power stations in the Hunter Region and their impacts on the river quality of the Hunter River. This chapter will review how some of the water quality management strategies discussed in Chapter 2 have been applied in the management of industrial discharge in the Hunter region. The purpose of this chapter is to investigate the origins, evolution and institutional arrangements of the pilot Hunter River Salinity Trading Scheme (PHRSTS).

Section 4.1 gives a synopsis of the historical policies and initiatives for management of industrial saline water discharge in the Hunter region, and refers to some important events. It not only documents the history of people's understanding of the Hunter River, but also to some extent reflects the evolution of people's concepts of river quality management. Six main historical policies and initiatives will then be described in greater detail in Section 4.2 to Section 4.7. This helps understand the features of the PHRSTS and identifies the conditions of alternative management strategies against which the PHRSTS can be compared in this study.

4.1  Overview of strategies for water quality management in the Hunter River

The strategies for managing industrial saline water discharge in the Hunter region have evolved over many years and have involved many entities, from local industries up to and including state authorities. At least four significant historical phases are apparent: the inclusion of quality management in quantity management; the domination of direct regulation; the exploration of flexible management strategies; and the application of economic instruments and regulations.

4.1.1  Inclusion of quality management in quantity management

Australia is a drought-prone continent. For decades prior to the 1970s, water management Australia-wide focused exclusively on water supply to meet the demands of various water needs. Water resources planning and development meant looking only for alternatives to increase supply through managing droughts and floods. There was no, or certainly little, attempt
to investigate what variables actually affect water uses. Management of river quality received scant attention until the late 1960s (Day 1986).

The practices of managing water resources in the Hunter River reflected the trend nation-wide. Before 1970 emphasis on water quality problems in the Hunter River was not as strong as on droughts and floods, and little was known about the characteristics of surface and groundwater in the Hunter River. There was neither a specific administrative authority nor any legislation dealing with water quality issues in the Hunter Valley. Although the State Planning Authority (SPA)\(^8\) was established under the *State Planning Authority Act 1963*, its major responsibility was limited to static and detailed land use, and it had less concern for environmental outcomes caused by changes of land use (Day 1986).

In the late 1960s, with the increasing demand for water, the emphasis on water quality for diverse uses increased and legislation relating to the environment was separated from the planning legislation. The *Clean Water Act 1970* was enacted: as the first legislation prescribing standards for ambient river quality and discharge limits of pollutants at outfalls, this Act represented a milestone, in highlighting water quality management in addition to water quantity management. It was then replaced by the *Clean Water Act 1975* (Day 1986). At the same time, the State Pollution Control Commission (SPCC) was established and initially given responsibility for environmental impact assessments related to the administration of the Clean Water Act. Responsibility for administering the function of environmental impacts was then taken over by the Department of Environment and Planning (DEP) (Day 1986).

### 4.1.2 Domination of direct regulations

The 1970s and 1980s saw the domination of Command and Control (CAC) approaches in policy regimes for environmental quality management, at international, national and state levels. That was also the case for water quality management for the Hunter River. Subject to the Clean Water Act since the 1970s, the SPCC started to adopt a traditional licensing strategy to regulate industrial wastewater discharge to the Hunter River (Day 1986). This licensing system is known as *Trickle Discharge* (TD) because it is allowed industrial wastewater to be discharged into the nearest watercourse on a continuous, low-rate basis all year around, but with little dynamicism, that is, with little regard to changing river flow and background salinity conditions.

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\(^8\) State Planning Authority (SPA) was replaced by the Department of Environmental and Planning (DEP), then by NSW EPA and now by the NSW Department of Environment and Climate Change (DECC).
Chapter 4 Historical Perspective of Water Quality Management in the Hunter River

(NSW EPA 1994a; Smith 1995). The TD gradually incorporated dynamicism into the system as it evolved over time from volume-based control to load-based control and incremental control, but was still an approximate static non-tradable permit (SN) program in the notation of Table 1.1 in Chapter 1 of this thesis. The TD is used as a base against which the PHRSTS is compared in Chapters 6 and 7.

4.1.3 Exploration of flexible management strategies

In the 1980s, enormous effort was put into exploring flexible strategies for managing the salinity of Hunter River as knowledge of the implications of the discharge of industrial wastewater for the river quality increased. A concept with a strong dynamic dimension, Staged Discharge, was developed. The underlying philosophy of Staged Discharge is to use the natural dilution capacity of the river to reduce the impact of industrial wastewater discharges on other water uses of the river (Croft & Associates 1983). Cost factors and conditions of the river, for the first time, were considered in the development of salinity management strategies. Staged Discharge will be described in Section 4.3.

Several initiatives built on the concept of Staged Discharge were subsequently developed. These included Dilution Discharge (see Section 4.4), which was to reduce the impact of industrial wastewater discharge via river flow augmentation; the Three-tier Scheme (see Section 4.5), an integration of Staged Discharge, Dilution Discharge and Trickle Discharge; and Prioritised Discharge (see Section 4.6), a supplementary option which was to rank the firms' discharge priorities and allow them to share their discharge entitlements. The concepts of all these initiatives laid the foundations for the development and design of the later PHRSTS.

4.1.4 Application of economic instruments

PHRSTS represents a milestone of salinity management strategy moving from a traditional regulatory system to a market-based instrument (MBI). It is therefore important to examine what caused this movement.

One of the major factors that catalysed a shift of policy focus from a regulatory system to a MBI was the influence of parallel international trends of application of MBIs in environmental management regimes, instruments which gained popularity in the US where trading schemes
and other market-based programs gained credibility as an efficient way of delivering environmental improvements (Hahn et al. 2003). At the national level in Australia, the shift to including MBIs in the field of environmental management reflected the same trend. The 1990s saw MBIs begin to attract unprecedented attention of governments in Australia, both at federal and state levels, as potential tools for managing environmental problems (Brunton 1999; Environment Australia 1997; Whitten et al. 2003; Henderson and Norris 2008). Efforts were made to develop environmental policies that satisfied government and community aspirations for higher environmental standards without constraining economic development.

At the same time and under the same influence, an interest grew in NSW government in application of MBIs to achieve environmental objectives at reduced cost (Environment Australia 1997; Action Salinity & Water Australia 2002). The application of tradable permits in water quality management in the US generated strong interest in using tradable permits as part of the mix of instruments to manage the industrial saline discharge for the Hunter River.

Another major factor that provided the necessary impetus for introduction of the PHRSTSS was the desire of governments to solve the long-standing and increasing tension between local agriculture and industries over water quality in the Hunter River, and their recognition of the ineffectiveness of the existing regulatory system of the Trickle Discharge (TD). Governments expected the PHRSTSS to achieve the environmental objectives in a cost-effective way for the overall community.

A third major factor that further stimulated the development of PHRSTSS was the Murray Darling Basin Salinity Scheme, a limited form of a tradable permits program managing the river salinity through salt credits and debits (Environment Australia 1997). In fact, a number of design elements of the PHRSTSS were borrowed from the Murray Darling Basin Salinity Scheme (NSW EPA 1994a).

A fourth contributing factor was the growing body of quantitative information about the Hunter River flows and salinity, as discussed earlier. This information helped provide a better and broader understanding of the implications of industrial discharge for the river salinity and its impacts on other water users, which in turn helped identify a range of water quality objectives. This information was a preliminary step towards designing the PHRSTSS, which is a hybrid system that incorporates a tradable permits system into a dynamic permits system on the basis
of a regulatory system (see Chapter 1). Initiation of the PHRSTS will be described in Section 4.7.

Table 4.1 shows where the historical strategies and initiatives fit into the classes of discharge permits that were set out in Table 1.1.

### Table 4.1 Classification of the water quality management strategies for the Hunter River

<table>
<thead>
<tr>
<th>Static permits</th>
<th>Non-tradable permits</th>
<th>Tradable permits</th>
</tr>
</thead>
<tbody>
<tr>
<td>SN = static non-tradable permits</td>
<td>Trickle Discharge</td>
<td>ST = static tradable permits</td>
</tr>
<tr>
<td>Prioritised Discharge</td>
<td></td>
<td>DT = dynamic tradable permits</td>
</tr>
<tr>
<td>DN = dynamic non-tradable permits</td>
<td>Staged Discharge</td>
<td>HRSTS</td>
</tr>
<tr>
<td>Dilution Discharge</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

SN = static non-tradable permits
ST = static tradable permits
DN = dynamic non-tradable permits
DT = dynamic tradable permits
HRSTS = Hunter River Salinity Trading Scheme

### 4.2 Trickle Discharge

Trickle Discharge (TD) was the dominant control system that was used to regulate industrial saline water discharge in the Hunter region from the 1970s to 1994. The licence conditions for industrial saline water discharge evolved over time as the NSW EPA increasingly recognised and gathered more knowledge about the effects of industrial saline water discharge on the water quality of the Hunter River. This evolution was also motivated by the growth in industries' applications for discharge licences, prolonged periods of drought, and uncertain prospects for future water quality changes in the Hunter River (Day 1986). Three stages were apparent in the licence conditions: limits on discharge volume; limits on discharge volume and concentration; and ambient and incremental limits (Smith 1995). The durations of these stages are displayed in Table 4.2.
Table 4.2 Evolution of license conditions under the Trickle Discharge system

<table>
<thead>
<tr>
<th>Limits</th>
<th>1970s</th>
<th>1980s</th>
<th>1990s – Present</th>
</tr>
</thead>
<tbody>
<tr>
<td>Discharge volume</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Discharge volume and concentration</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ambient and incremental concentrations</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Sources: Adapted from Smith (1995) and D. Collins ([Executive Director Economics of the then NSW EPA] 2005, pers. comm., 30 Nov).

4.2.1 Limits on discharge volume and concentration

Initially, licenses only put a limit on the maximum daily discharge volume or/and discharge rate from a mine site. The reason for these limits was given as a need for reusing saline waters on site to reduce discharge. The actual limits were usually based on installed control capabilities, and were negotiable between the mines and the EPA individually (Day 1986).

The licence conditions evolved further to include the control of discharge load, which is a combination of the limits of discharge volume and concentration. Licences stipulated not only a maximum daily discharge volume, but also the maximum salinity concentration of discharge. This was to ensure river salinity levels did not exceed the defined value immediately downstream of the discharge points. Limits on the discharge concentrations were set to prevent the discharge of highly saline wastewater produced by successive recycling in coal washeries (D. Chapman [Catchment manager, Hunter Catchment Trust] 2004, pers. comm., 1 March). The actual defined value of the river salinity immediately downstream of the discharge points for licensing purposes, nevertheless, was determined on a site-by-site basis in discussion between the EPA, DLWC (then the Department of Water Resources) and individual mines (Smith 1995). Up to then, the TD was a static non-tradable permits (SN) program (according to the notation of Table 1.1 in this thesis).

However, one problem arose with the operation of the load control: although the licensees were entitled to discharge saline water to the river on a continuous “trickle” basis, the discharge was done with insufficient regard for the dynamicism of river flow, and ambient salinity status of contiguous or adjacent hydrological systems or of the cumulative effects. As shown in Section 3.2, the Hunter River experiences large variations in river flow and in-river background salinity.
Chapter 4 Historical Perspective of Water Quality Management in the Hunter River

At times of very low flow when the Hunter River itself presented a naturally high salinity background and the need for agricultural irrigation was high, the discharge from industries formed a significant proportion of the total flow of the river, and river salinity could increase above the level recommended for irrigation.

Throughout the 1980s, many farms reported significant reductions in pasture growth and even crop failure following irrigation with water drawn from the Hunter River. As this coincided with the growth phase of intensive mining activities in the Hunter region (see Section 3.4.1), the farmers believed that the mining industries should take the blame. The effects of increased salinity and the limits on the use of river water for agricultural activities brought local industries and the community into conflict (Harris 1998). In the late 1980s there had been increasing pressure from agriculture on industries and government to find ways of resolving the conflict. As the expansion of mining activities in the Hunter region continued, local agricultural and environmental organisations even called for a policy of Nil Off-Site Discharge to be applied to the new industrial developments. Considerable debates revolved around entitlement of discharge rights to the industries (E. Harris [Manager, Resource Assessment & Planning, Hunter Region, the former NSW Department of Natural Resources] 2003, pers. comm., 7 July).

In the meantime, mining industries in the Hunter region were also confronting a dilemma. While some of the more modern mines had a water management system that would be able to comply with Nil Off-Site Discharge, this was not possible at all mines in the area. Such factors as high groundwater seepage into mine workings, on-site coal washing, dust suppression and limited capacity to build storage dams meant that these mines had more water than they could use or recycle at the mine sites (NSW EPA 1994a; Smith 1995). Croft & Associates (1983) estimated that 11 of the 19 coal mines in the Hunter Catchment at that time needed to periodically discharge excess saline water off-site into the Hunter River. In this situation, the industries called for practical and equitable arrangements for their wastewater discharge (Harris 1998).

The responses of various parties to this call were animated. Among others, three events in this regard were notable throughout the 1980s and early 1990s. One was that substantial research was conducted on the effect of industrial wastewater discharge on river salinity. In the early 1980s, the NSW Minerals Council (then the NSW Coal Association) commissioned Croft &

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9 NSW Minerals Council is an industry association representing mineral exploration and production in the State.
Associates to carry out a study of salinity problems in the Hunter River and to find possible solutions for managing the industrial saline water discharge. The general conclusion of the study recognised the strong linkage between geology and salinity. The significant contribution of Croft & Associates' study (1983) was that it represented the first attempt to quantify the salinity inputs that come from excess mine water, natural salt discharge and agricultural irrigation. It was found that the contribution by the coal mines to the total salt load of the Hunter River was not significant overall, but the timing of saline water discharge was of concern. Based on this finding, Croft & Associates (1983) concluded that Staged Discharge, which was to schedule salt discharge at times of higher flow when the salt dilution capacity of the River was greater and the environmental and social impacts were minimal, was a cheap and viable means of disposing of excess saline water. Croft & Associates' study (1983) was one of the earliest and most influential studies of addressing the salinity associated with mining. It was also the first time that the concept of dynamic permits appeared in literature on the Hunter salinity management. The findings of Croft & Associates' study and the proposal for Staged Discharge became an important stepping stone for later studies and strategies for management of industrial saline water discharge in the Hunter River, including the HRSTS.

Another notable event was that the NSW DLWC (then DWR) began research into salt behaviour in 1990, aiming to gain an understanding of agricultural practices and the implications of industrial discharge for river salinity (Harris 1998). The research made at least two main contributions to water quality management of the Hunter River. One of which is the detection of *salt spikes*. Previously it had been believed that a linear relationship existed between river flow and river salinity. As part of the research, probes were installed at key locations to record salinity continuously. Detection of salt spikes from the observations helped explain the cause of the high salinity of the irrigation water; this information helped improve the relationship between mines and water users. Since then, both parties have realised that they have a common goal: management of salt (Harris 1998). The other contribution was helping identify the value of real time monitoring, which in turn promoted the development of a more extensive salinity monitoring network in the Hunter catchment that provided complete daily salinity records in the key gauging stations (Harris 1998).

The third important event that occurred in response to the conflict between the local agriculture and industries was that the NSW EPA (then SPCC) refined the licence conditions for discharge of industrial saline water over time. As each new industrial development was proposed, the NSW EPA set tighter limits on their saline water discharge, while existing industrial
developments were permitted to continue to discharge with less rigid restrictions (M. Bennett [Head of Regional Operations Unit, Newcastle Office of the former NSW EPA] 2003, pers. comm., 8 March). Bias against new market entrants was apparent.

4.2.2 Ambient and incremental limits

In the early 1990s, the NSW EPA started to impose ambient and incremental controls on new industrial developments. The new licences specified:

- the limit of maximum river salinity after discharge as 700 EC, and
- the limit of maximum allowable increase in river salinity caused by the discharge as 40 EC (NSW EPA 1994a).

The explanation for these changes was that setting an incremental limit was to prevent dischargers in the upper reaches of the River consuming the available discharge opportunities prior to the water reaching those downstream (NSW EPA 1994a).

Although the ambient and incremental limits were introduced into the licensing system, they did not completely replace the initial limits placed on discharge volume and concentration. Rather, the ambient and incremental limits were among the components of the licensing system (as Table 4.2 illustrates). Consequently, by 1994 the licensing system was not really a coherent system, but an accumulation of historical licence conditions made for various firms at many different times, and which had generally not been changed over time. Even if all new licences comprised ambient and increment limits, most licences were still subject to the limits of discharge volume and concentration (D. Collins [Executive Director Economics of the then NSW EPA] 2005, pers. comm., 30 Nov). Therefore, the overall licensing system was still close to a static non-tradable (SN) control system. Only very few licences contained ambient and incremental limits and these were related to river flow. Any one particular firm may be subject to any components of the licence conditions. The Trickle Discharge (TD) will be used as a base against which the PHRSTS is compared in Chapters 6 and 7.

The introduction of ambient and incremental controls was the first time in the history of Hunter River water quality management that quantitatively related the allowable discharge to the river flow conditions. Ambient and incremental control can clearly be seen as a version of dynamic discharge permits. However, this appears not to have been well recognised by the regulators.
For example, the NSW EPA (2001:7) noted that “...there was the system of combined ambient and incremental limits, known as the trickle discharge system: licences entitled licensees to discharge a small volume of saline water to the river at all times, regardless of river flow and salinity conditions.” This gives an impression that the ambient and incremental control is a kind of static permit system. In order to clarify the nature of ambient and incremental control, I examined it in a rigorous way (see Appendix 4). This examination revealed that although limits of ambient concentration (700 EC) and incremental concentration (40 EC) are static in number, because river ambient conditions change, their effects on allowable discharges may be highly variable. Therefore, the control of ambient and incremental concentrations virtually embodied the nature of dynamic discharge control.

The design of ambient and incremental control seemed conceptually appealing. However, further examination of the system reveals that it had at least two operational problems. The first problem is in regard to the equity of discharge. The ambient and incremental limits imply that when the in-river salinity had already exceeded the threshold of 700 EC, discharge was not permitted. Such a constraint did not reflect the actual flow conditions of the Hunter River and therefore resulted in few opportunities for discharge. This was particularly the case for those mines located in the lower reaches in which periods when background salinity of less than 700 EC occurred were not long. By examining daily observations of river salinity of the Denman (in the upper sector of the Hunter River), Glennies/Hunter (in the middle sector of the Hunter River) and Singleton (in the lower sector of the Hunter River) gauging stations from 1995 to 2002, I found that while a period of river salinity of less than 700 EC occurred at Denman more than 89% of a year, the periods at Glennies/Hunter and Singleton occurred approximately 40% and 50% of the year, respectively. Caution needs to be applied in the interpretation of these figures, however, because the daily salinity observations examined were actually the ambient salinity in the river after the input of industrial discharges, rather than the natural background salinity in the Hunter River. Even so, it is apparent that the opportunities for saline water discharge in the lower river sectors are far fewer than those in the upper river sectors. Thus, the equity of discharge entitlements for the industries is an issue.

A second operational problem, which was identified by the NSW EPA (1994a), is in relation to the difficulty of monitoring and detecting non-compliance. Due to the limited monitoring of the Hunter River and the time delay between discharge into tributaries and full mixing in the river, it was almost impossible to accurately establish the size of the salinity increase caused by a
particular discharge. Thus, the ambient and incremental controls were difficult to regulate effectively.

In spite of these problems, from the standpoint of river quality management, the evolution from the initial limits placed on discharge volume and concentration to the later ambient and incremental controls represented a forward step, from a static permit system towards a dynamic permits system.

### 4.3 Staged Discharge

As noted in Section 4.2, an increasing concern in relation to mining water discharge gave rise to Croft & Associates' study (1983) on the generation, treatment and disposal of saline mine water. The study examined various technological options available for either treatment or disposal of saline water, such as a pipeline to the sea, evaporation basins, deep well injection, and desalination, and concluded that all these options were either too costly or would produce a highly concentrated brine solution as a by-product, whose disposal off-site would still pose problems. Accordingly, Croft & Associates (1983) advocated tackling the impact of industrial wastewater discharge on river quality with a strong dynamic dimension. Thus, the strategy of Staged Discharge was initiated.

Under the Staged Discharge scheme, mine water was proposed to be discharged in a controlled manner so that all the irrigation requirements of agricultural activities, reservoir releases, discharges of coal mines and water extraction of power stations would be coordinated with rainfall, river flow and in-river salinity. The study by Croft & Associates (1983) claimed that the Staged Discharge would be the most economically and environmentally efficient solution to dispose of saline water from coal mines among all the other options mentioned above. However, the study did not provide any in-depth cost analysis of the various technical options to substantiate the claim.

Following the recommendations of Croft & Associates' study, in 1983 the NSW Minerals Council (then NSW Coal Association) approached the NSW EPA (then the SPCC) with a proposal for a joint study with the DLWC (then Water Resources Commission) to develop an operational scheme of Staged Discharge. However, the proposal did not proceed because it was decided at that time that the relatively insignificant contribution by the mines to river salinity

97
did not warrant the effort and expense of developing the scheme (D. Chapman [Catchment manager, Hunter Catchment Trust] 2004, pers. comm., 1 March).

In the early 1990s, the need to develop strategies for the disposal of excess mine water was re-emphasised, driven by a number of factors. On one side, mining in the Hunter region, at both existing and proposed mines, especially for the open-cut mines, was continuing to expand since the rapid growth in the 1980s. The proportion of open-cut mines to the total number of mines in the Hunter Valley increased from 26% in 1980/81 to 47% in 1992/1993 (NSW Minerals Council 2001). As noted in Section 3.4.3, open-cut mines are more prone to production of excess saline water than underground mines. With the general trend of existing open-cut mines going to greater depths, higher groundwater inflows were being encountered. The situation was further exacerbated as the whole Hunter catchment experienced above-average rainfall in the late 1980s and early 1990s (AGC Woodward-Clyde 1992). The coal mines thus faced increasing difficulty in managing saline water on site and were eager to find a resolution to disposing of the saline water off site. One the other side, the then dominant application of Trickle Discharge (TD) was not curtailing the increase of the river salinity (see Section 4.2). The environmental effectiveness of the TD was questioned and the more effective strategy of managing industrial discharges, such as Staged Discharge, was invoked.

In this context, the NSW EPA approached the NSW Minerals Council (then NSW Coal Association) to commission AGC Woodward-Clyde to assess the feasibility of Staged Discharge of saline water from the mines (AGC Woodward-Clyde 1992). AGC Woodward-Clyde (1992) concluded that Staged Discharge would be feasible if there were no incremental limits on river salinity placed on the firms. Given the difficulties in meeting both ambient and incremental limits on river salinity by Staged Discharge alone, the study also conducted an investigation into the effectiveness of Dilution Discharge, which was to use releases from the Glenbawn Dam to dilute mine water discharges. The study asserted that the diluting effects of Glenbawn Dam water would increase the opportunities for discharging saline water (AGC Woodward-Clyde 1992).

Both Staged Discharge and Dilution Discharge are versions of Dynamic Discharge, whose underlying philosophy is to allow more wastewater to be discharged when river flow is high. The only difference is that, while Staged Discharge uses the natural dilution capacity of the river, Dilution Discharge increases the diluting effects of the river by artificially increasing the river flow.
4.4 A trial of Dilution Discharge

In spite of AGC Woodward-Clyde's study indicating the effectiveness of Dilution Discharge, the Hunter's farming community was suspicious about the applicability of this option in practice (Harris 1998). Coincidently, almost at the same time (late 1992), several mines expressed concerns that they would not have the capability to store their excess mine water under the existing TD system if a significant rainfall event occurred (Harris 1998). Both the suspicions and this concern gave rise to a trial of Dilution Discharge organised by DLWC (then DWR) for one month from January to February 1993 (NSW EPA 1994a; Harris 1998). The following description of the Dilution Discharge trial is summarised from the post-trial report produced by the Hunter Water Quality Task Group (1994).

The objectives of the trial were multiple. The foremost objective was to test the feasibility of the method of hands-on control of mine water discharge combined with the release of clean water from the storage dams in the catchment. Other objectives included testing the accuracy and effectiveness of procedures being developed in the Hunter Valley to monitor and forecast water quality factors such as salinity, and encouraging community involvement in developing solutions to regional water quality problems (Hunter Water Quality Task Group 1994).

The trial was designed to coordinate the releases from Glenbawn Dam, the rate of water extraction by the two power stations owned by the Macquarie Generation (then Pacific Power), and the discharge of saline water from coal mines, so as to keep river salinity at the Singleton gauging station below a threshold of 900 EC (Hunter Water Quality Task Group 1994). This threshold was set as a result of extensive community consultation. It later became the river salinity objective in the FHRSTS (see Section 5.1). Benefits were anticipated for both power stations and coal mines. Because the releases from the dam have a relatively low level of salinity compared to the Hunter tributaries, this would allow the power stations to access water at times of low salinity, and in turn reduce the cost of water treatment and the cost of disposal of salt residues. For the coal mines, this could dilute the high-salt mine water, allowing mines to discharge excessive amounts of water and thence reduce on-site storage to manageable levels.

The releases of fresh water from Glenbawn Dam began on 20 January 1993 (Hunter Water Quality Task Group 1994). The water was jointly paid for by mining companies. Macquarie Generation also ordered a high volume of water from Glenbawn Dam, which coincided with the trial, to assist with dilution of the saline mine water. Extensive rainfall occurred after these
releases in the Hunter tributary catchment. This resulted in an excess of water within the system. The trial started with all mines discharging to maximum capacity, and gradually reducing as the flows from the tributaries fell (Hunter Water Quality Task Group 1994). To facilitate the trial, the NSW EPA relaxed the restrictions on daily discharge rates, and ambient and incremental limits as specified in the discharge licenses for the duration of the trial (Harris 1998).

The trial helped elucidate the impact of industrial wastewater discharge on river quality and greatly improved understanding of the cause-and-effect relationship of river salinity. A salt spike was clearly observed in the trial: at the beginning, the storm plus the release of dilution water flushed accumulated salt from the river streams, resulting in short, sharp increases in salinity levels. As the trial progressed, the dilution of in-river salt became more effective, since the water flows gradually stripped salt out of the river (Harris 1998). This observation indicated the best timing of purchasing dilution water, suggesting that a significant amount of dilution water could be saved if it was used to extend the time of natural high flow events, called "freshes", instead of being used at the time of low river flow events (E. Harris [Manager, Resource Assessment & Planning, Hunter Region, the former NSW Department of Natural Resources] 2003, pers. comm., 7 July).

The trial also indicated that the Dilution Discharge approach was very promising from an environmental perspective. It allowed 1005 ML of mine water, equivalent to 2233 tonnes of salt, to be discharged. This represented 15% of the total mine water that needs to be discharged into the Upper Hunter every year\textsuperscript{10} (Hunter Water Quality Task Group 1994). Notwithstanding the fact that the strategy was environmentally promising, the operational feasibility of Dilution Discharge depended upon a number of factors.

One of the factors was the amount of water required for the dilution versus the amount of water available from the dam. AGC Woodward-Clyde (1992) estimated that a release of some 30,000 ML from Glenbawn Dam (15% of the annual demand for water from the reservoir for all uses) would be required to dilute a discharge of 1,500 ML of mine water (33% of the total annual requirement for all coal mines). Smith (1995) estimated that an additional 100% and 160% over total water allocation in 1995 would be required to dilute the industrial discharge to a salinity level of 900 EC at the Singleton gauging station. This volume of dilution water would not be reliably available.

\textsuperscript{10} Based on this figures, I calculated that the total saline water the mines need to discharge is approximate 6700 ML/year with an average salinity concentration of 3703 μS/cm.
Another factor is weighing up the value of the reservoir water for diluting flows to coal mines against other alternative water uses. Dilution water is only needed when river flow is low and background river salinity is high while the industries have to discharge excess saline water. This is often the time when agricultural activities have a high need of water. Although there is no information on the value of the dam water to coal mines versus that to other alternative water users, there is no doubt that the use of water for diluting industrial wastewater during droughts itself could be politically sensitive. Even if the required volume of dilution water is available, concerns could arise that water downstream of mine discharge points would be unsuitable for higher value uses.

Even if the above two factors were not in question, the cost of the provision of dilution water to the industries could become problematic. It was estimated that the firms could incur costs in the range of $0.36 million/year (NSW EPA 1994a) to $3.35 million/year (Smith 1995), contingent upon the security of water supply, as well as threshold of the river salinity to be achieved and the inclusion of power stations. For these reasons, both the NSW EPA and the industries believed that Dilution Discharge could only be used as a supplementary component of a broad salt management strategy.

Even so, the trial had a number of merits. It helped shift the community's opinion into accepting the concept of real time management for river quality, which consequently promoted the development of a monitoring network across the whole Hunter catchment (Harris 1998; NSW DLWC 2000b). The result of the trial also made the community agree to a higher salinity threshold in higher river flows (see Section 5.1.1). The salinity target of 900 EC (instead of previous 700 EC) at Singleton has been adopted since then. In addition, the continuous monitoring records collected in the trial provided an invaluable scientific base for the development of an Integrated Quantity and Quality Model (IQQM) for the Hunter River (Hunter Water Quality Task Group 1994). The IQQM is now used to predict discharge events and estimate the daily salt discharge permits in operation of the HRSTS, as will be discussed in Section 5.1.

4.5 Three-tier Scheme

Immediately after the trial of Dilution Discharge, as one of a wide range of options for mine water disposal investigated by the coal mining industry, the NSW Coal Association (1993) put
forward a proposal for a Three-tier Scheme in the coal mines of the upper Hunter. The Three-tier Scheme was not put into force for unknown reasons, but as many of its design elements form the basic design of the latter PHRSTS, it is worth describing this scheme here. Much of the remainder of this section is drawn from the proposal by the NSW Coal Association (1993).

The underlying consideration of the Three-tier Scheme was to allow site-specific regulations to be set according to the location and nature of the discharge points, through integration of a high rate of discharge, Trickle Discharge (TD), and Dilution Discharge. In the Three-tier Scheme, TD was proposed for those mines with a relatively small volume of excess saline water and which therefore had a small capacity in their existing storage dams. For those mines with substantial discharge requirements, the rule of “high rate of discharge during freshes” would be applied (NSW Coal Association 1993).

The definition of discharge rate and flow threshold of a “fresh” thus becomes the central issue of this rule. In principle, a “fresh” should be defined in a way that, when it occurs, dilution capacity is high, background river salinity is low, and little use of river water is required for agricultural irrigation, so that discharge of substantial amounts of saline mine water will not significantly affect river quality. Clearly, the higher the river flow rate chosen, the less frequent and reliable its recurrence interval will be, and the fewer will be the discharge opportunities. In other words, river flow rates with more frequent recurrent intervals will provide greater reliability and a higher safety factor for the disposal of industrial wastewater, while rates with a lower recurrence will cause greater reliance upon other disposal options. Therefore, a balance needs to be found that will provide a reasonable degree of reliability, but still accord with the basic tenet of mine discharges during periods when irrigation demand is low.

In the proposal for the Three-tier Scheme, two criteria for defining the threshold of a “fresh” were recommended. First, a river flow rate was to be chosen so that it would recur at least three times per year in average rainfall years. Correspondingly, a high discharge rate was to be set to ensure that mines have the opportunity to dispose of all excess water on at least three occasions per year in an average year (NSW Coal Association 1993). Second, a high river flow rate was required to ensure that mine discharges could be carried to the sea within a relatively short time and would not pool or back up in the estuary region (NSW Coal Association 1993). Based on these criteria, a river flow rate of 2000 ML/day at Singleton gauging station, along with corresponding flow rates at other key control points, was proposed as the threshold to define a
"fresh" for the purpose of industrial discharge. These thresholds of river flows were later adopted by the PHRSTS to define the High flow periods (see Section 5.1).

In using this design, the frequency of freshes should be sufficient to permit total discharge three times per year in average rainfall years. During extended drought periods, however, the frequency of freshes is reduced, and this may not meet the requirements of industrial discharge. That is where the third element, Dilution Discharge, was proposed to be brought into the Three-tier Scheme. Dilution Discharge was included to serve as a safety valve during extended periods of low river flow. This would allow the mines to plan an overall site water management strategy with the degree of certainty necessary to justify the capital expenditure (NSW Coal Association 1993). Based on the proposal, two ways of managing dilution water were proposed for the Three-tier Scheme, building on the observations of the trial of Dilution Discharge. One way allowed mines to coordinate their discharges with special releases of water from the Glenbawn Dam, to meet the supply requirements of the two power stations of Macquarie Generation (then Pacific Power) (NSW Coal Association 1993). The other way allowed mines to purchase dilution water to extend the duration of naturally occurring freshes, to provide sufficient time for the mines to discharge all excess water. Dilution water could be purchased by either a group of mines acting in concert or by a single mine, provided that the mine(s) could demonstrate a need for wastewater disposal that could not be met in any other way (NSW Coal Association 1993). Either way, the timing and flow rate of individual mine discharge needed to be carefully managed by the DLWC (then DWR) on the basis of the information generated by the monitoring network.

4.6 Prioritised Discharge

The proposition of Prioritised Discharge, described in the following, had little direct relevance to the PHRSTS. I cite it here for completeness, and to show the exploration of alternatives and the broadening orientation that occurred to address the salinity issue for the Hunter River.

The proposition of Prioritised Discharges was as an alternative to Dilution Discharge if a long interval between events of high river flow were to take place whilst the mines were incapable of managing their saline water on site (NSW EPA 1994a). It was initially suggested by an irrigator in a meeting of stakeholders organised by the Hunter Catchment Management Trust in early 1990s (D. Chapman [Catchment manager, Hunter Catchment Trust] 2004, pers. comm., 1
March). This idea might have arisen as a result of the extraction licences regulating water uses of the Hunter River. While the water is supplied on a security basis, the priority of wastewater discharge for this option was proposed to be based on the degree of reliability of use of the discharge permits. That is, coal mines with high priority permits were allowed to discharge first. If the level of river salinity is still below a specified ambient limit or an incremental limit, those coal mines holding the next level of priority permits can then follow. The process continues on until the river salinity reaches the limit. Noticeably, mines with high priority permits with no immediate need to discharge could transfer their permits to other coal mines (NSW EPA 1994a).

The Prioritised Discharge conveyed the idea of sharing the entitlements of discharge. However, it left a series of design and operational questions open. For example, how many salt load (for example, tonnes/day) is a firm allowed to discharge? How can the order of discharge priority be determined? How do mines transfer their permits of higher priority to those with lower level of priority permits? Clearly, answers to these questions are closely associated with the issues of discharge rights: defining discharge rights; allocating discharge rights; utilising discharge rights; and transferring discharge rights. As will be described in Section 5.1, PHRSTS provided answers to these questions.

### 4.7 Initiation of the pilot HRSTS

The scope of any environmental policies is determined to a large extent by the political views of the key governments of the time. With the PHRSTS, there exists arguably a political impetus to establish a water quality trading scheme. The international and domestic trend of the increasing application of market-based instruments (MBIs) in the regime of environmental management gave the NSW governments a strong interest in introducing the PHRSTS. The NSW Cabinet Office coordinated a study by NSW EPA (1994a) to investigate the potential use of tradable discharge permits to manage industrial discharge in the Hunter Region. In early 1994, the NSW EPA conducted a scoping study to examine the applicability of an effluent trading scheme for managing salinity of the Hunter River. The scoping study clarified the aim of the HRSTS. Borrowing the operational information from the Murray-Darling Scheme, the scoping study also developed the framework for the HRSTS. Six months later, a draft operational plan for the PHRSTS followed (NSW EPA 1994b). The concepts from all the strategies described previously were taken into the draft operational plan and blended to produce the comprehensive mechanisms of the PHRSTS, which will be elaborated in Chapter 5. This made the PHRSTS a
unique dynamic tradable permits (DT) control system. After some refinements of the draft optional plan, a Guideline and Rulebook (NSW EPA 1995) was released, and the PHRSTS was established.

The PHRSTS started on 1 January 1995. It was initially designed as a one-year pilot program. The purposes of the one-year trial were multiple, including allowing the participating firms to become familiar with the Scheme and enabling them to make necessary alterations in response to the new regulatory system, testing the technical requirements of the Scheme and trialling the salinity credits (NSW DLWC 1995). However, it was expected from the outset that the Scheme would continue beyond the one-year pilot period subject to the modifications set down by the review of its operation (NSW DLWC 1996). When the operation of the Scheme in its first year was reviewed, it was decided that the pilot period would be for an additional year, given that there had been no high rainfall periods to adequately test the Scheme in 1995 (NSW DLWC 1996).

After that, the mechanisms of the PHRSTS were reviewed and further refined, and the pilot period of the Scheme was extended several times. On 1 December 2002, the PHRSTS was made permanent via the Protection of the Environmental Operation Hunter River Salinity Trading Scheme Regulation 2002 [Regulation 2002] (NSW EPA 2002). The PHRSTS thus ran from 1 January 1995 to 30 November 2002.

4.8 Summary

The strategies of water quality management for the Hunter River have evolved over time as people's understanding of the Hunter River and the needs of industry developed. Before the inception of PHRSTS in 1995, the industrial saline water discharges in the Hunter River were predominately regulated by the Trickle Discharge (TD) system. This system allowed industrial wastewater to be discharged on a continuous and low-rate basis, but with little regard to the river conditions and cumulative effects of salinity. Although the TD evolved over time incorporating dynamicism into the system, it still approximated a static non-tradable permits (SN) control system. TD provides a base against which the PHRSTS is compared in Chapters 6 and 7.
Over time, a number of strategies were proposed for management of river salinity of the Hunter River. These include Staged Discharge, which was to utilise the natural dilution capacity of the river to reduce the impact of industrial discharge; Dilution Discharge, which was to reduce the impact of industrial discharge via river flow augmentation; the Three-tier Scheme, an integration of Staged Discharge, Dilution Discharge and TD; and Prioritised Discharge, which was to rank the firms' discharge priorities and allow them to share their discharge entitlements. Although these strategies were never put into force, they laid the foundations for development and design of the PHRSTS.
Chapter 5  Mechanisms and Operations of the PHRSTS

This chapter consists of three sections. Section 5.1 describes and interprets the main mechanisms and rules of the PHRSTS. Section 5.2 examines the operational aspects of the PHRSTS. The focus of this section is the analysis of the credit trading market, drawing on the evidence on trading volumes and on the pattern of PHRSTS participants’ trading activities. Section 5.3 describes two credit auctions conducted in 2004 and 2006, and provides a preliminary analysis of the implications of the PHRSTS participants’ behaviour in the auctions and the credit trading markets.

5.1  Mechanism of the PHRSTS

The PHRSTS began operating 1 January 1995. Its stated objective was “to manage the saline water discharges so as to minimise impacts on irrigation, other water uses and on the aquatic ecosystems of the Hunter River catchment, at least overall cost to the community; in an equitable and flexible manner; and in a way that provides ongoing financial incentives to further reduce pollution” (NSW EPA 1995:1). To achieve this objective, the PHRSTS was designed to take the concepts from the strategies described in Chapter 4 and blend them to produce a comprehensive mechanism. PHRSTS rules have been well documented in the Guideline and Rulebook of Hunter River Salinity Trading Scheme (NSW EPA 1995). This section outlines and explains the main elements of these rules.

5.1.1  Main elements in the PHRSTS rules: sectors, flow periods and blocks

As stated in Section 3.4.5, both the amount and timing of industrial discharges are important to the salinity level in the Hunter River. Thus, discharge permits are ideally defined in a heterogeneous way over space and time. The PHRSTS achieved this by regulating the industrial discharges from both spatial and temporal perspectives, which are summarised in Table 5.1. A worked example of how the industrial discharge controlled by the PHRSTS is illustrated in Appendix 6.
### Table 5.1 Summary of PHRSTS salinity targets and discharge rules

<table>
<thead>
<tr>
<th>River Sector</th>
<th>Location</th>
<th>Control stations</th>
<th>River Flow (ML/day)</th>
<th>Salinity target (EC)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Upper</td>
<td>Muswellbrook - Denman</td>
<td>Denman</td>
<td>Low: &lt; 600</td>
<td>High: 600-2,000</td>
</tr>
<tr>
<td>Middle</td>
<td>Denman – Glennies Creek/Hunter River junction</td>
<td>Glennies/Hunter</td>
<td>Low: &lt; 800</td>
<td>High: 1,800-6,000</td>
</tr>
<tr>
<td>Lower</td>
<td>Glennies Creek/Hunter River junction - Singleton</td>
<td>Singleton</td>
<td>Low: &lt; 2,000</td>
<td>High: 2,000-10,000</td>
</tr>
</tbody>
</table>

Discharge rules
- No discharge
- Dynamic tradable discharge (subject to the river salinity objectives being met)
- Effectively unlimited discharge (subject to the maximum volumetric discharge rates being met)

Source: Adapted from NSW EPA (1995)

In regard to the spatial perspective, the river is divided into three sectors: Upper, Middle and Lower; the gauging stations at Denman, Glennies/Hunter River and Singleton are the control stations for these river sectors, respectively.

Each of the river sectors has different salinity objectives (NSW EPA 1995). There is no detailed discussion in the literature justifying these threshold values. NSW EPA (1994b) noted that the salinity objectives being built on the long-term averages of river salinity were derived from 25 years of monitoring records, advice from the NSW EPA and State Fisheries, and consultation with water users and the community. The limit of 900 EC on river salinity at Singleton, the furthest downstream control station, was established to ensure it would not exceed values that could detrimentally affect riverine ecosystem and irrigated crops grown within the area. Obviously, these objectives were set up based on Marginal Social Damage (MSD) function alone as discussed in Section 2.1.3. The apparent (but not real) relaxation in environmental standards that the 900 EC limit entailed, compared to the 700 EC limit under TD mentioned in Section 4.2, is discussed in Section 5.1.2.
Chapter 5 Mechanisms and Operations of the PHRSTS

The salinity objective of the Upper river sector (600 EC) is lower than that of the Middle (900 EC) and Lower sectors (900 EC). The rationale for this is two-fold: firstly, the ambient salinity at the reference point of the upper sector is the background salinity of the downstream sector. A stricter salinity objective is set for the upper sector to ensure that upstream releases will not restrict discharge opportunities for downstream locations (NSW EPA 1994b). In this way, environmental bias towards the downstream sources (see Section 2.2.4.2) could be avoided. Secondly, because of a comparatively lower salinity background in the upper sector, restrictions on the industrial dischargers would be easier than the restrictions on the dischargers further downstream (NSW EPA 1994b).

In regard to flow perspective, the river flows are divided into three periods: Low, High and Flood flow periods, the rules for which are explained in Section 5.1.3. The threshold flows that define these periods vary by river sectors (NSW EPA 1995), as shown in Table 5.1. The NSW EPA (1994b) explained that these threshold values were developed in close consultation with water users and the community. The values for High and Flood river flows, during which industrial discharges are permitted, were determined based on advice from irrigators that flows of this magnitude usually follow general rain in the catchment and that irrigation was thus not needed. It is documented in the HRSTS Annual Report (NSW DLWC 1995) that, initially, the threshold value of High flow at Singleton was set at 3000 ML/day. After observing the flow events, some farmers commented that 3000 ML/day was too high and could be lowered. The comments were then examined at the end of 1995 by the NSW DLWC, and the threshold of the High flow period for Singleton was reduced to 2000 ML/day (NSW DLWC 1995). Since then, this figure has been kept for the remainder of the period of the PHRSTS.

