3. The nature of the salinity problem in the Hunter River

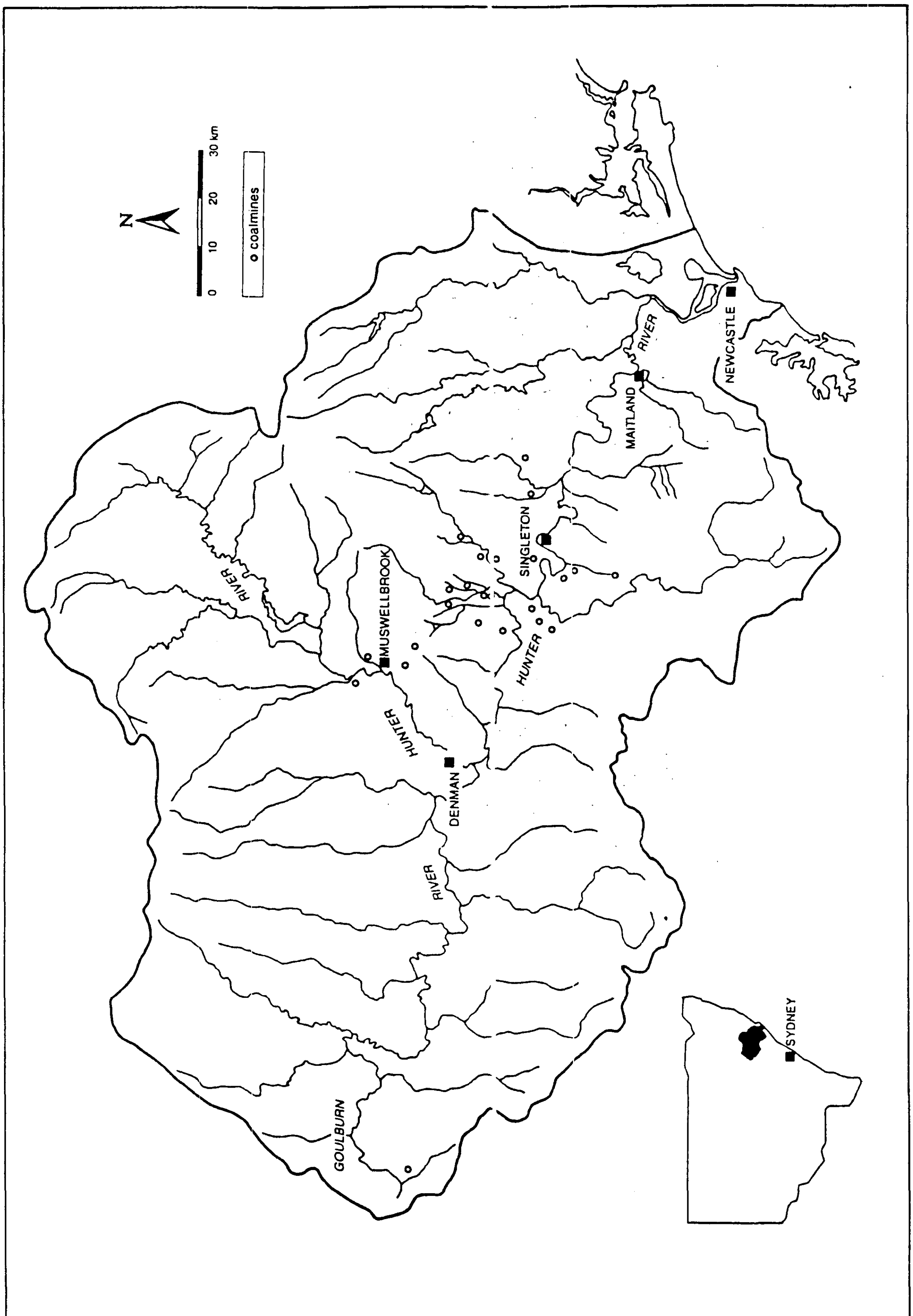
3.1 Introduction

This chapter provides background information on the location of the Hunter River System and the nature of the salinity problem. The problem of saline discharges from coal mines is reviewed and alternative methods of disposing of saline minewater presented. The final section of the chapter documents the policies which have been implemented in the past to control saline discharges from coal mines, and outlines the