In regard to the temporal perspective, the river flow is divided into blocks. A block is the amount of water that the Integrated Quantity and Quality Model (IQQM) predicted will pass the Singleton gauging station during a 24-hour period (NSW EPA 1995). Thus, there are 365 blocks within a year. As flows combine from different tributaries along the course of the Hunter mainstream, blocks will not be consistently Low, High or Flood flow as they move down the river, and may last more or less than 24 hours. For example, a block classified as Low flow in the Upper sector could be classified as High flow in the Middle sector and then as Flood flow in the Lower sector as the river gains water going downstream.
5.1.2 Why are some salinity objectives laxer under PHRSTS than TD?

As noted above following Table 5.1, from the increase of the salinity ambient limit from 700 EC under the Trickle Discharge (TD) to 900 EC under the PHRSTS, it appears paradoxically that the salinity objective could be regarded as being laxer under the PHRSTS than under the TD. But this is not actually the case, for the following reasons.

Firstly, the salinity objective of 900 EC only applies to some river sectors (for example, the Middle and Lower river sectors) at particular periods (for example, the High and Flood periods). Whilst the limit on discharge of industrial saline water during the Low flow period is much stricter under the PHRSTS (that is, no discharge was permitted) than the TD (that is, continuous low-rate discharge was permitted), the salinity objective for the Upper river sector (that is, 600 EC) is stricter under the PHRSTS than the TD (that is, 700 EC).

Secondly, relaxing some salinity ambient limits from 700 EC under the TD to 900 EC under the PHRSTS does not mean a major increase in social damage cost. Figure 5.1 below depicts how the social costs change in response to river conditions. It shows marginal social damage (MSD) functions under different river flow periods: $MSD_L$ applies to Low river flow periods, and $MSD_H$ and $MSD_F$ apply to High and Flood river flow periods, respectively. $MSD_L$ lies above $MSD_H$ and $MSD_F$, showing that for any given level of ambient salinity in the River, social damages are greater in Low flow than in High and Flood flows because there are more water users (for example, irrigation) in the lower river flows than in the higher river flows. Furthermore, $MSD_L$ is steeper than $MSD_H$ or $MSD_F$, showing that the impact of the increase of one unit of salinity in the River in Low flow period will be greater than in High and Flood flow periods. This Figure helps understand the movement of river salinity standard from 700 EC all year round under the TD to 900 EC in High and Flood flow periods under the PHRSTS. While the social cost of 700 EC under Low flow might be much greater than under High/Flood flows, the difference in social costs of High/Flood flows between the river salinity of 700 EC and 900 EC could be modest.
5.1.3 Rules of discharge for Low, High and Flood flow periods

The PHRSTS's embodiment of the nature of dynamic permits is implemented through defining different discharge rules in three flow periods: no discharge in the Low flow period; limited discharge in the High flow period; and potentially unlimited discharge in the Flood flow period, as already shown in Table 5.1 and now explained in more detail.

For any High flow block, the PHRSTS limits the industrial salt load discharges at two levels. At the aggregated level, total allowable discharge (TAD) in tonnes/day is introduced to specify "the maximum amount of salt which, if discharged, would result in the river not exceeding 900 EC at Singleton" (NSW EPA 1995:13). The TAD is dynamically determined by the flow conditions and in-river salinity levels and thus varies on a daily basis. At the individual firm level, the salt load discharge is subject to the salt credits held by the firm, which will be discussed in detail in Section 5.1.4 below.
Although the PHRSTS itself allows its participants to discharge an unlimited amount of salt load into any Flood flow block, the discharges are still required to meet a prevailing condition set down in each discharge licence that is formally called the maximum discharge rate (NSW EPA 1995). This can be confusing, and we call it here the maximum volumetric discharge rate, because it is measured in ML/day, and specifies only the maximum daily volume of saline water that is allowed to be discharged by a licence holder from each licensed point, but does not limit the salt load in that volume. Maximum discharge rate is usually set for mines as several mines usually discharge into the same tributary. The discharges could come simultaneously from the mines and the upstream catchment. The maximum discharge rate is set in accordance with the physical features and capacity of the river or tributary stream at the discharge point to avoid erosion and flooding problems (M. Bennett [Head of Regional Operations Unit, Newcastle Office of the former NSW EPA] 2003, pers. comm., 8 March). The maximum discharge rate overrides the PHRSTS rules (NSW EPA 1995). This means, no matter what flow period the river is in and how many credits a discharger holds, the maximum discharge rate cannot be exceeded.

In addition to the discharge rules presented above, licensed dischargers are also allowed to purchase dilution flow, termed a bonus discharge entitlement, to supplement their discharges under the PHRSTS (NSW EPA 1995). This is a special release of water from the NSW DLWC's storage that creates an additional discharge opportunity for those dischargers who have extra discharge needs. To ensure the normal discharge opportunities are not diminished and the salinity objectives of each sector are not breached, the minimum amount of an entitlement should be sufficient to dilute the extra discharge (NSW EPA 1995). To this end, while the bonus discharge will not affect credit holders who are not involved in the purchase, the full benefit will only accrue to those who have paid for it.

### 5.1.4 Initial allocation of salt credits

The total allowable discharge (TAD) is a key determinant of the aggregate level of discharges from all the PHSRTS participants. The way that it is allocated can potentially change the way in which the participants operate on a day-to-day basis and affects decisions regarding investigation into saline water reduction. Under the PHRSTS, salt credits are used to facilitate the sharing of the TAD. In 1995, NSW EPA created 1000 credits in total for the whole river, each of which entitles discharge of 0.1% of the total allowable salt load discharge into a specific block of river flow (NSW EPA 1995). Based on the discharge rules stated above, credits are
only required for the discharges into High flow blocks, but not for Low and Flood flow blocks. Moreover, because the TADs of High flow blocks are themselves dynamically determined by the river flow and in-river salinity conditions, the currency of a credit, defined here as the amount of salt load that one unit of salt credit allows the PHRSTS participants to discharge, also varies from time to time.

As discussed in Section 2.3.5.1, the method of allocating the initial permits is the most important and controversial aspect in the design of an effluent trading program. Like most environmental trading programs (OECD 1999), the NSW EPA also adopted the grandfathering approach to allocate initial discharge permits at the beginning of the PHRSTS in order to increase industries’ acceptance of the program and to ensure a high degree of government control over the distribution of the allocation. There were 20 participants (19 coal mines and two power stations of Macquarie Generation11) at the start of the PHRSTS (NSW EPA 1994b). The NSW EPA allocated 800 of the total credits (80%) to these participants using the grandfathering approach (NSW EPA 1995). Rather than using historical discharge records as the criteria for allocating the credits to the participants, the NSW EPA developed a comprehensive score system that took into account the production outputs, the social contribution (that is, employment), environmental performance, discharge needs and the control capacities of each participant (NSW EPA 1995).

The distribution of initial allocation of salt credits is set out in Appendix 5 and summarised in Table 5.2. Among these 800 credits, 571 credits (57% of the grand total) were allocated to the coal mines and 229 credits (23%) to the two power stations. The NSW EPA reserved the remaining 200 credits (20%) for future use for new developments (NSW EPA 1995). By doing so, according to the NSW EPA’s point of view, the economic bias against new sources was avoided (NSW EPA 2001). The NSW EPA did not accept the alternative point of view that since all dischargers, existing or new, have to pay the same opportunity cost for a credit, there is no marginal bias against new sources. The reserved 200 credits also served as a buffer for safety provision to ensure the achievement of the river salinity objectives by the PHRSTS.

The NSW EPA has since then allocated 70 out of these 200 reserved credits to two new developments (33 credits to Bengalla Coal Mine in 1998, and 37 credits to Redbank Power Station in 2001) free of charge under the PHRSTS. Also, during the period of the PHRSTS the

11 Two power stations of Macquarie Generation are treated as one participant.
NSW EPA allocated a total of 43 extra credits to some existing participants, such as Nardell Coal Mine and Hunter Valley Coal Mine, due to their progression to advanced mining stages. As a result, by 30 November 2002 (end of the PHRSTS), NSW EPA only held 87 reserved credits. This had a potential for a net increase of salt discharge from the industries into the Hunter River, in turn diminishing the environmental effectiveness of the PHRSTS.

Table 5.2 Initial allocation of credits

<table>
<thead>
<tr>
<th>Entity</th>
<th>Allocation</th>
<th>Year of allocation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coal mines</td>
<td>571</td>
<td>1995</td>
</tr>
<tr>
<td>Macquarie’s two power stations</td>
<td>229</td>
<td>1995</td>
</tr>
<tr>
<td>Redbank Power Station</td>
<td>37</td>
<td>2001</td>
</tr>
<tr>
<td>Bengalla Coal Mine</td>
<td>33</td>
<td>1998</td>
</tr>
<tr>
<td>To some existing participants</td>
<td>43</td>
<td>2001</td>
</tr>
<tr>
<td>EPA in hand</td>
<td>87</td>
<td>30/11/2002</td>
</tr>
<tr>
<td>Total</td>
<td>1000</td>
<td></td>
</tr>
</tbody>
</table>


Importantly, when the credits were allocated, not all of the coal mines that participated in the PHRSTS held discharge licences. The mines without discharge licences were the modern mines whose water management systems were able to manage excess saline water on-site and thus did not need to discharge it off-site (EPA 1994b). However, these mines were still allocated the same amount of credits as the smallest positive discharge source (NSW EPA 1994b). Based on the information provided in HRSTS Annual Report (NSW DLWC 1995), among the initial 800 salt credits, 599 were allocated to licensed mines and power stations, and 201 to mines without discharge licenses. This had an immediate effect on the utilisation rate and value of salt credits, which will be discussed at length in Sections 5.2.2 and 5.4.3.

As stated in Section 2.3.5.1 on the design elements of a tradable permit program, the life of the permits ideally needs to be clearly defined when they are initially allocated. However, the lifetime of the initially allocated salt credits appeared ambiguous and uncertain, at least for the first few years of the PHRSTS. The main reason for this was that the pilot period of PHRSTS
itself was uncertain when it was introduced and involved several extensions over time. Uncertainty over the lifetime of the initial salt credits may have had an effect on the performance of the credit trading market: this will be examined in depth in Section 5.3.

5.1.5 Rules of credit trading

Tradability of the salt credits is the other principal component of the PHRSTS, in addition to their dynamicism. While each PHRSTS participant is entitled to discharge a share of total allowable discharge (TAD) according to the number of salt credits that they hold, they are allowed to trade their credits with others subject to NSW EPA approval. Thus, the salt credits traded in the PHRSTS trading markets represent a one-time right to discharge saline water into a certain water block of the Hunter River. This differs from the salt credits in the Formalised Hunter River Salinity Trading Scheme (FHRSTS) auction markets, which represent an ongoing discharge right to the Hunter River over a ten-year period (see Section 5.4.2). Based on the HRSTS Guideline and Rulebook (NSW EPA 1995), participation in trading by individual firms is voluntary. Trades can be permanent (for the remaining life of the credits) or temporary (for a specified number of blocks), but must be of whole credits and for whole blocks. There are no specifications related to banking or borrowing in the PHRSTS rules.

As described in Section 2.3.5.2, effluent trading has the potential to induce the risk of “hot spots” where discharges are concentrated. The PHRSTS applied a sector credit discount factor (NSW EPA 1995) to ensure that the credit trading does not result in the salinity objective being exceeded in any river sector. As the currency of a credit (that is, permits) is dynamically changed in response to the actual river flow conditions and in-river salinity and is defined in a combined time (that is, a block of water) and space (that is, river sectors) unit, the salinity objectives are binding at every point in time and at any place.

To sum up, the PHRSTS represents a mix of conventional command-and-control (CAC) regulations and market-based instruments (MBIs): while the dynamicism of discharge permits defines the time of discharge and the amount of total allowable discharge within the time frame, their tradability determines both who can discharge and the amount of discharge from individuals through an exchangeable permits system. At the same time, licence conditions specify the volume limits of discharge, and the option of bonus discharge allows extra discharge through the augmentation of river flow. The integration of all these components distinguishes
the PHRSTS from any other water quality trading programs throughout the world, such as Fox River Wisconsin, Tar-Pamlico North Carolina and Lake Dillon Colorado, Cheery Creek Colorado and Chesapeake Bay (Jarvie and Solomon 1998; Woodward and Kaiser 2002; Kraemer et al. 2004).

5.1.6 Monitoring and reporting

Administration of the PHRSTS involved the NSW DLWC, NSW EPA and HRSTS Steering Committee, each playing a different role in the operation of the PHRSTS. The DLWC provided river monitoring, modelling and registering. The NSW EPA provided licensing regulation, the credit register and an exchange facility. The main rules of the Steering Committee included overseeing the financing of the Scheme, technical issues related to discharge monitoring, review of scheme performance, and integration of the Scheme with other water management programs (NSW EPA 1995).

As noted in Sections 3.2 and 3.3, the DLWC now has an extensive monitoring network in the Hunter River. To manage the PHRSTS, the DLWC developed a comprehensive real time monitoring system, known as the Hunter Integrated Telemetry Scheme (HITS). Four basic parameters, river flow, river salinity, water temperature, and rainfall intensity, at 24 gauging stations throughout the Hunter catchments were monitored in 5-10 minute intervals. These data, along with the integrated Quantity and Quality Model (IQQM) developed by the DLWC, were used in the PHRSTS to assess and forecast tributary inflow routing to predict river flows and salinity and to identify discharge events and calculate the total allowable discharge (TAD) of High flow blocks. This information was promptly communicated to the firms through an electronic communication system (NSW EPA 2001).

The DLWC also measured the rates and salinity concentrations of the industrial discharges at the discharge points of individual firms. In the meantime, the PHRSTS participants were required to continuously monitor their discharge volume and salinity concentration at discharge points in the event of discharge. They were also required to monitor river flow and salinity upstream and downstream of the discharge points in the Hunter mainstream and tributaries (NSW EPA 1995). Supply of these monitoring data to the DLWC was compulsory to assist river modelling calibration and discharge opportunity predictions.

12 The steering committee comprised the representatives from licence holders, irrigators, NSW EPA; DLWC and the Hunter River Catchment Management Trust.
In addition to these monitoring data, each participant was required to submit Annual Returns to the NSW EPA for recording and auditing purposes. The Annual Returns consist of: (1) a Discharge Worksheet for each block into which discharge was made, which records the block number, number of credits available, salt discharge permitted, volume discharge limit, actual discharge starting and ending time, and actual discharge volume and concentration; (2) a Credit Transfer Form and Trading Record Sheet for each trade made, which document trading date, the number of river blocks, number of credits traded, and trading partners; and (3) a Discharge Record Sheet for each discharge, which contains information on block number, TAD of the block, number of usable credits, duration of the discharge, discharge volume, and salinity concentration (NSW EPA 1995).

The NSW EPA reserved the right to refuse to approve any credit trade that could lead to an excessive concentration of credits in any one sector of the river. The EPA was able to punish PHRSTS participants who failed to abide by the Scheme rules or the licence conditions relating to the Scheme by revoking, suspending or reallocating their credit holdings (NSW EPA 1995). However, no punishment has been documented in the HRSTS Annual Reports (NSW DLWC 1995; 1996; 1997; 1999; 2000a; 2001a; 2002).

It is important to note that the costs for the credits traded, if any, are not required to be reported. As will be discussed in Section 5.3 and 5.4, because the information on trading prices was not available, this study has used other information, such as the auction prices of salt credits generated from the 2004 and 2006 auctions under the Formalised Hunter River Salinity Trading Scheme (FHRSTS), to examine the PHRSTS trading market. Under the FHRSTS, trading prices are still not disclosed. The information on trading prices of salt credits, along with the information on auction prices, is important to be able to understand the PHRSTS trading market and the FHRSTS auction market. So it is in the public interest to make this information publicly available.

5.2 Operation and implementation of the PHRSTS

This section examines the operational performance of the PHRSTS over its lifespan from 1 January 1995 to 30 November 2002, focusing on occurrences of Low, High and Flood river flow periods, the distribution of the salt load discharges, transaction costs, and the credit trading
market. Specific examination of environmental and economic performances of the PHRSTS will be presented in Chapter 6 and Chapter 7, respectively.

When undertaking the formalisation of the HRSTS, the NSW EPA (2001) reviewed the PHRSTS from a number of aspects, including environmental outcomes, new developments and credit trading. However, the NSW EPA's review only focused on the facts, rather than analysing the reasons behind the facts. The examination of the PHRSTS in this section focuses on various aspects of the trading market. The new additional findings from this study will enrich the evidence on the PHRSTS operation and contribute to the continuing development of the Formalised Hunter River Salinity Trading Scheme (FHRSTS). They will also help to interpret the research findings from Chapters 6 and 7.

For the examination, the hydrological and water quality data, such as daily river flow and salinity at the Singleton gauging station, were obtained from the NSW DLWC. The discharge data (that is, daily total allowable discharge permits and daily salt load discharge of each PHRSTS participant) and credit trading data were all obtained from the HRSTS Annual Reports of 1995-2002 (NSW DLWC 1995; 1996; 1997; 1998; 1999; 2000a; 2001a; 2002).

5.2.1 Occurrences of river flow periods

Examination of the PHRSTS operation in this study starts with an evaluation of the accuracy of Integrated Quantity and Quality Model (IQQM)'s prediction of river flow periods. This is important as the operation of the entire Scheme is based on the accurate and reliable measurement of river flow and salinity, from which the prediction of discharge opportunities and issue of discharge permits is derived.

The qualitative flow periods (Low, High and Flood) chosen for the PHRSTS operation on the basis of IQQM predictions over the period January 1996 to November 2002 (that is, 2526 days in total) are available in the HRSTS Annual Reports of 1995-2002 (NSW DLWC 1995; 1996; 1997; 1998; 1999; 2000a; 2001a; 2002). Comparing these data with the actual monitoring flow data obtained from the DLWC, this study found that, as shown in Table 5.3, during 1996-2002, the average accuracy rate of river flow prediction was 93.7%. In other words, the error rate of river flow prediction was as low as 6.3%. Among these errors, 3.4% comprise those that predicted Low flows, but where the actual events were either Flood or High flows. Though reducing the discharge opportunities, such errors guaranteed river salinity objectives will not be
breached. In comparison, 1.8% of errors comprise those that predicted High or Flood flows, where the actual events were Low flows. Such errors are of concern because they are likely to cause the exceedance of river salinity objectives. Closer examination of the monitoring data of river salinity reveals that none of exceedance of the salinity objectives was caused by such prediction errors.

From the above discussion, the IQQM model is considered accurate in a qualitative sense. As the forecasted river flow data is not available, this study is unable to make any comment on its quantitative accuracy.

**Table 5.3** Comparison of predicted and actual daily river flows at Singleton gauging station

<table>
<thead>
<tr>
<th>Predicted river flows</th>
<th>Actual river flows</th>
<th>Number of days</th>
<th>%</th>
<th>Accuracy/Errors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flood</td>
<td>Flood</td>
<td>46</td>
<td>1.82</td>
<td>Accurate</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>11</td>
<td>0.44</td>
<td>Risky error</td>
</tr>
<tr>
<td></td>
<td>Low</td>
<td>4</td>
<td>0.16</td>
<td>Risky error</td>
</tr>
<tr>
<td>High</td>
<td>Flood</td>
<td>17</td>
<td>0.67</td>
<td>Safe error</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>172</td>
<td>6.81</td>
<td>Accurate</td>
</tr>
<tr>
<td></td>
<td>Low</td>
<td>42</td>
<td>1.66</td>
<td>Risky error</td>
</tr>
<tr>
<td>Low</td>
<td>Flood</td>
<td>4</td>
<td>0.16</td>
<td>Safe error</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>81</td>
<td>3.21</td>
<td>Safe error</td>
</tr>
<tr>
<td></td>
<td>Low</td>
<td>2149</td>
<td>85.08</td>
<td>Accurate</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td></td>
<td><strong>2526</strong></td>
<td><strong>100.00</strong></td>
<td></td>
</tr>
</tbody>
</table>

Source: Compiled from data provided by the NSW DLWC in 2003

This study has examined the daily observations of river flow at Singleton provided by the NSW DLWC in 2003. It shows that over a total of 2891 days in January 1995 to November 2002, approximately 88% of the time (2545 days) the River was in the Low flow period, during which no industrial discharge was allowed. Industrial discharge was allowed for only 12% of the time, of which 2.33% (68 days) was in the Flood flow period and 9.58% (278 days) in the High flow period. The days of Flood, High and Low flow periods in each year from 1995 to 2002 are summarised in Table 5.4.
Table 5.4 Distribution of river flow periods against years

<table>
<thead>
<tr>
<th></th>
<th>Flood flow</th>
<th>High flow</th>
<th>Low flow</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Days (%)</td>
<td>Days (%)</td>
<td>Days (%)</td>
<td>Days</td>
</tr>
<tr>
<td>1995</td>
<td>1</td>
<td>0.27</td>
<td>14</td>
<td>3.84</td>
</tr>
<tr>
<td>1996</td>
<td>8</td>
<td>2.19</td>
<td>12</td>
<td>3.28</td>
</tr>
<tr>
<td>1997</td>
<td>5</td>
<td>1.37</td>
<td>11</td>
<td>3.01</td>
</tr>
<tr>
<td>1998</td>
<td>26</td>
<td>7.12</td>
<td>84</td>
<td>23.01</td>
</tr>
<tr>
<td>1999</td>
<td>1</td>
<td>0.27</td>
<td>32</td>
<td>8.77</td>
</tr>
<tr>
<td>2000</td>
<td>21</td>
<td>5.74</td>
<td>58</td>
<td>15.85</td>
</tr>
<tr>
<td>2001</td>
<td>6</td>
<td>1.64</td>
<td>58</td>
<td>15.89</td>
</tr>
<tr>
<td>200213</td>
<td>0</td>
<td>0.00</td>
<td>9</td>
<td>2.69</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>68</strong></td>
<td><strong>2.35</strong></td>
<td><strong>278</strong></td>
<td><strong>9.62</strong></td>
</tr>
</tbody>
</table>


5.2.2 Salt load discharge

This study also examined the discharge of salt load from individual firms documented in the HRSTS Annual Report of 1995-2002 (NSW DLWC 1995; 1996; 1997; 1998; 1999; 2000a; 2001a and 2002). It found that during January 1995 to November 2002, the PHRSTS participants discharged an aggregated amount of 93420 tonnes salt load to the Hunter River. This represents 5.74% of the total salt in the River during the same period. The total salt in the River, denoted $S_R$, is calculated based on the monitoring data of daily river salinity and flow provided by the DLWC, using the following formula:

$$S_R = f \sum_{t=1}^{2891} (ec_t \times flow_t)$$ \[5.1\]

where $f$ is the conversion factor between EC and mg/L. In general, 1 EC = 0.6-0.68 mg/L. For the Hunter River, $f = 0.6$ applies (NSW EPA 1995). $ec_t$ is daily river salinity at Singleton (EC), and $flow_t$ is daily river flow at Singleton (ML/day). $t$ refers to any day during the PHRSTS. There were 2891 days in total during the PHRSTS from 1 January 1995 to 30 November 2002.

13 2002 refers to the period from 1 January 2002 to 30 November 2002.
The distribution of the aggregated salt load discharged from the PHRSTS participants among the river flow periods and across the years is provided in Table 5.5. Within this aggregated salt load, 53% was discharged in the Flood flow period, 43% in the High flow period and 4% in the Low flow period. The occurrence of industrial discharge in the Low flow period was due to the errors of flow event prediction shown in Table 5.3.

### Table 5.5 Actual salt discharge versus allowable salt load discharge

<table>
<thead>
<tr>
<th>Year</th>
<th>Low flow (L)</th>
<th>High flow (H)</th>
<th>Flood flow (F)</th>
<th>Total (L+H+F)</th>
<th>Total allowable discharge of High flow (tones) (TAD) = H/TAD</th>
</tr>
</thead>
<tbody>
<tr>
<td>1995</td>
<td>1050</td>
<td>1985</td>
<td>453</td>
<td>3488</td>
<td>13264</td>
</tr>
<tr>
<td>1996</td>
<td>1140</td>
<td>487</td>
<td>2580</td>
<td>4207</td>
<td>2403</td>
</tr>
<tr>
<td>1997</td>
<td>0</td>
<td>1315</td>
<td>2634</td>
<td>3949</td>
<td>5752</td>
</tr>
<tr>
<td>1998</td>
<td>36</td>
<td>8642</td>
<td>25055</td>
<td>33733</td>
<td>66573</td>
</tr>
<tr>
<td>1999</td>
<td>734</td>
<td>3837</td>
<td>894</td>
<td>5466</td>
<td>32152</td>
</tr>
<tr>
<td>2000</td>
<td>280</td>
<td>7572</td>
<td>11817</td>
<td>19669</td>
<td>66239</td>
</tr>
<tr>
<td>2001</td>
<td>80</td>
<td>16185</td>
<td>6071</td>
<td>22337</td>
<td>44562</td>
</tr>
<tr>
<td>2002</td>
<td>55</td>
<td>517</td>
<td>0</td>
<td>571</td>
<td>6204</td>
</tr>
<tr>
<td><strong>Total</strong> (tonnes)</td>
<td><strong>3375</strong></td>
<td><strong>40541</strong></td>
<td><strong>49504</strong></td>
<td><strong>93420</strong></td>
<td><strong>237149</strong></td>
</tr>
<tr>
<td>%</td>
<td>4%</td>
<td>43%</td>
<td>53%</td>
<td>100%</td>
<td></td>
</tr>
</tbody>
</table>


Comparing the daily total allowable discharge (TAD) and daily actual salt load discharge during the High flow period documented in the HRSTS Annual Reports of 1995-2002 by NSW DLWC (1995; 1996; 1997; 1998; 1999; 2000a; 2001a; 2002), this study calculated the utilisation rate of credits, which is defined as the percentage of the actual aggregated salt discharged by the PHRSTS participants on any High flow day over the TAD issued by the NSW EPA on that day. The result is presented in Table 5.5, showing that the average utilisation rate of credits during the PHRSTS was only 17%. The low utilisation rate may explain the low credit auction prices, which will be discussed in Section 5.4.3.
Chapter 5 Mechanisms and Operations of the PHRSTS

The low utilisation rate can be attributed to a number of factors. First, as the NSW EPA reserved 20% of the credit at the initial credit allocation (see Section 5.1.3), even if the PHRSTS participants had used their credits to their full extent, their utilisation rates of the total credits would have been only 80%. Second, as noted in Section 5.1.3, among the credits (800 credits) that were initially allocated to the PHRSTS participants, 25% (201 credits) were allocated to mines without discharge licences due to having no need for discharge. Some of these credits were not used during discharge events. This fact might be responsible for the low utilisation rate of credits to some extent. The above two factors are the result of the method used to allocate initial salt credits. The third factor is that the TAD was only required for discharges into High flow periods. As the majority (53%) of the salt discharges occurred in Flood flow periods when unlimited discharges of salt were allowed, the need for discharging salt load in a High flow period was reduced. This further decreased the utilisation rate of credits.

The low utilisation rate of salt credits eventually led to an initiative by the NSW EPA to reallocate the credits, which were initially allocated to the firms using the grandfathering approach, through periodic public auctions after 2002 under the FHRSTS (NSW EPA 2001; NSW EPA 2002). Section 5.3 will review the credit auctions in detail.

5.2.3 Transaction costs

As discussed in Section 2.3.5.4, information searching, bargaining and decision-making, and monitoring and enforcement are the three main sources of transaction costs associated with environmental trading programs. In the case of the PHRSTS, the major part of the transaction costs arose from the initial cost of setting up the monitoring network and installing an electronic communication system, as well as the continuing operational costs associated with all the administration activities described Section 5.1.6, such as monitoring, information flow and reporting.

In the first three years of the PHRSTS, the DLWC notified the PHRSTS participants of the discharge events and TAD through radio and telephone systems. NSW EPA processed credit trading based on paper-work (NSW DLWC 1995; 1996; 1997). Since 1998, the information about discharge events and TAD was released in a daily River Register maintained on the HRSTS official website, enabling participants to promptly identify their needs for discharge against their permitted discharge (NSW DLWC 1998). The EPA also developed a 24-hour online credit exchange programme. This assisted the firms in identifying potential trading partners.
quickly and easily and enabled the firms to trade their credits at any time in response to changing river flows and the need to discharge (NSW EPA 2003a). Provision of the on-line credit exchange programme has reduced the search and information transaction costs and in turn contributed to the active PHRSTS trading market.

The NSW DLWC and EPA bore the initial cost of setting up the monitoring network and installing the electronic community system, but information on these costs is not available. The PHRSTS participants were required to share the continuing operational cost in the form of a contribution fee based on their credit holdings (NSW DLWC 1995). Table 5.6 provides a comparison of contribution fees paid by the PHRSTS participants and the actual cost of operating the PHRSTS from 1997 to 2002. The contribution fee then moved from cost-sharing toward cost-recovery with the formalisation of the HRSTS. In line with the Regulation 2002, the operational cost of the Scheme has been shared equally between credit holders and discharge licence holders since 1 December 2002. The contribution fees in relation to savings in the damage costs and total control cost derived from the PHRSTS will be discussed in Sections 6.5 and 7.2.4.

Table 5.6 Sharing of transaction costs under the PHRSTS

<table>
<thead>
<tr>
<th>Financial year</th>
<th>Contribution fee ($)</th>
<th>Total credits of PHRSTS participants</th>
<th>Cost per credit ($/credit)</th>
<th>Actual cost of operating the scheme ($)</th>
<th>Cost recovery (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1997/1998</td>
<td>125,000</td>
<td>833</td>
<td>150</td>
<td>168,524</td>
<td>74%</td>
</tr>
<tr>
<td>1998/1999</td>
<td>125,000</td>
<td>833</td>
<td>150</td>
<td>179,175</td>
<td>70%</td>
</tr>
<tr>
<td>1999/2000</td>
<td>167,000</td>
<td>833</td>
<td>200</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>2001/2002</td>
<td>355,996</td>
<td>913</td>
<td>390</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>2002</td>
<td>200,200</td>
<td>913</td>
<td>219</td>
<td>200,200</td>
<td>100%</td>
</tr>
<tr>
<td>Total/average</td>
<td>973,196</td>
<td>913</td>
<td>221.8</td>
<td>NA</td>
<td>NA</td>
</tr>
</tbody>
</table>

NA = not available

Source: Compiled from data provided by the NSW DLWC in 2003

5.3 Credit trading market

An important reason for using discharge trading rather than conventional command-and-control (CAC) regulation of individual firms is that the flexibility provided by trading can reduce the overall cost of achieving the required reduction in aggregate discharge. The key to a successful
water quality trading market is the existence of a large number of pollution sources whose costs of pollution abatement vary from relatively low to very high for participating in the trading activities in a competitive manner. Accordingly, in examining the PHRSTS trading market in this study, two main questions are posed in relation to the functioning of the trading market:

- Did the PHRSTS trading appear active?
- Did the PHRSTS appear efficient? In other words, did the pattern of trades on the PHRSTS credit trading market appear to shift from the firms with higher MACs to those with lower MACs?

To answer the first question, this study examined the PHRSTS credit trading market from three aspects: volume of trades; the number of trading participants; and pattern of the trades. The analysis in this section was based on data drawn from the HRSTS Annual Reports of 1995-2002 (NSW DLWC 1995; 1996; 1997; 1998; 1999; 2000a; 2001a; 2002). However, it is difficult to accurately trace individual participants' trading behaviour for several reasons. One important reason is the inconsistency in the names of trading participants. The names of some participants documented in the trading records varied from year to year. While the parent companies' names were used in some years, the firms' names were used in other years. This situation became more complicated as from time to time some participants changed ownership, some changed their names, some merged with others and some ceased operations.

Another reason is the unclear documentation of credit movements in the trading records. Based on the trading rules stated in Section 5.1.5, credit trades can be permanent (for the whole life of the credits) or temporary (on the basis of blocks). While the vast majority of the actual trades were temporary ones between participants, some permanent credit transfers occurred between firms owned by the same parent company. In addition, while new firms were free to enter into the PHRSTS trading market through private negotiations directly with other firms, NSW EPA allocated 70 out of its 200 reserved credits to two new developments (37 credits to Redbank Power Station; 33 credits to Bengalla Coal Mine) free of charge (see Table 5.2). Also, during the period of the PHRSTS the NSW EPA allocated a total of 43 extra credits to some existing participants (see Table 5.2). The fact that these credit movements were not explicitly indicated in the trading records increases the difficulty of using the data for this analysis.

14 A parent company may own several firms.
This study made every attempt to reconcile inconsistencies with the data and to present the most accurate portrait of the PHRSTS trading behaviour. For example, the changes of names and ownership of participants were clarified through cross-checks with documents in the NSW Coal Profile (NSW DMR 1995; 1996; 1997; 1998; 1999; 2001), their annual reports and website, as well as the NSW Minerals Council (P. Smith [Assistant Director, Environment, NSW Minerals Council] 2003, pers. email. 16 September). Some “suspicious” trading records were cross-checked with the original trading data spreadsheet obtained from the NSW DLWC and the participants’ Annual Returns stored at the NSW EPA Newcastle office.

Before moving on to the detail of the analysis, note that in this analysis of the trades, a single trade refers to one transfer of credits from one participant to another. Various trading activities were shown on the trading records. For example, there were cases where one participant transferred a group of credits belonging to more than one block (known in the trade as “bulk” of credits) to another participant. In these cases, it is defined as one trade. There were also occasions where one participant transferred credits on the same block to several other participants. On these occasions, this is defined as several trades. In this analysis, a permanent credit transfer is also treated as one trade. Table 5.7 presents the number of trades and credits traded in each year during the whole lifespan of the PHRSTS, based on data drawn from the HRSTS Annual Reports of 1995-2002. The numbers of High-flow and Flood-flow days are also included in the Table for comparison purposes.

Table 5.7  Trading records in 1995 -2001

<table>
<thead>
<tr>
<th>Year</th>
<th>Number of trades</th>
<th>Number of credits traded</th>
<th>Flood flow days</th>
<th>High flow days</th>
</tr>
</thead>
<tbody>
<tr>
<td>1995</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>14</td>
</tr>
<tr>
<td>1996</td>
<td>2</td>
<td>55</td>
<td>8</td>
<td>12</td>
</tr>
<tr>
<td>1997</td>
<td>0</td>
<td>0</td>
<td>5</td>
<td>11</td>
</tr>
<tr>
<td>1998</td>
<td>20</td>
<td>904</td>
<td>26</td>
<td>84</td>
</tr>
<tr>
<td>1999</td>
<td>18</td>
<td>953</td>
<td>1</td>
<td>32</td>
</tr>
<tr>
<td>2000</td>
<td>31</td>
<td>1540</td>
<td>21</td>
<td>58</td>
</tr>
<tr>
<td>2001</td>
<td>69</td>
<td>2638</td>
<td>6</td>
<td>58</td>
</tr>
<tr>
<td>2002</td>
<td>16</td>
<td>720</td>
<td>0</td>
<td>9</td>
</tr>
<tr>
<td>Total</td>
<td>157</td>
<td>6811</td>
<td>68</td>
<td>278</td>
</tr>
</tbody>
</table>

The above table shows few trades occurring during the first three years of the PHRSTS. This observation has been explained by the participants’ lack of experience with the Scheme (NSW EPA 2001), which is generally supported by the experience of Australian water trading market where trading activities in initial years were inactive and increased rapidly as irrigators gained familiarity of the market (Young 2004). However, at least other two possible reasons exist for this inactivity of the credit trading market of the PHRSTS. One is the participants’ uncertainty about the lifetime of salt credits in the initial years and about the evolution of regulatory rules, as noted in Sections 4.7 and 5.1.4. The experience of Dutch Nutrients market has suggested that policy uncertainty in the early stage of the Nutrient Quota System, such as the uncertainty of continuance of the quota system and of future constraints on quota use, led to a wait-and-see attitude of the participants, limiting the tradability and impeding the efficient functioning of the quota market (Wossink 2004). The uncertainty of the salt credits and the Scheme itself in the first three years of the PHRSTS could inspire caution of the participants towards the trading market.

The other reason for the inactivity of the credit trading market at the initial stage of PHRSTS could be the lack of the need to discharge. An evaluation of Australian water trading market (Bjornlund and Rossini 2005) has shown that the volume of water allocation trades for temporary water rights heavily depended upon seasonal conditions, such as rainfall and river flow. The rationale for this factor in the case of the PHRSTS is that the few Flood-flow and High-flow days between 1995 and 1997 suggest fewer high rainfall events in the Hunter region over this period. Fewer High-flow days could lead to a lower generation of saline water, in particular for the open-cut mines and thus less need for industrial discharge.

No trade in 1997 is also noticeable in Table 5.7. NSW EPA (2001) commented that a lack of discharge opportunities in 1997 accounted for no trade occurring in that year. However, a closer examination of river flow data of 1997 in this study, which shows 11 days of High flow period in that year (compared to 9 High-flow days with 16 trades in 2002), does not support the NSW EPA’s assumption.

From Table 5.7, it appears that the number of trades and credits traded has substantially increased since the fourth year of the Scheme. At least three factors have contributed to this increase: participants’ increasing experience with the Scheme; the establishment of an online trading system, which has lowered the transaction costs and time delay and in turn induced
more trades (see Section 5.2.3); and the increasing need for discharge from the industries due to higher rainfall events, for example, in 2000 and 2001.

The implication of the number of High-flow and Flood-flow days for the number of credits traded is of interest. Intuitively, the number of credits traded may have an inverse relationship with the number of Flood-flow days. The intuition is that the longer the duration of a given Flood flow period, the fewer trades would occur in the following High flow period, due to unlimited discharge in Flood flow and thus less need of discharge in High flow period. Conversely, the number of credits traded may have a positive relationship with the number of High-flow days. More trades would be expected over a longer duration of High flow. However, closer examination of Table 5.7 reveals that the number of transactions is neither related to the number of High flow days nor to the number of Flood flow days. This reflects the complexity of the factors determining the needs for discharge of the industries as discussed in Section 3.4.3

In total there were 157 trades involving 6811 credits traded during the seven-year span of the PHRSTS, which is on average 45 credits per trade and approximately 22 trades per year. The most successful US water quality trading programs in terms of the number of trades have been the Grassland Area Farmers (39 trades over a two-year period, that is, approximately 20 trades/year), Long Island Sound (63 trades over a two-year period, that is, approximately 32 trades/year), Truckee River (33 trades over an eight-year period, that is, approximately 4 trades/year) and the Red Cedar River (22 trades each year since 2001) (Morgan and Wolverton 2008). Compared with these programs, the number of trades in the PHRSTS may not be very impressive. However, considering that during 1995-2002, on average less than 10% of the time (278 days) was a High flow period, in which both discharges and trades were allowed, the PHRSTS trading market appeared quite active.

Another indicator of the activity of the trading market is the number of participants. The trading records showed that by 2002, 20 out of 22 PHRSTS participants\(^\text{15}\) had been involved in credit trading activities, as shown in Table 5.8, which also gives further information about the trading participants. This shows that the majority of firms (ten mines) were involved in credit trading only as sellers and never as buyers. Only one firm was a buyer but never a seller of credits: the new entrant, Redbank Power Station. A significant proportion of firms (eight mines and one power station) were both sellers and buyers, of which five were net sellers (more credits sold

\(^{15}\) The initial 20 participants in 1995 plus two later new developments
than bought), three were net buyers (more credits bought than sold), and one bought and sold
the same amount of credits.

Table 5.8 Information about the PHRSTS credit trading participants

<table>
<thead>
<tr>
<th>PHRSTS participants</th>
<th>Only sellers (mines)</th>
<th>Only buyers (power station)</th>
<th>Both sellers and buyers (8 mines and 1 power station)</th>
<th>Neither sellers nor buyers (mines)</th>
<th>Total (20 mines &amp; 2 power stations)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Upper</td>
<td>2</td>
<td>0</td>
<td>2</td>
<td>2</td>
<td>6</td>
</tr>
<tr>
<td>Middle</td>
<td>4</td>
<td>1</td>
<td>5</td>
<td>0</td>
<td>10</td>
</tr>
<tr>
<td>Low</td>
<td>4</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>6</td>
</tr>
<tr>
<td>Mining methods</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Open-cut</td>
<td>7</td>
<td>0</td>
<td>7</td>
<td>0</td>
<td>14</td>
</tr>
<tr>
<td>Underground</td>
<td>2</td>
<td>0</td>
<td>1</td>
<td>2</td>
<td>5</td>
</tr>
<tr>
<td>Open-cut/underground</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
</tbody>
</table>


Table 5.8 also shows the distribution of the trading participants among river sectors. All the firms located in the middle and lower river sectors participated in the credit trading activities. Two firms that were never involved in trading were the mines located in the upper river sector. This suggests that the firms located in the middle and lower river sectors tend to have more difficulty in managing their wastewater on site and thus tend to derive more benefit from the trading market.

A closer look at the mining methods of the 20 coal mine participants in Table 5.8 shows that all the open-cut mines had participated in credit trading. The two mines that had never been involved in any trading activity were underground mines. This observation reflects the situation noted in Section 3.4.3, that open-cut mines tend to generate more saline water on-site than underground mines, and thus experience more difficulty in managing saline water on-site than underground mines. This observation also suggests that open-cut mines might have derived more advantages from the trading market than underground mines.

In terms of the relationships between the trading parties, trading in the PHRSTS market can be grouped into three categories: intra-company trades, which refers to the trades between the
Chapter 5 Mechanisms and Operations of the PHRSTS

mines owned by the same parent company\textsuperscript{16}; \textit{arm-relationship trades}, which occur between the power stations and their coal suppliers; and \textit{inter-company trades}, which refers to trades between the mines owned by different companies, and those between the power stations and the mines that had no supply relationship with the power stations. Among the total 157 trades during 1995-2002, half were intra-company trades, followed by inter-company trades (44\%) and arm-relationship trades (6\%). The high proportion of intra-company trades can be attributed to high cross-company ownerships of the coal mines. As noted in Section 3.4.1, more than 63\% of the coal mines belonged to two big companies: Xstrata Coal and Coal & Allied.

To answer the second question of “did the pattern of trades on the PHRSTS credit trading market appear to shift from the firms with higher MACs to those with lower MACs”, obtaining information about saline water control cost at individual firm level is essential. However, this information was not available and thus prevented this study from examining the implications of firms’ control cost for their behaviours in the credit trading market. As an alternative, the study attempted to use the auction prices of the credits to conduct the examination because the auction prices at least provide an indication of the magnitude of the firms’ saline water control costs. For this reason, although FHRSTS is mostly beyond the scope of the study, the next section will conduct a preliminary analysis on the credit auctions of 2004 and 2006. A credit auction was also conducted on 2 April 2008; however, the information on this auction was not available until 18 September 2008, after this cut-off date for detailed analysis, so only brief notes on the 2008 auction are included below.

\textbf{5.4 Analysis of the 2004 and 2006 Credit Auctions}

The Pilot Hunter River Salinity Trading Scheme (PHRSTS) faced a number of risks and pressures in its later stage of implementation. Availability of the credits and efficiency of the credit use was the main concern. On one hand, the NSW EPA faced a high possibility of running out of credits for new developments within a short time if the grandfathering system for credit allocation continued to be used (NSW EPA 2001). For example, by 30 November 2002 NSW EPA only held 87 reserved credits, comparing with the initial 200 reserved credits (NSW DLWC 2002). On the other hand, a large number of credits, which were initially allocated to those mines without discharge licences, remained unused during discharge events, and some

\textsuperscript{16} See Section 5.1.3, salt credits were allocated to firms (that is, coal mines and power stations) rather than companies. There are a number of large companies that operate more than one firm.
were never used at all (NSW EPA 2001). The utilisation rate of credits during the period of PHRSTS was as low as 17%, as found in Section 5.2.2.

One of the key changes that accompanied the introduction of the Formalised Hunter River Salinity Trading Scheme (FHRSTS) by the Regulation 2002 was a re-allocation through periodic public auctions of the salt credits that were initially distributed to the PHRSTS participants by means of grandfathering. Another key change was increasing the Flood flow thresholds in the Upper and Middle river sectors to address the anticipated impacts of the increasing industrial growth in the upper Hunter River on the attainment of river salinity objectives in these sectors (NSW EPA 2002).

This section provides a preliminary analysis of the credit auctions held in 2004 and 2006 under the FHRSTS. The purpose is to use this analysis to explain firms' behaviours in relation to the PHRSTS credit trading market and to help interpret the findings reported in Chapter 7 on the economic performance of the PHRSTS. This section contains three subsections. Section 5.4.1 will provide a brief overview of the key formats for auctioning multiple units of homogeneous items and compare their properties, drawing on the auction literature. This subsection provides knowledge to help understand the FHRSTS credit auctions. Section 5.4.2 will outline the key elements of the FHRSTS credit auctions. Section 5.4.3 will provide a simple analysis of the outcomes of FHRSTS auctions. Instead of analysing firms' behaviour in the actions (such as, strategic bidding and market domination), which falls into the field of game theory, this subsection focuses on the implications of the observations on the auctions for those in credit trading markets. This subsection will also discuss the implications of auction prices for market prices and firms' marginal abatement costs (MACs) of saline water control.

5.4.1 Key auction formats

Auctions have been used as a form of economic transaction for a long time. Auctions are being increasingly used as an economic instrument in natural resource and environmental management. Such auctions have included a diverse array of rights, such as timber harvests, for example, US. Forest Service timber auctions (Athey and Levin 2001); resource extractions, for example, fishery and water (Chan et al. 2003); emission allowances, for example, US SO2 emission permits (Svendsen and Christensen 1999), carbon permit auctions (Cramton and Kerr 2002); and discharge permits, for example, FHRSTS salt credits (NSW EPA 2002). Auctions have also been used by governments to meet conservation goals. Examples include the US
Conservation Reserve Program Auctions (Vukina et al. 2006) and Australian Bush Tender Trial in Victoria (Stoneham et al. 2003).

For any auction, achieving allocative efficiency and maximising revenues are the two fundamental goals, where allocative efficiency is measured by whether or not the auction results in the items auctioned being distributed to those who value them the most. The efficiency and revenue-raising capacity of an auction are closely related. It is often the case that an auction with a high level of efficiency will also generate high revenues, and vice versa (Chan et al. 2003). For an environmental auction where an active spot market for an environmental resource does not exist or where the information on market values or user costs of resources is not readily available, as in the case of HRSTS, price discovery is often an additional primary goal of the auction. The auction prices could provide an indication of these values and costs (Chan et al. 2003).

Auctions can be one-sided where there is one seller or two-sided where there are several buyers and sellers. The items auctioned can be a single indivisible unit; or multiple homogeneous or heterogeneous units. In a multi-unit auction, the bidders confront more complicated decisions than in the single-unit auction settings. Rather than simply deciding whether to buy and at what price, the bidders must decide which goods to buy, how many of each and at what prices (Klemperer 1999). Since a fixed number of identical salt credits are sold in each single FHRSTS auction, the following review focuses on the multi-unit homogeneous auctions.

The approaches of conducting multi-unit homogeneous auctions can be categorised in two basic dimensions. The first dimension is the number of auction rounds. In a single-round auction (known as a sealed-bid auction), there is only one round of bidding in which the bidders simultaneously submit their demand quantities at one or more price levels without knowing the bids of other bidders. These bids are added to form the aggregated demand curve. The intersection between the demand curve and available supply determines the clearing price and allocation. In a multiple-round auction (known as open auction), an open, iterative bidding procedure allows bidders in each bidding round to express a series of bids, taking into account the information revealed by other earlier bids. Ascending price and its ascending-clock variant are the most prominent open bidding procedure. In this type of auction, the price is raised until the final round is reached or the demand meets supply (Klemperer 1999).
The second dimension is pricing rules. There are two common pricing rules. *Uniform pricing* requires all the winners to pay the same price, that is, the clearing price, for every item they acquire, while *discriminatory pricing* (also called *pay-as-you-bid*) requires successful bidders to pay their own bid prices (Klemperer 1999). Combinations of these two dimension forms four common auction formats: uniform price sealed-bid, discriminatory sealed-bid, uniform price open and discriminatory open. These auction formats have different ranks against the criteria of achieving economic efficiency, raising revenue and revealing the true values of the auctioned items in the setting of incomplete information (Klemperer 1999).

Incomplete information is a key characteristic of almost any auction environment. The bidders might not know how much the items are worth. In the literature, two basic models regarding bidder’s information have evolved and their approaches are very different (Klemperer 1999). The *private-values model* applies to the situation where each bidder knows exactly the worth of the item to her but may not know how much it is worth to others. In this situation, the valuations of individual bidders are considered different, and one’s valuation does not depend upon others’ valuations. The *common-values model* applies to the situation where the item auctioned is worth the same or very similar to every bidder but no one knows that value with certainty. In this situation, a bidder’s value may depend to some extent on other bidders’ signals (Klemperer 1999). In reality, it is quite often the case that an auction has characteristics of both private values and a common value (as the case of the FHRSTS auctions). In a situation in which bidder’s signals are affiliated (Milgrom and Weber 1982), that is, bidders’ signals depend upon not only their private values but also their opposing bidders’ common values, a general model encompassing both the private-values and common-values models needs to be used to analyse bidders’ individual behaviours and their interactions.

The Revenue Equivalence Theorem is valid in both private-values (Vickrey 1961) and common-values settings (Myerson 1981). This states that under the assumption of risk neutrality, independence of valuations and symmetry among bidders, all four of the common auctions yield the same allocation outcomes and thus the same expected revenues. However, relaxing any of these assumptions invalidates the general equivalence.

Cramton (1998), Ausubel and Cramton (2002) and Ausubel (2004) emphasised the difference in allocative inefficiencies between discriminatory pricing and uniform pricing. For sealed-bid auctions, bidders’ behaviours are different under different pricing rules. With the discriminatory price format, there is a close linkage between the bidders’ actual payments and their bids.
Strategic bidders tend to bid under their true values to avoid a *Winner's Curse*, where a bidder pays far more than would have been necessary to win the auction due to incomplete information regarding the common value of the item. With the uniform price format, large bidders may have an incentive to bid under their true values in an attempt to bring down the clearing price, so *demand reduction* is prone to occur (Ausubel and Cramton 2002). Thus, neither discriminatory pricing nor uniform pricing is held to be fully efficient.

In the literature, open ascending auctions are regarded superior to sealed-bid auctions in terms of achieving auction goals (Cramton 1998; Klemperer 1999; Ausubel 2002). Price discovery is the primary attribute and advantage of open ascending auctions over sealed-bid auctions, particular in the situation where bidder’s signals are affiliated. In such a situation, an open ascending auction may induce bidders to bid more aggressively without fear of the Winner’s Curse. This is because bidders can infer greater information about the common values and refine their valuation estimates through a competitive process where each bidder has the opportunity to change losing bids in the previous round to win bids in the next round. The price information generated as an open auction proceeds not only helps in discovering the true values, but also increases the probability for efficient outcomes and expected seller’s revenues. For this reason, an open ascending format is often recommended for auctioning multi-unit objectives, in particular for auctions whose objectives include the discovery of price information.

Collusion and entry deterrence are the main barriers to achieving the goals of auctions, and thus the major elements of concern of practical auction design (Klemperer 2002). *Collusion* refers to a phenomenon where bidders coordinate their strategies to shade their bid prices down through an implicit or explicit agreement (Klemperer 2002). Different auction formats vary in their vulnerability to collusion. Generally, open bidding procedures, where repeated signals of value and demand are available to all bidders, are regarded as being more vulnerable to tacit collusion than are sealed-bid. Bidders can use the early stages to signal who would win which objectives and then tacitly agree to stop pushing prices up (Robinson 1985; Klemperer 2002). A number of strategies have been provided in the literature to help curtail collusive bidding, such as keeping the reserve price secret (Chan *et al.* 2003); including a final sealed-bid stage into an otherwise-ascending auction to create an “Anglo-Dutch” auction (Klemperer 2002); and imposing a time limit on the bidding process if an open auction is used (Chan *et al.* 2003). As will be discussed in Section 5.4.2, some of these strategies have been applied to the FHRSTS credit auctions in order to reduce the potential for collusion.
An auction with too few bidders is unlikely to succeed in either revenue or efficiency terms due to the lack of competition. Potential entrants appear to believe that an open auction is more likely than a sealed-bid auction to be won by the strongest bidder. Open auctions are thus thought to be more vulnerable to lack of entry than sealed-bid auctions (Klemperer 2002). Other factors such as the high expense of entry and bidding and large asymmetries between bidders also discourage entry of potentially weaker bidders (Chan et al. 2003).

The strengths and weaknesses of the key auction formats for multi-unit homogeneous items are summarised below in Table 5.9.

**Table 5.9 Comparison of attributes of various auction approaches**

<table>
<thead>
<tr>
<th>Uniform price</th>
<th>Discriminatory price</th>
<th>Sealed-bid auction</th>
<th>Open auction</th>
</tr>
</thead>
<tbody>
<tr>
<td>Efficiency achievement</td>
<td>-</td>
<td>-</td>
<td>*</td>
</tr>
<tr>
<td>Revenue maximisation</td>
<td>-</td>
<td>-</td>
<td>*</td>
</tr>
<tr>
<td>Information discovery</td>
<td>-</td>
<td>+</td>
<td>-</td>
</tr>
<tr>
<td>Prevent collusions</td>
<td>*</td>
<td>*</td>
<td>+</td>
</tr>
<tr>
<td>Promote entry</td>
<td>*</td>
<td>*</td>
<td>+</td>
</tr>
</tbody>
</table>

Note: + = strength; - = weakness; * = contingent on the specific situation.

Sources: Adapted from Cramton (1998); Klemperer (1999); Ausubel and Cramton (2002); Klemperer (2002); Chan et al. (2003); Ausubel (2004).

There are many ways to structure an auction. The purposes of an auction determine the choice of auction format and influence the auction design. In practice, these auction formats are often jointly used to maximise a specific auction objective (Svendsen and Christensen 1999; Holt et al. 2007). The US SO\(_2\) emission permits auction was the first auction ever to be applied in environmental regulation (Svendsen and Christensen 1999). Following this auction, a number of auctions for the sale of emission permits are being proposed, as listed in Table 5.10.
Table 5.10  Environmental trading programs and their auction formats

<table>
<thead>
<tr>
<th>Program/proposal</th>
<th>Items auctioned/to be auctioned</th>
<th>Features of auction approaches</th>
<th>Status</th>
</tr>
</thead>
<tbody>
<tr>
<td>US Acid Rain Programme</td>
<td>SO₂ emission permits</td>
<td>two-sided pay-as-you-bid call auction</td>
<td>In place</td>
</tr>
<tr>
<td>Virginia</td>
<td>NOₓ allowances</td>
<td>In place</td>
<td></td>
</tr>
<tr>
<td>UK Emission Trading Scheme</td>
<td>GHG emissions reduction</td>
<td>A descending clock auction</td>
<td>Proposed</td>
</tr>
<tr>
<td>Irish auction</td>
<td>GHG emission reduction</td>
<td>Sealed-bid uniform pricing auction</td>
<td></td>
</tr>
<tr>
<td>US Regional Greenhouse Gas (GHG) Initiative</td>
<td>Initial sale of CO₂ allowances</td>
<td>Single-round sealed-bid uniform-auction with a reserve price</td>
<td>Proposed</td>
</tr>
<tr>
<td>Hungarian auction</td>
<td>GHG emission reduction</td>
<td>Uniform pricing auction</td>
<td></td>
</tr>
</tbody>
</table>

Sources: Adapted from Svendsen and Christensen (1999); Holt et al. (2007); Smith and Swierzbinski (2007)

While there have been a number of emission trading programs involving auctions of emission permits, the FHRSTS salt credits auctions were the first one used for sale of discharge permits. However, their influence has not been as great as the Scheme itself: while the FHRSTS has been frequently cited in the literature (for example, OECD 1999; Kraemer et al. 2004; US EPA 2004; Henderson and Norris 2008), formal research on the FHRSTS auctions is still lacking. This study gives a preliminary examination of these auctions, starting with reviewing the key design elements of the FHRSTS credit auctions.

5.4.2  Rules of the FHRSTS credit auctions

For any auction, it is crucial to be clear what is to be auctioned and the purposes of the auction. Under the Regulation, every two years from 2004 to 2014, 20% of the credits (that is, 200 credits) that were initially allocated to the FHRSTS participants by means of grandfathering are repossessed by the authority, and reallocated through public auctions. This means that from 2004 to 2012, 200 credits are sold at auction every two years, but the total number of active credits still remains at 1000. The new, auctioned credits expire 10 years after the date of the auction (as shown in Figure 5.2). It is important to highlight the difference between the salt...
credits in the PHRSTS trading market and in FHRSTS auction market: while what was traded on the trading market represent the one-time right to discharge saline water into a certain water block of the Hunter River, what is traded in the auction market is an ongoing right to discharge saline water into the Hunter River over a ten-year period.

Figure 5.2 Lifespan of credits

[Diagram showing the lifespan of credits]

Source: NSW EPA (2001)

The primary objectives of the FHRSTS credit auctions were to “distribute credits to those who value them the most” (NSW EPA 2004a:3) and “provide equal access for all bidders, including new entrants to the market” (NSW EPA 2004a:3). Another primary objective of the FHRSTS auction is to learn the market value of credits (NSW EPA 2003b). Maximising revenue is not the prioritised purpose of the credit auctions. The revenue from the auctions was proposed to pay the auction costs and offset the costs of the FHRSTS. The latter will reduce the annual contributions payable by the FHRSTS participants (NSW EPA 2004a).

The NSW EPA proposed an auction design in 2003. Following a trial auction on 24 March 2004, the first formal FHRSTS credit auction was conducted on 7 April 2004 and the second on 5 April 2006. Both auctions were multi-round, sealed-bid, discriminatory pricing format without reserve price. The detailed auction rules and operations of these two auctions have been well documented in the NSW EPA reports, such as NSW EPA (2004a), NSW EPA (2004b) and NSW DEC (2006). The rest of this subsection will outline the key elements of the FHRSTS auctions.
The FHRSTS auctions were one-sided auctions. There was the only seller, the NSW EPA, whereas there were multiple buyers, the coal mining and power generating firms. The FHRSTS auctions adopted the multi-round sealed-bid auction approach. In each round buyers tendered a sealed bid consisting of the bid price and the amount of credits they were bidding for. At the end of each round, bids were sorted from highest price to lowest price to produce a bid stack. The top bidders whose aggregated bids were 200 credits were the provisional successful bidders. Rounds continued until there were no new bids or the time limit (3 hours) was reached whichever came first (NSW EPA 2004b). Incorporating the sealed-bid format into the multi-round procedures would have strengthened the capability of the auctions in price discovery.

The discriminatory pricing rule was applied to the payment of successful bidders. Successful bidders were required to pay what they bid for their final successful bids, rather than all bidders paying the same price for all credits, such as the lowest successful price (NSW EPA 2004b). A reserve price is often set in an auction to ensure a desired level of revenue and to reduce the chance of collusion. As maximising revenue for the FHRSTS auctions was not sought, no reserve was set (NSW EPA 2004b).

A number of rules were applied in the FHRSTS auctions to reduce the potential for collusion. One of the rules regards the release of bidding information during the auctions. For example, each registered bidder was allocated an identification number (ID). At the end of each round, only bidder IDs and the number of credits they had been provisionally allocated were announced (NSW EPA 2004b). Therefore, each bidder was able to see how many credits they had won provisionally, but they did not know who the other successful bidders were. The lowest fully allocated bid price and the lowest partially allocated bid price were also released, but excluded the prices offered by the individual bidders (NSW EPA 2004b). This information allowed bidders to see what they needed to bid in subsequent rounds to secure credits. Only at the end of the auctions were the names of successful bidders, as well as their final allocation of credits and prices offered, announced (NSW EPA 2004b).

Another auction rule specified the offer of bids. All bidders were required to bid in the first round. For each credit there was a requirement that the prices of new bids must be higher than the lowest partially allocated bid price from the previous round after the first round. For each round, while bidders were allowed to lodge up to three bids, there was no limit on the amount of credits a single bidder could purchase (NSW EPA 2004b).
A three-hour time limit was also imposed on both the 2004 and the 2006 credits auctions (NSW EPA 2004c; NSW DEC 2006). This limit was set not only to help in curtailing collusion, but also to ensure a practical way of ending the auction if bidding was progressing by very small increments for a long time (NSW EPA 2003b). In addition, the conditions imposed on the successful bidders’ payment and those dealing with allocation of residual credits (the credits that were not sold after successful bids had been settled) (NSW EPA 2004b) provide further prevention of strategic bidding behaviours.

The main operational information and outcomes of the 2004 and 2006 auctions are summarised in Table 5.11.

### Table 5.11 Summary of operational information about 2004 and 2006 auctions

<table>
<thead>
<tr>
<th></th>
<th>2004 auction</th>
<th>2006 auction</th>
</tr>
</thead>
<tbody>
<tr>
<td>Participants</td>
<td>10</td>
<td>11</td>
</tr>
<tr>
<td>Number of auction rounds</td>
<td>19</td>
<td>21</td>
</tr>
<tr>
<td>Winners</td>
<td>8</td>
<td>11</td>
</tr>
<tr>
<td>Maximum credit price ($/credit)</td>
<td>551</td>
<td>658</td>
</tr>
<tr>
<td>Minimum credit price (market clearing price) ($/credit)</td>
<td>478</td>
<td>535</td>
</tr>
<tr>
<td>Average price ($/credit)</td>
<td>507</td>
<td>564</td>
</tr>
<tr>
<td>Revenue ($)</td>
<td>101,465</td>
<td>112,770</td>
</tr>
</tbody>
</table>

Sources: Compiled data from NSW EPA (2004b); NSW DEC (2006).

The 2004 auction generated a revenue of $101K (that is, $507/credit) and the 2006 auction generated a revenue of $112K (that is, $564/credit) (NSW EPA 2004b; NSW DEC 2006). For a very rough estimate, the total capital cost of 1000 credits, each of which is with 10-year lifetime, would be worth in the order of $500K. This suggests that the industries do not value the salt credits much, which is in line with the observations of this study (see Section 5.2.2) that the industries appeared to take advantage of unlimited discharge in the Flood flow period (for example, see Table 5.5, 53% of total salt load was discharged in the Flood flow period) and only use the tradability of salt credits as a backup alternative to discharge excess saline water in the High flow period.

From the recently available 2008 Auction Report (DECC 2008), the average price of a credit in the 2008 auction was $947, with a maximum price of $1019/credit and a minimum of
$869/credit. This auction generated total revenue of $189,442, representing an increase of 68% over that from the 2006 auction. DECC (2008:3) provided an apparently straightforward explanation of the rising auction price: "the decreased number of bid rounds (10 in 2008 compared with 21 in 2006) may have encouraged bidders to place few but higher value bids rather than waiting until the final round of a greater number of bid rounds to secure credits". There could also have been other reasons, such as increasing credit prices on the FHRSTS trading market; fewer Flood flow days for unlimited discharge of salt in the past two years, and in turn more credits in demand to discharge salt on High flow days; the effect of the increased Flood flow threshold meaning more high flow discharge opportunities; or more value put on the credits by the firms as the amount of the initial allocated credits decreases.

The above plausible reasons are generally supported by anecdotal evidence in NO\textsubscript{x} and SO\textsubscript{x} markets in the US (Ellerman 2004), the fisheries market in New Zealand (Kerr 2004) and the water markets in Australia (Bjornlund and Rossini 2005; 2007). For example, it has been found from Australian water market that the prices paid for permanent water entitlement has a significant positive correlation to those for temporary water allocation (Bjornlund and Rossini 2005; 2007). This finding could be valid for the credit trading market of the PHRSTS.

Evidence in Australian water markets has also suggested that water prices predominantly are driven by water scarcity determined by the seasonal availability of water (Bjornlund and Rossini 2005; 2007). Fewer flood flow days in the context of the PHRSTS represent scarcity. While the purchase of salt credits from the trading market during the period of scarcity is a short-term action of the firms to minimise non-compliance risk, it is expected the credit prices in both trading and auction markets are significantly driven by this scarcity. When the occurrence of flood flow days is frequent and duration is long, the firms can discharge their excess saline water using their existing allocation of credits. As scarcity and prices of salt credits in the trading market increases, it becomes increasingly difficult and uncertain to secure the necessary credits at a reasonable price from trading market. Firms thus tend to buy more permanent credits from auction market and by doing so they increased demand and prices in market. Further investigation of the FHRSTS trading market and its implications for the auction market are therefore suggested to be topics for further study of the FHRSTS.
5.4.3 Simple analysis of the auction outcomes

In spite of the prolific literature on auctions, it is only recently that the literature on environmental permits has blossomed, driven by the increasing applications of tradable permits in controlling greenhouse gas emissions. Research in the field of auctioning environmental permits has been concentrated in two main areas: the evaluation of existing auctions (for example, Kline and Meneses 1999; Cason et al. 2003; Brookshire and Burness 2001), and the optimal auction design for a particular tradable permits program (for example, Klemperer 2002; Sunnevag 2003; Holt et al. 2007; National Emissions Trading Taskforce 2007). The following simple analysis of the 2004 and 2006 FHRSTS credit auctions, which was the earliest attempt in this regard, will contribute to the analysis of the former area.

In spite of various differences in design elements between the FHRSTS credit auctions and US SO₂ emission permits auctions, they have key elements in common. That is, trading had existed before the auctions were conducted, and furthermore the trading and auction markets still work in parallel. Nevertheless, in contrast to the US SO₂ emission permit markets where both auctions and trading markets deal with the long term right to emit SO₂, in the case of the HRSTS the trading market deals with one-time discharge rights, while the auction market deals with ongoing discharge rights over a ten-year period. In this regard, the HRSTS credit trading market seems more similar to Australian water markets where both markets for temporary water allocations and permanent water entitlements are in place. The literature on empirical analysis of the US SO₂ emission permits auctions (such as, Kline and Meneses 1999; Brookshire and Burness 2001; Cason et al. 2003) and the Australian water markets (Bjornlund and Rossini 2005; 2007) thus offers a helpful reference to examine the FHRSTS auctions.

For a tradable permits program where both the trading market and the auction market are active and both markets for temporary and permanent permits exist, as the case of the HRSTS, their interaction can be examined from two perspectives: the connection between the trading market and auction market, and the connection between the market for temporary permits and that for permanent permits. Since the auction prices are derived in this study for examining the PHRSTS credit trading market and estimating the net cost saving of the PHRSTS, the analysis focused on the implications of the credits traded in the PHRSTS trading market (representing one-time discharge rights) for the those traded in the FHRSTS auctions (representing ongoing discharge rights). The auction information revealed who were the successful bidders, as well as the number and prices of the credits they won. Drawing on this information and in conjunction with
the credit trading information, this study examines the implications of the observations on the auction market for those on the credit trading market from three aspects: the nature of the successful bidders; the amount of the credits won in the auctions; and their prices. In analysing the auction information, this analysis focuses on the final round of each auction because it is the binding offer round that determines the winners and losers for that auction, so it is obviously the most important round in terms of the price signals.

5.4.3.1 Successful bidders in the auctions versus their roles in the trading market

Coal mining and power generating firms were the only bidders in both the 2004 and the 2006 auctions. There were no registrations from non-industrial bidders in either of these two auctions (NSW EPA 2004; NSW DEC 2006). In the 2004 auction ten bidders representing 11 licensed mining and power generation facilities participated: all of which were already PHRSTS participants and held discharge licences (NSW EPA 2004). In the 2006 auction, 11 bidders representing mining and power generation facilities competed for credits. All but one bidder represented the firms that had already participated in the PHRSTS and held discharge licences before the auction. The only one who had not participated in the Scheme was Centennial Coal, a new development that was being proposed (NSW DEC 2006). The constitution of the bidders suggests that the credits were general worth more to those HRSTS participants who held discharge licences than to the HRSTS participants who did not hold discharge licences and other water users.

There were eight successful bidders in the 2004 auctions and 11 successful bidders in the 2006 auctions (NSW EPA 2004b; NSW DEC 2006). Six of these were successful bidders in both auctions, as shown in Table 5.12. Intuitively, it appears that the successful bidders are perceived to be the buyers or net buyers in the credit trading market. This study looked into the successful bidders' trading records immediately before the auctions in which they participated. For example, for the successful bidders of the 2004 auction, the study reviewed their trading records between January 1995 and March 2004. For the successful bidders of the 2006 auction, the study reviewed their trading records between January 1995 and March 2006.
Successful bidders in FHRSTS auctions and their roles in the trading markets

<table>
<thead>
<tr>
<th>Successful bidders</th>
<th>Auction(s) participated in</th>
<th>Roles in the trading markets</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hunter Valley Operations</td>
<td>2004</td>
<td>net seller</td>
</tr>
<tr>
<td>Warkworth Colliery Holding</td>
<td>2004</td>
<td>net seller</td>
</tr>
<tr>
<td>Bulga Coal Management</td>
<td>2004 and 2006</td>
<td>Not involved in trading</td>
</tr>
<tr>
<td>Liddell Coal Operations</td>
<td>2004 and 2006</td>
<td>net buyer</td>
</tr>
<tr>
<td>Macquarie Generation</td>
<td>2004 and 2006</td>
<td>net seller</td>
</tr>
<tr>
<td>Ravensworth Operations</td>
<td>2004 and 2006</td>
<td>net buyer</td>
</tr>
<tr>
<td>Redbank Power Station</td>
<td>2004 and 2006</td>
<td>buyer</td>
</tr>
<tr>
<td>Wambo Mining Corporation</td>
<td>2004 and 2006</td>
<td>seller</td>
</tr>
<tr>
<td>Bengalla Mine</td>
<td>2006</td>
<td>net seller</td>
</tr>
<tr>
<td>Centennial Hunter</td>
<td>2006</td>
<td>NA</td>
</tr>
<tr>
<td>Rio Tinto Coal Australia</td>
<td>2006</td>
<td>NA</td>
</tr>
<tr>
<td>United Collieries</td>
<td>2006</td>
<td>seller</td>
</tr>
<tr>
<td>Mt Owen Coal Mine</td>
<td>2006</td>
<td>seller</td>
</tr>
</tbody>
</table>


Individual successful bidders' roles in the trading market are presented in Table 5.12. Interestingly, the successful bidders in the auctions were a mix of sellers and buyers in the credit trading market. More interestingly, sellers and net sellers, rather than buyers and net buyers, dominated the successful bidders. The large proportion of sellers and net sellers among the successful bidders appears to contradict the perception that buyers and net buyers would be the ones value credits most and thus put higher prices on bids.

There are two possible explanations for the high proportion of sellers and net sellers among the successful bidders. One is that the prices the bidders paid do not reflect the true private value of the credits to them. A further discussion on the implication of auction prices for market prices and MACs will be provided in Section 5.4.3.3. Another possible explanation could be that some successful bidders may actually represent their parent companies in competing for credits. As a company may own more than one firm, once a firm won the bids, it would transfer the credits through intra-company trades. While the firm names were registered in the trading records, the company names were registered for both the 2004 and the 2006 auctions. So for a specific firm,
there might not exist any connection between its behaviours on the trading market and in the auctions.

One observation about the trading records during December 2002 and March 2006 is worthy of note. Whilst the number of triggered trades increased, a few in-advance trades occurred. For example, on 29 April 2003, Lemington Coal Mine Colliery Holding sold four separate bulks of credits to Hunter Valley Operation (see Table 5.13). Whilst the durations (that is, river blocks) for individual bulk of credits traded were different, three bulks of credits were for future use. The variety of the trading formats suggests that the firms had become more experienced with the credits trading with their increasing knowledge about the currency of credits and their discharge needs. It also reflects the firms’ certainty about the lifetime of the credits under the FHRSTS.

Table 5.13 Trading record of Hunter Valley Operation on 29 April 2003

<table>
<thead>
<tr>
<th>Date and time of trade</th>
<th>Seller</th>
<th>Buyer</th>
<th>First block</th>
<th>Last block</th>
<th>Number of credits traded</th>
</tr>
</thead>
<tbody>
<tr>
<td>29 April 2003, 15:50</td>
<td></td>
<td></td>
<td>2008/183</td>
<td>2010/181</td>
<td>16</td>
</tr>
<tr>
<td>29 April 2003, 15:49</td>
<td></td>
<td></td>
<td>2006/182</td>
<td>2008/182</td>
<td>24</td>
</tr>
<tr>
<td>29 April 2003, 15:49</td>
<td></td>
<td></td>
<td>2004/183</td>
<td>2006/181</td>
<td>32</td>
</tr>
<tr>
<td>29 April 2003, 15:49</td>
<td></td>
<td></td>
<td>2003/119</td>
<td>2004/182</td>
<td>40</td>
</tr>
</tbody>
</table>

Source: NSW DEC (2007)
5.4.3.2 Quantity of the credits won in the auctions versus trading volume in the trading market

No literature was identified systematically analysing the number of permits being traded in the trading market for those being traded in the auction market for a tradable permit program. Based on anecdotal evidence on the water allocation and entitlement markets in Australia (Bjornlund and d Rossini 2007) and in the fisheries quota market in New Zealand (Kerr 2004) and theoretical expectations, it could be hypothesized that under the HRSTS for a firm who was an active buyer in the trading market, the number of credits it purchased in the auction market should reflect its trading level in the trading market, even if the credits on these two markets represent different lifetime of discharge rights (that is, one-time discharge rights in the trading market versus ongoing discharge rights in the auctions). To examine the connection between the trading volume in the trading market and quantity of the credits won in the auctions, let \( X \) denote a firm's initial allocation of credits in 1995, \( Y \) denote the firm's history of traded-in credits (for example, mean, maximum or minimum number of the credits traded-in), and \( Z \) denote the quantity of credits bid in the auction. These three quantities would be perceived to be related by accounting: \( Z = Y + 20\% \times X \). This means the amount of credits bid by the firm in an auction might equal the sum of the number of expired credits and the number of credits traded in on the market. Therefore, there is a reason to compare these figures.

For each successful bidder, this study has compared their 20% of expired credits with the number of credits they obtained from the auctions and the historical trading records. The information is presented in Tables 5.14 and 5.16.
### Table 5.14 Credits obtained from 2004 auctions versus trading volume of 1995-2004

<table>
<thead>
<tr>
<th>Firms</th>
<th>Credits held by 2003 (X)</th>
<th>Credits expired (20% X)</th>
<th>2004 auction</th>
<th>Number of traded-in credits</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Credits</td>
<td>Minimum</td>
</tr>
<tr>
<td>Bulga Coal</td>
<td>50</td>
<td>10</td>
<td>7</td>
<td>-3</td>
</tr>
<tr>
<td>Hunter Valley</td>
<td>120</td>
<td>24</td>
<td>30</td>
<td>6</td>
</tr>
<tr>
<td>Liddell Coal Operations</td>
<td>65</td>
<td>13</td>
<td>14</td>
<td>1</td>
</tr>
<tr>
<td>Macquarie Generation</td>
<td>230</td>
<td>46</td>
<td>50</td>
<td>4</td>
</tr>
<tr>
<td>Ravensworth Operations</td>
<td>100</td>
<td>20</td>
<td>20</td>
<td>0</td>
</tr>
<tr>
<td>Redbank Power</td>
<td>35</td>
<td>7</td>
<td>47</td>
<td>40</td>
</tr>
<tr>
<td>Wambo Coal</td>
<td>35</td>
<td>7</td>
<td>12</td>
<td>5</td>
</tr>
<tr>
<td>Warkworth</td>
<td>40</td>
<td>8</td>
<td>20</td>
<td>12</td>
</tr>
<tr>
<td>TOTAL</td>
<td>675</td>
<td>135</td>
<td>200</td>
<td>65</td>
</tr>
</tbody>
</table>

Table 5.15 Credits obtained from 2006 auction versus trading volume of 1995-2006

<table>
<thead>
<tr>
<th>Firm</th>
<th>Credits held by 2003 (X)</th>
<th>Credits expired (20%X)</th>
<th>2006 auction</th>
<th>Number of traded-in credits</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bulga Coal</td>
<td>50</td>
<td>10</td>
<td>9</td>
<td>-1</td>
</tr>
<tr>
<td>Liddell Coal Operations</td>
<td>65</td>
<td>13</td>
<td>22</td>
<td>9</td>
</tr>
<tr>
<td>Macquarie Generation</td>
<td>230</td>
<td>46</td>
<td>50</td>
<td>4</td>
</tr>
<tr>
<td>Ravensworth Operations</td>
<td>100</td>
<td>20</td>
<td>25</td>
<td>5</td>
</tr>
<tr>
<td>Redbank Power</td>
<td>35</td>
<td>7</td>
<td>15</td>
<td>8</td>
</tr>
<tr>
<td>United Collieries</td>
<td>10</td>
<td>2</td>
<td>2</td>
<td>0</td>
</tr>
<tr>
<td>Wambo Coal</td>
<td>35</td>
<td>7</td>
<td>12</td>
<td>5</td>
</tr>
<tr>
<td>Xstrata Mt Owen</td>
<td>15</td>
<td>3</td>
<td>3</td>
<td>0</td>
</tr>
<tr>
<td>Centennial Hunter(^\text{17})</td>
<td></td>
<td>10</td>
<td>10</td>
<td></td>
</tr>
<tr>
<td>Rio Tinto</td>
<td></td>
<td>40</td>
<td>40</td>
<td></td>
</tr>
<tr>
<td>Bengalla</td>
<td>35</td>
<td>7</td>
<td>12</td>
<td>12</td>
</tr>
<tr>
<td>TOTAL</td>
<td>575</td>
<td>115</td>
<td>200</td>
<td></td>
</tr>
</tbody>
</table>


From the above Tables 5.14 and 5.15, the number of credits that the firms won in the auctions appears to have little relationship to their pre-auction trade history. Except for the new firms, most of the existing firms seemed to use the auction for recovering their expired credits, rather than for obtaining extra credits for discharging excessive wastewater. The following reasons offer explanations for this.

First, according to Regulation 2002, firms need to pay a contribution fee based on the number of credits they hold to fully cover the operational cost of the Scheme, instead of sharing the cost with the NSW DLWC and EPA (see Section 5.2.3). Therefore, anyone purchasing credits at the auction will be required to pay a FHRSTS contribution fee for each credit they buy on top of the credit purchase price. Due to the uncertainty of the need for the discharge of saline water and discharge opportunities, as well as the changing currency of a unit of credit over time, firms

\(^{17}\) Centennial Hunter is a new development. No trading records are available. Rio Tinto Coal Australia manages Coal & Allied's operations at Bengalla, Mt Thorley, Wakeworth and Hunter Valley. From the auction reports, it is impossible to identify which coal operation Rio Tinto stands for there.
may be prone to buying the credits they need in the trading market, unless the auction prices are lower than market prices. Second, regardless of the allocation of credits on the auctions, a firm could recover its loss of credits in the auctions with later purchases in the trading markets for credits. This may also give firms an incentive to place their bids below the true values of the credits. The next section will provide further discussion in this regard.

5.4.3.3 Auction prices versus market prices versus firms' marginal abatement costs

Information released on US SO\textsubscript{2} emissions permit auctions shows that in every year, the auction price has been nearly coincident with the spot trading market prices in the surrounding months, or has been in line with a trend in prices, suggesting that auction prices have close relationship to trading market prices (Kline and Meneses 1999; Cason \textit{et al.} 2003; Brookshire and Burness 2001). However, caution may be required to use this information in analysis of credit prices in the HRSTS trading and auction markets. In the US SO\textsubscript{2} market, both trading and auction markets dealt with the same product – the long term right to emit SO\textsubscript{2}. In contrast, under the HRSTS, what was traded in the trading market represents an one-time right to discharge saline water into a certain water block of the Hunter River, while what was traded in the auctions represents an ongoing right to discharge saline water into the Hunter River over a ten-year period. Thus, the observations in US may not easily apply to the case of HRSTS.

In this regard, the significant insights gained from Australian water markets are referred, where both markets for temporary water allocations and permanent water entitlements exist in parallel. An evaluation of the Australian water markets conducted by Bjornlund and Rossini (2005; 2007) has revealed that water prices in the (temporary) allocation market fluctuate widely in response to short-term water scarcity, and water prices in the (permanent) entitlement market, as a capital investment, reflect long-term factors in the economy and long-term trends in supply factors. Even so, there is a strong relationship between these two kinds of prices. The prices for water entitlements follow the same general trend of water allocations. On average the price of water entitlements increases when the price of water allocation increases, holding other variables constant. In some cases, the market for allocations has been leading the market for entitlements.

For an auction in which a trading market co-exists with the auction market, bidders take both common value and private value into consideration. In the case of the FHRSTS credit auctions,
the common value of a unit of credit relates to its market price. Although the market price was not publicly available, those firms that have been active players in the trading market might have gained this information from their trading activities. Those firms who had limited experience in credit trading might have very little information on the market price before they attended the auctions. However, they learned this information from others' price signals through the iterative auction procedures and reflected it in their bids.

In addition to the common value of a credit, a firm may also have its independent private value. This private value relates to its marginal abatement cost (MAC). However, while the firm may know the cost of reducing a unit of saline water or salt load, it might find it hard to value a unit of credit because of the uncertainty in the need to discharge saline water and the discharge opportunities, as well as the changing currency of a unit of credits. In this case, firms' marginal prices of bidding for a unit of salt credit in the auction could be more closely related to the firms' information about the market price of a credit than to the firms' own MACs. Regardless of their initial allocation of credits, firms may raise the bidding prices up to the market price.

The highest, average and lowest successful credit prices offered for each of the rounds in the 2004 and 2006 auctions are shown in Figures 5.3 and 5.4, respectively. Figure 5.3 shows that deviations between the highest price and lowest prices gradually reduced as the rounds went on, reflecting the firms' learning process of the common value. The price curves show a steeper gradient over the last three rounds compared to the general trend in the previous 16 rounds. This suggests that most bidders raised their price increments in these last rounds, possibly to the maximum they were willing to pay, because they knew the auction was drawing to a close.
From the 2006 auction (see Figure 5.4), while the highest bids increased in a stepwise manner, the rate of increase in the average price curve remained relatively constant throughout the auction, in contrast to the 2004 auction (see Figure 5.3) when it increased more steeply in the final rounds. This suggests that the bidders had more symmetrical information on the market value of a credit in the 2006 auction than in the 2004 auction, either from the trading market or from the price information of the 2004 auction.
Comparing the auction prices between the 2004 and 2006, auctions listed in Table 5.16, the average auction price of the 2006 ($564/credit = 112770/200) is higher than that of the 2004 auction ($507/credit = 101467/200). This suggests that the firms placed more value on the credits as the amount of the initial allocated credits was decreasing. Nevertheless, converting the average auction price ($507) into a 2006 price using the price indexes $I_{2004} = 146.0$ and $I_{2006} = 154.7$ released by Australian Bureau of Statistics (ABS, 2008), the magnitude of the difference between the 2006 auction price ($564 in 2006 prices) and the 2004 price ($537 in 2006 prices) is modest.

Table 5.16   Comparison of the prices of the firms who won in both auctions

<table>
<thead>
<tr>
<th>Firms</th>
<th>2004 auction</th>
<th>2006 auction</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Number of credit obtained</td>
<td>Total cost ($)</td>
</tr>
<tr>
<td>Hunter Valley</td>
<td>30</td>
<td>15,780</td>
</tr>
<tr>
<td>Warkworth</td>
<td>20</td>
<td>11,020</td>
</tr>
<tr>
<td>Bulga Coal</td>
<td>7</td>
<td>3,657</td>
</tr>
<tr>
<td>Liddell Coal Operations</td>
<td>14</td>
<td>6,874</td>
</tr>
<tr>
<td>Macquarie Generation</td>
<td>50</td>
<td>24,300</td>
</tr>
<tr>
<td>Ravensworth Operations</td>
<td>20</td>
<td>10,170</td>
</tr>
<tr>
<td>Redbank Power</td>
<td>47</td>
<td>23,666</td>
</tr>
<tr>
<td>Wambo Coal</td>
<td>12</td>
<td>6,000</td>
</tr>
<tr>
<td>United Collieries</td>
<td>2</td>
<td>1,080</td>
</tr>
<tr>
<td>Xstrata Mt Owen</td>
<td>3</td>
<td>1,656</td>
</tr>
<tr>
<td>Centennial Hunter</td>
<td>10</td>
<td>5,550</td>
</tr>
<tr>
<td>Rio Tinto</td>
<td>40</td>
<td>23,640</td>
</tr>
<tr>
<td>Bengalla</td>
<td>12</td>
<td>6,518</td>
</tr>
<tr>
<td><strong>TOTAL/Average</strong></td>
<td><strong>200</strong></td>
<td><strong>101,467</strong></td>
</tr>
</tbody>
</table>

Sources: NSW EPA (2004c) and NSW DEC (2006)

The narrow dispersion of the average auction prices of 2004 and 2006 can be explained by two possible factors. One is the assumption that under the competitive PHRSTS trading market with good information, a sufficient number and frequency of trades have established a readily recognisable market price of the salt credits, even if this price is not publicly available. Since there would be no point in bidding above and it would be futile to bid below the market price, the auction prices are expected to cluster around the market price. If this assumption is valid, the auction prices then provide a reliable indicator of the trading market value of salt credits.

150
Chapter 5: Mechanisms and Operations of the PHRSTS

The other factor is the assumption that the price information on the trading market was poor. The auction prices would then be expected to reflect the bidders' MAC. As storing excess saline water on site is the prevailing control option for the coal mines, the closing auction prices would have suggested small variation in the MACs among bidders due to the similar saline water control option. Whatever the reason, it suggests that significant cost saving from permit trading under the PHRSTS should not be expected, as will be revealed in Chapter 7.

In the meantime, for both the 2004 and 2006 auctions, firms had a potential incentive to bias their offer below their true value (that is, market price or marginal cost of credit). Clearly this potentially biased bidding incentive stems from the discriminatory pricing rule applied in both auctions, where firms’ payments are closely related to their bids. Firms have an incentive to make a sophisticated bid lower than a bid that reveals their true values with the hope of paying less. They would then face the risk that their bids will be turned down. However, as firms can recover their credits lost in the auction with later purchases in the trading market, losing the bids in the auction may not have a significant impact on their overall capability of environmental compliance. In this situation, firms are prone to bid lower prices. Firms would also have an incentive to attempt arbitrage, that is, to buy cheaply at the auction and sell at the market price afterwards. For each unit that the firm buys in the auction, it would then make an additional profit equal to the difference between the auction price and the market price. This may also explain why the successful bidders are dominated by sellers and net sellers on the trading market.

5.5 Summary

This chapter has looked into a number of operational aspects of the PHRSTS. It finds that the PHRSTS participants tended to take more advantage of unlimited discharges in Flood flows than of tradable discharges in High flows. Around 53% of total salt load was discharged in the Flood flow periods (2.3% of the total time), whilst 43% was discharged in the High flow period (9.6% of the total time). This reflects the feature of industrial saline water generation — the volume of saline water generated on site has a generally positive relationship with rainfall and river flows — and also offers an explanation for the low average utilisation rate of salt credits and the low auction prices of salt credits, because the scope for credit trading is limited by the flow conditions.
This chapter also provides a preliminary analysis of the credit trading market and its implications for the first two FHRSTS credit auctions in 2004 and 2006. It finds that the credit trading market of the PHRSTS was active in terms of both volume of trades and number of participants, in spite of the high proportion of intra-company trades due to high cross-company ownership of the PHRSTS participants. There were 157 credit transactions involving 6811 credits traded during the PHRSTS period, 50% of which were intra-company trades. All the PHRSTS participants were involved in credit trading activities. A significant proportion of firms (9 out of 22 firms) were both sellers and buyers, and the successful bidders of the 2004 and 2006 credit auctions during the FHRSTS were a mix of sellers and buyers in the credit trading market. However, while the low prices at these auctions suggest that the firms did not value the salt credits much, the narrow range of auction prices, which indicate little diversity of salt control costs across the firms, suggests that the gains of the industries from the credit trading is not great.
Chapter 6 Examination of the Environmental Effectiveness of the PHRSTS

6.1 Introduction

On various occasions, the NSW EPA has claimed that the PHRSTS is environmentally effective. Two pieces of evidence have often been cited by the NSW EPA, drawn from Figure 6.1 below (NSW EPA 2001: 11). The first is that “the mean monthly salinity at Singleton has shown a declining trend since the Scheme’s inception in 1995” (NSW EPA 2001:10). The second, already cited in Box 1.1, is “in the fifteen years before the Scheme, salinity at Singleton exceeded 900 EC around 35% of the time. Since the Scheme’s commencement, this has been reduced to 4% of the time” (NSW EPA 2001:11). The NSW also believed that “the lower salinity under the Scheme has occurred despite the occurrence of drier weather that would previously have been associated with increasing salinity levels” (NSW EPA 2001:11), meaning that the PHRSTS improved the river salinity even though river flows were lower.

At first glance, the above evidence seems convincing that the decline of river salinity after 1995 is the direct result of the PHRSTS. However, it is also possible that the decrease in river salinity was at least partly caused by changes in other factors, such as rainfall, river flow conditions, irrigation runoff and underground inflows, given that the river salinity levels are jointly determined by the industrial salt load inputs which are affected by the PHRSTS, and the above-mentioned factors which are not.
This chapter assesses the environmental effectiveness of the PHRSTS against the Trickle Discharge (TD) through examining the effects of two factors, river flow and industrial salt load inputs, on salinity in the Hunter River. The examination of other factors, such as irrigation runoff, groundwater flows, requires further and more intensive investigation and was not pursued in this study. Analysis of the environmental effectiveness of the PHRSTS is done from two perspectives:

(1) **Before-vs-after perspective**: how has the salinity of the Hunter River changed since 1995?

(2) **With-vs-without perspective**: how does the salinity of the Hunter River under the PHRSTS compare with what it would have been without the PHRSTS (that is, under the previous Trickle Discharge system), and what is the benefit of the difference in the river quality to society? Although it is more analytically demanding, this with-vs-without perspective should ideally be adopted because it appears to be the conceptually correct basis for evaluating the net environmental effect of the PHRSTS.

To address these questions, this chapter is divided into two parts. Though related, they have different focuses. The first part examines the trends in river salinity and flow before and after the introduction of the PHRSTS. The purpose of this examination is to detect any difference in...
the river salinity between the PHRSTS and TD and any change of river flow that has affected the river salinity.

The second part focuses on the simulation of river salinity without the PHRSTS to quantify the difference in river salinity caused by the PHRSTS. The challenge in such an analysis is to develop a consistent, accurate characterisation of the without-PHRSTS conditions. That characterisation must link estimates of pollutant discharges (that is, loadings) without-PHRSTS to their impacts on the river quality, the services provided by the water of the Hunter River, and the social value of these services. Those metrics are then compared to those under without-PHRSTS conditions.

As presented in the remainder of this chapter, this chapter addresses the first main research question posed in Chapter 1 – *did the PHRSTS significantly improve the river salinity of the Hunter River?* The analysis leads to a conclusion that *the PHRSTS had a minimal effect on the improvement of river salinity in the Hunter River.* In addition, there are two main contributions of the findings of this chapter to knowledge. Firstly, this study for the first time takes into account the autocorrelation effect in water quality analysis for the Hunter River. This new methodology provides a general reference for hydrological and water quality analysis. Secondly, the relationship of river salinity, river flow and discharge of industrial salt load simulated in this chapter, which can be further tested and calibrated using data collected in more recent years, could contribute to ongoing improvement in the operation of FHRSTS, such as providing earlier and more accurate prediction of daily Total Allowable Discharge (TAD) for High flow days.

To measure the environmental performance of the PHRSTS aims, the ambient salinity concentration at the Singleton gauging station is chosen as the indicator. This choice was made for several reasons: (1) Singleton is the control point for defining river blocks, river salinity objective and total discharge permits in the PHRSTS regulation and operation (see Section 5.1); (2) hydrological and water quality records at the Singleton station are of sufficient duration and quality for statistical analysis; and (3) both daily data for ambient salinity at Singleton and daily salt load discharges from the PHRSTS participants are available for the period 1995-2002 when the PHRSTS was implemented.
Data required for this study include river flow and salinity at the Singleton station, and industrial salt load discharges of the PHRSTS participants. The available data are summarised in Table 6.1; all data were obtained from the NSW DLWC. As noted previously in Section 3.2, the flow data have been gathered by the DLWC at numerous gauging locations throughout the Hunter catchment. As one of the oldest gauging stations, Singleton has accumulated an abundance of historical flow data, which can be traced back as early as late 1981. However, it was only from 1993 that the daily flow data with very few missing observations became available.

Monitoring salinity at Singleton gauging station began comparatively recently, and the frequency and integrity of the salinity observations vary between different periods. During the period 1980-1993, only monthly mean salinity records with considerable discontinuities are available, based on sporadic daily observations. After February 1993, daily river salinity observations are available as a result of a trial of flow augmentation and afterward the Hunter Integrated Telemetry Scheme (HITS) in the Hunter catchment (see Section 3.3). However, there have been a few failures due to equipment malfunction and technical problems, such as electrical shorting, damage to cables and wiring, failure of the new radio telemetry and faults in data transformation.

Daily records of the salt load discharge of the PHRSTS participants for the period 1995-2002 have been documented and maintained by the NSW EPA Newcastle office. The NSW DLWC releases these data in the HRSTS Annual Reports. The daily data on salt load discharge in this study were obtained from the HRSTS Annual Reports for the years from 1/1995 to 11/2002.

<table>
<thead>
<tr>
<th></th>
<th>Monthly mean</th>
<th>Daily mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Salt load discharge (tonnes/day)</td>
<td>Not available</td>
<td>1/1995~11/2002</td>
</tr>
</tbody>
</table>
6.2 Comparison of the monthly mean EC and flow before and after the PHRSTS

This section examines the difference in river salinity and flow before and after the PHRSTS implementation. That is, comparing the river salinity and flow under the PHRSTS and the TD. In order for the examination to be effective, long-term historical records have been used. To match the data series of monthly mean river salinity (denoted EC), which is only available after 1980, the figures for monthly mean river flow (denoted Flow) from 1/1980 to 11/2002 are used in the comparison. Basic information on the monthly data under the PHRSTS and Trickle Discharge (TD) is listed in Table 6.2. Figure 6.2 depicts the EC against the time period. It shows a big gap in the EC data from 12/1987 to 2/1991 (before the PHRSTS), and seven shorter gaps between 1/1980 and 11/2002 (mainly before the PHRSTS as well).

<table>
<thead>
<tr>
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<td>180</td>
<td>95</td>
</tr>
<tr>
<td>Available data</td>
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<td>91</td>
</tr>
<tr>
<td>Missing data</td>
<td>46</td>
<td>42</td>
<td>4</td>
</tr>
<tr>
<td>EC</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Available data</td>
<td>274</td>
<td>179</td>
<td>95</td>
</tr>
<tr>
<td>Missing data</td>
<td>1</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td>Flow</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Source: Data obtained from the NSW DLWC in 2003
6.2.1 Exploratory analysis: normality and autocorrelation of the data

This analysis starts with an examination of the characteristics of monthly mean river salinity ($EC$) and monthly mean river flow ($Flow$) data. The (probability) density plot and quantile-quantile (qq)-plot against a normal distribution for the $EC$ data are displayed in Figure 6.3 (a) and (b). The $EC$ data appear to have an approximately normal distribution, although this is slightly skewed toward the right, indicating the presence of a number of higher $EC$ values. The density plot and qq-plot for the $ln$-transformation of $EC$, that is, $ln(EC)$, are presented in Figure 6.3 (c) and (d), respectively. They show that the distribution of $ln(EC)$ is closer to a normal distribution than to the distribution of $EC$. 

Source: Data obtained from the NSW DLWC in 2003.
Figure 6.3  Data distributions for $EC$ and $ln(EC)$, monthly mean, 1/1980-11/2002

(a) Density for $EC$

(b) qq-plot for $EC$

(c) Density for $ln(EC)$

(d) qq-plot for $ln(EC)$

$EC$ = monthly mean river salinity

From Figure 6.4 (a) and (b), it can be seen that the Flow observations are not normally distributed and are clearly skewed toward the left due to the presence of a large number of low flows. Figure 6.4(c) and (d) show that the normality of ln-transformed Flow, denoted $ln(Flow)$, is much more normally distributed.
Figure 6.4  Data distributions for Flow and ln(Flow), monthly mean, 1/1980-11/2002

Flow = monthly mean river flow

For most environmental data, such as river flow and salinity in this case, measured sequentially over time, the observations are usually not independent of each other. Rather, the observations made at a particular time are related to those made at immediately preceding times. For example, low flow days tend to follow other low flow days, and high flow days tend to follow other high flow days. This phenomenon is called autocorrelation or the time series effect (Diggle 1990) in statistical terms. Autocorrelation can have important implications for predicting water quality impacts. It is thus necessary to investigate whether or not autocorrelation is present in the EC and Flow data, because if it is present, the method of analysis for autocorrelated data must be used. Using analyses appropriate for independent data will give the wrong answers.

Autocorrelation function (ACF) and Partial autocorrelation function (PACF) plots are useful and straightforward graphical tools for detecting the presence of autocorrelation in a data set. ACF measures the correlation between observations of a time series separated by time units. Suppose there are $n$ time based observations $x_1, x_2, x_3 \ldots x_n$. When lag = 1, ACF gives a value of
the correlation between \((x_0, x_1, x_2 \ldots x_n,1)\) and \((x_1, x_2, x_3 \ldots x_n)\). When \(\text{lag} = 2\), ACF then gives a value of the correlation between \((x_0, x_1, x_2 \ldots x_n,2)\) and \((x_2, x_3, x_4 \ldots x_n)\), and so on. PACF finds a correlation between some components of a data series, eliminating the contribution of other components. It removes the effect of shorter lag autocorrelation from the correlation estimate at longer lags (Millard and Neerchal 2001).

The ACF for \(\ln(\text{EC})\) and \(\ln(\text{Flow})\) are plotted in Figure 6.5. The height of the vertical lines in the ACF plots represents the autocorrelation estimate at each lag. It measures the strength of similarity at each time lag among the observations, and usually decreases as the lag increases. The horizontal band around zero, which is defined by the two dotted lines, represents the approximate 95% confidence limits for the acceptance of a hypothesis of zero autocorrelation. That is, if no "height" falls outside the band, it can be safely assumed that there is no autocorrelation effect in the tested data. Otherwise, the autocorrelation should be taken into account in the data analysis (Ginevan and Splitstone 2004). The PACF varies between -1 and +1, with values near ±1 indicating stronger correlation (Millard and Neerchal 2001).

Figure 6.5 below clearly indicates a strong autocorrelation in \(\ln(\text{EC})\), and the peak at lag 11 in month indicates a seasonal cycle. The autocorrelation in \(\ln(\text{Flow})\) is not as strong as that in the \(\ln(\text{EC})\), as it dies out sooner, at around lag 4 in month.
6.2.2 Graphical comparisons

In order to understand the difference in monthly mean river salinity and flow under the Pilot Hunter River Salinity Trading Scheme (PHRSTS) and the Trickle Discharge (TD), \( \ln(\text{EC}) \) and \( \ln(\text{Flow}) \) are first compared graphically. This is because graphical displays can be especially helpful for presenting a large volume of data and help develop a "big picture" understanding of the data before proceeding to analysis.

First, \( \ln(\text{EC}) \) and \( \ln(\text{Flow}) \) under the PHRSTS and the TD are compared using boxplots. Boxplots are widely regarded as a very effective and concise method for comparing important distributional characteristics of two data sets. A central box shows the location of the 25th percentile (1st Q) and 75th percentile (3rd Q) with a central locating median (50th percentile) (Millard and Neerchal 2001).
Let $EC_0$ and $EC_1$ be the monthly mean river salinity under the TD and the PHRSTS respectively, and let $Flow_0$ and $Flow_1$ be the monthly mean river flow under the TD and PHRSTS respectively. The boxplots allowing comparison of $ln(EC_0)$ and $ln(EC_1)$ are displayed in Figure 6.6 below. By visual inspection, the medians, 1st Q and 3rd Q of $ln(EC_0)$ are all higher than those of $ln(EC_1)$. While there are a few of extreme high $ln(EC_0)$, there are a few of extreme low $ln(EC_1)$. This seems to support the NSW EPA's claim that the river salinity declined after the introduction of the PHRSTS, although it does not indicate how much of this decline is the effect of the PHRSTS, and how much is the effect of other determining factors.

**Figure 6.6** Box plots for $ln(EC_0)$ and $ln(EC_1)$, monthly mean, 1/1980-11/2002

The boxplots for $ln(Flow_0)$ versus $ln(Flow_1)$ are shown in Figure 6.7 below. The variation in $ln(Flow_0)$ and $ln(Flow_1)$ are similar. The median, 1st Q and 3rd Q of $ln(Flow_1)$ are all higher than those of $ln(Flow_0)$. This indicates that the overall river flow under the PHRSTS is higher than that under the TD.
6.2.3 Exceedances

Another common way to compare hydrology and water quality data at a specified location on a river is to compare their frequencies of exceedance, that is, the percentage of time for which any selected value may be equalled or exceeded. It is often performed by delineating so-called occurrence/exceedance curves (Brooks et al. 2003). For any point on the curve, the vertical coordinate indicates the values of a hydrological or water quality parameter, while the horizontal coordinate shows the percentage of time that the particular value has been equalled or exceeded. These curves can provide a good indication of a river’s hydrological and water quality history. A curve with a very flat slope indicates little variation in hydrological or water quality values, while that with a steep slope indicates the opposite.

The exceedance of $\ln(EC)$ and $\ln(Flow)$ under TD and PHRSTS are compared in Figure 6.8 and Figure 6.9 respectively. Note that the percentages of exceedances for $\ln(EC)$ are identical to those for $EC$, but the scale of $\ln(EC)$ can show the difference in the river salinity before and after the PHRSTS implementation more clearly than that of $EC$. Figure 6.8 plots the exceedance...
Chapter 6 Examination of the Environmental Effectiveness of the PHRSTS

curves for $ln(EC_0)$ and $ln(EC_1)$. The corresponding numerical comparison is provided in Table 6.3. It shows that the general distribution of $ln(EC_1)$ is lower than that of $ln(EC_0)$. The proportion of observations in which EC exceeds 900 EC is reduced from 28.3% for $ln(EC_0)$ to 2.5% for $ln(EC_1)$. This finding also supports the NSW EPA’s claim that the river salinity has decreased since the introduction of the PHRSTS, though does not mean this is a result of the PHRSTS.

Figure 6.8 Comparison of the exceedances of river salinity

TD = Trickle Discharge
PHRSTS = Pilot Hunter River Salinity Trading Scheme
$EC_0$ = monthly mean river salinity under the TD
$EC_1$ = monthly mean river salinity under the PHRSTS
Table 6.3  Numerical comparison of the exceedances of river salinity

<table>
<thead>
<tr>
<th>EC</th>
<th>ln(EC)</th>
<th>TD</th>
<th>PHRSTS</th>
</tr>
</thead>
<tbody>
<tr>
<td>1500</td>
<td>7.31</td>
<td>0.72</td>
<td>0.00</td>
</tr>
<tr>
<td>1200</td>
<td>7.09</td>
<td>4.35</td>
<td>0.00</td>
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<tr>
<td>900</td>
<td>6.80</td>
<td>28.26</td>
<td>3.33</td>
</tr>
<tr>
<td>800</td>
<td>6.68</td>
<td>47.83</td>
<td>10.00</td>
</tr>
<tr>
<td>700</td>
<td>6.55</td>
<td>72.46</td>
<td>37.78</td>
</tr>
<tr>
<td>600</td>
<td>6.40</td>
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<td>67.78</td>
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<tr>
<td>500</td>
<td>6.21</td>
<td>97.83</td>
<td>91.11</td>
</tr>
<tr>
<td>400</td>
<td>5.99</td>
<td>99.28</td>
<td>97.78</td>
</tr>
<tr>
<td>300</td>
<td>5.70</td>
<td>99.28</td>
<td>100.00</td>
</tr>
</tbody>
</table>

EC = monthly river salinity
TD = Trickle Discharge
PHRSTS = Pilot Hunter River Salinity Trading Scheme

Figure 6.9 compares the exceedances for ln(Flow0) and ln(Flow1). Again, numerical comparisons are provided in Table 6.4. Both clearly indicate that the overall distribution of ln(Flow1) is slightly higher than ln(Flow0). While the percentage exceedance of 50 ML/day (base flow) is almost the same, the percentage exceedance of a flow of more than 5000 ML/day for the PHRSTS period is almost double that of the TD period. Based on the general inverse relationship between river flow and salinity (see Section 3.3), a plausible inference is that the decline of river salinity under the PHRSTS may be the result of higher river flows.
Figure 6.9  
Comparison of the exceedances of river flow

TD = Trickle Discharge
PHRSTS = Pilot Hunter River Salinity Trading Scheme
Flow0 = monthly mean river flow under the TD
Flow1 = monthly mean river flow under the PHRSTS
Table 6.4 Numerical comparison of exceedances of river flow

<table>
<thead>
<tr>
<th>Flow</th>
<th>ln(Flow)</th>
<th>Exceedances (%)</th>
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<tbody>
<tr>
<td></td>
<td></td>
<td>TD</td>
</tr>
<tr>
<td>25000</td>
<td>10.12</td>
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</tr>
<tr>
<td>20000</td>
<td>9.90</td>
<td>1.12</td>
</tr>
<tr>
<td>15000</td>
<td>9.62</td>
<td>2.23</td>
</tr>
<tr>
<td>10000</td>
<td>9.21</td>
<td>5.03</td>
</tr>
<tr>
<td>5000</td>
<td>8.52</td>
<td>7.26</td>
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<tr>
<td>3000</td>
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<td>11.17</td>
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<td>2000</td>
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<td>84.92</td>
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<td>4.61</td>
<td>96.09</td>
</tr>
<tr>
<td>50</td>
<td>3.91</td>
<td>98.88</td>
</tr>
</tbody>
</table>

TD = Trickle Discharge  
PHRSTS = Pilot Hunter River Salinity Trading Scheme  
Flow = monthly mean river flow

6.3 Regression models with autocorrelated errors

This study first used simple statistical methods, such as the Student's t-test, the Wilcoxon Rank Sum test and the Quantile test, to examine whether there is a statistical difference in river salinity and river flow between the PHRSTS and TD. However, all these methods assume that the data are independent of each other (Ginevan and Splitstone 2004), which is not a valid assumption for the river salinity and flow data of this study. The results are presented in Appendix 7. These simple statistical tests conclude that the river salinity under the PHRSTS is significantly different from that under the TD, but lead to different conclusions as to whether there has been a change in the river flow since the PHRSTS implementation. These conclusions contrast strongly with conclusions drawn from appropriate testing methods, which are presented here.
To be rigorous, the comparison of a continuous run of data before and after an intervention must account for the effect of autocorrelation within the data. In the literature, such an analysis falls broadly into the category of interrupted time series (Box and Tiao 1975; Manly 1992). The aim of interrupted time series analysis is to detect whether there is any evidence of a change in the process part way through the observation period. Interrupted time series analysis is often applied in environmental impact studies, behaviour modification experiments, and studies of the effect of government legislation on economic indicators, where the intervention may be planned or unplanned (Box and Tiao 1975; Manly 1992). In this section the data are analysed in a more rigorous way that accounts for the autocorrelation in the data analysis.

6.3.1 Methodology

An underlying analytical framework that can handle a wide range of interrupted time series problems is given in Manly (1992). In a case where the effect of autocorrelation is trivial and a series appears to exhibit a linear trend, a linear model can be fitted

\[ Y_t = \alpha + \beta t + \varepsilon_t \]  

[6.1]

where \( Y_t \) is the observed value at time \( t \), \( \alpha \) and \( \beta \) are constants, and \( \varepsilon_t \) represents a random disturbance. For intervention analysis, a dummy (or indicator) variable, denoted \( I \), is introduced with values 0 (zero) at all times before the intervention event, and 1 at all times after the event. An intervention effect that shifts the mean level of the series by the amount \( \delta \) can be allowed for by fitting the model

\[ Y_t = \alpha + \beta t + \delta I + \varepsilon_t \]  

[6.2]

An intervention effect that changes the trend by the amount \( \tau \) can be allowed for by fitting the model

\[ Y_t = \alpha + \beta t + \delta I + \tau (It) + \varepsilon_t \]  

[6.3]

where \((It)\) represents the product of \( I \) and \( t \). By using these models, the effect of the intervention can be assessed based on the significance of the estimates of \( \delta \) and \( \tau \). In regression analysis, when the \( \varepsilon_t \) are assumed to be independent or when the autocorrelation in \( \varepsilon_t \) is assumed to be
small enough to ignore, \( \varepsilon_t \) has a mean of zero and does not change with \( t \). However, when the \( \varepsilon_t \) are serially correlated, this correlation has to be accounted for in the models. In this situation,

\[
\varepsilon_t = \tau_1 \varepsilon_{t-1} + \tau_2 \varepsilon_{t-2} + \ldots + \tau_k \varepsilon_{t-k} + \delta_t
\]  

[6.4]

where \( \delta_t \) is a random error, and \( k \) indicates the order of autocorrelation, which can be chosen according to the needs of each individual study.

This method was initially developed by Box and Tiao (1975) to assess the effect of two interventions on the oxidant pollution level in downtown Los Angeles, and the effect of government controls on the monthly inflation rate in the United States. Bhattacharyya and Layton (1979) also used this method to study the effectiveness of seat-belt legislation in Queensland, Australia.

### 6.3.2 Fitted models for monthly mean river salinity (EC)

To examine the effect of PHRSTS on river salinity, a dummy variable, PHRSTS, has been introduced to indicate the values in the period of the Pilot Hunter River Salinity Trading Scheme (PHRSTS). PHRSTS=0 is for the Trickle Discharge (TD) period, and PHRSTS=1 is for the PHRSTS period. Using S-plus statistical software, the models were fitted for the river salinity data in accordance with Equations [6.1], [6.2] and [6.3]. The simple model without a linear temporal (monthly) trend but accounting for the effect of PHRSTS was also fitted. The model description, coefficients of the estimates and the indicators for model selection are summarised in Table 6.5 below.

<table>
<thead>
<tr>
<th>Table 6.5</th>
<th>Fitted models for ( \ln(\text{EC}) )</th>
</tr>
</thead>
<tbody>
<tr>
<td>Independent</td>
<td>Models</td>
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<tr>
<td>ln(EC).m1</td>
<td>ln(EC)~Month</td>
</tr>
<tr>
<td>ln(EC).m2</td>
<td>ln(EC)~PHRSTS</td>
</tr>
<tr>
<td>ln(EC).m3</td>
<td>ln(EC)~Month+PHRSTS</td>
</tr>
<tr>
<td>ln(EC).m4</td>
<td>ln(EC)~Month+PHRSTS+Month*PHRSTS</td>
</tr>
<tr>
<td>AR(1)</td>
<td>ln(EC).ar1.m1</td>
</tr>
<tr>
<td></td>
<td>ln(EC).ar1.m2</td>
</tr>
<tr>
<td></td>
<td>ln(EC).ar1.m3</td>
</tr>
<tr>
<td></td>
<td>ln(EC).ar1.m3</td>
</tr>
</tbody>
</table>

\( EC \) = monthly mean river salinity  
AR(1) = first autoregressive model  
AIC = Akaike Information Criterion
In Table 6.5 above and throughout the whole Chapter 6, $AR(1)$ means the first autoregressive model. $m1, m2, m3$ and $m4$ represent inclusion of various variables in the models. The Akaike Information Criterion (AIC) measures how close the fitted values under a model are to the observed data. It is a popular criterion for choosing between competing statistical models. The model with the lowest (most negative) AIC is best, but there is no associated test of how much better (Millard and Neerchal 2001). Models were fitted up to AR(11), but only the first few models are shown in the table. In terms of the AIC, AR(1) models are generally better than those with higher AR levels. Based on Table 6.5, the model with a linear temporal trend better accounts for changes in $EC$ than the simple model of constant $EC$ before and after the PHRSTS implementation. The AR(1) models provide better fit than those that assume no autocorrelation. Furthermore, among the AR(1) models, $ln(EC).arl.ml$, which is expressed as Model [6.5] below, is the best fitting model for the trend in river salinity observations because of its smallest AIC value.

$$ln(EC) = 6.78 - 0.016Month \quad [6.5]$$

This model fits a simple line with a negative slope for time (month) but without the variable PHRSTS, indicating there is no evidence of any change in the monthly mean river salinity of the Hunter River attributable to the introduction of the PHRSTS.

To check that Model [6.5] fits the data, several diagnostic procedures were applied to the residuals from this model, including an autocorrelation check, a cumulative periodogram check and residual plots.

A common diagnostic technique to assess the fit of a model to the autocorrelated data is to examine the ACF and PACF for its residuals. If no autocorrelation in the residuals is found, it can be assumed that the model adequately fits the autocorrelation in the data. Otherwise, the model is inadequate (Millard and Neerchal 2001). For comparison purposes, the ACF and PACF plots for the residuals from both $ln(EC).i.ml$ (not accounting for autocorrelation effect) and $ln(EC).arl.ml$ (accounting for autocorrelation effect) are displayed in Figure 6.10. At small lags, it is obvious that, while several autocorrelation estimates for the residuals of $ln(EC).i.ml$ fall outside the strip defined by the two dotted lines, almost all the autocorrelation estimates for the residuals of $ln(EC).arl.ml$ sit well within the strip. This indicates $ln(EC).arl.ml$ has eliminated the autocorrelation effect, and thus is adequate for the $EC$ observation trend.
Figure 6.10  ACF and PACF of $\ln(EC).i.m1$ and $\ln(EC).ar1.m1$

(a) ACF of $\ln(EC).i.m1$

(b) PACF of $\ln(EC).i.m1$

(c) ACF of $\ln(EC).ar1.m1$

(d) PACF of $\ln(EC).ar1.m1$

ACF = autocorrelation function
PACF = partial autocorrelation function
$\ln(EC).i.m1 = $ fitted model for monthly mean river salinity without accounting for autocorrelation effect
$\ln(EC).ar1.m1 = $ fitted model for monthly mean river salinity accounting for autocorrelation effect

A cumulative periodogram provides an effective means for the detection of periodicities in the residuals. The diagonal lines in the cumulative periodogram plots define the 95% confidence bands for an uncorrelated relationship between two observations in a stationary time series (Millard and Neerchal 2001). Figure 6.11 presents the cumulative periodograms for the residuals from the $\ln(EC).i.m1$ and $\ln(EC).ar1.m1$ fits. The residual cumulative periodogram curve for $\ln(EC).ar1.m1$ lies within the 95% band, while that for $\ln(EC).i.m1$ lies outside the band at more than 5% of the frequencies. This provides extra evidence on the goodness of fit of $\ln(EC).ar1.m1$. 

172
Cumulative periodograms of $ln(EC).i.m1$ and $ln(EC).ar1.m1$

Residual plotting is another common tool used to reveal any weaknesses in a linear model (Millard and Neerchal 2001). The residual plots for $ln(EC).ar1.m1$ are displayed in Figure 6.12. The normal quantile-quantile plot of residuals shown in Figure 6.12 (a) provides a visual test of the assumption that the model’s errors are normally distributed. The ordered residuals of $ln(EC).ar1.m1$ lying along the qq-line reveal a normal distribution of its errors. In addition, the residuals plotted against PHRSTS, month and the fitted values EC are presented in Figure 6.12 (b), (c) and (d). Compared to the total 229 observation values, the number of outliers is minimal. The residuals plotted against PHRSTS indicate more variability for the TD period than the PHRSTS period perhaps because of a number of large values of river salinity present under the TD.

$ln(EC).i.m1$ = fitted model for monthly mean river salinity without accounting for autocorrelation effect

$ln(EC).ar1.m1$ = fitted model for monthly mean river salinity accounting for autocorrelation effect
Figure 6.12 Residual plots of \(\ln(EC).ar1.m1\)

\(\ln(EC).ar1.m1\) = fitted model for monthly river salinity accounting for autocorrelation effect

The diagnostic checks show no lack of fit or violation of assumptions in Model [6.5], confirming that [6.5] is the best model of those tested. The fitted line for Model [6.5] has been plotted in Figure 6.13 to show the EC trends for both before and after PHRSTS periods.
Figure 6.13  Fitted lines of AR(1) models of $\ln(EC)$

Fitted line of $\ln(EC).ar1.m1$

$\ln(EC).ar1.m1 =$ fitted model for monthly mean river salinity accounting for autocorrelation effect
TD = Trickle Discharge
PHRSTS = Pilot Hunter River Salinity Trading Scheme

Although the level of river salinity for the PHRSTS period is lower than that for the TD period, any differences in trend of river salinity between the periods are not statistically significant. The decrease of the monthly mean river salinity of the Hunter River is caused by a general declining trend of the river salinity over time, rather than by any effect of the PHRSTS. This leads to the conclusion that the PHRSTS has had little effect on the monthly mean river salinity. Comparing this finding with those from the simple statistical methods presented in Appendix 7, when the autocorrelation effect in the data is taken into account, the analysis no longer shows differences in river salinity flow under the PHRSTS and the TD. This illustrates that erroneous conclusions can be made if an analysis uses an inappropriate statistical method.
6.3.3 Fitted model for monthly mean river flow (Flow)

In the same way, the models for ln(Flow) up to lag 4 in month were fitted based on the insight obtained from examination of the data. As shown in Figure 6.5(c), autocorrelation effect in ln(Flow) appears up to lag 4. Similar to ln(EC), the AR(1) models are generally better than others. Only AR(0) and AR(1) models are presented in Table 6.6.

Table 6.6 Fitted models of ln(Flow)

<table>
<thead>
<tr>
<th>Models</th>
<th>AR(1)</th>
<th>Intercept</th>
<th>Month</th>
<th>PHRSTS</th>
<th>Month*PHRSTS</th>
<th>AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>ln(Flow).i.m1</td>
<td>ln(Flow)-Month</td>
<td>5.9188</td>
<td>0.0320</td>
<td></td>
<td></td>
<td>910.57</td>
</tr>
<tr>
<td>ln(Flow).i.m2</td>
<td>ln(Flow)-Flow</td>
<td>6.1839</td>
<td>0.2896</td>
<td></td>
<td></td>
<td>914.99</td>
</tr>
<tr>
<td>ln(Flow).i.m3</td>
<td>ln(Flow)-Month+PHRSTS</td>
<td>5.8400</td>
<td>0.0461</td>
<td>-0.2393</td>
<td></td>
<td>911.85</td>
</tr>
<tr>
<td>ln(Flow).i.m4</td>
<td>ln(Flow)-Month+PHRSTS+Month*PHRSTS</td>
<td>5.8991</td>
<td>0.0362</td>
<td>-1.3192</td>
<td>0.0619</td>
<td>912.80</td>
</tr>
<tr>
<td>AR(1)</td>
<td>ln(Flow).ar1.m1</td>
<td>ln(Flow)-Month</td>
<td>0.4863</td>
<td>5.9207</td>
<td>0.0314</td>
<td>838.58</td>
</tr>
<tr>
<td>ln(Flow).ar1.m2</td>
<td>ln(Flow)-PHRSTS</td>
<td>0.4948</td>
<td>6.1771</td>
<td>0.2938</td>
<td>839.96</td>
<td></td>
</tr>
<tr>
<td>ln(Flow).ar1.m3</td>
<td>ln(Flow)-Month+PHRSTS</td>
<td>0.4848</td>
<td>5.8658</td>
<td>0.0414</td>
<td>-0.1698</td>
<td>840.44</td>
</tr>
<tr>
<td>ln(Flow).ar1.m4</td>
<td>ln(Flow)-Month+PHRSTS+Month*PHRSTS</td>
<td>0.4828</td>
<td>5.9086</td>
<td>0.0354</td>
<td>-0.8988</td>
<td>-0.0422</td>
</tr>
</tbody>
</table>

Flow = monthly mean river flow
AR(1) = first autoregressive model
AIC = Akaike Information Criterion

It can be seen from Table 6.6 that the AIC value of ln(Flow).ar1.m1 is the smallest among those of other fitted models. ln(Flow).ar1.m1, expressed in Model [6.6], is the best fitted model for ln(Flow).

\[
\text{ln(Flow)} = 5.92 + 0.03\text{Month} \quad [6.6]
\]

To check the fit of the model ln(Flow).ar1.m1, diagnostic checks were undertaken and are displayed in Figures 6.14, 6.15 and 6.16. The diagnostic checks for the independent model ln(Flow).i.m1 are provided for comparison purposes. The fitted line for the final model ln(Flow).ar1.m1 is presented in Figure 6.17.
Figure 6.14   ACF and PACF for $ln(\text{Flow}).i.m1$ and $ln(\text{Flow}).ar1.m1$

(a) ACF of $ln(\text{Flow}).i.m1$

(b) PACF of $ln(\text{Flow}).i.m1$

(c) ACF of $ln(\text{Flow}).ar1.m1$

(d) PACF of $ln(\text{Flow}).ar1.m1$

ACF = autocorrelation function
PACF = partial autocorrelation function
$ln(\text{flow}).i.m1$ = fitted model for monthly mean river flow without accounting for autocorrelation effect
$ln(\text{flow}).ar1.m1$ = fitted model for monthly mean river flow without accounting for autocorrelation effect
Chapter 6 Examination of the Environmental Effectiveness of the PHRSTS

Figure 6.15  Cumulative periodograms of \( \ln(\text{Flow}).i.m1 \) and \( \ln(\text{Flow}).ar1.m1 \)

\[ \begin{align*}
\text{(a) } \ln(\text{Flow}).i.m1 & \\
\text{(b) } \ln(\text{Flow}).ar1.m1 & 
\end{align*} \]

\( \ln(\text{flow}).i.m1 \) = fitted model for monthly mean river flow without accounting for autocorrelation effect
\( \ln(\text{flow}).ar1.m1 \) = fitted model for monthly mean river flow without accounting for autocorrelation effect

Figure 6.16  Residual plots of \( \ln(\text{Flow}).ar1.m1 \)

\[ \begin{align*}
\text{(a) qq-line of Residuals} & \\
\text{(b) Residuals-Month} & \\
\text{(c) Residuals-Fitted} & \\
\text{(d) Residuals-PHRSTS} & 
\end{align*} \]

\( \ln(\text{flow}).ar1.m1 \) = fitted model for monthly mean river flow accounting for autocorrelation effect
The above analysis reveals that the higher river flow under the PHRSTS than under the TD was caused by a general increasing trend of the river flow over time. However, the monthly mean river flow under the PHRSTS does not significantly differ from that under the TD in a statistical sense.

6.3.4 Sensitivity analyses

Further analyses were conducted to examine whether there are foresight or transitional effects in monthly mean river salinity $\ln(EC)$ or monthly mean river flow $\ln(Flow)$. As noted in Chapter 5, initiation of the PHRSTS started in 1993 after intensive consultation with local industries. The anticipation of the inception of the PHRSTS may have caused these industries to alter their discharge behaviour some years before 1995 when the PHRSTS was brought into force. Therefore it would be of interest to examine the effect of anticipation on the results of analyses. To do so, a one-year foresight effect was examined by employing a dummy variable $PHRSTS_{-1}$, while $PHRSTS_{-1} = 0$ is for $1/1980$ to $12/1993$ and $PHRSTS_{-1} = 1$ is for $1/1994$ to $11/2002$. That is, it is assumed the PHRSTS effectively came into force after 1994. A two-year foresight effect was also examined. A dummy variable $PHRSTS_{-2}$ is used to represent a two-year anticipation period.

The opposition to the foresight effect is the transitional effect. The PHRSTS implementation may not have had an immediate influence on behaviour. Industries might take several years to reform their saline water management, become accustomed to the operational system of PHRSTS, and change their discharges. To assess such possible transitional effects on the results of the analysis, two dummy variables, $PHRSTS_{-1}$ and $PHRSTS_{-2}$, were introduced to represent a one-year and two-year transitional period, respectively. By using $PHRSTS_{-1}$, it is assumed that the PHRSTS effectively started in 1996. That is, $PHRSTS_{-1} = 0$ for $1/1980-12/1995$, while $PHRSTS_{-1} = 1$ is for $1/1996-11/2002$. Likewise, for an effective start date of 1997, $PHRSTS_{-2} = 0$ for $1/1980-12/1996$, while $PHRSTS_{-2} = 1$ is for $1/1997-11/2002$.

Using the same procedures as elaborated in Section 6.3.2 above, the fitted models for $ln(EC)$ with different indicating variables from $PHRSTS_{-2}$ to $PHRSTS_{-2}$ were estimated. The results are shown in Appendix 8. The results show that there is no evidence of foresight or transitional effects in the data. The best fitting models for $ln(EC)$ are the same as Model [6.5]. Again, this confirms that the PHRSTS had little effect on the overall trend of the monthly mean river salinity of the Hunter River.

Up to this point, the analysis has shown that although the overall salinity of the Hunter River was lower after the PHRSTS than before, the PHRSTS is not the contributing factor. This finding, though disappointing, is straightforward to explain. The PHRSTS was designed to achieve the river salinity objectives by *redistributing* the saline water discharges over time and among industries, not by reducing total discharge. Therefore, if the PHRSTS had been environmentally effective, this may not have been reflected in the overall trend of the river salinity. Bearing this in mind, the next step of the study is to analyse the effect of the inception of PHRSTS on high salinity, which usually occurs in the Low flow periods.

### 6.3.5 Comparison of the monthly mean river salinity in Low flow periods

To assess the effect of the PHRSTS on the monthly mean river salinity in the Low flow periods, only data corresponding to flows less than 2000 ML/day were used to fit the models. Following
the same procedures as elaborated in Section 6.3.2 above, models for $\ln(EC)$ in the Low flow period ($\text{Flow} < 2000 \text{ ML/day}$) were fitted; these are shown in Table 6.7.

### Table 6.7  Fitted models of $\ln(EC)$ of Low flow period

<table>
<thead>
<tr>
<th>Models</th>
<th>$\text{AR(1)}$</th>
<th>Intercept</th>
<th>Month</th>
<th>PHRSTS</th>
<th>Month*PHRSTS</th>
<th>AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\ln(EC)_{l.m1}$ $\sim$ Month</td>
<td>0.4573</td>
<td>6.8029</td>
<td>-0.0174</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>$\ln(EC)_{l.m2}$ $\sim$ PHRSTS</td>
<td>0.5455</td>
<td>6.6535</td>
<td>-0.2063</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>$\ln(EC)_{l.m3}$ $\sim$ Month+PHRSTS</td>
<td>0.4571</td>
<td>6.7993</td>
<td>-0.0168</td>
<td>-0.0147</td>
<td></td>
<td></td>
</tr>
<tr>
<td>$\ln(EC)_{l.m4}$ $\sim$ Month+PHRSTS+Month*PHRSTS</td>
<td>0.4442</td>
<td>6.7882</td>
<td>-0.0153</td>
<td>0.8189</td>
<td>-0.0418</td>
<td>-101.52</td>
</tr>
</tbody>
</table>

$EC_l = \text{monthly mean river salinity of low flow period}$  
$\text{AR(1)} = \text{first autoregressive model}$  
$\text{AIC} = \text{Akaike Information Criterion}$

From Table 6.7, the best model for the river salinity of the Low flow periods is $\ln(EC)_{l.ar1.m1}$. That is,

$$\ln(EC)_l = 6.80 - 0.017 \text{Month} \quad [6.7]$$

where $\ln(EC)_l$ represents the river salinity in the Low flow period, accounting for the autocorrelation effect in the errors at lag 1 in month. Model [6.7] is very close to Model [6.5], the best fitted model for the overall monthly mean river salinity. Model [6.7] does not include the variable PHRSTS either, indicating that there is no difference in the river salinity during the Low flow period before and after the PHRSTS implementation.

It would be of interest to fit models to $\ln(EC)$ in both the High and Flood flow periods. However, there are insufficient monthly EC records in either of these periods to conduct the study. For example, while 35 monthly records of the river flow fall into the category of High flow, only one monthly record of the river salinity is available. There are 13 monthly records of the river flow falling into the category of the Flood flow, but only six monthly records of river salinity are available, of which two records are for the TD period, and four are for the PHRSTS period.

So far, analyses of the river flow and salinity have been separated, rather than combined. The analyses have revealed that the difference in the monthly river flow and salinity before and after...
the inception of PHRSTS is not statistically significant. Moreover, the difference in monthly mean river salinity of the Low flow period before and after the inception of PHRSTS is modest. However, two limitations of this method may prevent drawing a concrete conclusion without further quantitative analysis.

One limitation is that the above analyses are all based on the monthly mean of both river salinity and river flow observations. However, the observation records for river salinity are incomplete before 1993. While there were no observation records for the 42 months before 1993, only a few daily observations are available for many months. This may affect the reliability of the comparison results. Even if the monthly mean data are complete without any missing records, they can only provide the general features of river flow and salinity in an indicative way. Loss of extreme daily values in the averaging might also have an additional effect on the reliability of the comparison results.

The other limitation is that as a general problem with comparative time series, the time series of river salinity and flow under the PHRSTS and TD may not in fact be very comparable, and may be affected in different ways by various interrupted events during the time, although superficially they seem similar. As pointed out at the beginning of this chapter, it is assumed that all the conditions, such as irrigation runoff, land use and groundwater inflows in the Hunter River, are unchanged under the PHRSTS compared to the TD. Under this assumption, the inception of PHRSTS and river flow are regarded as the only interventions. However, other possible interventions may have some negative effects on the river salinity; these can offset any positive effects of the PHRSTS. Without extra information, it is hard to make a judgment and reach a concrete conclusion.

In response to these limitations, a convincing way of examining the net effect of the PHRSTS on the river salinity of the Hunter River is to compare the river salinity with and without the PHRSTS. Ideally, if the relationship between the river flow, river salinity and industrial salt load inputs could be modelled from readily obtained data, the salinities under a variety of hydrological conditions and management strategies could be easily predicted, and then the net environmental effect of the PHRSTS on the river salinity could be quantified. However, again, the complexity of the salinity process of the Hunter River suggests a need for examining other intensive information on land use, irrigation practices and groundwater movement. This information is less easy to obtain and beyond the scope of the present research. So the next section models the relationship between river salinity, river flow and industrial salt inputs, based
on the available daily observations during 1995-2002, with the aim of quantifying the response of river salinity to the change of flow conditions and the salt discharge from the PHRSTS participants.

### 6.4 Comparison of the environmental performance with versus without-PHRSTS using daily observed data

Statistical analysis to identify variables affecting river salinity must include river flow and industrial salt discharge. Other variables, such as salt load from irrigation and groundwater inflows, could be added but the data are not available. In this section, the relationship between river salinity, river flow and industrial discharge are explored. This relationship will be used to simulate the river salinity under the scenario without-PHRSTS and then assess the environmental effectiveness of the PHRSTS through comparing the river salinity with versus without the PHRSTS.

To do this, the daily observations of river salinity (denoted $ec$), river flow (denoted $flow$)\(^{18}\) and aggregated industrial salt discharge (denoted $d$) at Singleton for the period of 1/1995-11/2002 are used here. These observations are not available for the period before the PHRSTS implementation. The basic information about $ec$, $flow$ and $d$ is provided in Table 6.8.

<table>
<thead>
<tr>
<th>Table 6.8</th>
<th>Basic information about the daily data of $ec$, $flow$ and $d$ (1/1995-11/2002)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sample size</td>
<td>$ec$ = 2891</td>
</tr>
<tr>
<td>Available data</td>
<td>$ec$ = 2633</td>
</tr>
<tr>
<td>Missing data</td>
<td>$ec$ = 258</td>
</tr>
</tbody>
</table>

$ec$ = daily river salinity  
$flow$ = daily river flow  
$d$ = daily aggregated industrial salt discharge

\(^{18}\) The lower case distinguishes the daily from the monthly data, which have been denoted as $EC$ and $Flow$, respectively.
6.4.1 Exploratory analysis

The density plots and qq-plots for \( ec, \) \( flow \) and \( d, \) and for their corresponding \( \ln \)-transformations, denoted as \( \ln(ec), \) \( \ln(flow) \) and \( \ln(d+1) \) respectively, are presented in Figures 6.18, 6.19 and 6.20. Adding 1 to \( d \) values is to avoid the undefined \( \ln(0) \) when there is no discharge from the PHRSTS participants. Also, the ACF and APCF plots for \( ec \) and \( \ln(ec) \) are presented in Figure 6.1.

Figure 6.18 Distribution of \( ec \) and \( \ln(ec) \), daily river salinity, 1/1995-11/2002
Chapter 6 Examination of the Environmental Effectiveness of the PHRSTS

Figure 6.19  Distribution of flow and $\ln(\text{Flow})$, daily river flow, 1/1995-11/2002

(a) Histogram of flow

(b) Histogram of $\ln(\text{flow})$

Figure 6.20  Distribution of $d$ and $\ln(d+1)$, daily industrial salt discharge, 1/1995-11/2002

(a) Histogram of $d$ when $d>0$

(b) Histogram of $\ln(d+1)$ when $d>0$
Figure 6.21  ACF and PACF of ec and ln(ec), daily river salinity, 1/1995-11/2002

![Graphs showing ACF and PACF of ec and ln(ec)]

ACF = autocorrelation function  
PACF = partial autocorrelation function

The figures show that

- The distribution of ec is approximately normal, and follows a normal distribution more closely than ln(ec) does.

- Strong autocorrelation is seen in both ec and ln(ec).

- The distribution of flow is far from normal. This is a problem, since to use flow as an explanatory variable in models for ec, its distribution should be relatively even. The evenness of the distribution of flow is greatly improved by transforming it into ln(flow).

- The distribution of d is also non-normal. A considerable number of nil discharges in the Low flow period and occasionally large values of salt discharge in the Flood flow period are a direct effect of the PHRSTS. Recall that the PHRSTS prohibits discharge during Low flow periods, and allows virtually unlimited discharge during Flood flow periods. As d is to be used as an explanatory variable in models for ec, its distribution should be relatively even. Thus, transforming d to ln(d+1) improves the evenness of the distribution of salt.
Accordingly, $ec$, $ln(flow)$ instead of $flow$, and $ln(d+1)$ instead of $d$ are used in the models. The autocorrelation effect between successive observations is also taken into account.

### 6.4.2 Relationship between $ec$, $ln(flow)$ and $ln(d+1)$

The relationships between daily river salinity $ec$, daily river flow $ln(flow)$ and daily aggregated industrial salt discharge $ln(d+1)$ are explored graphically, two at a time, below.

#### 6.4.2.1 Relationship between $ln(d+1)$ and $ln(flow)$

Figure 6.22 plots daily aggregated industrial salt discharge $ln(d+1)$ against daily river flow $ln(Flow)$ at various river flow periods. As noted in 5.1.3, the PHRSTS regulations regulated the industrial discharge of saline water on the basis of river flow periods. Discharge is only allowed when the river flow is predicted to be High (over 2000 ML/day). This is reflected in Figure 6.22. In Figure 6.22(a), the amount of industrial salt discharge shows generally increasing relationship with increasing river flow when the river flow exceeds the threshold of $ln(2000) \approx 7.6$. Another interesting finding from Figure 6.22(b) is that some discharge events in the Low flow periods should not have occurred. This is due to flow prediction errors. As described in Section 5.2.1, the PHRSTS relies on an Integrated Quantity and Quality Model (IQQM) to predict the flow at Singleton and issues the discharge permits in advance. Although IQQM is reliable, it still generated a relatively low error rate (that is, 1.82%) during 1995-2002 when High or Flood flows were predicted, but actual events were Low flows. This kind of error resulted in the discharge events as shown in Figure 6.22(b). Figure 6.22 (c) and (d) show a number of nil discharges during the High and Flood flow periods. This may be caused by errors in model predictions, or it is possible that there was no need for industry to discharge.
Figure 6.22 Relationship between $\ln(d+1)$ and $\ln(flow)$, daily data, 1/1995-11/2002

(a) All flow periods
(b) Low flow period
(c) High flow period
(d) Flood flow period

$d = \text{daily aggregated industrial salt discharge}$

$flow = \text{daily river flow}$

6.4.2.2 Relationship between $ec$ and $ln(d+1)$

Figure 6.23 plots daily river salinity $ec$ against daily aggregated industrial salt discharge $ln(d+1)$ for the period 1995-2002. No clear relationship between $ec$ and $ln(d+1)$ can be detected from the scatter plot. However, when a smoothing spline is fitted, it can be seen that when $ln(d+1)$ is greater than 2, $ec$ gradually decreases as the discharge increases. Since a large amount of industrial discharge occurs in High and Flood flow periods as the PHRSTS specifies, this suggests industrial discharge has little effect on the river salinity. It appears that river flow is the dominating factor affecting the river salinity levels in the Hunter River.

19 Smoothing spline is an exploratory operation, a means of gaining insight into data without precisely formulated models or hypotheses (Diggle 1990).
Figure 6.23  Relationship between $ec$ and $ln(d+1)$, daily data, 1/1995-11/2002

$ec = \text{daily river salinity}$  
$d = \text{daily aggregated industrial salt discharge}$

6.4.2.3 Relationship between $ec$ and $ln(flow)$

An exploratory method suitable for detecting temporal trends in daily river flow ($flow$) and daily river salinity ($ec$) is to plot them through time, as shown in Figure 6.24. This shows large day-to-day and year-to-year fluctuations in both $flow$ and $ec$. A generally inverse relationship is observed between $ec$ and $flow$. In a model that has diffused high salinity groundwater mixing with fresher surface water, such an inverse relationship is expected and well known (see Section 3.3). Nonetheless, the inverse relationship between $ec$ and $flow$ appears to be different in different periods.
Figure 6.24 \textit{flow} and \textit{ec} at Singleton, daily data, 1/1995-11/2002

\textit{flow} = daily river flow  \\
\textit{ec} = daily river salinity

The daily time series of river salinity (\textit{ec}) shown in Figure 6.24 exhibits two distinct processes. First, an underlying sinusoidal pattern generally peaks during summer Low-flow periods. This pattern represents the receiving water's response to the steady background salt load and high evaporation rate (see Section 3.3), which together produce maximum concentrations during Low flow. Second, sudden peak concentrations sometimes occur. These peaks represent the occurrences of shock loads from the runoff in the first-flush washoff at the start of a rainfall runoff (see Section 3.3) or starts of intensive salt discharges from the industries.

Figure 6.25 shows a scatter plot of \textit{ec} against \textit{ln(flow)} and a fitted smoothing spline. From the graph, dilution of salinity is evident at flows higher than 600 ML/day where \textit{ln(flow)} = 6.4. However, the relationship between \textit{ec} and \textit{ln(flow)} at flows lower than 600 ML/day is complex with a wide range of variation. This could be due to complex salt discharge processes (for example, diffuse groundwater inflows, and salt from agricultural irrigation), and/or the distribution of rainfall across the catchment, or instrumental and data processing errors.
Figure 6.25  Relationship between $ec$ and $\ln(\text{flow})$, daily data, 1/1995-11/2002

![Scatter plot](image)

![Smoothing spline](image)

$flow = \text{daily river flow}$  
$ec = \text{daily river salinity}$

Figure 6.25 highlights two simple hydrological trends of the Hunter River. First, the average river salinity is, at least qualitatively, inversely proportional to the river flow. In general, higher river flows are characterised by lower average salinity. Second, at lower river flows, the variability in average salinity is much higher than that at higher flows.

### 6.4.2.4 Model to predict $ec$ from $flow$ and $d$

As observed in Section 6.4.1, the river salinity of any one day not only directly depends upon the salt inputs from various sources and the river flow conditions of that day, but is also indirectly affected by the in-river salinity on the preceding days. The autocorrelation inherent in the river salinity is also present in the river flow, which varies over time. Using S-plus, the following model was fitted, allowing autocorrelation of various orders up to AR(8) in $\epsilon$, where $b$ and $c$ measures the effect of $flow$ and $d$ on $ec$ separately, $d$ measures the joint effect of $flow$ and $d$ on $ec$.

\[
ec = a + b \ln(flow) + c \ln(d + 1) + d \ln(flow) \times \ln(d + 1) + \epsilon, \quad [6.8]
\]
In such modelling, the main effects are usually first fitted, followed by the AR terms. Running the models showed that a possible convergence problem arises from $ec.ar5$. This implies there is no further improvement in the result of the models that allow autocorrelation of orders higher than AR(4). Therefore, only the coefficients of the estimates and the indicators for model selection for the models up to ec.ar4 are summarised in Table 6.9.

Table 6.9  Fitted model of ec.flow and $d$

<table>
<thead>
<tr>
<th></th>
<th>AR(1)</th>
<th>AR(2)</th>
<th>AR(3)</th>
<th>AR(4)</th>
<th>a</th>
<th>b</th>
<th>c</th>
<th>d</th>
<th>AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>ec.i</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>33646.20</td>
</tr>
<tr>
<td>ec.ar1</td>
<td>0.96</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>27627.62</td>
</tr>
<tr>
<td>ec.ar2</td>
<td>0.98</td>
<td>-0.03</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>27618.57</td>
</tr>
<tr>
<td>ec.ar3</td>
<td>0.98</td>
<td>-0.12</td>
<td>0.11</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>27588.72</td>
</tr>
<tr>
<td>ec.ar4</td>
<td>0.97</td>
<td>-0.11</td>
<td>0.04</td>
<td>0.07</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>27579.52</td>
</tr>
</tbody>
</table>

AIC = Akaike Information Criterion
ec.i = fitted model without accounting for autocorrelation effect
ec.ar1, ec.ar2, ec.ar3, ec.ar4 = fitted model accounting for autocorrelation effect at lag1, lag2, lag3 and lag4 respectively

Comparing the AICs in the above Table, it can be seen that the last four models which consider the autocorrelation effect are superior to the one without considering the autocorrelation effect as their considerable lower AIC values. Among them, ec.ar4, the model that includes the effect of river salinity of the previous four days, has the smallest value of AIC and is thus the best model to fit the data. The residual cumulative periodogram curves in Figure 6.26 provide evidence of the goodness of fit of ec.ar4. Figure 6.26 shows that ec.ar1, ec.ar2, ec.ar3 and ec.ar4 are much more adequate than that which ignores autocorrelation (that is, ec.i). Furthermore, the fitness of ec.ar4 is the best among all.
Chapter 6 Examination of the Environmental Effectiveness of the PHRSTS

Figure 6.26  Cumulative periodograms of fitted models for ec

\[ ec = 1061.19 - 65.28 \ln(\text{flow}) + 12.17 \ln(d + 1) - 1.57 \ln(\text{flow}) \times \ln(d + 1) + \varepsilon, \]

[6.9]

In [6.9], the significance of the coefficients of \( \ln(\text{flow}) \) and \( \ln(d+1) \) is less visible. When \( d = 0 \), the coefficient of the \( \ln(\text{flow}) \) parameter is -65.28, showing that as flow increases, ec decreases. At higher \( d \) values, because \( \ln(d+1) \) is positive, the negative coefficient (that is, -1.57) on the interaction term \( \ln(\text{flow}) \times \ln(d+1) \) makes ec decrease faster than at \( d = 0 \). At lower flow, when \( d \) increases, ec increases. In contrast, at higher flow, when \( d \) increases, ec decrease. This means
that while industrial salt discharge has an effect on river salinity, such an effect is more evident in Low flow than in High flow.

6.4.2.5 Comparison of river salinity with and without the PHRSTS

As stated earlier, one way to evaluate the net environmental effectiveness of the Pilot Hunter River Salinity Trading Scheme (PHRSTS) is to compare the river salinity with and without the PHRSTS. Since the Trickle Discharge (TD) system dominated the NSW EPA’s regulations on industrial saline water discharge before the introduction of the PHRSTS in 1993, this study stimulates the daily river salinities at Singleton under the TD using Model [6.9] and further assumptions below, and compares the simulated salinities with the actual salinities under the PHRSTS.

Under the TD system, industries were allowed to discharge saline water to the Hunter River at a low rate on a more-or-less continuous basis all year round, subject to various licensing conditions (see Section 4.2). Because information about discharge frequency, discharge concentration and volume on a daily basis from each mine and power station is not available, and given that the underlying philosophy of this simulation is to re-distribute the industrial discharge within the entire period of the PHRSTS so as to capture the net environmental benefits of the PHRSTS, the following restrictive assumptions were made in this study:

- In the period of 1/1995-11/2002, the aggregated amounts of industrial salt discharged under the PHRSTS and TD system are the same;
- Under the TD system, the industrial salt is discharged at a continuous, stable and even rate. This assumption enables the daily salt discharge under the TD system to be estimated by averaging the aggregated amount of salt discharge under the PHRSTS.

Let \( d_i \) (tonnes/day) be the aggregated salt load discharged by the PHRSTS participants on any day during the PHRSTS period. The daily salt load discharged by the participants under the TD, denoted \( d \), is

\[
d = \frac{\sum_{t=1}^{T} d_{it}}{T}
\]  

[6.10]

\( T \) is the number of days during 1/1995-11/2002, which is 2981 in total. Putting \( d \) into [6.9], the daily river salinity under the TD system, denoted \( \hat{e}_c \), is predicted as follows:
\[
\hat{c} = 1061.19 - 65.28 \ln(\text{flow}) + 12.17 \ln(d + 1) - 1.57 \ln(\text{flow}) \times \ln(d + 1) + \varepsilon_t \quad [6.11]
\]

By the definition in [6.4] and the result in Table 6.9, in the analysis here

\[
\varepsilon_t = 0.97 \varepsilon_{t-1} - 0.11 \varepsilon_{t-2} + 0.04 \varepsilon_{t-3} + 0.07 \varepsilon_{t-4} + \delta_t \quad [6.12]
\]

where \( \delta_t \) is a random error on day \( t \). By including the term \( \varepsilon_t \), the effect of the river salinity in the previous four days is included in the prediction.

The observed river salinity under the PHRSTS (that is, \( ec \)) and the stimulated river salinity under the TD system (that is, \( \hat{c} \)) are compared using smoothing splines in Figure 6.27. These two lines intersect at approximately \( \ln(\text{flow}) = 6.55 \) where \( \text{flow} = 700 \text{ ML/day} \). This is the direct effect of prediction errors generated by the IQQM. If the IQQM had predicted river flow at Singleton as precisely as it should have occurred, and there was no actual discharge from the industries in the Low flow period as the PHRSTS specified, the two lines are expected to intersect at \( \ln(\text{flow}) = 7.6 \) where \( \text{flow} = 2000 \text{ ML/day} \). This procedure clearly shows that without the PHRSTS the river salinity at lower flow period (\( \text{flow} < 700 \text{ ML/day} \)) would be much higher than that with the PHRSTS, and the river salinity at higher flow periods (\( \text{flow} > 700 \text{ ML/day} \)) would be lower than those with the PHRSTS. This is the anticipated result — the low values of river salinity (which occurred in higher flow period) under the TD are lower than those under the PHRSTS, and high values of river salinity (which occurred in lower flow period) under the TD are higher than those under the PHRSTS — since under the PHRSTS no industrial salt was discharged at Low flows and more salt was discharged at High and Flood flows. However, the Figure shows a modest difference between \( ec \) and \( \hat{c} \).
Chapter 6 Examination of the Environmental Effectiveness of the PHRSTS

Figure 6.27  Observed ec of the PHRSTS versus stimulated ec of the TD

\[ ec = \text{daily river salinity} \]
\[ flow = \text{daily river flow} \]
PHRSTS = Pilot Hunter River Salinity Trading Scheme
TD = Trickle Discharge

The exceedance curves for the river salinity under the PHRSTS and TD are plotted in Figure 6.28. The Figure shows that the curves under these two situations are too close to be seen separately. Table 6.10 provides a comparison of the exceedance at some key thresholds of river salinity for both. It shows that the PHRSTS would have reduced the number of days that river salinity exceeded 900 EC by 0.57%, compared to without the PHRSTS.
Chapter 6 Examination of the Environmental Effectiveness of the PHRSTS

Figure 6.28  Exceedance curves for $ec$ under the PHRSTS and under the TD

$ec = \text{daily river salinity}$  
PHRSTS = Pilot Hunter River Salinity Trading Scheme  
TD = Trickle Discharge

<table>
<thead>
<tr>
<th>Daily river salinity $ec$</th>
<th>Exceedances (%)</th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>PHRSTS</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>200</td>
<td>99.85</td>
<td>99.96</td>
<td>0.11</td>
<td></td>
</tr>
<tr>
<td>500</td>
<td>84.31</td>
<td>87.81</td>
<td>3.49</td>
<td></td>
</tr>
<tr>
<td>600</td>
<td>64.34</td>
<td>66.54</td>
<td>2.20</td>
<td></td>
</tr>
<tr>
<td>900</td>
<td>4.60</td>
<td>5.17</td>
<td>0.57</td>
<td></td>
</tr>
<tr>
<td>1000</td>
<td>0.87</td>
<td>1.06</td>
<td>0.19</td>
<td></td>
</tr>
<tr>
<td>1100</td>
<td>0.04</td>
<td>0.08</td>
<td>0.04</td>
<td></td>
</tr>
<tr>
<td>1200</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td></td>
</tr>
</tbody>
</table>
Using a regression with autocorrelated errors, this study has predicted river salinity subject to the same river flow conditions as under the PHRSTS, but assuming that industrial discharge takes place under the TD system. The analysis shows that the difference in the river salinity under the PHRSTS and TD is minor. So the data show little if any effect of PHRSTS on the improvement of water quality in the Hunter River.

However, several factors may cause some profound effects on these predictions. Firstly, regression with autocorrelated errors assumes the explanatory variables in a model are independent of each other. However, in this case, the industrial discharge is not a completely independent variable from the river flow. Despite the fact that the amount of actual salt discharge is not closely related to the river flow, the occurrence of the discharge (that is, the time that industrial saline water is discharged into the Hunter River) is determined by the river flow periods. Therefore, there is a connection between flow and \( d \) in the model.

Secondly, the nil salt discharge from the PHRSTS participants in the Low flow period could mask the effect of the discharge on the river salinity under a Low flow period to some extent. Strong dilution in the High flow period could also weaken the effect of the industrial discharge. These two aspects would jointly produce a biased model.

Thirdly, in this prediction, the industrial salt discharge under the TD is estimated by averaging the aggregated amount of discharge under the PHRSTS during the whole period of 1/1995-11/2002. This may not reflect the real situation of TD. For example, in the real situation, especially in its later stage, the TD not only specified the ambient level of the River, but also imposed a limit of incremental river salinity on some industrial dischargers. It is plausible that the total industrial salt under the TD system would be less than that under the PHRSTS. The rough estimate of industrial discharge under the TD may have some effect on the result of river salinity prediction.

### 6.5 Estimation of environmental benefit

The above analysis has revealed that the river salinity of the Hunter River under the PHRSTS is not significantly different from that under the TD. As such, no considerable saving in social damage cost generated by the PHRSTS is expected. Even so, the cost saving is still estimated here for completeness and comparison. The methodology of this section could provide a
reference for other studies that evaluate the environmental effectiveness of water quality management programs.

As noted in Section 3.6, there is still a considerable gap in the knowledge of the costs associated with elevated salinity in the Hunter River. Given that investigation of the social costs of the Hunter salinity is beyond the scope of this study, and the AGC Woodward-Clyde’s study (1992) represents the best estimation in this respect, this study borrows the figures from the latter study to estimate the social benefits derived from the PHRSTS in improving the water quality in the Hunter River.

AGC Woodward-Clyde (1992) estimated that the annual cost (in 1992 prices) of an increase of 1 EC, starting from a conservative threshold of 600 EC, was $10,000 for the Hunter River in 1992 prices, of which $7000 would be borne by urban users, $1200 by non-urban users and $1800 by irrigators. The daily social cost when the river salinity exceeds 600 EC by one unit can then be estimated in 2002 prices using the following formula:

\[
DC_{2002} = \frac{10000}{365} \times \frac{I_{2002}}{I_{1992}}
\]

where \(DC_{2002} (\$/EC.day)\) is the daily social cost of one incremental EC in 2002 prices. \(I_{1992}\) and \(I_{2002}\) are price indexes of 1992 and 2002 respectively. Based on Australian Bureau of Statistics (ABS, 2008), \(I_{1992} = 107.5\) and \(I_{2002} = 138.1\). Accordingly, an increase in one unit of EC starting from 600 EC would cause a social cost of \(DC_{2002} = \$35/EC.day\) in 2002 prices.

Thus, the total social cost during the entire period of the PHRSTS can be approximately calculated by the below Equation [6.14]:

\[
TC_{2002} = DC_{2002} \sum_{N=1995}^{2002} \sum_{r=1}^{n} (c_r - 600)(1+r)^{2002-N} \quad (c_r > 600)
\]

where \(TC_{2002} (\$)\) is total social cost during the PHRSTS period in 2002 prices. \(c_r (EC)\) represents the daily river salinity that exceeds 600 EC. \(N\) represents any year during 1995-2002. \(n\) is the total number of days that daily river salinity exceeded 600 EC within year \(N\). \(r\) is the discount rate used to convert a series of costs into a present value in 2002 prices.

In the literature, there has been a long debate on what discount rate is appropriate for a regulatory program or public investment project that provides benefits and costs to the general
public. While social time preference and opportunity cost of capital are the two most common bases for the setting of the discount rate, there has been no universally accepted correct discount rate for the cost-benefit analysis of regulatory policies and public projects (Morrison 1998). This study has adopted a central real discount rate of 7% per year, as recommended by the NSW Treasury (2007).

Putting the observed and simulated river salinity from Section 6.4.2.5 into [6.14] separately, the total social costs under the PHRSTS and TD system have been calculated and are presented in Table 6.11. They show that the PHRSTS would save approximately $0.33 million (in 2002 prices) in total social cost that would have been incurred under the TD system during 1995-2002, which is equivalent to $41K per annum.

<table>
<thead>
<tr>
<th>Table 6.11 Comparison of damage costs under the PHRSTS and the TD system</th>
</tr>
</thead>
<tbody>
<tr>
<td>Social costs</td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td>Trickle Discharge (TD)</td>
</tr>
<tr>
<td>Pilot Hunter River Salinity Trading Scheme (PHRSTS)</td>
</tr>
<tr>
<td>Cost saving Savings</td>
</tr>
</tbody>
</table>

As discussed in Section 3.6, AGC Woodward-Clyde assumed a linear MSD function for the elevated river salinity of the Hunter River. This linear function is regarded as under-estimating the social costs of the impact of those river salinities that are over 1500 EC. Nevertheless, as none of the observed and stimulated river salinities exceed 1500 EC (see Table 6.10), the under-estimation of the MSD has no effect on the estimates of the saving in the social cost generated by the PHRSTS.

The saving in the social cost as a result of the river salinity improvement under the PHRSTS is minimal, as expected from the statistically insignificant impact of the PHRSTS (compared with the TD) on salinity found in the previous section.
6.6 Summary

This chapter has examined the environmental effectiveness of the Pilot Hunter River Salinity Trading Scheme (PHRSTS) from two perspectives. Firstly, the long-series monthly mean observations of the river flow and salinity have been examined using advanced regression models that account for the time-series (autocorrelation) effect in the data. It has been revealed that the differences in the overall river salinity and flow between the PHRSTS and the Trickle Discharge (TD) are not statistically significant. Furthermore, even for the Low flow period, the difference in the river salinity between the PHRSTS and the TD is modest. Secondly, the relationship of river salinity, flow and industrial salt load discharge has been established by a regression model with autocorrelated errors using available daily observations of 1995-2002. Based on this relationship, the river salinity of without-PHRSTS scenario has been predicted. Comparison of the river salinity with and without-PHRSTS indicates that PHRSTS had a minimal effect on the improvement of river salinity in the Hunter River. It is estimated that the PHRSTS saved only about $0.33 million (in 2002 prices) of total social cost of river salinity during its whole lifetime.

This chapter concludes that the PHRSTS, although attaining the overall river salinity objectives, did not significantly improve the river salinity of the Hunter River. As such, the PHRSTS only generated a trivial saving in the social cost of the river salinity, contradicting the NSW EPA’s claim reported in Box 1.1 of Chapter 1.
Chapter 7 Examination of the Economic Effectiveness of the PHRSTS

This chapter evaluates the economic performance of the Pilot Hunter River Salinity Trading Scheme (PHRSTS). It addresses the second main research question of this study set out in Chapter 1— did the PHRSTS generate substantial saving to its participants in control costs? The chapter proceeds as follows. Section 7.1 presents an innovative, purpose-built, theoretical model for estimating the total control cost of the PHRSTS to coal mines and power stations under various discharge permits schemes, following the theoretical framework constructed in Section 2.4. The lack of the required data is the largest barrier to the application of this theoretical model, and indeed it cannot be used empirically in this thesis. Even so, the methodology of this theoretical model will offer a useful reference for other research on water quality management, in particular for cost estimation for storable river pollutants, an area which is sparse in the literature. Quite separately, in Section 7.2 an operational, empirical model is developed to estimate the saving in the control cost of the PHRSTS over the TD for its 20 initial participants and the two new developments. The overall saving is apportioned between the two main elements of the PHRSTS, namely the dynamicism and the tradability of its discharge permits. The significance of the total cost saving to the PHRSTS participants is also measured. To this end, this section provides the first set of quantitative measurements of the economic effectiveness of the PHRSTS.

7.1 Theoretical model for estimating saline water control costs of four regulatory schemes

7.1.1 Hypothetical regulatory schemes

Estimating the cost effectiveness of the PHRSTS requires, in part, characterising the costs of saline water control required in the absence of the PHRSTS. These without-PHRSTS cost estimates, combined with an estimate of the with-PHRSTS conditions, provide the basis for estimating the cost savings of the PHRSTS over other schemes.

In this study, four regulatory schemes for managing industrial saline water discharge have been developed following the analytical framework constructed in Section 2.4. The definitions of these regulatory schemes are consistent with the general definitions of SN, ST, DN and DT in
Table 2.2, but reflect features of historical water quality management strategies for the Hunter River as presented in Chapter 4. So the four acronyms SN, ST, DN and DT have slightly different meanings here from those in Table 2.2. The four schemes are defined below.

(1) Static non-tradable permits (SN) - approximately corresponding to the Trickle Discharge (TD) scheme

- The total permitted daily discharge of salt (measured as tonnes/day) for the whole river is defined based on the lowest daily river flow and its corresponding river salinity background concentrations from the historical observations. Once this discharge is defined, it remains unchanged over time.

- Daily permitted discharge of salt for an individual discharger (that is, coal mine or power station) is defined in proportion to the PHRSTS initial credit allocation. Therefore, for each discharger, the amount of salt discharged is fixed.

- Discharge permits cannot be traded. The salt load of each discharger is not allowed to exceed its corresponding discharge permits.

(2) Static tradable permits (ST) – hypothetical

- The daily discharge permits of salt load are defined in the same way as the above SN.

- Dischargers are allowed to trade permits as long as their aggregated daily discharge of salt load does not exceed the fixed daily permit.

(3) Dynamic non-tradable permits (DN) - hypothetical

- The total permitted daily discharge of salt load is determined by conditions in the river. In principle, it should be defined based on instantaneous daily river flow and its corresponding salinity background concentrations. It therefore varies from day to day. To be consistent with the dynamic component of the PHRSTS, total permitted daily salt load is defined as zero when the river flow is Low, infinite when the river is in Flood flow, and could be any intermediate values when the river is in High flow (as described in Chapter 4).
- The permitted daily salt load for an individual discharger is defined in accordance with the PHRSTS initial credit allocation, that is, a fixed share allocated initially (in percentage) multiplied by a variable daily total allowable salt load permits results in a variable daily salt load for an individual discharger.

- Discharge permits cannot be traded. The salt load of each discharger is not allowed to exceed that stipulated in its corresponding discharge permits.

(4) Dynamic tradable permits (DT) - corresponding to the PHRSTS

- The discharge permits of salt load are defined in the same ways as the above DN.

- Dischargers are allowed to trade permits as long as their aggregated daily salt load discharged does not exceed the fixed daily total allowable salt load.

For a firm producing outputs and discharging regulatory pollutants, there are two basic ways to control its discharges: investing in pollution control facilities, and scaling down its production activities (and therefore reducing outputs). In either case, discharge reduction entails costs. The theoretical framework that applies to the firm’s trade-off between these two processes is total cost function, which combines both production cost and pollution abatement cost. As elaborated by Carlson et al. (2000), any economically sensitive firm tends to choose general inputs of capital $k$, materials $m$, and labour $l$ to minimise the total cost ($TC$) of producing output $q$ and achieving a discharge rate $d$ in time period $t$, subject to production and discharge constraints. This process is expressed by the following equation:

$$\text{min } TC = p_k k + p_m m + p_l l$$ \hspace{1cm} [7.1]

Subject to

$$q(k,m,l) \geq q^*$$ \hspace{1cm} [7.2]

$$d(k,m,l) \leq d^*$$ \hspace{1cm} [7.3]

where $p_k$, $p_m$ and $p_l$ are the price of $k$, $m$ and $l$, $q^*$ represents the target of minimum outputs, $d^*$ represents the discharge permits expressed in the load of a pollutant. In Equation [7.1], unit and time indexes are suppressed for convenience.
In a situation where the generation of discharge is not very relevant to the level of productivity (for example, the coal mines in this study), or where firms do not tend to control their discharge through reducing the level of their outputs (for example, the power stations in this study), production costs may be excluded from the cost analysis of discharge control activities. The total cost function can then be simplified to an abatement cost function that relates inputs of capital, materials and labour to discharge reduction. As the firms’ abatement behaviours in response to regulatory policies are the concern of this study, the abatement cost is expressed by the following function that explicitly incorporates actual discharge $d$ as an independent variable

$$C = C(p_k, p_m, p_l, d, t)$$ \[7.4\]

In the abatement cost function [7.2], $d$ is an endogenous variable. It is subject to discharge permits $d^*$ that may be different under different regulatory schemes. In other words, different $d^*$ may require and result in different effects on firms’ abatement activities and performance, and in turn on their associated costs.

In the case study presented here, coal mines and power stations adopt different options for saline water control. As presented in Sections 3.4 and 3.5, while storing excess water on-site without any treatment is a viable option for the coal mines, desalination is a common technique used by power stations to extract the salt. Accordingly, theoretical models for estimating the costs of saline water control for coal mines and power stations are developed separately here.

### 7.1.2 Storage costs for coal mines

Storage of pollutants, that is, treatment and storage of pollutants for a period of time and releasing them into the environment later when adverse effects are minimal, represents an important aspect of the pollution management strategy. Storage, by temporally separating pollutant generation and discharge, has a theoretical attraction when the relative benefits of generating pollutants and costs of discharging pollutants vary through time (Lee 1977). However, so far there is little theoretical research on this area, nor are there empirical studies on estimating the costs of storage of pollutants.

Loosely following Lee (1977), Lewis (1981) and O’Neill (1983a), this study has developed a theoretical model for estimating the costs of storing mine water on-site under various regulatory
schemes as follows. Assuming there were no on-site dams constructed before the PHRSTS implementation, if the data on excess mine water for individual coal mines were available, the storage requirement of a mine under the four regulatory schemes (SN, ST, DN and DT) can be simulated. Furthermore, once information on the marginal construction and operational costs of storage for an individual mine is available, the optimal capacity of on-site storage dams with a minimised total cost for the whole PHRSTS coal mines can be estimated.

The notation used in the model, and its meaning, is as follows:

\( i \) Any firm of mining industry.

\( m \) The number of all firms of mining industry, \( i = 1, \ldots, n \).

\( T \) Number of days during 1/1/1995 to 30/11/2002 under the PHRSTS. In total, \( T = 2891 \).

\( v_i \) Volume of the saline water generated by firm \( i \) on day \( t \) (ML).

Its cumulative generation of the saline water up to \( t \) days is

\[
V_i = \sum_{x=1}^{t} v_i 
\]

\( c_i \) Salinity concentration of the saline water generated by firm \( i \) (µS/cm). For simplicity, it is assumed constant over time for firm \( i \), but may vary from mine to mine.

\( f \) Conversion factor between EC and mg/L. In general, \( 1 \text{EC} \approx 0.6-0.68 \text{ mg/L} \). For the Hunter River, \( f = 0.6 \) applies (NSW EPA 1995).

\( g_i \) Firm \( i \)'s generation of salt load on day \( t \) (tonnes).

Then, \( g_i = f v_i c_i \) [7.6]

Its cumulative generation of salt load up to \( t \) days is

\[
G_i = \sum_{x=1}^{t} g_i 
\]

\( w_i \) Volume of actual discharge from firm \( i \) on day \( t \) (ML).

Its total volume of actual discharge up to \( t \) days is

\[
W_i = \sum_{x=1}^{t} w_i 
\]

\( d_i \) Firm \( i \)'s actual discharge of salt load on the day \( t \) (tonnes).

Then, \( d_i = f w_i c_i \) [7.9]

The total discharge from the mining industry, denoted \( m \), on day \( t \) is

\[
D_m = \sum_{i=1}^{m} d_i = f \sum_{i=1}^{m} w_i c_i 
\]
Chapter 7 Examination of the Economic Effectiveness of the PHRSTS

$A_i$ Firm $i$'s storage capacity for saline water (ML).

$K_i(A_i)$ Firm $i$'s annualised capital cost of storage construction ($$/year).

The cost of storing saline water on-site is made up of the capital cost incurred to construct dams, and the cost associated with operating and maintaining the dams. Therefore, separate equations are needed to specify the cost components.

The annualised capital cost of storage construction $K_i(A_i)$ is the function of storage capacity $A_i$. It is reasonable to believe that $K_i$ is an increasing, though not necessarily linear, function of $A_i$. That is, the larger the capacity is, the more the capital cost would be ($K_i'(A_i) > 0$). Then the capital cost of storage construction for the mining industry, denoted $KC_m$, is

$$KC_m = \sum_{i=1}^{m} K_i(A_i) \quad [7.11]$$

Operational activities of storage dams include, but may be not limited to, water pumping, monitoring the water quality in the storage dams under a range of flows, identifying periods when they may be discharged, and monitoring the discharge rates to ensure the downstream waterways are in fact being protected. Storage dams also need regular maintenance: inlet and outlet structures need to be kept clear of debris; litter needs to be collected and removed regularly; and water levels need to be altered at particular periods to prevent algal blooms from forming (Cullen et al. 1988; Lawrence and Breen 1998). Therefore, it is reasonable to divide the operation and maintenance (O&M) cost of any mine into two separate parts: fixed O&M cost (denoted $FC_i$) and variable O&M cost (denoted $OC_i$).

$FC_i$ refers to those O&M costs that neither change with the size of storage dam $A_i$, nor change with duration and volume of the saline water stored. The above listed monitoring cost is one of the examples. However, $FC_i$ may vary from mine to mine. So, the annual fixed O&M cost for the whole industry, denoted $FC_m$, is

$$FC_m = \sum_{i=1}^{n} FC_i \quad [7.12]$$

In contrast to $FC_i$, $OC_i$, at least in principle is the function of the number of days that the saline water is stored and the volume of the saline water stored. The most representative cost falling into this category is the cost (for example, electricity consumption) associating with water
pumping. Let \( a_{w0} \) be the accumulation inherited at the start of day 1. It reflects the history of stored water before the day being studied. Then

The accumulation at the end of day 1 is \( a_{w1} = a_{w0} + v_{w1} - w_{w1} \)

The accumulation at the end of day 2 is \( a_{w2} = a_{w1} + v_{w2} - w_{w2} = a_{w0} + v_{w1} - w_{w1} + v_{w2} - w_{w2} \)

The accumulation at the end of day \( t \) is \( a_{wt} = a_{w(t-1)} + v_{wt} - w_{wt} \)

\[
\begin{align*}
&= a_{w0} + v_{w1} - w_{w1} + v_{w2} - w_{w2} + \ldots + v_{wt} - w_{wt} \\
&= a_{w0} + \sum_{t=1}^{t} (v_{wt} - w_{wt}) \\
&= a_{w0} + V_{wt} - W_{wt}
\end{align*}
\]

So, the variable O&M cost on day \( t \) for mine \( i \) is \( OC_{wt} = O_i a_{wt} = O_i (a_{w0} + V_{wt} - W_{wt}) \)

The variable O&M costs over all \( T \) days for all \( m \) mines (that is, the annual industry cost) denoted \( OC_m \), is

\[
OC_m = \sum_{i=1}^{m} \sum_{t=1}^{T} OC_{wt} = \sum_{i=1}^{m} \sum_{t=1}^{T} O_i a_{wt}
\]

[7.13]

Since \( O_i \) is not dependent on time \( t \), it can be taken outside the second summation, giving

\[
OC_m = \sum_{i=1}^{m} \sum_{t=1}^{T} O_i a_{wt} = \sum_{i=1}^{m} O_i \sum_{t=1}^{T} (a_{w0} + V_{wt} - W_{wt})
\]

[7.14]

Combining Equations [7.11], [7.12] and [7.14], the total annual cost of storing mine water on-site for the mining industry \( TC_m \), is

\[
TC_m = KC_m + FC_m + OC_m = \sum_{i=1}^{m} K_i (A_i) + \sum_{i=1}^{m} FC_i + \sum_{i=1}^{m} O_i \sum_{t=1}^{T} (a_{w0} + V_{wt} - W_{wt})
\]

[7.15]

It could be envisaged that the marginal cost \( OC_{wt} \) in response to the time and amount of saline water stored would virtually not be of significance, and could be omitted from [7.15]. Then [7.15] can be simplified as

\[
TC_m = \sum_{i=1}^{m} K_i (A_i) + \sum_{i=1}^{m} FC_i
\]

[7.16]
7.1.3 Treatment costs of power stations

As described in Section 3.5.1.3, the power stations directly or indirectly withdraw water from the Hunter River for boiler feed water and make-up water and for cooling condensers and reactors. The water is pre-treated to eliminate salt and other contaminants, then recycled many times for plant use and eventually discharged to the Hunter River. In this case, this study can treat each power station as an individual wastewater treatment plant that involves a desalination process.

Given this simplification, the method of analysing costs for wastewater treatment plants is applicable to the power stations on saline water treatment. The engineering model is a prevailing method used to analyse the cost associated with water treatment. Borrowing the idea from Fraas and Munley's study (1984), the costs of any water treatment system depend upon the volume of the flow, concentration of influent stream, and concentration of effluent stream. The following general form of the cost function applies

\[ C = f(F, I, E, P) \]  \hspace{1cm} [7.17]

where \( C \) is the cost of wastewater treatment, \( F \) is flow size of the waste stream, \( I \) is the concentration of pollutant in the influent stream, \( E \) is the concentration of pollutant in the effluent stream, and \( P \) is the vector of prices for factor inputs.

Following [7.17], the capital cost and operational cost for desalination processes of each power station need to be estimated separately. This is because the magnitude of capital cost is expected to relate to design flow and performance, while operating cost would be related to actual flow and performance.
For the desalination treatment facility of any power station $j$, let

- $F_{d_j}$: Designed influent capacity (ML/day)
- $F_{a jt}$: Actual flow of influent on day $t$ (ML/day)
- $I_{d}$: Designed salinity concentration of influent (EC)
- $I_{a t}$: Actual salinity concentration of influent on day $t$ (EC)
- $E_{d_j}$: Design salinity concentration of effluent (EC)
- $E_{a t}$: Actual salinity concentration of effluent on day $t$ (EC)
- $P_{d_j}$: Vector of annualised prices for capital inputs ($/year)
- $P_{a t}$: Vector of prices for the inputs of actual performance ($/day)

The annual capital cost for power station $j$ is:

$$K_j = f(I_{d_j}, E_{d_j}, F_{d_j}, P_{d_j})$$  \[7.18\]

The annual capital cost for the power generating industry with $n$ power stations is:

$$K_g = \sum_{j=1}^{n} K_j$$  \[7.19\]

The operating cost on day $t$ for power station $j$ is:

$$OC_{jt} = (I_{a t} - E_{a t})F_{a t}P_{a t}$$  \[7.20\]

The operating cost over all $T$ days ($T=2891$) for all $n$ power stations is:

$$OC_g = \sum_{j=1}^{n} \sum_{t=1}^{T} OC_{jt} = \sum_{j=1}^{n} \sum_{t=1}^{T} (I_{a t} - E_{a t})F_{a t}P_{a t}$$  \[7.21\]

Thus, the total treatment cost for the power generating industry is

$$TC_g = K_g + OC_g = \sum_{j=1}^{n} K_j + \sum_{j=1}^{n} \sum_{t=1}^{T} (I_{a t} - E_{a t})F_{a t}P_{a t}$$  \[7.22\]
From [7.18] and [7.20], the treatment cost for any power station is positively related to the influent flows \((F_d \text{ and } F_{ai})\) and the pollutant concentration of influent \((I_d \text{ and } I_{ai})\), and negatively related to the pollutant concentration of effluent \((E_d \text{ and } E_{ai})\). That is, increases in flow size and influent concentration raise treatment costs, but increases in the level of effluent concentration (lower effluent quality) reduce treatment costs. Within a power station’s microeconomic framework, an obvious trade-off is between obtaining cleaner influent and minimising treatment cost. This is especially true for the Liddell Power Station, which uses Lake Liddell as a buffer for storing intake water and withholding discharge. The salinity of its effluent may directly affect that of its influent, and in turn their treatment costs.

Let \(d_j\) be power station \(j\)’s actual salt load discharge on day \(t\), then

\[
d_{jt} = E_{aij} F_{aij} \tag{7.23}
\]

The total salt load discharge from the power generating industry, which includes \(n\) power stations, on day \(t\) is

\[
D_{gt} = \sum_{j=1}^{n} d_{jt} = \sum_{j=1}^{n} E_{aij} F_{aij} \tag{7.24}
\]

For simplicity, it is assumed for each power station

- The salinity concentration of influent \(I_{ai}\) is exogenous; meaning the salinity of influent is out of the power stations’ control and may change over time.

- The influent flow \(F_{ai}\) has some implications for the amount of electricity generated and the times of recycling. However, power stations do not reduce their electricity product or increase the recycling times in order to reduce the volume of water treated.

- The effluent flow is approximately the same as the influent flow in volume on day \(t\). That is, the loss of water in the power generating and cooling processes is omitted.

Under these restrictive assumptions, power stations may change their operating costs by choosing different levels of treatment, that is, by altering \(E_{ai}\), subject to variations in the salt load discharge permits under different regulatory schemes. When \(F_{ai}\) is fixed, the more discharge permits a power station obtains, the higher the salinity effluent it will discharge, and the less cost it would incur in the desalination processes. However, the capital cost of treatment
does not change in response to different regulatory schemes, since the treatment facilities have to be constructed to account for the higher effluent quality to meet the rigid salt load discharge permits in the worst situations (for example, no discharge is allowed in the Low flow periods under the HRSTS) and to accommodate large swings in effluent quality responding to salt load discharge permits.

7.1.4 Minimisation of total control costs

From Equations [7.10] and [7.24], the total salt load discharged from the whole industry (including mines and power stations) on any day $t$ is

$$D_t = D_m + D_p = \left( \sum_{i=1}^{n} w_{pi} c_{ij} \right) + \sum_{j=1}^{n} E_{aqf} F_{aqj}$$  \hspace{1cm} [7.25]

From Equations [7.15] and [7.22], the total control costs of the whole industry over $T$ days is

$$TC = TC_m + TC_p$$

$$= \left( \sum_{i=1}^{n} K_i (A_i) + \sum_{i=1}^{n} FC_i + \sum_{i=1}^{n} O \sum_{t=1}^{T} (a_{t0} + V_{t} - W_{t}) \right) + \sum_{j=1}^{n} K_j + \sum_{j=1}^{n} \sum_{t=1}^{T} (I_{aqj} - E_{aqj}) F_{aqj} P_{aqj}$$  \hspace{1cm} [7.26]

$TC$ is minimised by actions of both firms in response to regulations, and by the regulator’s choice of regulatory scheme and the parameters within that scheme. Assuming that each mine treats its $v_{t}$ (volume of saline water generated) as exogenous, that is, beyond its control, and assuming that power stations are able to operate flexibly their levels of removal of desalination facilities in response to different regulations, and further assuming that:

(1) Using the notations of the schemes set out in 7.1.1, **firms** collectively minimise either $TC_{SN}$ (for static non-tradable permits), $TC_{DN}$ (for dynamic non-tradable permits), $TC_{ST}$ (for static tradable permits), or $TC_{DT}$ (for dynamic tradable permits), depending on which scheme is chosen by the regulator, and

(a) for the mines, by choosing their dam sizes $A_i$ and daily discharge volumes $w_{pi}$, subject to the following constraints:

- Neither generation nor discharge of mine water can be negative
$v_i \geq 0, \; w_i \geq 0, \; \text{for all } i \text{ and } t$ \hfill [7.27]

- The amount of mine water that mine $i$ stores in its dam on day $t$ can never be negative, or greater than its dam capacity $A_i$:

$0 \leq a_i \leq A_i, \; \text{for all } i \text{ and } t$ \hfill [7.28]

(b) for power stations, by choosing the salinity concentration $E_{aq}$ and flow $F_{aq}$ of their effluent, subject to the following constraints:

- Flow of effluent can never be negative

$F_{aq} \geq 0 \; \text{for all } j \text{ and } t$ \hfill [7.29]

- Actual flow of effluent can never exceed its designed capacity

$F_{aq} \leq F_{aq}$ \hfill [7.30]

- The quality of effluent, measured by salinity concentration, can never be better than its designed quality

$I_{aq} \geq I_{aq}$ \hfill [7.31]

- The quality of effluent, measured by salinity concentration, can never be worse than that of the influent

$E_{aq} \leq I_{aq}$ \hfill [7.32]

(c) The discharge permits are set to control the salt load discharge ($d_s$ and $D_{mf}$ for mines; $d_p$ and $D_{gf}$ for power stations), rather than the volume of discharge ($w_q$ for mines; $F_{aq}$ for power stations), into the River.
Chapter 7 Examination of the Economic Effectiveness of the PHRSTS

Let

\( s_i, s_j \)  
Mine \( i \) or power station \( j \)'s daily permit for salt load discharge under the system of static discharge permits. It is fixed throughout the time (tonnes).

The industry discharge permit for any day is

\[ S = \sum_{i=1}^{m} s_i + \sum_{j=1}^{n} s_j \]  \[ (7.33) \]

\( s_{it}, s_{jt} \)  
Mine \( i \) or power station \( j \)'s discharge permit on day \( t \) (tonnes) under the system of dynamic discharge permits. Under the PHRSTS, no discharge is permitted in a Low flow period. In a High flow period, the permits vary from day to day, depending on the assimilative capacity of the river, which is determined by river flow and in-river salinity concentration. No limit is set on salt discharge during a Flood flow period (we assume the limits on volumetric discharge noted in Section 5.1.3 are never binding). Those are,

\[ s_{it} = 0 \text{ and } s_{jt} = 0 \]  
Low flow

\[ 0 < s_{it} < \infty \text{ and } 0 < s_{jt} < \infty \]  
High flow

\[ s_{it} = \infty \text{ and } s_{jt} = \infty \]  
Flood flow

The industry discharge permit on day \( t \) is

\[ S_t = \sum_{i=1}^{m} s_{it} + \sum_{j=1}^{n} s_{jt} \]  \[ (7.34) \]

Constraints under the four different discharge permit schemes defined in Section 7.1.1 are shown in Table 7.1. Thus, a mine's actual daily discharge of salt load \( d_{it} \) (tonnes) and the actual aggregated daily discharge of salt load from the whole mining industry \( D_{mi} \) (tonnes) are translated into the volume of daily discharge of saline water \( w_{it} \) (ML) so as to make it clearer that the salinity discharge limits translate into constraints on the mines' choice variables \( \{w_{it}\} \).
### Table 7.1 Constraints under the four different schemes

<table>
<thead>
<tr>
<th>Scheme</th>
<th>Static</th>
<th>Dynamic</th>
</tr>
</thead>
<tbody>
<tr>
<td>Non-tradable</td>
<td>$(SN)$</td>
<td>$(DN)$</td>
</tr>
<tr>
<td>$d_{it} \leq s_i \forall i, t$</td>
<td>$d_{it} \leq s_i \forall i, t$</td>
<td></td>
</tr>
<tr>
<td>$d_{jt} \leq s_j \forall j, t$</td>
<td>$d_{jt} \leq s_j \forall j, t$</td>
<td></td>
</tr>
<tr>
<td>Tradable</td>
<td>$(ST)$</td>
<td>$(DT)$</td>
</tr>
<tr>
<td>$D_t = \frac{\sum_{i=1}^{m} w_i c_i + \sum_{j=1}^{n} E_{mt} F_{mt}}{S_t} \leq S_t$</td>
<td>$D_t = \frac{\sum_{i=1}^{m} w_i c_i + \sum_{j=1}^{n} E_{mt} F_{mt}}{S_t} \leq S_t$</td>
<td></td>
</tr>
</tbody>
</table>

Constraint [7.35] is set to ensure that the salt load discharge of each firm ($d_{it}$ or $d_{jt}$) does not exceed its corresponding static permit ($s_i$ or $s_j$), which is defined based on the worst river conditions (that is, lowest level of river flow) and remains constant throughout the years. The purpose of Constraint [7.36] is basically the same as that of Constraint [7.35], except that the permits for a specific firm ($s_i$ or $s_j$) may vary with the daily river flow and salinity conditions.

Constraints [7.37] and [7.38] are set to make sure that the aggregated discharge from the firms ($D_t$) does not exceed the total discharge permits ($S$ or $S_t$) on any day, although the permits under the static and dynamic permit systems are different. The relaxation of the constraint that occurs moving from [7.35] and [7.36] to [7.37] and [7.38], respectively, reflects the tradability of permits.

(2) Assuming that the regulator knows how firms collectively minimise their costs as in (1) above, for each scheme it will minimise $TC$ by choosing

- which scheme (SN, DN, ST, DT) to use

- within each scheme, a full schedule of discharge permits, respectively $\{s_i\}, \{s_h\}, \{s_j\}, \{s_f\} \{S\}$ or $\{S_t\}$

- subject to the constraint that salinity in the river never exceeds 900 EC at any time or place.

215
(3) Assuming cost-minimising actions both by all firms and by the regulator as in (1) and (2) above, the cost-saving potential of the PHRSTS is then either \( TC_{DT} - TC_{SN} \), \( TC_{DT} - TC_{DN} \) or \( TC_{DT} - TC_{ST} \), depending on which control scheme is considered to be the alternative.

This purpose-built theoretical model is conceptually comprehensive and appealing. However, there are obvious difficulties in using it empirically. The full range of required data about each PHRSTS participants is not available, and where some data exist, those data are generally limited in scope. The parameters required for exercising the theoretical model and the data available for this study are compared in Figure 7.1 below and described fully in Appendix 9. Because of problem of the lack of data, this study has developed an alternative operational model to estimate the cost effectiveness of the PHRSTS over the TD using the empirical data from the PHRSTS operation.
Figure 7.1 Data availability of the theoretical model for the coal mines

Grey box means the data is available
White box means the data is not available
The dotted rectangle shows the data needed to determine the discharge requirement for an individual mine.
Chapter 7 Examination of the Economic Effectiveness of the PHRSTS
7.2 An operational model for estimating the cost saving of the PHRSTS over the TD

This section develops an alternative, practical model that uses data available from the actual operation of the PHRSTS from 1995 to 2002 to evaluate empirically its economic performance. This so-called operational model is comparatively simple and straightforward. Its underlying philosophy is calculation of (1) the total amount of saline water storage capacity avoided for the coal mines, and (2) the total amount of saline water treatment avoided for the power stations, under the Pilot Hunter River Salinity Trading Scheme (PHRSTS) in relation to the Trickle Discharge (TD). This will allow us to estimate the cost saving of the PHRSTS over the TD without the need of knowing the data on industrial salt discharges under the TD, which are unavailable. Using the operational model, the cost savings of the PHRSTS over the TD derived from the dynamicism and from the tradability of discharge permits can be estimated separately. The cost savings to the initial PHRSTS participants and the two new developments (those are, Bengalla Coal Mine and Redbank Power Station) can also be estimated separately.

7.2.1 Cost savings of the PHRSTS for the coal mines

7.2.1.1 Discharge behaviour of coal mines

The NSW EPA regulates mine water discharges based on the mines’ discharge of salt load. What the mines are really concerned about, however, is the volume of discharge, rather than or in addition to salt load. Therefore, mines have to account for both water quality and water quantity in their on-site mine water management. Figure 3.7 in Section 3.4.3 presents a comprehensive picture of the mine water generated and excess water, showing the implications of different control systems for on-site storage capacity of the coal mines.

In response to different regulatory schemes, mines’ decisions on capital investment are particularly important, since the costs associated with capital works usually constitute a relatively high proportion of the costs of excess water management. If a mine chooses to store excess water on-site without any treatment, as is the case for most coal mines in the Hunter Valley, determination of storage capacity is a crucial and challenging issue confronting any decision-makers who are sensitive to economic efficiency and averse to the risks of non-compliance with the regulations.
Under a static non-tradable permits (SN) control system daily permits are specified in advance and do not change with the river conditions. There is a relatively high degree of certainty that firms will have to undertake cost-efficient investment in management of excess water. In designing their storage capacity, firms only need to quantify the volumes of excess water that must be withheld on-site. This can be carried out through a site-specific mine water balance model.

To operate this model, a long-term local catchment rainfall-runoff model is often used to calculate daily runoff originating at the mine site from historical daily rainfall data. The mine water generated on-site on any day is the difference between the sum of runoff originating from rainfall and groundwater inflow, and the sum of evaporation and net mine water use on the site on that day (see Appendix 9). Daily runoff originating at the mine site and the daily volume of net mine water use are the main components of the mine water balance model. The daily volume of net mine water use may be relatively easy to quantify based on the operational experience.

However, the volume of mine water generated is hard to predict with a high level of confidence. This is because daily on-site runoff largely depends on local rainfall conditions. Even with a sophisticated rainfall-runoff model in hand, the inherent natures of rainfall (such as, discontinuous, with random occurrence, duration and intensity) still make the precise prediction of the volume of runoff difficult.

In this situation, mines often design their dam capacity based on the criterion that it should be sufficient to accommodate the duration and intensity of a rainfall event with a certain recurrence interval, accounting for the fixed daily allowable discharge permits. Given that the operational lifetime for most coal mines in the Hunter Valley is designed to be 21 years, the 20-year Average Recurrence Interval (ARI) is the prevailing reasonable criterion used in the designation of dam capacity. Firms may choose a lower ARI (for example, 10-year) to lessen the capital costs of dam construction (Ascervatham 1997). However, as mines bear all the risks associated with non-compliance with river quality regulations, they have to trade off the costs of additional storage capacity against the consequence of non-compliance.

Under a dynamic permits scheme (either dynamic non-tradable or dynamic tradable permits), daily discharge permits vary with river conditions: more discharge is allowed when the river flow is higher, and less discharge or even no discharge is allowed when the flow is lower. In
this situation, both the NSW EPA and the mines face some uncertainties and would bear some risks, though different in nature. For the NSW EPA, it is their responsibility to detect the window of discharge opportunities. Moreover, they have to determine the total salt load discharge permits issued to the firms so that the river salinity standards at reference gauging stations are not exceeded. In other words, the NSW EPA has to translate the ambient river salinity standards into the discharge limits defined in tonnes of salt with some degree of accuracy.

However, given the nature of highly variable flow and salinity background concentration in the Hunter River, this may not be an easy task. As already shown in Figure 6.25, although the average daily river salinity is inversely related to the daily flow in the Hunter River, at lower river flows the average salinity is more variable than that experienced at higher flows. It therefore seems impossible to identify the allowable discharge of salt load with a high level of confidence in the Low flow period. This may partly explain why the dynamic permits of the PHRSTS were designed using a trigger value approach mentioned in Section 2.2.4. By defining flow-variable permits based on both ranges of flows (that is, Low, High and Flood flow periods) and individual instantaneous flows with a certain range of flows (that is, daily discharge permits in High flow period), the trigger value approach provides a middle ground between conditional permits and periodic permits.

Under a typical dynamic non-tradable permits (DN) control system, mines have to answer a set of crucial questions in designing their storage capacity, which include, but may not be limited to

- How often do the windows of discharge opportunities (that is, High flows and Flood flows defined by the EPA) occur? In other words, what would be the intervals of discharge opportunities?

- What variability in the allowable total salt load can be expected at a given location (for example, Singleton gauging station)?

- What variability would exist in the length (days) of these opportunities?

- If so, what dam capacity would be required to take sufficient advantage of High flow conditions when discharges are allowed?

- What would be the variability of salinity concentration in the mine water?
Would local extreme rainfall events (such as, 10-year ARI or 20-year ARI) be likely to occur during a period of Low flow in the Hunter River defined by the EPA, and thus fail to comply with the regulations? (A detailed discussion about such a situation will be provided below).

These questions require considerable information and analysis, though working out the answers is not completely impossible. To address these questions, in addition to using an on-site mine water balance model, mines have to model the river flow in response to rainfall. Linking a mine water balance model to the rainfall-flow model superimposes the activities of mines onto the hydrological cycles (Aseervatham 1997). A comparison of the mine water generated in mines and flows in the Hunter River at the relevant gauging stations can be carried out to identify the incidence of potential discharges during Low, High and Flood flow periods respectively and in turn the storage requirement of the mines.

A long-term local rainfall-runoff analysis, combined with a mine’s individual water balance model and compared to the corresponding Hunter River flows is thus essential for managing risk as it enables the average risk exposure of the mine to be quantified. In addition to managing the average risk, mines also have to be able to manage the risk caused by spatial variability in the local rainfall and flow periods within the river. When some storms occur in the immediate locality of a mine site, the opportunity to release may not exist, because the larger Hunter River catchment has not been exposed to a similar storm and the river flow is still at a low level (J. Pola [Water manager, Coal & Allied Operation] 2003, pers. comm., 8 July). Mines are therefore required to maintain some short-term absorption capability on-site for such a contingency. A filtered analysis of site-specific rainfall against, respectively, Low, High and Flood flow events in the river is needed to determine the size of the threshold rainfall event above which the likelihood of a non-correlated flow event in the Hunter River decreases (J. Pola [Water manager, Coal & Allied Operation] 2003, pers. comm., 8 July). The identification of these threshold rainfall values could result in the calculation of desired minimum buffer volumes to be maintained in the storage dams of a mine.

If the answers to the above questions cannot be expressed in quantitative terms with a high level of certainty, mines may be exposed to the risk of either non-compliance with regulations or adverse impacts on mine operations (as they may have to store excess water in the existing mining pits). The risks associated with these uncertainties are alleviated to some extent by the tradability of PHRSTS salt credits, as this tradability allows a mine to offset its short-term risks
through buying extra discharge permits from those mines in low-risk positions with surplus permits during a given period.

Under the PHRSTS, however, tradable salt credits only work during the High flow scenario when discharges are allowed. There is no latitude for offsetting short-term risks of discharge during Low flow periods as no discharge is allowed for any firms at those times. Conversely, there is no requirement to offset any risks when a Flood flow presents because there are no discharge constraints. Thus, it should be recognised that while tradability of salt credits can reduce some mines’ storage requirements for the periods of High river flow, it will not remove the requirement for a certain minimum level of storage for withholding excess water on-site for the periods of Low river flow. As such, the use of PHRSTS could alleviate the risks to the mines, but is in no way a substitute for a risk-based water management strategy.

### 7.2.1.2 A numerical example

A numerical example is given in Table 7.2 to explain the method of estimating the storage capacity that a coal mine avoided under the PHRSTS, compared to the Trickle Discharge (TD).

#### Table 7.2 Illustration of avoided withholdings for Mine i under the PHRSTS

(days of AWEs for Mine i, defined below, are marked in *bold italic.*)

<table>
<thead>
<tr>
<th>Day (t)</th>
<th>Flow periods</th>
<th>TD Permit (s₁)</th>
<th>PHRSTS Permit (s₂)</th>
<th>PHRSTS Discharge (d₁)</th>
<th>Avoided discharge with permit trading (Δd₁)</th>
<th>Avoided discharge without permit trading (Δd′₁)</th>
</tr>
</thead>
<tbody>
<tr>
<td>...</td>
<td>Low</td>
<td>10</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>x</td>
<td>Low</td>
<td>10</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>x + 1 = M₁(0)</td>
<td>High</td>
<td>10</td>
<td>15</td>
<td>5</td>
<td>0 (5&lt;10)</td>
<td>0</td>
</tr>
<tr>
<td>x + 2</td>
<td>Flood</td>
<td>10</td>
<td>∞</td>
<td>45</td>
<td>35 (=45-10)</td>
<td>35 (=45-10)</td>
</tr>
<tr>
<td>x + 3</td>
<td>Flood</td>
<td>10</td>
<td>∞</td>
<td>60</td>
<td>50 (=60-10)</td>
<td>50 (=60-10)</td>
</tr>
<tr>
<td>x + 4</td>
<td>High</td>
<td>10</td>
<td>40</td>
<td>50</td>
<td>40 (=50-10)</td>
<td>30 (=40-10)</td>
</tr>
<tr>
<td>x + 5 = M₂(0)</td>
<td>High</td>
<td>10</td>
<td>20</td>
<td>35</td>
<td>25 (=35-10)</td>
<td>10 (=20-10)</td>
</tr>
<tr>
<td>...</td>
<td>Low</td>
<td>10</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>y + 1 = M₃(0)</td>
<td>High</td>
<td>10</td>
<td>5</td>
<td>25</td>
<td>15 (=25-10)</td>
<td>0 (5&lt;10)</td>
</tr>
<tr>
<td>y + 2 = M₄(0)</td>
<td>High</td>
<td>10</td>
<td>45</td>
<td>30</td>
<td>20 (30-10)</td>
<td>20 (30-10)</td>
</tr>
<tr>
<td>...</td>
<td>Low</td>
<td>10</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>

223
Chapter 7 Examination of the Economic Effectiveness of the PHRSTS

The notations and definitions of the variables in the Table follow those in earlier sections. Those are

\( s_i \) \hspace{1cm} Mine \( i \)'s daily permitted discharge of salt load under the TD, which remains unchanged throughout days (tonnes/day).

\( s_u \) \hspace{1cm} Mine \( i \)'s permitted discharge of salt load on day \( t \) under the PHRSTS. The definition of Flood, High and Low flow days and the corresponding discharge rules are as introduced in Section 5.1.1. That is,

\[
\begin{align*}
    s_u &= 0 \quad \text{Low flow} \\
    0 < s_u < \infty &\quad \text{High flow} \\
    s_u &= \infty \quad \text{Flood flow}
\end{align*}
\]

\( d_u \) \hspace{1cm} Mine \( i \)'s actual discharge of salt load on day \( t \) under the PHRSTS (tonnes/day).

\( \Delta d_u \) \hspace{1cm} Mine \( i \)'s avoided withholding of salt load on day \( t \) under the PHRSTS with permits trading (tonnes/day).

\( \Delta d'_u \) \hspace{1cm} Mine \( i \)'s avoided withholding of salt load on day \( t \) under the PHRSTS without permits trading (tonnes/day).

To calculate the avoided storage capacity, we need first to define:

(a) an Avoided Withholding Event (AWE) for mine \( i \) is any set of consecutive days \( t, t+1, \ldots \) on which its discharge is allowed; and

(b) let mine \( i \) experience \( z(i) \) AWEs during the entire PHRSTS period from \( t=1, \ldots, T \) (\( T=2891 \)); let these AWEs be indexed by \( k = 1, \ldots, z(i) \); and let the first and last days of the \( k \)th AWE be respectively \( M_{k(i)} \) and \( N_{k(i)} \) (the values of all the \( z(i), M_{k(i)} \) and \( N_{k(i)} \) are found by processing the empirical data sets).

The dynamicism and tradability of discharge permits, which lie at the heart of the PHRSTS and form the major difference between the PHRSTS and the TD, allow the mine to make an excess discharge during High and Flood flow days. Our admittedly rough estimate of the cost advantage of the PHRSTS over the TD is based on:
(a) the *observation* that during the period of 2891 days from 1/1/1995-30/11/2002 inclusive for which the PHRSTS was implemented, which is indexed by $t = 1, \ldots, T$ where $T = 2891$, no enlargement of mine water storage dams took place to accommodate excess discharge, in particular during the Low flow days on which no discharge was made; and:

(b) the *assumption* that during this same period, any seller of permits – which must be discharging less than its permits, and therefore *withholding* (not discharging) more saline water than it needs to – is just using up spare dam capacity, and does not incur any cost of dam enlargement as a result; and lastly

(c) the *assumption* that any buyer of permits would otherwise have had to *store* any excess of discharge over permits, $d_0s_0 + d_{(0)}s_{(0)}d_{(1)} + \ldots$ which is *accumulated* over successive days $t, t+1, \ldots$, and so gains a benefit from the PHRSTS of avoiding the cost of dam enlargement.

Assumption (b) is based directly on observation (a), but assumption (c) is more speculative and can only be approximate at best. One reason for this is that in general, and contrary to our assumption, the regulator would not choose the same optimal set of firms’ permit levels under the PHRSTS as under the TD, because the firms’ responses to permits differ between the schemes. Another reason is that assumption (b) excludes what we call *precautionary excess discharge*. This happens whenever a mine chooses to make an excess discharge even when it still has spare dam capacity, because it suspects that reducing storage volume now will avoid the need to buy permits on future days when they will be more expensive. However, a mine with a well developed water management strategy and a sound understanding of their water balance, hydrological processes and risk will realise the economical benefits of storing water for use on-site and reducing water withdrawal (at a price) from the Hunter River, as noted in Section 3.4.3. This leads to a well-defined balance between retention and discharge, backed by the appropriate level of analysis and rigour. Our assumptions allow the cost saving of the PHRSTS (compared to TD) from the Hunter mining industry to be approximated by the sum of avoided storage costs of all mines during the PHRSTS period, as follows.

In Table 7.2, suppose the mine has two AWEs over the entire PHRSTS period. AWE$_1$ runs from $t = x+1 = M_{(0)}$ to $t = x+5 = N_{(0)}$, comprising 1 High flow days, followed by 2 Flood flow days, and lastly another 2 High flow days (typically followed by a set of long consecutive Low flow
days). AWE₂ runs from \( t = y+1 = M_{2(0)} \) to \( t = y+2 = N_{2(0)} \), comprising only 2 High flow days. The next calculations are comprised of three parts.

\[ \text{(1) Avoided storage capacity with permit trading} \]

For both Avoided Withholding Events (AWEs), any actual daily discharge \((d_{it})\) that is more than its TD permit \((s_i)\) on a particular High or Flood flow day is the discharge that would otherwise have to be stored on site. In other words, on any day of an AWE, as long as \( d_{it} > s_i \), the mine avoids withholding of an extra volume of mine water equivalent to the salt load of \((d_{it} - s_i)\) on that day as a result of dynamicism and tradability of the permits of the PHRSTS.

For AWE₁, on day \( x+1 \) the mine does not avoid any withholding of discharge because \( d_{it} = 5 \) \(< (s_i=10) \). From day \( x+2 \) to \( x+5 \), the avoided withholding amounts are \((45-10=35)\), \((60-10=50)\), \((50-10=40)\) and \((35-10=25)\) in order. Thus, the accumulated withholding of salt that the mine avoids during its AWE₁ is \((45-10)+(60-10)+(50-10)+(35-10)=150\) tonnes. For AWE₂, the accumulated avoided withholding of salt is \((25-10)+(30-10)=35\) tonnes. The overall withholding of salt that the mine avoids as a result of dynamicism and tradability of the permits during the entire PHRSTS period is the \(\text{sum} \) of avoided withholding over these two AWEs, that is, \(150+35=185\) tonnes.

Assuming that the salinity concentrations of mine water for both AWEs are approximately the same, the storage capacity that the mine avoids building during the entire PHRSTS period is the volume equivalent to \(\text{maximum, cumulative, avoided saline water} \) over these two AWEs; that is, the volume equivalent to the salt of 150 tonnes.

\[ \text{(2) Avoided storage capacity without permit trading} \]

In the above analysis, the tradability of discharge permits allows the mine to make discharge over its PHRSTS permits on some particular High flow days (that is, \( d_{it} > s_{it} \) on days \( x+4, x+5 \) and \( y+1 \) for the mine) through buying salt credits from other firms. Of course, some other firms must discharge less than their permits \((d_{it} < s_{it} \) for other firms) so that the industry-wide Constraint [7.38] in Section 7.1.4 is obeyed. If the permits are not tradable, Mine \( i \) would have to only discharge up to its PHRSTS permit \( s_{it} \). In other words, the mine would have to withhold more saline water on site (that is, less avoided withholding). The mine’s avoided storage capacity in the scenario without permit trading is thus calculated as follows.

226
Referring again to Table 7.2, during AWE₁, from day $x+1$ to $x+3$, the mine's daily avoided withholdings are still 0, $(45-10 = 35)$ and $(60-10 = 50)$ – this is exactly the same as Calculation (a) where the permits are tradable. However, on day $x+4$, the avoided withholding of salt load is now $(40-10 = 30)$ instead of $(50-10 = 40)$, because the mine is not allowed to make a discharge more than its PHRSTS permit (that is, $s_\text{it}=40$). For the same reason, on day $x+5$, the avoided withholding of salt load is $(20-10 = 10)$ instead of $(35-10 = 25)$. Thus, the accumulated withholding of salt that the mine avoids during its AWE₁ is $(45-10) + (60-10) + (40-10) + (20-10) = 125$ tonnes.

During AWE₂, on day $y+1$, the avoided withholding is 0; this is because given the permits are not tradable, the mine is not allowed to make a discharge more than its PHRSTS permit (that is, $s_\text{it}=5$). As the PHRSTS permit is less than the TD permit (that is, $(s_\text{it}=5) < (s_\text{t}=10)$), the mine does not avoid any withholding on this day. On day $y+2$, the avoided withholding is $(30-10 = 20)$ – although the actual discharge does not exceed its PHRSTS permit (that is, $(d_\text{it}=30) < (s_\text{it}=45)$), but the actual discharge is more than its TD permit (that is, $(d_\text{it}=30) > (s_\text{t}=10)$). So, the avoided withholding is $d_\text{it}$ minus $s_\text{t}$, that is, $(30-10 = 20)$. So, the accumulated withholding of salt that the mine avoids during its AWE₂ is $0 + (30-10) = 20$ tonnes.

Therefore, without permit trading, the total salt load that mine $i$ avoids withholding during the entire PHRSTS period would have been $125+20=145$ tonnes. But what matters for saving costs is the storage capacity that the mine avoids building during the entire PHRSTS period. That is the volume equivalent to the maximum, cumulative, avoided saline water over both AWEs for these two, that is, the volume equivalent of 125 tonnes of salt.

(3) Avoided storage capacity due to permit trading

From the above calculations (a) and (b), the withholding of salt during the entire PHRSTS period that the mine avoids due to permit trading is $20 (=185-165)$ tonnes. The storage capacity that the mine avoids building during the same period is a volume of saline water equivalent to the salt of $25 (=150-125)$ tonnes.
### 7.2.1.3 Method for calculation of avoided storage capacity

The above analysis of the numerical example can be generalised to the following methodology applicable to all PHRSTS coal mines for calculating their avoided storage capacities under the Pilot Hunter River Salinity Trading Scheme (PHRSTS) over the Trickle Discharge (TD).

**1. Avoided storage capacity with permit trading**

The salt load that firm $i$ avoids withholding on any day $t$ of its AWE (denoted $\Delta d_i$) due to both the dynamicism and the tradability of permits under the PHRSTS is

$$\Delta d_i = 0 \quad \text{(when } d_i \leq s_i) \text{ or } \Delta d_i = d_i - s_i \quad \text{(when } d_i > s_i)$$  \[7.39\]

The accumulated withholding of salt that firm $i$ avoids during its Avoided Withholding Event (AWE) $k$ (denoted $\Delta D_i$) is then

$$\Delta D_i = \sum_{t=M_{k(i)}}^{N_{k(i)}} \Delta d_i$$  \[7.40\]

The corresponding accumulated withholding of mine water with an average salinity concentration $c_i$ (denoted $\Delta V_i$) is

$$\Delta V_i = \sum_{t=M_{k(i)}}^{N_{k(i)}} \left( \frac{\Delta d_i}{fc_i} \right)$$  \[7.41\]

So the storage capacity which $i$ avoids building due to the tradability of permits during the entire PHRSTS period is the volume equivalent to maximum, cumulative, avoided saline water over its $z(i)$ AWEs:

$$\Delta A_i := \max_{k=1,...,z(i)} \Delta V_i = \frac{1}{fc_i} \max_{k=1,...,z(i)} \left( \sum_{t=M_{k(i)}}^{N_{k(i)}} \Delta d_i \right)$$  \[7.42\]

The total storage capacity that the mining industry (consisting of $n$ mines) avoid under the PHRSTS compared to the TD is thus:

$$\Delta A := \sum_{i=1}^{n} \left( \frac{1}{fc_i} \right) \max_{k=1,...,z(i)} \left( \sum_{t=M_{k(i)}}^{N_{k(i)}} \Delta d_i \right)$$  \[7.43\]
(2) *Avoided storage capacity without permit trading*

The salt that mine \( i \) avoids withholding on any day \( t \) of its AWE (denoted \( \Delta d''_u \)) due to the dynamicism but no tradability of permits under the PHRSTS is

When \( d_u \leq s_i \), \( \Delta d''_u = 0 \)

When \( d_u > s_i \)

If on Flood flow days, then \( \Delta d''_u = d_u - s_i \)

If on Flood flow days, then

\[
\Delta d''_u = d_u - s_i \quad \text{(when } s_i < d_u < s_d) \\
\Delta d''_u = s_u - s_i \quad \text{(when } s_i < s_u < d_u) \\
\Delta d''_u = 0 \quad \text{(when } s_u < s_i < d_u) \\
\]

The accumulated withholding of salt load that firm \( i \) avoids during its AWE \( k \) (denoted \( \Delta D''_u \)) is then

\[
\Delta D''_u = \sum_{t=M_{AWE}(k)}^{N_{AWE}(k)} \Delta d''_u \\
[7.45]
\]

The corresponding accumulated withholding of mine water with an average salinity concentration \( c_i \) (denoted \( \Delta V''_u \)) is

\[
\Delta V''_u = \sum_{t=M_{AWE}(k)}^{N_{AWE}(k)} \left( \frac{\Delta d''_u}{f c_i} \right) \\
[7.46]
\]

So the storage capacity which \( i \) avoids building due to the dynamicism but no tradability of permits during the entire PHRSTS period is the volume equivalent to maximum, cumulative, avoided saline water over its \( z(i) \) AWEs:

\[
\Delta A''_i = \text{Max}_{k=1 \ldots z(i)} \Delta V''_u = (1/f c_i) \text{Max}_{k=1 \ldots z(i)} \sum_{t=M_{AWE}(k)}^{N_{AWE}(k)} \Delta d''_u \\
[7.47]
\]

The total storage capacity that the mines avoid under the PHRSTS compared to the TD is thus:

\[
\Delta A'' := \sum_{i=1}^{n} (1/f c_i) \text{Max}_{k=1 \ldots z(i)} \sum_{t=M_{AWE}(k)}^{N_{AWE}(k)} \Delta d''_u \\
[7.48]
\]
(3) Avoided storage capacity **due to permit trading**

So the storage capacity which $i$ avoids building due to permit trading during the entire PHRSTS period, denoted $\Delta A''$, is the difference between $\Delta A$ and $\Delta A'$, that is, $\Delta A'' = \Delta (A_i - A')$. Accordingly, the total storage capacity that the mining industry avoids due to permit trading under the PHRSTS compared to the TD, denoted $\Delta A''$, is the difference between $\Delta A$ and $\Delta A'$, that is,

$$\Delta A'' = \Delta A - \Delta A'$$  \hspace{1cm} [7.49]

### 7.2.1.4 Data and results

The data for daily total allowable discharge (TAD) of salt load (denoted $TAD_i$) during the PHRSTS period was obtained from the HRSTS Annual Report 1995-2002 (NSW DLWC 1995; 1996; 1997; 1998; 1999; 2000a; 2001a; 2002). In conjunction with the number of salt credits that the mines were initially allocated (denoted $\hat{\delta}_i$), the daily permitted discharge of salt load for individual mines on any day ($s_i$) can be calculated by

$$s_i = \hat{\delta}_i \times \frac{TAD_i}{1000}$$  \hspace{1cm} [7.50]

The daily salt load that an individual mine actually discharged ($d_i$) was also obtained from the HRSTS Annual Report 1995-2002. The discharge salinity concentration for an individual mine ($c_i$) is the average of the available discharge concentrations recorded in its Annual Returns.

The data for maximum daily release rates for individual mines that were permitted under the TD (denoted $v_i$) were obtained from the NSW EPA report (1995). If we assume that the salinity concentration of discharge for each individual mine ($c_i$) remains the same under both the PHRSTS and the TD systems, the daily permitted discharge of salt load under the TD (that is, $s_i$) was calculated by $s_i = f_c v_i$.

Recall that Bengalla Coal Mine is one of the two new developments that occurred under the PHRSTS. In the discussion in Section 4.2, it was stated that because of the increasing river salinity under TD system, the NSW EPA had to move in 1992 to a position of allowing no further salt discharge to the Hunter River. If the TD had remained in place and the Nil Discharge policy had applied, no discharge from Bengalla Coal Mine would have occurred. In
the calculation, \( s_i \) is treated as zero for Bengalla Coal Mine (and Redbank Power Station, see Section 7.2.3) to reflect this Nil Discharge situation.

The avoided withholdings of salt load and mine water, as well as the avoided storage capacity of the coal mines under the PHRSTS with and without permit trading, have been calculated based on the above-stated method and data. The break-down of results by the mines are set out in Table 7.3 below. The existing storage capacity of each mine before the inception of the PHRSTS, which is extracted from the NSW EPA report (1995), is also listed in the Table for comparison purposes.

Table 7.3  Mines' avoided withholdings and storage capacities under the PHRSTS

<table>
<thead>
<tr>
<th>Mines</th>
<th>With permits trading</th>
<th>Without permits trading</th>
<th>Due to permits trading</th>
<th>( A_{TD} ) (ML)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>( \Delta D_i ) (tonnes)</td>
<td>( \Delta V_i ) (ML)</td>
<td>( \Delta A_i ) (ML)</td>
<td>( \Delta D_i' ) (tonnes)</td>
</tr>
<tr>
<td>Bengalla</td>
<td>588</td>
<td>724</td>
<td>276</td>
<td>350</td>
</tr>
<tr>
<td>Dartbrook</td>
<td>191</td>
<td>37</td>
<td>9</td>
<td>191</td>
</tr>
<tr>
<td>Howick</td>
<td>2702</td>
<td>1560</td>
<td>254</td>
<td>1831</td>
</tr>
<tr>
<td>Hunter Valley</td>
<td>356</td>
<td>122</td>
<td>32</td>
<td>291</td>
</tr>
<tr>
<td>Lemington</td>
<td>191</td>
<td>107</td>
<td>76</td>
<td>191</td>
</tr>
<tr>
<td>Liddell</td>
<td>5282</td>
<td>1863</td>
<td>1505</td>
<td>5247</td>
</tr>
<tr>
<td>Muswellbrook</td>
<td>386</td>
<td>178</td>
<td>88</td>
<td>307</td>
</tr>
<tr>
<td>Mt Owen</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Ravensworth</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Saxonvale/Bulga</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Warnbo</td>
<td>3102</td>
<td>768</td>
<td>383</td>
<td>3102</td>
</tr>
<tr>
<td>Workworth</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Sum</td>
<td>12798</td>
<td>5359</td>
<td>2621</td>
<td>11510</td>
</tr>
<tr>
<td>Percentage (%)</td>
<td>19</td>
<td>10</td>
<td>16</td>
<td>2</td>
</tr>
</tbody>
</table>

\( \Delta D_i, \Delta D_i', \Delta D_i'' \) = the accumulated withholding of salt that Mine \( i \) avoids with permit trading, without permit trading and due to permit trading respectively.

\( \Delta V_i, \Delta V_i', \Delta V_i'' \) = the accumulated withholding of mine water that Mine \( i \) avoids with permit trading, without permit trading and due to permit trading respectively.

\( \Delta A_i, \Delta A_i', \Delta A_i'' \) = the storage capacity that Mine \( i \) avoids with permit trading, without permit trading and due to permit trading respectively.

\( A_{TD} \) = the existing storage capacity of the mines before the inception of the PHRSTS.
From Table 7.3, the avoided storage capacities of the mines vary across a wide range. According to the model estimates, the coal mines avoided withholding a total 12798 tonnes salt equivalent to a volume of 5359 ML mine water during the entire PHRSTS period that they would otherwise have had to store on site under the previous TD control system, due to both the dynamicism and the tradability of the permits under the PHRSTS. This leads the mines to save an overall 2621 ML storage capacity, accounting for 19% (= 2621 / (2621+11211)) of the total storage capacity that would otherwise be required by the mines under the previous TD.

If the permits had not been traded under the PHRSTS, the coal mines would have avoided a withholding of 11510 tonnes salt equivalent to 4492 ML mine water during the same period. The mines would have saved a slightly lower capacity of 2577 ML. Accordingly, permit trading under the PHRSTS made the mines avoid withholding 1289 tonnes of salt, equivalent to 867 ML of mine water and saving 44 ML of storage capacity. Permit trading contributed 10% (= 1289/12798) to the total withholding of salt load and 16% (= 867/5359) of the total withholding of mine water that the mines avoided during the entire PHRSTS period (see Table 7.3).

However, permit trading only contributed 2% (= 44/2621) of the total avoided storage capacity. This is because for most of the mines, the maximum cumulative avoided saline water over its AWE actually occurred on Flood flow days, rather than on High flow days. As each of these mines’ avoided storage capacity is ultimately determined by its discharges on these Flood flow days, their excess discharges (through permit trading) on the High flow days have little, if any, effect on the size of their avoided storage capacities $\Delta A$. Given that the mines’ cost savings from the PHRSTS virtually come from the avoided costs associated with construction of extra storage capacity, the low contribution of permit trading to the total avoided storage indicates that the cost saving from permit trading under the PHRSTS is minimal, as will be discussed further in the next section.

### 7.2.1.5 Indicative cost saving of the mining industry

The information of the marginal cost of storing saline water for individual mines is crucial to an estimation of the total cost saving of the PHRSTS from the Hunter mining industry. Appendix 10 provides a review of various approaches to estimation of MAC curves. It also contains a detailed description of a survey that I conducted to collect the information about saline water control costs from the PHRSTS participants. There were few responses to the survey, which prohibits this study estimating the cost of storing saline water for the coal mines in a precise...
way. So I have estimated the total cost saving based on empirical information in the following indicative way.

Indicative costs of constructing storage dams in the Hunter Region were provided by the NSW EPA (2001) when assessing the effect of altering Flood flow threshold on the storage capacity required by the PHRSTS mines. The costs associated with various levels of water storage requirements are listed in Table 7.4 below.

### Table 7.4 Water storage requirements and their costs

<table>
<thead>
<tr>
<th>Water storage requirement</th>
<th>Estimated cost ($ Million)</th>
<th>Incremental cost above base ($ Million)</th>
<th>Marginal cost $/ML</th>
</tr>
</thead>
<tbody>
<tr>
<td>230 ML</td>
<td>1.5</td>
<td>0</td>
<td>—</td>
</tr>
<tr>
<td>~ 480 ML</td>
<td>2.8</td>
<td>1.3</td>
<td>5200</td>
</tr>
<tr>
<td>~ 600 ML</td>
<td>3.3</td>
<td>1.8</td>
<td>4865</td>
</tr>
<tr>
<td>970 ML</td>
<td>4.5</td>
<td>3.0</td>
<td>4687</td>
</tr>
</tbody>
</table>

Source: NSW EPA (2001:30)

As for new development, in 2000 Bengalla Coal Mine constructed its staged discharge dam with a capacity of 280 ML for storing excess mine water. This volume was designed sufficient to store water for all but the most extreme wet weather conditions accounting for the tradability of discharge permits (Bengalla Mining Company 2003). Bengalla Coal Mine estimated that the capital cost for its construction was approximately $2 million based on tenders received, which is equivalent to $7143/ML in 2000 prices (Bengalla Mining Company 2003). This average construction cost is converted into $7696/ML in 2002 prices using the price indexes $I_{2000} = 129.2$ and $I_{2002} = 139.2$ released by the Australian Bureau of Statistics (see Table 7.5). Comparing Table 7.5 with Table 7.4, it appears that Bengalla’s cost estimation is not unreasonable.

### Table 7.5 Bengalla’s water storage and its construction cost (used in the estimation)

<table>
<thead>
<tr>
<th>Water storage</th>
<th>Total construction cost</th>
<th>Average marginal cost (in 2000 prices)</th>
<th>Average marginal cost (in 2002 prices)</th>
</tr>
</thead>
<tbody>
<tr>
<td>280 ML</td>
<td>$2 Million</td>
<td>$7143/ML</td>
<td>$7696/ML</td>
</tr>
</tbody>
</table>

However, an empirical formula for estimating the operating costs of storage dams in the Hunter region does not exist. As noted in Section 3.4.3, in addition to having a staged dam for storing
saline water, most Hunter coalmines have some other types of dams, such as a raw water dam, a washery dam and a ramp dam, for the mining operations. Bengalla is no exception. Bengalla budgeted for their water management system, including dams, pipelines, pumps, valves and power, as a whole, but not for individual dams. As a very broad figure, Bengalla estimated that the cost for the ongoing management of its 280 ML staged dam is approximately $10,000-$15,000 per annum (Bengalla Mining Company 2003).

It may not be proper to directly apply the above capital and operating costs of Bengalla Coal Mine to any specific PHRSTS mines. However, in the absence of other more precise information, these costs indicate the likely order of magnitude of costs avoided by the mines under the PHRSTS.

My study assumes that (1) the space required for storage extension for any individual PHRSTS mine is readily available; (2) cost of storage extension for existing mines is proportional to the increase in cost of storage requirements for new mines; and (3) the incremental operating cost due to the storage extension for an individual mine is trivial and therefore can be ignored. Based on the avoided storage capacity presented in Table 7.3, the total avoided cost by the mines for storage extension during the entire PHRSTS period (see Table 7.6) is approximately $20.2 million in 2002 prices.

As in the earlier calculation, the avoided storage capacity of the mines due to permit trading accounts for 2% of the overall avoided storage capacity during the entire PHRSTS period. Accordingly, the cost saving of permits trading for the mining industry during the same period is $0.4 million (= 20.2 x 2%) in 2002 prices.
Chapter 7 Examination of the Economic Effectiveness of the PHRSTS

Table 7.6  Avoided storage cost of the coal mines under the PHRSTS (in 2002 prices)  
(assuming $7696/ML capital cost)

<table>
<thead>
<tr>
<th>Mines</th>
<th>$\Delta A_i$ (ML)</th>
<th>Cost saving ($Million)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bengalla</td>
<td>276</td>
<td>2.1</td>
</tr>
<tr>
<td>Dartbrook</td>
<td>9</td>
<td>0.1</td>
</tr>
<tr>
<td>Howick</td>
<td>254</td>
<td>2.0</td>
</tr>
<tr>
<td>Hunter Valley Operation</td>
<td>32</td>
<td>0.2</td>
</tr>
<tr>
<td>Lemington</td>
<td>76</td>
<td>0.6</td>
</tr>
<tr>
<td>Liddell</td>
<td>1505</td>
<td>11.6</td>
</tr>
<tr>
<td>Muswellbrook</td>
<td>0</td>
<td>0.0</td>
</tr>
<tr>
<td>Mt Owen</td>
<td>88</td>
<td>0.7</td>
</tr>
<tr>
<td>Ravensworth</td>
<td>0</td>
<td>0.0</td>
</tr>
<tr>
<td>Saxonvale/Bulga</td>
<td>0</td>
<td>0.0</td>
</tr>
<tr>
<td>Wambo</td>
<td>383</td>
<td>2.9</td>
</tr>
<tr>
<td>Warkworth</td>
<td>0</td>
<td>0.0</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>2621</strong></td>
<td><strong>20.2</strong></td>
</tr>
</tbody>
</table>

$\Delta A_i = \text{Mine } i \text{'s avoided storage capacity during entire PHRSTS period}$

The above approach to the cost estimation is oversimplified by assuming that the total capital cost of incremental storage for an individual mine is just proportional to the size of storage extension (See $TC_i$ in Figure 7.2). In reality, however, the total capital cost function might well be a stepped line like $TC$ in the Table. Thus, the cost tends to be under-estimated (for example, point A) or over-estimated (for example, point B) by an unknown amount.
Figure 7.2 Assumed and actual total capital cost functions of storage construction

$TC_A$ = assumed total capital cost function of storage construction
$TC$ = actual total capital cost function of storage construction

7.2.2 Cost savings of the PHRSTS for the power stations

The description in Sections 3.4 and 3.5 shows that whilst the volume of mine water is the main concern of the coal mines, the mass of salt is a greater worry for the power stations. Thus, for the power stations it is meaningful to calculate their avoided treatment costs of salt under the Pilot Hunter River Salinity Trading Scheme (PHRSTS) over the Trickle Discharge (TD).

Under the TD, the permitted daily discharge of salt load did not vary in response to the river flow conditions. Under the PHRSTS, however, on Low flow days, the power stations have to treat all salt load produced on these days because no discharge is allowed, which in turn might induce an incremental treatment cost. On High and Flood days, the power stations are allowed to discharge more salt (depending on actual flows), resulting in a reduced treatment cost. Therefore, the net cost saving of the power stations under the PHRSTS over the TD depends upon the reduced cost during High and Flood flows in relation to the incremental cost during Low flows.

My estimation of the salt that the power stations avoided treating under the PHRSTS is based on the following two assumptions. (1) The power stations can withhold the incremental salt load
treatment on Low flow days, incurring no or minimal additional cost. This assumption is valid because, as described in Section 3.5.1.3, in practice Macquarie's two power stations use Lake Liddell as a buffer to withhold their freshwater and wastewater. The large storage capacity (152 GL) of the Lake Liddell makes their withholding of saline water flexible. (2) All power stations treat their daily discharges of saline water in excess of the corresponding permits (by extracting the salt using the existing desalination technology) on any one day. These assumptions allow this study to treat any additional salt load from a power station beyond its daily permit as avoided treatment, and allows the total saving in treatment under the PHRSTS (compared to the TD) from the power generation industry to be approximated by the sum of avoided treatment of individual power stations, as follows.

7.2.2.1 Calculation of avoided saline water treatment

Following the notations set out in Section 7.2.1.3, the salt load that power station \( j \) avoids treating on any one day under the PHRSTS against the TD, denoted \( \Delta d_{jt} \), can be calculated based on either Equation [7.39] or Equation [7.44], depending upon the scenario with or without permit trading.

The volume of saline water that the power station avoids treating on that day is then

\[
\frac{\Delta d_{jt}}{f_{c_j}}
\]  

[7.51]

So the total salt (\( \Delta SR_j \)) and volume of saline water (\( \Delta VR_j \)) which \( j \) avoids treating during the entire PHRSTS period (\( T = 2891 \) days) are the sum of daily salt load and sum of saline water avoided treating over that period as expressed by [7.52] and [7.53] respectively:

\[
\Delta SR_j := \sum_{t=1}^{T} \Delta d_{jt}
\]  

[7.52]

\[
\Delta VR_j := \left(\frac{1}{f_{c_j}}\right) \sum_{t=1}^{T} \Delta d_{jt}
\]  

[7.53]

The overall salt load (\( \Delta SR \)) and volume of saline water (\( \Delta VR \)) that the power stations avoids under the PHRSTS compared to under TD are thus [7.54] and [7.55] respectively:

\[
\Delta SR := \sum_{i=1}^{n} \sum_{t=1}^{T} \Delta d_{jt}
\]  

[7.54]
\[
\Delta VR := \sum_{i=1}^{n} \left(1 / f_{c_j}\right) \sum_{t=1}^{T} \Delta d_{jt}
\]

[7.55]

Using the same data set as in Section 7.2.1.3, the total salt load and volume of saline water that the power stations avoided treating during the entire PHRSTS period with and without permits trading are calculated and presented in Table 7.7. Since Redbank Power Station is another new development under the PHRSTS, we treat it in the same way as Bengalla Coal Mine. So in our calculation, Redbank Power Station’s daily permitted discharge under the TD is treated as zero to reflect the Nil Discharge regulation it would have been subject to under the TD.

<table>
<thead>
<tr>
<th>Table 7.7</th>
<th>Avoided salt load and volume of saline water treatment during PHRSTS period</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Power Stations</strong></td>
<td><strong>With permits trading</strong></td>
</tr>
<tr>
<td></td>
<td>(\Delta SR_j)</td>
</tr>
<tr>
<td>Macquarie Power</td>
<td>27034</td>
</tr>
<tr>
<td>Redbank Power</td>
<td>582</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>27616</strong></td>
</tr>
</tbody>
</table>

\(\Delta SR_j\) = Power Station j’s total salt load avoided treating over the entire PHRSTS period
\(\Delta VR_j\) = Power Station j’s total saline water avoided treating over the entire PHRSTS period

As Table 7.7 shows, the power stations avoided 11811 ML of saline water, which contains 27616 tonnes of salt, being treated during the entire PHRSTS period, thanks to the dynamicism and tradability of permits. But only 5\% (= 586/11811) of the total avoided saline water, equivalent to 6\% (= 1563/27616) of the total avoided salt, is attributable to permit trading. This suggests that most of the cost saving of the power stations from the PHRSTS also derives from the dynamicism of permits.

### 7.2.2.2 Indicative treatment cost avoided by the power generation industry

For the power stations, the cost saving comes from reduced treatment cost due to lower quantity or quality of saline water treated under the PHRSTS than would have been needed under the TD. My study assumes that (a) power stations used the same Reverse Osmosis (RO) desalination technique to treat their saline water under the PHRSTS as under the TD; and (b) the capacity of the existing desalination technique for each individual power station is sufficient to accommodate excess saline water for treatment, and therefore no extra investment (that is,
incremental capital costs) would incur. Borrowing the following empirical information on operating costs of brackish RO desalination plants in the literature, this study has estimated the treatment cost avoided by the power generation industry under the PHRSTS as follows.

Commissioned by the Murray-Darling Basin Commission and the National Land and Water Resources Audit, URS Australia (2002) undertook a study, assessing the technical and financial aspects of desalination as a source of fresh water and a salinity management tool in the National Action Plan for Salinity and Water Quality regions of Australia, including the NSW region. This study examined various desalination techniques and provided a generic estimation of depreciated capital and operational costs for these techniques. Among others, the information on brackish RO operating costs of various feedwater salinity and product flow rate is relevant to this study and is thus presented in Table 7.8. The Table shows that the per ML operating costs of desalination processes is positively related to the feedwater salinity level (that is, influent concentration) and negatively related to product flow rate (that is, volume of effluent). The higher influent salinity level is, the more operating costs would be. Conversely, the lower effluent volume, the more operating costs would be.

Table 7.8  
RO operating costs by feedwater salinity and product flow rate (in 1999 price)

<table>
<thead>
<tr>
<th>Feed salinity in TDS</th>
<th>Product water flow rate</th>
<th>5 KL/day ($/ML)</th>
<th>15 KL/day ($/ML)</th>
<th>50 KL/day ($/ML)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000 mg/L (3333 EC)</td>
<td>1000</td>
<td>900</td>
<td>650</td>
<td></td>
</tr>
<tr>
<td>10000 mg/L (16667 EC)</td>
<td>1500</td>
<td>1300</td>
<td>930</td>
<td></td>
</tr>
<tr>
<td>35000 mg/L (55000 EC)</td>
<td>2200</td>
<td>2000</td>
<td>1890</td>
<td></td>
</tr>
</tbody>
</table>

Source: URS Australia (2002)

Neither the average feedwater salinity nor the treatment flow capacity of Macquarie Generation’s two stations is known. As mentioned in Section 3.1.2.3, the NSW EPA (2001: 6) estimated that the discharge of saline water could save Macquarie Generation treatment costs in the order of $100/ML to $400/ML, without specifying the level of feedwater salinity and effluent volume. Obviously, the NSW EPA’s estimate is much lower than the figures in Table 7.8. For an optimistic estimation, the upper boundary of the operating cost at the feed salinity of 55000 EC in Table 7.8, that is, $2200/ML (in 1999 prices), has been used in the calculation of the treatment cost avoided by Macquarie Generation under the PHRSTS.
The average feedwater salinity of Redbank Power Station is approximately 33000 EC (NSW EPA 2001). Again, without information on its treatment flow capacity, the upper boundary of the operating cost at the feedwater salinity of 55000 EC, that is, $2200/ML (in 1999 prices), has also been used in the calculation of the treatment cost avoided by Redbank Power Station under the PHRSTS.

In the calculation, the average treatment cost of $2200/ML in 1999 prices is converted into $2478/ML in 2002 prices based on the price indexes $I_{1999} = 123.6$ and $I_{2002} = 139.2$ released by the ABS (2008). In conjunction with the estimated avoided volume of saline water ($\Delta VR$) as presented in Table 7.7, the total avoided treatment cost by the PHRSTS power stations is $29$ million (in 2002 prices), as shown in Table 7.9. As the avoided treatment of saline water due to permit trading only accounts for 5% of the total avoided volume of treatment, the cost saving of the power stations from permit trading is approximately $1.46 (= 29.2 \times 5\%)$ million over the whole the lifetime of the PHRSTS.

<table>
<thead>
<tr>
<th>Table 7.9</th>
<th>Avoided treatment cost by the power generation industry (in 2002 prices)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Power stations</td>
<td>$\Delta VR_j$ (ML)</td>
</tr>
<tr>
<td>Macquarie power stations</td>
<td>11756</td>
</tr>
<tr>
<td>Redbank Power</td>
<td>55</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>11811</strong></td>
</tr>
</tbody>
</table>

$\Delta VR_j =$ Power Station j’s total saline water avoided treating over the entire PHRSTS period
7.2.3 Cost saving of the PHRSTS from the new development

As in Box 7.1, the NSW EPA claimed that a significant cost saving from the PHRSTS was from allowing new development.

**Box 7.1 NSW EPA’s claim on the PHRSTS contribution to the new development**

"The pilot Scheme has enabled the development of new mining and power generation projects that might not otherwise have been possible under the previous [Trickle Discharge] system because that system restricted new salt discharge into the catchment (as discharges were already adversely affecting river salinity and threaten agriculture)…”

“One of these new developments is Bengalla Coal Mine, which was developed at a cost of around $300 million. This mine employs 300 people and has a projected coal output of nearly 6 Mt per year. The value of this output is around $300 million per year at current price.

Another new development that may not have been possible without the Scheme is Redbank Power Station, which was also established at a cost of $300 million. The power station will employ 50 full-time employees and will have a power output of 130 MW…”

“The benefits of avoiding the cost of foregone development are also significant, with each mine likely to employ over 300 people and to generation output worth over $300 million per year.”

Source: NSW EPA (2001: 12; 13; 51)

However, the high rate of saving from development not forgone cannot be attributed to the PHRSTS. In principle, the new mine and power station are technically able to achieve zero discharge through desalinisation technology or building large storage capacity or other alternatives. The PHRSTS just provides an option for the new development to achieve the stricter effluent standard (such as, Nil Discharge) at a lower cost than other control measures. Bearing this in mind, the cost effectiveness of the PHRSTS to the new developments is the avoided control cost that would have otherwise been incurred under other control systems such as the TD.

This study estimated the cost saving of the PHRSTS to the new development via calculation of the avoided storage capacity for Bengalla Coal Mines and the avoided saline water treatment for
Redbank Power Station, as have been presented in the earlier sections of this chapter. In the calculations, the daily permitted discharge \((s_i \text{ or } s_j)\) for both Bengalla and Redbank under the TD is treated as zero to reflect the Nil Discharge requirement as if the PHRSTS had not been in place. Thus, any discharges from Bengalla and Redbank under the PHRSTS are counted as their savings in storage capacity enlargement or saline water treatment under the PHRSTS over the TD. The cost savings of Bengalla Coal Mine and Redbank Power Station were included in Table 7.6 and Table 7.9, respectively, and are summarised in Table 7.10 below. As the Table shows, the total cost saving of the new development under the PHRSTS is $2.2 million (in 2002 prices). The small portion of Redbank Power Station’s cost saving is partly due to the fact that it did not join the PHRSTS until 21 December 2000.

Table 7.10  Avoided control cost for new developments (in 2002 prices)

<table>
<thead>
<tr>
<th>Power stations</th>
<th>Avoided control cost saving (ML)</th>
<th>cost saving ($ million)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bengalla Coal Mine</td>
<td>276 (avoided storage capacity)</td>
<td>2.1</td>
</tr>
<tr>
<td>Redbank Power</td>
<td>55 (avoided saline water treatment)</td>
<td>0.1</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td></td>
<td><strong>2.2</strong></td>
</tr>
</tbody>
</table>

7.2.4  Total saving in control cost and its significance

7.2.4.1  Total cost saving of the PHRSTS

The total cost saving of the Pilot Hunter River Salinity Trading Scheme (PHRSTS) over the Trickle Discharge (TD) is the sum of avoided cost of storage enlargement by the mining industry ($20.2 million) and avoided treatment cost by the power generating industry ($29.2 million), equalling $49.4 million in 2002 prices (see Table 7.11). Table 5.6 shows the contribution fees of the PHRSTS across the years from 1995 to 2002. Accordingly, the total contribution fee during the entire PHRSTS period is calculated based on the price index released by the ABS (2008) and the real discount rate of 7%/yr recommended by the NSW Treasury (2007). The total transaction cost borne by the PHRSTS participant is $2.2 million, accounting for approximately 4.5% (= 2.2/49.4) of the total saving in control cost generated by the PHRSTS. Taking into account this transaction cost, the total net saving in control cost to the PHRSTS participants is $47.2 million.
Among the total net cost saving, $2.2 million or 4.5% of the saving comes from the new development. Thus, most of the cost saving of the PHRSTS comes from the existing mines and power stations. Furthermore, only $1.85 million — $0.4 million from the mines and $1.45 million from the power stations — comes from permit trading. The cost saving from permit trading only accounts for 3.7% (=1.85/49.4) of the total cost saving (see Table 7.12). So the vast majority of the cost saving of the PHRSTS actually derives from the dynamicism of the permits. This finding provides an important explanation of the low values of the FHRSTS salt credits indicated by their auction prices (see Section 5.4.3).

<table>
<thead>
<tr>
<th>Cost saving</th>
<th>Mining industry</th>
<th>20.2</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Power generating industry</td>
<td>29.2</td>
</tr>
<tr>
<td>Sum</td>
<td></td>
<td>49.4</td>
</tr>
<tr>
<td>Transaction cost</td>
<td></td>
<td>(-2.2)</td>
</tr>
<tr>
<td>Net cost saving</td>
<td></td>
<td>47.2</td>
</tr>
</tbody>
</table>

At first glance, the modest cost saving of the PHRSTS from permit trading seems conflict with the observation of the high volume of trading and large number of participants on the PHRSTS credit trading market (see Section 5.3). Contemplation offers a plausible explanation; the actual daily discharge record \( (d_t) \) used in the calculations above gives 100% accurate hindsight of discharge events. Reliance on that discharge record carries an implicit assumption that all events in the record are accurately forecast ahead of time and that sufficient advance notice was given for dischargers to make use of those opportunities. In reality, however, the firms conducted a large number of what I called precautionary trading to meet their precautionary excess discharge. As noted in Section 5.3, precautionary excess discharge occurred whenever a mine,
facing a high degree of uncertainty in the time and duration of the next discharge event, chose to make an excess discharge one day even when it still had spare dam capacity. This precautionary trading behaviour can be regarded as one of their strategies to address the inherent risk that the firms have to bear under any dynamic permits system, as discussed in Section 2.2.4. This explanation is well supported by the observation of the low utilisation rate of salt credits (that is, 17%) during the entire PHRST period, as in Section 5.2.2.

### 7.2.4.2 Significance of the total cost saving

Ideally, the significance of the total cost saving of the PHRSTS should be measured by its magnitude in relation to the total control cost of the coal mining and power generating industries under the TD. However, lack of data on the latter prevented this study proceeding in this way. The conclusion drawn from the earlier analysis in Section 7.2.1 is that the PHRSTS saved the coal mines up to 19% of the construction cost of storage capacity that would otherwise have been needed under the TD.

As an alternative, this study measures the significance of the PHRSTS total cost saving by calculating its magnitude in relation to the coal mines and power stations’ revenues. This study estimated the revenue for the PHRSTS mines during 1995-2002 based on their saleable production and average export price of the individual year sourced from the NSW Coal Industry Profile. Each year’s revenue is converted into 2002 prices using the price indexes released by the ABS (2008). The real discount rate of 7% (NSW Treasury 20007) is then used to convert a flow of revenues into a present value in 2002 prices. The total revenue of the coal mines during the PHRSTS period is then $32981 million in 2002 prices, as shown in Table 7.13.
Table 7.13  

<table>
<thead>
<tr>
<th>Year</th>
<th>Saleable production (Mt)</th>
<th>Average value ($/tonne)</th>
<th>Revenue ($ Million)</th>
<th>Price index</th>
<th>Revenue ($ Million) (in 2002 price)</th>
<th>Present value ($ Million)</th>
</tr>
</thead>
<tbody>
<tr>
<td>94/95</td>
<td>45.81</td>
<td>47.05</td>
<td>2155</td>
<td>113.0</td>
<td>2617</td>
<td>4202</td>
</tr>
<tr>
<td>95/96</td>
<td>45.91</td>
<td>53.09</td>
<td>2437</td>
<td>118.7</td>
<td>2818</td>
<td>4229</td>
</tr>
<tr>
<td>96/97</td>
<td>50.03</td>
<td>50.36</td>
<td>2520</td>
<td>120.4</td>
<td>2872</td>
<td>4028</td>
</tr>
<tr>
<td>97/98</td>
<td>61.26</td>
<td>52.62</td>
<td>3224</td>
<td>120.5</td>
<td>3670</td>
<td>4810</td>
</tr>
<tr>
<td>98/99</td>
<td>63.40</td>
<td>49.15</td>
<td>3116</td>
<td>122.5</td>
<td>3490</td>
<td>4276</td>
</tr>
<tr>
<td>99/00</td>
<td>64.97</td>
<td>42.69</td>
<td>2774</td>
<td>125.4</td>
<td>3034</td>
<td>3474</td>
</tr>
<tr>
<td>00/01</td>
<td>67.04</td>
<td>50.55</td>
<td>3389</td>
<td>133.2</td>
<td>3490</td>
<td>3734</td>
</tr>
<tr>
<td>01/02</td>
<td>69.55</td>
<td>60.79</td>
<td>4228</td>
<td>137.2</td>
<td>4228</td>
<td>4228</td>
</tr>
<tr>
<td>Total</td>
<td>467.97</td>
<td>23842</td>
<td></td>
<td></td>
<td>32981</td>
<td></td>
</tr>
</tbody>
</table>


In fact, approximately 75% of coal produced in the Hunter Region is exported overseas, with only 25% actually consumed by domestic users, primarily by the power stations in the Region (HVRF 2003). For instance, in 2001/2002, 20 Mt out of the 70 Mt total saleable productions was consumed by the two large power stations located in the Liddell area (HVRF 2003). It is reasonable to assume that the export prices should be higher than domestic prices because of the additional costs of transport and tariff for coal exporting. However, the absence of data on coal production supplied to domestic markets and their associated sale prices prohibit this study from separating the revenue of production exported from that sold in the domestic markets. The revenue may thus have been overestimated.

This study has calculated the revenue of Macquarie Generation during the PHRSTS period based on its annual electricity outputs and NSW average pool prices. Redbank Power Station joined the PHRSTS in December 2000. Information on its annual electricity outputs during 2001 and 2002 is unavailable. So the revenue of Redbank is not included in this calculation. The total revenue of the power generating industry during the entire PHRSTS is $5416 million, as presented in Table 7.14.
Table 7.14  Revenue of the Hunter power generating industry during the PHRSTS period

<table>
<thead>
<tr>
<th>Year</th>
<th>Electricity output (GWh)</th>
<th>Pool price ($/MWh)</th>
<th>Revenue ($ Million)</th>
<th>Price index</th>
<th>Revenue in 2002 prices ($ Million)</th>
<th>Present value ($ Million)</th>
</tr>
</thead>
<tbody>
<tr>
<td>96/97</td>
<td>24,501</td>
<td>21.9</td>
<td>536</td>
<td>120.4</td>
<td>629</td>
<td>943</td>
</tr>
<tr>
<td>97/98</td>
<td>23,894</td>
<td>14.3</td>
<td>341</td>
<td>120.5</td>
<td>400</td>
<td>400</td>
</tr>
<tr>
<td>98/99</td>
<td>24,096</td>
<td>24.2</td>
<td>583</td>
<td>122.5</td>
<td>672</td>
<td>672</td>
</tr>
<tr>
<td>99/00</td>
<td>25,059</td>
<td>28.3</td>
<td>708</td>
<td>125.4</td>
<td>797</td>
<td>797</td>
</tr>
<tr>
<td>00/01</td>
<td>25,984</td>
<td>37.7</td>
<td>979</td>
<td>133.2</td>
<td>1,037</td>
<td>1,037</td>
</tr>
<tr>
<td>01/02</td>
<td>26,084</td>
<td>34.8</td>
<td>907</td>
<td>137.2</td>
<td>932</td>
<td>932</td>
</tr>
<tr>
<td>02/03</td>
<td>24,333</td>
<td>32.9</td>
<td>801</td>
<td>141.1</td>
<td>801</td>
<td>801</td>
</tr>
<tr>
<td>Total</td>
<td><strong>173,951</strong></td>
<td><strong>4,856</strong></td>
<td></td>
<td></td>
<td><strong>5,583</strong></td>
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Source: Compiled from data provided by the Macquarie Generation in 2003

Table 7.15 compares the cost saving of the PHRSTS from the mining industry and Macquarie Generation in relation to their respective revenue during 1995-2002. The total net cost saving of the PHRSTS accounts for 0.12% (= 47.2/38564) of the overall PHRSTS participants’ production revenue. The cost savings of the PHRSTS from the coal mine and power stations account for 0.06% (= 0.2/32981) and 0.52% (= 29.2/5583) of their revenue, respectively. This means that the cost saving is marginally more significant to the power stations than to the coal mines. This is because the main sources of cost saving for the coal mines are different from those of the power stations. As discussed earlier, while the savings mainly derive from capital costs of storage dams for the coal mines, the savings are mainly from the operational costs of the desalinisation processes for the power stations. The PHRSTS might not be able to reduce the coal mines’ capital cost, as most of the storage dams had been constructed prior to the inception of PHRSTS. It can reduce the operation cost for the power stations by allowing variation of low treatment rates in response to changes of river conditions.

---

20 1 GWh (gigawatt hour) = 1000 MWh (megawatt hour)

246
Chapter 7 Examination of the Economic Effectiveness of the PHRSTS

### Table 7.15 Cost savings of the PHRSTS versus industrial revenue

<table>
<thead>
<tr>
<th></th>
<th>Cost saving ($ million)</th>
<th>Production revenue ($ million)</th>
<th>Saving / revenue (%)</th>
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</thead>
<tbody>
<tr>
<td>Mining industry</td>
<td>20.2</td>
<td>32981</td>
<td>0.06</td>
</tr>
<tr>
<td>Power generating industry</td>
<td>29.2</td>
<td>5583</td>
<td>0.52</td>
</tr>
<tr>
<td>Transaction cost</td>
<td>(-2.0)</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>47.2</strong></td>
<td><strong>38564</strong></td>
<td><strong>0.12</strong></td>
</tr>
</tbody>
</table>

### 7.3 Summary

Using an operational model and available empirical data, this chapter has estimated the avoided storage enlargement by the mining industry and the avoided saline water treatment by the power generating industry under the Pilot Hunter River Salinity Trading Scheme (PHRSTS) over the Trickle Discharge (TD), respectively. This enabled estimates of the cost savings of these industries to be made from the dynamicism and tradability of the PHRSTS separately, and of the significance of the cost savings. It revealed that the PHRSTS enables the mining industry to avoid a storage capacity enlargement of 2624 ML and the power generating industry to avoid a salt treatment of 27616 tonnes, which would otherwise be needed under the TD. This generates a cost saving of $20.2 million to the mining industry and $29.2 million to the power generating industry, of which the new development saves $2.2 million. Together, the total net cost saving during the entire PHRSTS period (1995-2002) is $47.2 million after a $2.2 million transaction cost is taken into account.

The total cost saving may be worthwhile in relation to the control cost of the industries (for example, a 19% saving is worthwhile to the coal mines in relation to their storage costs), but it is very modest by other measures, being only 0.12% of their overall production revenue. Furthermore, of the total net cost saving, only 3.7% ($1.85 million) actually derives from permit trading.

In conclusion, the PHRSTS did not generate a substantial saving in the total cost of saline water management to either its initial participants or the two new developments over its entire period. Tradability of the discharge permits produced only a very modest cost saving, and the
dynamicism of the permits was the main source of the cost saving of the PHRSTS. This conclusion is, again, contrary to the NSW EPA’s claim reported in Box 1.1 of Chapter 1.
Chapter 8 Conclusions

8.1 Main findings and contributions of this thesis

This thesis has examined the application of dynamic tradable discharge permits (DT) in managing the discharge of conservative pollutants from industrial point sources into a river system. Its main contributions lie at two levels. Conceptually, it has constructed an analytical framework that allows comparisons of various discharge permits schemes from both environmental and economic perspectives (Section 2.4). This framework enriches the theoretical foundation of DT as the most cost-effective pollution control scheme. Empirically, it has evaluated many aspects of the pilot Hunter River Salinity Trading Scheme (PHRSTS) which ran from 1995 to 2002, and some aspects of the formalised Hunter River Salinity Trading Scheme (FHRSTS) which succeeded it in 2002. It has investigated the origins, evolution and institutional arrangement of the PHRSTS; examined the performance of the credits trading market of the PHRSTS, and the 2004 and 2006 auctioning markets of the FHRSTS; and as its major contribution to knowledge, measured the environmental and economic effectiveness of the PHRSTS. Altogether, it provides the first comprehensive quantitative assessment for many aspects of the HRSTS. To this end, the study not only offers the interim feedback necessary to shape the evolution of the Scheme over time, but also enriches the empirical evidence on operation of tradable permit programs, adding knowledge about how well a tradable permit system can work and the design features that contribute to desirable outcomes.

To provide this assessment, it proved necessary to undertake a comprehensive review of the environmental and engineering science of industrial saline discharges to the Hunter River in New South Wales (Chapter 3). It was also necessary to carry out an extensive investigation of the long and varied institutional history of how these discharges have been managed, and of the initiation, development and evolution of the PHRSTS (Chapter 4). While not providing any new data analyses or policy recommendations, these reviews and investigations make an interdisciplinary contribution to knowledge by bringing together for the first time a wide range of information on the HRSTS, much of which was obtained from oral sources and thus previously unavailable in the written record.
This study found that the PHRSTS participants tended to take the advantage of the effectively unlimited discharge during the Scheme's Flood flows to the Hunter River instead of the tradable discharge in the High flows. Around 53% of total salt load was discharged in the Flood flow period (2.3% of the total time), whilst 43% was discharged in the High flow period (9.6% of the total time) (Section 5.2). This reflects the feature of industrial saline water generation that the volume of the saline water generated on mine sites has a generally positive relationship with river flows. It also helps explain the low average utilisation rate of salt credits (Section 5.2), and the low prices of salt credits in the 2004 and 2006 auctions of the FHRSTS (Section 5.4).

Another finding was that the credit trading market of the PHRSTS was active in terms of both volume of trades and number of participants, in spite of the high proportion of intra-company trades due to the high cross-company ownership of the PHRSTS participants. There were 157 credit transactions involving 6811 credits traded during the PHRSTS period; 50% of these were intra-company trades. All the PHRSTS participants were involved in credit trading activities, and a significant proportion of firms (9 out of 22 firms) were both sellers and buyers (Section 5.3). The successful bidders in the 2004 and 2006 credit auctions of the FHRSTS were a mix of sellers and buyers on the credit trading market of the PHRSTS. The low prices of the salt credits in the 2004 and 2006 auctions suggest that the firms did not value the salt credits very highly. The narrow range of these auction prices also indicates little diversity in the salt control costs across the firms, and suggests that the gains of the PHRSTS participants from credit trading are not great (Section 5.4).

Environmental effectiveness and economic effectiveness are the two important criteria used for evaluating the PHRSTS in this thesis and formed the bases of the two research questions posed in Chapter 1. Examination of the environmental effectiveness of the PHRSTS, measured against the Trickle Discharge (TD) control system that it replaced in 1995, has provided one of the main contributions to knowledge, and has been conducted in various ways (Chapter 6). Regression models that account for the autocorrelation (time-series) effect in the data were used to examine the long-series monthly mean observations of river salinity and flow. They showed that neither the overall river salinities and flows nor the river salinity of Low flow period between the PHRSTS and the TD are significantly different in a statistical sense. This finding was confirmed by analysis using a simulation model, which showed that the difference in the river salinity between the (actual) with-PHRSTS and (hypothetically modelled) without-PHRSTS situations is minor. Thus, the PHRSTS saved only an insignificant amount of the social cost of the river salinity (approximately $0.3 million at 2002 prices) during its nearly 8-year lifetime. These
findings lead to the conclusion that the PHRSTS has had little, if any, effect on the improvement of river salinity, contrary to the NSW EPA’s claim regarding the environmental effectiveness of the PHRSTS, as reported in Box 1.1.

Evaluation of the economic (cost) effectiveness of the PHRSTS compared to the TD is another major contribution of this study to knowledge here. The new theoretical model developed in this thesis would have demonstrated the cost saving potential of the PHRSTS over other regulatory schemes if the required data had been available (Chapter 7.1). Since they were not, the cost effectiveness of the PHRSTS was evaluated with a simple operational model, by estimating the avoided storage capacity for the mining industry and the avoided treatment of saline water for the power generating industry under the PHRSTS that would otherwise have been needed under the TD (Chapter 7.2). The model estimates that during its lifetime, the PHRSTS generated a cost saving of $20 million to the mining industry and $29 million to the power generating industry, of which $2.2 million derived from a new coal mine and a power station, while maintaining the river salinity objectives. The total net cost saving of the PHRSTS was approximately $47 million when the transaction cost of $2 million is taken into account. Furthermore, of the total cost saving, only 3.7% ($1.85 million) came from the credit trading. This finding of modest cost saving of the PHRSTS resulting from allowing the permits to be traded is consistent with the above observations of low average utilisation rate under the PHRSTS and low auction prices of salt credits under the FHRSTS. It also shows that the great majority of the cost saving of the PHRSTS in fact came from its dynamicism, that is, its allowing the total permitted discharge to vary over time in response to changing (Low, High or Flood) flows in the Hunter River. These findings, again, are contrary to the NSW EPA’s claim regarding the economic effectiveness of the PHRSTS, as reported in Box 1.1.

Overall, the finding of this study is that the PHRSTS attained the river salinity objectives of the Hunter River whilst allowing new development. However, it did not significantly improve the river salinity. Nor did it generate a substantial cost saving to its participants. These conclusions do not support the NSW EPA’s claim that the PHRSTS achieved significant environmental and cost (economic) effectiveness. The insights derived from this study on the merits of tradable permit system in terms of environmental and cost effectiveness in a broader sense highlight the potential benefits of integrating various regulatory instruments for managing natural resources and environmental quality at a time where the prevailing wisdom seems to be that the market is
a panacea to all problems. This could have strong implication for other fields of resource and environmental management beyond water quality management for river systems.

The experimental design features and operational experiences of permit trading being built into a dynamic permits control system are perhaps the most important benefits of the PHRSTS. These experiences include, but are not limited to, that:

- An environmentally and economically effective water quality management program requires the combination of multiple policy tools and the cooperation of multiple parties involving government, market, communities and polluters.

- Legislative and regulatory backing is essential for any market-based environmental management program.

- Fundamental scientific understanding of the nature of the pollutant and the characteristics of the river being regulated is essential for designing an effective river quality management program.

- Clearly defined and well agreed environmental objectives, and the stakeholder’s participation and support, are cornerstones for the effective implementation of any river quality management program.

- Appropriate institutional mechanisms for monitoring, reporting and enforcement are necessary for effective enforcement.

- Adequate information flow and low transaction costs are important to a smooth effluent trading market.

These experiences have formed a basis for the Scheme’s evolution, development and formalisation. They will prove a useful reference for developing innovative and effective water quality management programs in Australia and elsewhere, even for other tradable permit programs in the areas beyond water quality trading.

Further, less important contributions to knowledge from this thesis are:
Chapter 8 Conclusions

• The theoretical framework for comparing various discharge permits schemes from both environmental and economic perspectives (Section 2.4) will help empirical studies in devising strategies for river quality management.

• The new approach to accounting for the time series effect in analysis of the river flow and salinity of the Hunter River (Section 6.3) can provide a reference for hydrologic and water quality analysis for other rivers.

• The flow-discharge-salinity model developed for the Hunter River (Section 6.4) can contribute to the improvement of the ongoing FHRSTS operation by providing a more advanced model to predict the discharge events and calculate the salt discharge permits.

• The operational model for estimation of the cost saving for the coal mines (Section 7.2) can be extended to cost estimation for storable river pollutants, which is sparse in the literature.

8.2 Recommendations for the FHRSTS

The HRSTS has evolved since it was formalised and made permanent on 1 December 2002. There were a number of changes accompanying the formalisation of the HRSTS that built on the PHRSTS's operational experience, such as re-distributing the salt credits via progressive public auctions, and re-defining the thresholds of the river flow periods to address the increasing growth of industrial growth in the upper Hunter region (see Section 5.4). While these changes are expected to improve the environmental and economic effectiveness of the FHRSTS, this study still has two recommendations for improving the performance of the FHRSTS, as follows.

The information on the trading prices of salt credits is important for understanding the HRSTS credit trading and auctioning markets. Trading prices, along with the auction prices, could provide the best evidence of true value of the salt credits and more general insights into this aspect. This information is also important for empirical research on the HRSTS. However, reporting the trading prices is not required under either the PHRSTS or the current FHRSTS. So it is in the public interest to publish this information.

Looking ahead, a significant challenge exists to the FHRSTS in the context of improvement of the overall river salinity in the Hunter River. There has been clear evidence that nonpoint sources (NPS), such as irrigation runoff, are also major contributors to the salinity in the Hunter River. It is conceivable that in some cases reducing or even eliminating industrial salt discharge
would still be inadequate to achieve the river salinity objectives, given that the industrial salt inputs only account for a small proportion (some 5%) of the total salt received by the Hunter River. This suggests that improvement of the overall river salinity of the Hunter River also relies on management of the NPS in the region. In some instances, it is likely that the costs of reducing salt inputs from NPS can be much lower than reduction from well managed point sources. While the current Scheme is still confined to point-to-point trading, the greater benefits of effluent trading under the HRSTS would be realised when trades are made between point and nonpoint sources. In recognition of more difficulties in water quality trading involving NPS (such as, set up appropriate trading ratios, monitoring and enforcement), exploring the potential for inclusion of NPS in the Scheme would be more challenging but more rewarding.

8.3 Prospects for future study

Several follow-up topics in need of further research have been identified during the course of this study. These are:

- Improving understanding of the social costs associated with various levels of river salinity in the Hunter River (see Section 3.7); and

- Using recent-year data to test and calibrate the simulation model of discharge-flow-river salinity that has been developed in this study based on short-series historical observations (see Section 6.4.2).

It has been six years since the FHRSTS was made permanent in December 2002. While this study has partially investigated the 2004 and 2006 auctions of the FHRSTS, so far the performance of the FHRSTS has not been fully investigated. Thus, genuine opportunities exist for evaluating various aspects of the FHRSTS. Further studies of the areas given below are of particular interest and would be welcomed.

- Examining the relationship between the salt credit trading market and the auction market under the FHRSTS with the information generated from subsequent credit auctions when the trading price information becomes public available;

- Investigating the effect of the FHRSTS on changes of firms’ strategies for management of saline water;
Chapter 8 Conclusions

- Investigating the effect of the FHRSTS on firms’ incentives for further reduction of saline water discharge;

- Measuring empirically the degree of environmental effectiveness achieved by the FHRSTS;

- Estimating the cost saving generated by the FHRSTS, in particular, for the new development during the period, and measuring its significance; and

- Undertaking a feasibility study of extending the FHRSTS to include NPS.
## Appendices

### Appendix 1  List of the HRSTS stakeholders consulted for this study

<table>
<thead>
<tr>
<th>Stakeholders</th>
<th>Contact</th>
<th>Position</th>
<th>Consultation methods</th>
<th>Site</th>
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<tbody>
<tr>
<td>Department of Land and Water Conservation (DLWC)</td>
<td>David Hoey</td>
<td>Environmental Officer</td>
<td>Emails; interviews; group meetings</td>
<td>NSW Department of Natural Resources (DNR)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Newcastle Office; Canberra</td>
</tr>
<tr>
<td>DLWC</td>
<td>Eddie Harris</td>
<td>Manager Resource Assessment &amp; Planning Hunter Region</td>
<td>Interviews; group meetings</td>
<td>NSW DNR Newcastle Office</td>
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<tr>
<td>DLWC</td>
<td>Sandra Mitchell</td>
<td>HRSTS Administrator</td>
<td>Emails; interviews; group meetings</td>
<td>NSW DNR Newcastle Office</td>
</tr>
<tr>
<td>DLWC</td>
<td>Mark Simons</td>
<td>Resource analyst</td>
<td>Emails; interview</td>
<td>NSW DNR Newcastle Office</td>
</tr>
<tr>
<td>NSW EPA (Newcastle Office)</td>
<td>Mitchell Bennett</td>
<td>Head of Regional Operations</td>
<td>Emails; interview; group meeting</td>
<td>NSW EPA Newcastle Office</td>
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<tr>
<td>Hunter Catchment Trust</td>
<td>Dean Chapman</td>
<td>Catchment Manager – Land &amp; Water</td>
<td>Interview</td>
<td>Newcastle</td>
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<td>Hunter River Research Foundation</td>
<td>W.E.J. Paradice</td>
<td>Chairman</td>
<td>Emails</td>
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<td>Coal &amp; Allied Coal Mine</td>
<td>John Pola</td>
<td>Environmental Specialist – Water Manager</td>
<td>Interview</td>
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<td>Liddell Coal Operation</td>
<td>Edward Wegner</td>
<td>Environmental Co-coordinator</td>
<td>Interview; emails</td>
<td>Credit auctions at Newcastle</td>
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<td>NSW Minerals Council</td>
<td>Peter Smith</td>
<td>Assistant Director, Environment</td>
<td>Emails; interviews; Group meetings</td>
<td>Sydney; Singleton; Newcastle</td>
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<td>HRSTS participants</td>
<td>Representives of coal mines and power stations</td>
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<td>Credit auctions at Singleton</td>
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<td>NSW Department of Environment and Conservation</td>
<td>Michele Weight</td>
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<td>Macquarie Generation</td>
<td>Jim Devine</td>
<td>Public Affairs Manager</td>
<td>Emails</td>
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<td>Coal &amp; Allied</td>
<td>Scott Oakley</td>
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### Appendix 2  Available data and their sources for this study

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<td>Prices saleable coal</td>
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<td>1990-2002</td>
<td>NA</td>
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<td><strong>HRSTS participants (power stations)</strong></td>
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<td>Electricity outputs (Yearly)</td>
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<td><strong>PHRSTS operation</strong></td>
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<td>TAD (Daily)</td>
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<td>Salt load discharge (Daily)</td>
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<td>Contribution fee (yearly)</td>
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<tr>
<td>Prices of bids</td>
<td>2004 and 2006</td>
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</table>

Note: NA = Not available.
Appendices

Appendix 3  System of coal-fired power stations

To help better understand water consumption by and wastewater discharge from the power stations in the Hunter Valley, a brief description of the thermoelectric generation process is provided here. In general, almost all coal-fired power stations work in such a way that coal and fuels are burned inside a boiler to heat water and generate steam. The steam is then used to turn turbines that drive the power generators to make electricity.

The prevailing work process of coal-fired power stations is shown as Figure A3.1. Coal transported from the nearby open-cut and underground mines is crushed before being transported to the power station coal handling area. The coal is fed by conveyor into the stations’ mills, and pulverized into a fine consistency. The pulverized coal is blown into the boiler in a stream of pre-heated air. The coal is then burned in the boiler furnace chamber and produces great heat to convert water, circulating in the boiler tubes, into high-pressure steam. An extensive fabric filter system between the boiler and emission stack extracts virtually all fine ash particles, a by-product of burnt coal, from boiler exhaust gasses (Macquarie Generation 2005).

Steam, injected at very high pressure into the turbine, turns the fan-like turbine blades mounted along the main drive shaft of the turbine. For maximum efficiency, the stream is reheated in the boiler after its energy is partially spent driving the first stage of the turbine (higher pressure cylinder where the blades are smallest in diameter). It is then passed to the second stage of the turbine (intermediate pressure cylinder-large blades) and, as heat and pressure dissipate, on to the two low pressure cylinders, where the blades are largest. The spent steam is then cooled back to water for re-use as it passes over a series of condenser tubes through which cold water from the water cooling system is circulated. Electricity produced by the generators then passes into the adjacent power station switchyard through a transformer (Macquarie Generation 2005).

Thus, the generation of electricity in coal-fired power stations requires large amounts of water. There are two separate water systems involved: one is for the steam production cycle, and the other is for cooling purposes in the cycle. Most of the water in both systems is used over and over again. For the steam cycle, the quality of the feedwater used to produce the steam is critical. This is because impurities in the feedwater can cause problems such as scaling that will reduce the amount of electricity produced. They can also cause erosion of equipment and reduce
the efficiency of operations. Therefore, power stations usually treat the feedwater before it is used in any processes. By removing any contaminated content from the feedwater, the number of cycles before the deleterious effects of scaling are felt is maximised (Macquarie Generation 2005).

Figure A3.1 Flow diagram of coal-fired power station system

The salinity level in the Hunter River is naturally high. Before the water extracted from the River is used as feedback for steam generation and cooling, both power stations apply reverse osmosis (RO) to remove the salt from the water. RO is one of the most common desalination technologies used primarily with brackish waters (as opposed to seawater). While the RO treatment capacity of the Liddell Power Station is unknown, Bayswater Power Station owns a 35 ML/day RO plant, the largest desalination plant in Australia (Macquarie Generation 2005).

In RO (see Figure A3.2), saline feedwater is pumped at high pressure through permeable membranes, separating salts from the water. To this end, the saline water is separated into two streams: the product steam with a low salinity concentration (freshwater stream) and the effluent stream with a concentrated brine solution. When the non-salt-free freshwater stream is used by the power stations for steam generation and cooling, the soluble salt is concentrated, and the water becomes even more saline. The prevailing methods for desalination plants dealing with the brine stream include discharging it to the ocean, producing salt products or filling in land after it has dried out. However, Macquarie Generation stores the saline water in its dam, Lake Liddell, for management or disposal (Macquarie Generation 2005).
Figure A3.2  Flow diagram of a reverse osmosis system

Source: Macquarie Generation (2005)
Appendices

Appendix 4 Examination of incremental discharge

This appendix examines the ambient and incremental controls in a rigorous way. The purpose of this examination is to find out whether the control is a static permits scheme or dynamic permits scheme.

Let \( Q \) be river flow or industrial saline water discharge. \( C \) is the discharge or river salinity. The suffixes \( r \) and \( d \) refer to river and discharges, respectively, and the suffix \( rd \) represents the river status after the inputs of industrial discharge (see Figure A4.1).

Figure A4.1 River salinity before and after discharge input

\[
\begin{align*}
\text{Discharge} & \quad Q_d \quad C_d \\
\text{River} & \quad Q_r \quad C_r \\
\text{River} & \quad Q_{rd} \quad C_{rd}
\end{align*}
\]

where

\( Q_r \): river flow background at discharge point (ML/day).

\( C_r \): river salinity background concentration at discharge point (\( \mu \text{S/cm} \))

\( Q_d \): volume of industrial discharge (ML/day)

\( C_d \): salinity concentration of industrial discharge (\( \mu \text{S/cm} \))

\( Q_{rd} \): river flow after the inputs of industrial discharge (ML/day),

\( C_{rd} \): river salinity concentration after the inputs of industrial discharge (\( \mu \text{S/cm} \)).
Appendices

Since salt is a conservative pollutant, assuming that industrial saline water immediately and uniformly mixes with river flow once it is discharged into the river, it follows a mass balance, whose basic formula is:

\[ Q_{rd} \times C_{rd} = Q_r \times C_r + Q_d \times C_d \]  \hspace{1cm} [A4.1]

where \( Q_{rd} = Q_r + Q_d \)  \hspace{1cm} [A4.2]

[A4.1] and [A4.2] together give

\[ C_{rd} = \frac{Q_r C_r + Q_d C_d}{Q_r + Q_d} \]  \hspace{1cm} [A4.3]

This, the downstream salinity, faces two constrains:

1. \( C_{rd} \leq 700 \), which after simple algebra,

\[ \Rightarrow Q_d \leq \frac{(700 - C_r)Q_r}{C_d - 700} \]  \hspace{1cm} [A4.4]

provided \( C_d > 700 \) (if not the constraint cannot be binding)

2. \( C_{rd} \leq C_r + 40 \), which after simple algebra,

\[ \Rightarrow Q_d \leq \frac{40Q_r}{C_d - C_r - 40} \]  \hspace{1cm} [A4.5]

provided \( C_d > C_r + 40 \) (if not the constraint cannot be binding)

So, assuming both \( C_d > 700 \) and \( C_d > C_r + 40 \),

\[ Q_d \leq \min\{ \frac{(700 - C_r)Q_r}{C_d - 700}, \frac{40Q_r}{C_d - C_r - 40} \} \]  \hspace{1cm} [A4.6]

[A4.6] indicates, the maximum volume of saline water allowed to be discharged (\( Q_d \)) has a progressive linear relationship with the river flow (\( Q_r \)), other things being equal. Assuming the salinity concentration of discharge (\( C_d \)) of a firm’s discharge is fixed, this means that the higher the river flow, the greater the volume of saline water (and in turn the more salt load) allowed to be discharged. From this standpoint only, the ambient and increment control is actually a version of dynamic discharge permits.
### Appendix 5  Summary of credits held by the PHRSTS participants

<table>
<thead>
<tr>
<th>Location</th>
<th>Initial allocation in 1995</th>
<th>Credits by 2002</th>
<th>Credits by 2003</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bayswater/Mt Arthur</td>
<td>26</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>Camberwell</td>
<td>16</td>
<td>15</td>
<td>15</td>
</tr>
<tr>
<td>Cumnook</td>
<td>15</td>
<td>15</td>
<td>15</td>
</tr>
<tr>
<td>Dartbrook **</td>
<td>13</td>
<td>15</td>
<td>25</td>
</tr>
<tr>
<td>Drayton</td>
<td>27</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>Howick</td>
<td>37</td>
<td></td>
<td>0</td>
</tr>
<tr>
<td>Hunter Valley</td>
<td>55</td>
<td>120</td>
<td>120</td>
</tr>
<tr>
<td>Mt Thorley</td>
<td>27</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Leminton</td>
<td>42</td>
<td>40</td>
<td>40</td>
</tr>
<tr>
<td>Liddell **</td>
<td>56</td>
<td>55</td>
<td>65</td>
</tr>
<tr>
<td>Mt Owen</td>
<td>14</td>
<td>15</td>
<td>15</td>
</tr>
<tr>
<td>Muswellbrook</td>
<td>11</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>Ravensworth/Naruma</td>
<td>72</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>Rixs Creek</td>
<td>26</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>Saxonvale/Bulga **</td>
<td>38</td>
<td>40</td>
<td>50</td>
</tr>
<tr>
<td>Swamp Creek</td>
<td>15</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>United</td>
<td>8</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>Wambo</td>
<td>33</td>
<td>35</td>
<td>35</td>
</tr>
<tr>
<td>Wakeworth</td>
<td>40</td>
<td>40</td>
<td>40</td>
</tr>
<tr>
<td>Pacific Power</td>
<td>229</td>
<td>230</td>
<td>230</td>
</tr>
<tr>
<td>Bengalla *</td>
<td>0</td>
<td>35</td>
<td>35</td>
</tr>
<tr>
<td>Redbank Power Station *</td>
<td>0</td>
<td>35</td>
<td>35</td>
</tr>
<tr>
<td>Nardell *</td>
<td>0</td>
<td>30</td>
<td>30</td>
</tr>
<tr>
<td>Mt Arthur North **</td>
<td>0</td>
<td>0</td>
<td>40</td>
</tr>
<tr>
<td>Glennies Creek **</td>
<td>0</td>
<td>0</td>
<td>15</td>
</tr>
<tr>
<td>EPA reserve</td>
<td>200</td>
<td>85</td>
<td>0</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td>1000</td>
<td>1000</td>
<td>1000</td>
</tr>
</tbody>
</table>

* refers to the new firms that the NSW EPA allocated credits to between 1995 and 2002.
** refers to the firms to which the NSW EPA allocated extra credits between 2002 and 2003.

Sources: Compiled from data from NSW DLWC (2002); NSW EPA (1995; 2003a)
Appendices

Appendix 6  Example of discharge controlled by credits

The following example, taken from a publication prepared by the NSW EPA (2003a), illustrated how the PHRSTS worked in both spatial and temporal dimensions.

Table A6  River block moves along spatial and temporal dimensions

<table>
<thead>
<tr>
<th></th>
<th>Firm A</th>
<th>...</th>
<th>Firm B</th>
<th>Singleton</th>
</tr>
</thead>
<tbody>
<tr>
<td>15/7/2003</td>
<td>198</td>
<td>197</td>
<td>196</td>
<td>195</td>
</tr>
<tr>
<td>16/7/2003</td>
<td>199</td>
<td>198</td>
<td>197</td>
<td>196</td>
</tr>
<tr>
<td>17/7/2003</td>
<td>202</td>
<td>201</td>
<td>200</td>
<td>199</td>
</tr>
</tbody>
</table>

Source: Adapted from NSW EPA (2003a)

Suppose that Block 198 was the block of water that passed Singleton on the 198th day of 2003 (17/7/2003). Firm A with 25 credits was located at upper sector of the Hunter River, and Firm B with 15 credits was at middle sector of the River. Block 198 flowed pass the firms on different days (see above Table). The following procedures were followed:

Based on the real-time monitoring data of flow level and the ambient salinity generated by the HITS, DLWC classified the block as Flood, High or Low flow period. If the block was classified as a Flood flow period, the industrial wastewater could be discharged freely subject only to the volumetric conditions of individual licences. If the block was classified as Low flow period, no industrial wastewater discharge was allowed. If the block was classified as High flow period, DLWC calculated the total allowable salt load, by the IQQM, that could be added to the block so that salinity objective of 900 EC at Singleton was attained. Suppose Block 198 held 100 tonnes of salt, then each credit allowed discharge of 0.1 (= 100 x 0.1%) tonnes of salt into the River.

On 15/7/2003 when the block passed Firm A, Firm A was granted permission to discharge up to 2.5 (= 200 x 25 x 0.1%) tonnes of salt load. On 16/7/2003 when Block 198 passed to Firm B, Site B was granted permission to discharge up to 1.5 (= 200 x 15 x 0.1%) tonnes of salt load. The firms performed a second calculation to convert the permitted salt load into the volume of its discharge water based on its salinity concentration of effluent.
In the extreme case that Firm A did not discharge at all and traded 25 credits to Firm B, then Firm B could discharge 40 \((= 15 + 25)\) credits of salt, which was equivalent to 4 \((= 200 \times 40 \times 0.1\%)\) tonnes. In the case that Firm A discharged 20 credits of salt and traded the rest of credits to Firm B, then Firm B could discharge 20 \((= 15 + 5)\) credits of salt. The total salt load discharged from these two firms was still 4 tonnes.

In this way, the total allowable salt load of a block was controlled under the “cap”. On 17/7/2003 when Block 198 passed Singleton, the river salinity was less than 900 EC.
Appendices

Appendix 7  Simple statistical methods without autocorrelation

This section uses simple statistical methods to test whether there is a difference in river salinity and river flow before and after the PHRSTS implementation. Three methods are used: the Student's t-test, the Wilcoxon Rank Sum test and the Quantile test. All these methods assume that the data are independent of each other. However, the exploratory data analysis has shown that considerable autocorrelation is evident in both data sets, so none of these simple statistical methods is actually appropriate to test for a difference before and after the HRSTS implementation. The purpose of this Appendix is to derive conclusions from these erroneous tests so that they can be compared with conclusions drawn from appropriate testing methods, which are presented in Section 6.2 of the thesis.

Statistical methods are generally divided into two classes: parametric and nonparametric. While the former relies on the underlying assumption that the data have a particular distribution, no assumption about the form of the distribution is required by the latter. The Student's *t*-test is perhaps the most widely-used parametric method for comparing two groups of data. This test compares the mean of one group of data with the mean of the other. The following assumptions are made: (1) the two groups of data come from distributions with the same standard deviations; and (2) the data within each group are independent of each other, that is, no autocorrelation exists in the data; (3) in a small sample (no more than 30) the data must be normally distributed.

In the statistical literature, such as Venables and Ripley (1999), Millard and Neerchal (2001), Helsel and Hirsch (2002) and Ginevan and Splitstone (2004), a robust nonparametric method, the Wilcoxon Rank Sum (WRS) test, is recommended as an alternative to the Student's *t*-test when the samples are small and the data are non-normal. The WRS test tests the same null hypotheses as the Student's *t*-test. However, the WRS test is based on the ranks of the data, not the data themselves. The WRS performs by first pooling the two groups of observations, and assigning each observation a rank (the smallest has rank 1, the 2nd smallest rank 2, and so on). The ranks for the observations from each of the groups are then added up, and the probability of distribution of the rank sum determined. The WRS test requires both groups of data to have the same distribution, but is not affected by outliers.

Another method sometimes used to test the hypothesis of no difference between two groups of data while avoiding concerns about the data distribution is the Quantile test. Although the WRS
and Quantile test are both nonparametric tests, which do not make assumptions about the distributions of the data: they have different focuses from the statistical testing point of view. The WRS test is designed to detect a shift in the whole distribution of a data set relative to the distribution of another. The Quantile test is used to detect a difference between the two sample distributions where only a portion of the distribution of one sample is shifted relative to the distribution of another (US EPA 1994; Millard and Neerchal 2001; Ginevan and Splitstone 2004). The US EPA (1994) once adopted the Quantile test in conjunction with WRS test to assess the effectiveness of cleanup programs in remediated soils and solid media. The Quantile test was then proposed by Millard and Neerchal (2001) and Ginevan and Splitstone (2004) to undertake interrupted time series analysis for general environmental studies, where the autocorrelation is minimal and can be ignored.

In this research, the Student’s t-test, the WRS test and the Quantile test were used separately to test the following null hypotheses ($H_0$):

For the river salinity, $H_0$: $\ln(EC_0) = \ln(EC_1)$, meaning there is no difference between the monthly mean river salinity observations under the PHRSTS and under the TD.

For the river flow, $H_0$: $\ln(Flow_0) = \ln(Flow_1)$, meaning there is no difference between the monthly mean river flow observations under the PHRSTS and under the TD.

The testing results are listed in Table A7.1. For $\ln(EC_0)$ and $\ln(EC_1)$, all these simple statistical methods strongly reject the null hypothesis at the 5% level, as the p-values are far less than the $\alpha$ value of 0.05. However, for $\ln(Flow_0)$ and $\ln(Flow_1)$, the results of these simple statistical methods are inconsistent. Only the WRS test rejects the null hypothesis.
In summary, all three simple statistical methods agree that river salinity of the PHRSTS is significantly different from that of the TD. However, the tests lead to different conclusions as to whether there has been a change in river flow after the PHRSTS implementation. This is in part because the tests have different powers, that is, probability of finding a difference if there is one.
Appendix 8  Models considering foresight and transitional effects

Follows are the fitted models for \( \ln(EC) \) considering foresight and transitional effects of the PHRSTS.

Table A8.1  Fitted models of \( \ln(EC) \) accounting for one-year foresight effect

<table>
<thead>
<tr>
<th>Independent</th>
<th>Models</th>
<th>ar1</th>
<th>Intercept</th>
<th>Month</th>
<th>PHRSTS (_1)</th>
<th>Month*PHRSTS (_1)</th>
<th>AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>( \ln(EC) )1.i.m1</td>
<td>( \ln(EC) )-Month                                      6.7802</td>
<td>-0.0157</td>
<td>-32.02</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )1.i.m1</td>
<td>( \ln(EC) )-PHRSTS (_1)                                6.6896</td>
<td>-0.1883</td>
<td>-19.09</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )1.i.m2</td>
<td>( \ln(EC) )-Month+PHRSTS (_1)                          6.7818</td>
<td>-0.0161</td>
<td>0.0065</td>
<td>-30.03</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )1.i.m3</td>
<td>( \ln(EC) )-Month+PHRSTS (_1)+Month*PHRSTS (_1) 6.7441</td>
<td>-0.0095</td>
<td>0.2427</td>
<td>-31.82</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>AR(1)</td>
<td>( \ln(EC) )1.ar1.m1 ( \ln(EC) )-Month 0.4497</td>
<td>6.7815</td>
<td>-0.0161</td>
<td>-78.94</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )1.ar1.m2</td>
<td>( \ln(EC) )-PHRSTS (_1)                                0.4751</td>
<td>6.6825</td>
<td>-15.76</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )1.ar1.m3</td>
<td>( \ln(EC) )-Month+PHRSTS (_1)                          0.4511</td>
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<td>-0.0184</td>
<td>0.0361</td>
<td>-77.08</td>
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<td></td>
</tr>
<tr>
<td>( \ln(EC) )1.ar1.m4</td>
<td>( \ln(EC) )-Month+PHRSTS (_1)+Month*PHRSTS (_1) 0.4420</td>
<td>27.8786</td>
<td>-0.0107</td>
<td>0.3111</td>
<td>-0.0207</td>
<td>-77.08</td>
<td></td>
</tr>
</tbody>
</table>

Table A8.2  Fitted models of \( \ln(EC) \) accounting for two-year foresight effect

<table>
<thead>
<tr>
<th>Independent</th>
<th>Models</th>
<th>ar1</th>
<th>Intercept</th>
<th>Month</th>
<th>PHRSTS (_2)</th>
<th>Month*PHRSTS (_2)</th>
<th>AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>( \ln(EC) )2.i.m1</td>
<td>( \ln(EC) )-Month                                      6.7802</td>
<td>-0.0157</td>
<td>-32.02</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )2.i.m2</td>
<td>( \ln(EC) )-PHRSTS (_2)                                6.6529</td>
<td>-0.1977</td>
<td>-15.76</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )2.i.m3</td>
<td>( \ln(EC) )-Month+PHRSTS (_2)                          6.7677</td>
<td>-0.0136</td>
<td>-0.0429</td>
<td>-30.78</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )2.i.m4</td>
<td>( \ln(EC) )-Month+PHRSTS (_2)+Month*PHRSTS (_2) 6.7578</td>
<td>-0.0124</td>
<td>0.4835</td>
<td>-0.0272</td>
<td>-31.72</td>
<td></td>
<td></td>
</tr>
<tr>
<td>AR(1)</td>
<td>( \ln(EC) )2.ar1.m1 ( \ln(EC) )-Month 0.4497</td>
<td>6.7815</td>
<td>-0.0161</td>
<td>-78.94</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )2.ar1.m2</td>
<td>( \ln(EC) )-PHRSTS (_2)                                0.4872</td>
<td>6.6504</td>
<td>-0.2017</td>
<td>-72.74</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )2.ar1.m3</td>
<td>( \ln(EC) )-Month+PHRSTS (_2)                          0.4480</td>
<td>6.7673</td>
<td>-0.0137</td>
<td>-0.0490</td>
<td>-77.34</td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln(EC) )2.ar1.m4</td>
<td>( \ln(EC) )-Month+PHRSTS (_2)+Month*PHRSTS (_2) 0.4412</td>
<td>6.7537</td>
<td>-0.0120</td>
<td>0.5611</td>
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<td>-77.11</td>
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</tr>
</tbody>
</table>
### Table A8.3  Fitted models of ln(EC) accounting for one-year transitional effect

<table>
<thead>
<tr>
<th>Models</th>
<th>ar1</th>
<th>Intercept</th>
<th>Month</th>
<th>PHRSTS₂</th>
<th>Month*PHRSTS₂</th>
<th>AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Independent</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ln(EC)₂.i.m1</td>
<td>ln(EC)-Month</td>
<td>6.7802</td>
<td>-0.0157</td>
<td></td>
<td></td>
<td>-32.02</td>
</tr>
<tr>
<td>ln(EC)₂.i.m2</td>
<td>ln(EC)-PHRSTS₂</td>
<td>6.6529</td>
<td>-0.1977</td>
<td></td>
<td></td>
<td>-15.76</td>
</tr>
<tr>
<td>ln(EC)₂.i.m3</td>
<td>ln(EC)-Month+PHRSTS₂</td>
<td>6.7677</td>
<td>-0.0136</td>
<td>-0.0429</td>
<td></td>
<td>-30.78</td>
</tr>
<tr>
<td>ln(EC)₂.i.m4</td>
<td>ln(EC)-Month+PHRSTS₂+Month*PHRSTS₂</td>
<td>6.7578</td>
<td>-0.0124</td>
<td>0.4835</td>
<td></td>
<td>-31.72</td>
</tr>
<tr>
<td>AR(1)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ln(EC)₂.ar1.m1</td>
<td>ln(EC)-Month</td>
<td>0.4497</td>
<td>6.7815</td>
<td>-0.0161</td>
<td></td>
<td>-78.94</td>
</tr>
<tr>
<td>ln(EC)₂.ar1.m2</td>
<td>ln(EC)-PHRSTS₂</td>
<td>0.4872</td>
<td>6.6504</td>
<td>-0.2017</td>
<td></td>
<td>-72.74</td>
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<td>ln(EC)₂.ar1.m3</td>
<td>ln(EC)-Month+PHRSTS₂</td>
<td>0.4480</td>
<td>6.7673</td>
<td>-0.0137</td>
<td>-0.0490</td>
<td>-77.34</td>
</tr>
<tr>
<td>ln(EC)₂.ar1.m4</td>
<td>ln(EC)-Month+PHRSTS₂+Month*PHRSTS₂</td>
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<td>6.7537</td>
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<td>0.5911</td>
<td>-0.0332</td>
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</table>

### Table A8.4  Fitted models of ln(EC) accounting for two-year transitional effect

<table>
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<tr>
<th>Models</th>
<th>ar1</th>
<th>Intercept</th>
<th>Month</th>
<th>PHRSTS₄</th>
<th>Month*PHRSTS₄</th>
<th>AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Independent</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ln(EC)₄.i.m1</td>
<td>ln(EC)-Month</td>
<td>6.7802</td>
<td>-0.0157</td>
<td></td>
<td></td>
<td>-32.02</td>
</tr>
<tr>
<td>ln(EC)₄.i.m2</td>
<td>ln(EC)-PHRSTS₄</td>
<td>6.6609</td>
<td>-0.1909</td>
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<td></td>
<td>-16.36</td>
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<tr>
<td>ln(EC)₄.i.m2</td>
<td>ln(EC)-Month+PHRSTS₄</td>
<td>6.7722</td>
<td>-0.0142</td>
<td>-0.0267</td>
<td></td>
<td>-30.29</td>
</tr>
<tr>
<td>ln(EC)₄.i.m3</td>
<td>ln(EC)-Month+PHRSTS₄+Month*PHRSTS₄</td>
<td>6.7580</td>
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<td>0.3898</td>
<td>-0.0226</td>
<td>-31.39</td>
</tr>
<tr>
<td>AR(1)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ln(EC)₄.ar1.m1</td>
<td>ln(EC)-Month</td>
<td>0.4497</td>
<td>6.7815</td>
<td>-0.0161</td>
<td></td>
<td>-78.94</td>
</tr>
<tr>
<td>ln(EC)₄.ar1.m2</td>
<td>ln(EC)-PHRSTS₄</td>
<td>0.4844</td>
<td>6.6573</td>
<td>-0.1912</td>
<td></td>
<td>-72.42</td>
</tr>
<tr>
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<td>ln(EC)-Month+PHRSTS₄</td>
<td>0.4489</td>
<td>6.7741</td>
<td>-0.0147</td>
<td>-0.0250</td>
<td>-77.04</td>
</tr>
<tr>
<td>ln(EC)₄.ar1.m4</td>
<td>ln(EC)-Month+PHRSTS₄+Month*PHRSTS₄</td>
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<td>6.7550</td>
<td>-0.0122</td>
<td>0.4897</td>
<td>-0.0281</td>
</tr>
</tbody>
</table>
Appendices

Appendix 9  Parameters of theoretical models and their data availability

Following the parameters of theoretical model in Section 7.1, the required data for and their availability are discussed as below.

Estimates of the mine water generated

To apply to coal mines, the model is constructed with the prerequisite that mine water generation by an individual coal mine is clearly known. Theoretically, mine water generation can be estimated using the water balance principle. That is,

\[ v_{it} = I_{it} + R_{it} - E_{it} - U_{it} \]  \hspace{1cm} [A9.1]

where

- \( v_{it} \) = Volume of mine water generated by mine \( i \) on day \( t \) (ML)
- \( I_{it} \) = volume of groundwater inflow (ML)
- \( R_{it} \) = rainfall runoff accumulated in the pits (ML)
- \( U_{it} \) = on-site net water use (ML)
- \( E_{it} \) = evaporation (ML)

The estimation of the model parameters is explained.

Groundwater inflow \( (I_{it}) \)

Although the groundwater inflows are a constant by-product of mine activities, they vary from mine to mine due to the differences in mining geological conditions, and vary from time to time for the same mine due to the changes in depth and area of mining work over its lifetime. In general, the groundwater inflows to open-cut mines are less significant than those to the underground mines. AGC Woodward-Clyde (1992) estimated that on average, the inflow rate for the open-cut mines is less than 1 ML/day, and is up to 4 ML/day for the underground mines. These figures can be used to estimate the groundwater inflows for individual PHRSTS dischargers.

Rainfall runoff \( (R_{it}) \)

The ability of a surface to generate runoff is often referred to as that surface’s runoff potential. This potential is complex and depends on several factors ranging from the nature of the surface, its porosity and vegetative cover, to the spatial distribution of rainfall taking into accounting existing moisture conditions. However, in simpler applications where overall potential for runoff is evaluated rather than specific short duration events, this potential can be described in
terms of a runoff coefficient, expressing the fraction of rainfall that will run off. The runoff coefficient is a number between 0 and 1.

In a fixed time frame, the volume of rainfall runoff accumulated in working pits is a function of rainfall intensity, the areas of the mine catchments and their corresponding runoff coefficients, given by the following simple linear relationship for any given time period (like one day):

$$ R = rc \times r \times B $$

where

- $R$ = runoff (ML)
- $rc$ = runoff coefficient
- $r$ = rainfall intensity (mm)
- $B$ = area of mine catchments (km$^2$)

Rainfall stations with long-term records going back at least 100 years occur throughout the Hunter Valley. The choice of rainfall stations that can represent the rainfall conditions of the PHRSTS coal mines is based on the literature, and some coal mines’ Environmental Impact Statements (EIS) and environmental reports. The relevant stations for the coal mines in different river sectors as defined in the PHRSTS is shown in Table A9.1.

<table>
<thead>
<tr>
<th>River sector</th>
<th>River flow and EC gauging stations</th>
<th>Meteorological stations</th>
<th>Years of record available</th>
</tr>
</thead>
<tbody>
<tr>
<td>Upper</td>
<td>Denman</td>
<td>Muswellbrook</td>
<td>1990-2003</td>
</tr>
<tr>
<td>Middle</td>
<td>Glennies/Hunter</td>
<td>Jerrys Plains</td>
<td>1990-2003</td>
</tr>
<tr>
<td>Lower</td>
<td>Singleton</td>
<td>Singleton</td>
<td>1991-2003</td>
</tr>
</tbody>
</table>

For some new PHRSTS coal mines, the runoff coefficients can be obtained from their Environmental Impact Statements. Where site-specific data are unavailable, the representative coefficients that were used for mine water balance studies by the AGC Woodward-Clyde (1992) can be borrowed. These representative coefficients are for typical mine areas when long-term rainfall data are considered. However, the lack of data on mine area distributions (that is, disturbed and undisturbed areas) even in an indicative way makes it impossible to estimate the rainfall runoff for individual mines. Hence estimating on-site mine water generated for individual PHRSTS dischargers is also impossible.
Evaporation ($E_a$)

The evaporation data represent what is known as pan evaporation. To convert to the actual evaporation rate taking place in mining catchments, they must be multiplied by the so-called pan factor, which in the Hunter Valley is 0.8 (AGC Woodward-Clyde 1992). There are no evaporation data for the above meteorological stations mentioned above. Although the evaporation data at nearby stations can be borrowed, they should be used with caution since doing so may introduce a wide range of errors and inaccuracy.

On-site net water use ($U_{it}$)

Mine water is used as one of the priority water sources for coal washing and dust suppression. Since the water for coal washing is recyclable, only the net water use is of concern. The net water use for coal washing depends on the tailings production. The amount of water used for dust suppression depends upon the areas of haulage roads and stockpile or/and underground mining workings, and fluctuates with climatic conditions.

Compared with the amount of water used for coal washing, the amount used for dust suppression accounts for a small portion and can be ignored. Estimates of tailings production for individual PHRSTS coal mines can be made from the differences between their saleable and raw coal production. The long series of saleable and raw production data going back to the 1980s has been well documented in the NSW Coal Profile (NSW DMR 1983; 1994; 1995; 1996; 1997; 1998; 1999; 2000; 2001; 2002). AGC Woodward-Clyde (1992) estimated that the overall moisture rate of tailing production is in the range of 4%-5%. These, together with the estimates of tailing production, allow estimates of net use of mine water on-sites using the following formula:

$$U = (RC - SR) \times MR$$

[A9.3]

$U$ = on-site net water use for a mine (ML/day)
RC = raw coal production (tonnes/day)
SR = average daily saleable coal production (tonnes/day)
MR = moisture rate of tailing production

This all means that, for any mine, the generation of mine water ($v_{it}$) is impossible to estimate to any useful degree.
Appendix 10 Approaches to estimation of marginal abatement cost functions for individual firms

To estimate the size of the cost saving of PHRSTS over other alternative river quality management schemes, it is necessary to estimate marginal control cost functions for the firms. These functions can be used to calculate the least-cost solution for achieving an aggregate level of discharge, as well as the expected costs of alternative schemes.

An abatement cost function describes the cost of pollution abatement activities. In general, there are a number of options open to firms to reduce their pollution, including end-of-pipe clean-up, changing production processes, and curtailing production. The costs of the inputs that have to be devoted to reducing each additional unit of pollution are measured by marginal abatement cost functions/curves (MACs). The position and slope of the MAC function are affected by factors such as the scale and sectoral composition of production; the average operating efficiency of the firm; the available process technologies; and the efficiency of treatment technologies (Dasgupta et al. 1996). There are a number of approaches to constructing plant-level MAC functions, including the survey approach; the engineering model-based approach; the cost function approach; the total factor productivity approach; and the distance function approach.

The survey is one of the standard methods used to estimate the costs associated with pollution reduction. This approach usually asks target firms to estimate how much their abatement costs were/would be corresponding to various pollution reduction levels. One of the examples is the cost estimate for BOD reduction for the UK Forth Estuary (Hanley et al. 1998). Although the survey approach can reflect the real world to a great extent, there are several factors that could affect its accuracy. On one hand, firms are generally prone to overestimate the abatement costs to show their efforts in compliance with environmental objectives. On the other hand, there is the potential to underestimate because of the difficulty in estimating the increase of capital expenditure embedded in production processes caused by abatement activities. It is also often very difficult to get a large, representative sample if answering the survey is voluntary.

The engineering model-based approach may be the most common one for estimating the costs associated with end-of-pipe control options. However, it bears several significant constraints as follows: firstly, where one end-of-pipe measure reduces more than one pollutant (which is common in practice), it is very difficult or even impossible to isolate the abatement cost...
associated with one particular pollutant from the costs of another. In such circumstances, the abatement cost of a particular pollutant is often overestimated. Secondly, engineering models restrict the set of existing abatement options to end-of-pipe controls. However, for each pollutant there are many potential pollution reduction techniques. It is impossible to identify all the options in order to determine the best one. Furthermore, no interdependencies among the options are allowed. All options are assumed to be separate with respect to abatement costs as well as the amount of pollutant reduction, which may not apply to the real world. Finally, failure to use behavioural data could lead to large estimation errors.

The cost function approach is frequently employed in empirical studies of MACs. Usually, a cost function of pollution abatement, which relates abatement costs to the treatment volume, the characteristics of influent and effluent stream and the prices of input used in the pollution abatement activity, is pre-established. Then a set of pooled data is used to fit the abatement cost function econometrically, to identify the importance of variables and estimate their parameters. Using the cost function approach, Dasgupta et al. (1996) estimated a joint abatement cost function for China's major industrial water pollutants with the aim of evaluating the economic efficiency of current regulations and assessing the impacts of the emission charge system used in China. In a similar way, Goldar et al. (2001) derived a form of abatement cost function from the production function associated with pollution abatement activities, to estimate the MACs of small-scale factories in an industrial estate in India. In this study, they defined the output of a pollution treatment plant as the reduction in pollution load. Moreover, they assumed that plants minimise the total treatment cost under the given prices of inputs, the volume of waste water, and the reduction in the concentration levels of pollution load.

Traditionally, the engineering model and cost function approaches have been based on plants' reports on the direct costs of installing and operating pollution control equipment to estimate pollution abatement costs. However, as presented above, firms can adjust to the threat of higher pollution-related costs along many dimensions, such as introducing new processing technologies and/or improving working efficiency. Thus, the above approaches may provide incomplete measures of the costs of pollution reduction.

In order to overcome shortcomings of this kind, the total factor productivity approach has been developed to estimate a total cost function which combines both production and pollution control costs. The marginal cost of pollution abatement can be derived by taking derivation of
the total cost with respect to a specific pollutant. This marginal cost estimation takes into consideration pollution reduction in both the production process and the end-of-pipe treatment (Wang 2000). Though conceptually appealing, this method has rarely been used in empirical studies because of its intensive requirement for data.

Wang (2000) attempted to use a demand function approach to estimate MACs of China’s industrial organic water pollution (for example COD). Firstly, using a plant-level dataset with detailed information on discharge fees collection from 1500 industrial firms in 1994, Wang constructed a discharge function which related the total amount of a pollutant discharge to such variables as discharge fees, firm characteristics, input prices and community characteristics. Then the discharge levy can be obtained through the inverse of the demand function. Secondly, in China’s current discharge levy system, the total cost of pollution discharge to a firm is the sum of the costs of pollution abatement activities (including the expenditure on production process and the pollution treatment facilities) and the payment of the fees of actual pollution discharge. Under the assumption that firms are minimising the total cost of pollution discharge, it is plausible to infer that a firm abates its pollution up to the point where the MAC equals the marginal discharge levy of pollution discharge. Therefore, a MAC function can be obtained from the inverse of the demand function. Like the total factor productivity approach, this approach can provide a relatively complete estimation of industrial firms’ pollution abatement costs because it takes both the production process and end-of-pipe treatment into account. Nevertheless, the demand function approach of Wang’s study relies on information about prices of pollution discharges, which seems to limit its wider application.

All the methods presented above assume that pollution abatement activities are separable from the activities associated with producing the marketed output/production. The distance function approach, also known as the joint-production function approach or the shadow price approach, provides an alternative way in which to address this issue, and appears a promising approach in recent cost studies.

Duality theory offers the basis for the application of the distance function approach in estimating marginal costs of pollution abatement activities. The idea of deriving shadow prices using distance functions and duality theory originated from Shephard (1970 cited in Fare and Primont 1995). The underlying concept of this approach is treating pollutants as one type of output, that is, undesirable or bad outputs. These undesirable outputs are distinguished from desirable or good outputs in a number of ways, such as disposability and negative price. Once
pollutants are treated as outputs, the opportunity costs of pollution abatement activities, that is, MAC, can be measured by the foregone production of the desirable outputs that is caused by re-allocation of inputs from producing the desirable outputs to pollutant abatement activities (Fare et al. 1993).

There are at least two advantages in making distance function superior to other MAC estimate approaches. First, distance function does not require detailed information about pollution abatement technologies and their associated costs. Instead, the total cost of pollution abatement activities is measured by the reduced production of the good output due to the decrease in the bad output (that is, the increase in the reduction of the bad output). Second, this approach can capture the whole actual cost of pollution abatement activities, which may be the joint costs of the abatement processes of two or more pollutants, and/or the joint costs associated with both 'end-of-pipe' techniques and 'changes in the production process' techniques (Fare et al. 1993). Nevertheless, the last point may be a drawback as well, because it is very hard to separate the costs associated with a particular pollutant from the joint costs of all pollution abatement activities. Even so, the outstanding merits and modest data requirements discussed above may partly explain the increase in employing distance functions to estimate MAC of pollution control activities in empirical studies in during past decade.

Abatement costs can be estimated based on either an output distance function or an input distance function. The abatement costs based on an output distance function account for the cost of all pollution abatement methods used by the firms to meet the prescribed standards, but firms have resource constraints such that they have to reduce pollution at the cost of reducing the production of good outputs. In contrast, abatement costs based on an input distance function place no restriction on the source of funding the abatement activities of firms. This characterisation explains the fact that firms can achieve their pollution abatement through the use of additional input while still increasing or maintaining the desirable outputs (Murty et al. 2001).

Fare et al. (1993) produced the first study deriving the shadow prices of undesirable output using the output distance function. Thereafter, there have been a number of studies applying the Fare et al. (1993) methodology to estimate pollution abatement costs. For instance, Coggins and Swinton (1996) estimated the MACs of reducing sulfur dioxide emissions by 14 power plants in Wisconsin using data from 1990 to 1992. Also, Carlson et al. (2000) estimated the MACs faced
Appendices

by electric utilities in the reduction of sulfur dioxide emissions with the purpose of estimating the gains from emission trading against command-and-control regulation. Fare and Jr (2002) have also used the distance function approach to estimate pollution abatement costs incurred by US power plants, aiming to identify the extent of any divergence between the survey and modelling estimates and to investigate the sources of the divergence. Other studies used an input distance function to estimate abatement costs. For example, Hailu and Veeman (2000) used a similar approach to the Canadian pulp and paper industry; Murty et al. (2001) applied this to the sugar industry in India; and Rao and Kumar (2002) measured the productive efficiency of water pollution in India.

For the research presented here, the distance function approach was originally proposed to estimate the MACs of the PHRSTS members in saline water control. Identifying production inputs and outputs is the first and most important step of the distance function approach, and a good understanding of the production processes of PHRSTS members is a prerequisite for such identification. Through reviewing the available annual reports, environmental protection licences and other relevant documents, the main inputs and outputs of coal mines and power stations in the Hunter Valley were identified, as presented in Tables A10.1 and A10.2.

Table A10.1  Production inputs and outputs of coal mines

<table>
<thead>
<tr>
<th>Category</th>
<th>Items</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inputs</td>
<td>Annual capital inputs</td>
</tr>
<tr>
<td></td>
<td>Capital depreciation</td>
</tr>
<tr>
<td></td>
<td>Energy (electricity and fuel)</td>
</tr>
<tr>
<td></td>
<td>Labour (number of employee)</td>
</tr>
<tr>
<td>Desirable outputs</td>
<td>Saleable coal</td>
</tr>
<tr>
<td>Regulated undesirable salt outputs (Pollutants)</td>
<td>Salt load</td>
</tr>
<tr>
<td></td>
<td>Fluoride compounds</td>
</tr>
<tr>
<td></td>
<td>NO$_2$</td>
</tr>
<tr>
<td></td>
<td>Particulate Matter</td>
</tr>
<tr>
<td></td>
<td>CO</td>
</tr>
</tbody>
</table>
Once the production inputs and outputs have been identified, the next step is to collect the data required. After an intensive information search in publicly available and accessible sources, such as annual reports published on websites, I found that such crucial information as capital depreciation, energy and coal consumption is lacking, which suggested the necessity of a survey. A survey was conducted from November 2003 to February 2004 through questionnaires, and with the help of the NSW Minerals Council. The survey targeted all PHRSTS participants, which included 19 coal mines and two power generators. The main information sought was:

- Production inputs and outputs: producing material, number of employees, labour costs, output production, market price per unit of output, production revenue;

- Water intake and saline water discharge: annual volume of water intake from the Hunter River, conditions set in the discharge licences, and volume and concentration of annual saline water discharged to the Hunter River;

- Saline water control costs: saline water control activities that occurred in each firm during 1995-2002 and their associated costs; cost estimations of achieving hypothetical salt load

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**Table A10.2 Production inputs and outputs of power stations**

<table>
<thead>
<tr>
<th>Category</th>
<th>Items</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Inputs</strong></td>
<td>Ignition oil usage</td>
</tr>
<tr>
<td></td>
<td>Coal consumption</td>
</tr>
<tr>
<td></td>
<td>Biomass</td>
</tr>
<tr>
<td></td>
<td>Annual capital depreciation</td>
</tr>
<tr>
<td></td>
<td>Labour (number of employee)</td>
</tr>
<tr>
<td><strong>Desirable outputs</strong></td>
<td>Electricity</td>
</tr>
<tr>
<td><strong>Saleable by-production</strong></td>
<td>Fly ash</td>
</tr>
<tr>
<td></td>
<td>Bottom ash</td>
</tr>
<tr>
<td></td>
<td>Lime</td>
</tr>
<tr>
<td></td>
<td>Gypsum</td>
</tr>
<tr>
<td><strong>Regulated undesirable outputs</strong></td>
<td>Salt load</td>
</tr>
<tr>
<td>(Pollutants)</td>
<td>SO₂</td>
</tr>
<tr>
<td></td>
<td>NO₂</td>
</tr>
<tr>
<td></td>
<td>CO</td>
</tr>
<tr>
<td></td>
<td>Particulate matter</td>
</tr>
</tbody>
</table>
reduction targets, which are expressed as percentages (for example, 10%, 25%, 50% and 80% reduction of the salt load in 2002).

Unfortunately, the survey elicited few responses. At least two factors could account for the PHRSTS members not responding to the questionnaire. First, since the information the survey requested may not necessarily be readily available, the survey would impose an extra working burden on the participants for reviewing the instructions, searching existing data sources, gathering the data needed, and completing and reviewing the questionnaires. In some cases, completing the survey may have required the joint efforts of firms' financial and environmental staff. The firms may have been reluctant or could not afford the time and effort to do this. Second, completing the survey may expose the firms to the risk of disclosure of their commercially confidential information. In particular, the PHRSTS salt credits auction was scheduled for April 2004. The disclosure of information about the costs of saline water control could have been extremely sensitive for the PHRSTS members as rivals in the auction. This situation prevented this research from estimating the MACs by a distance function approach.
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289


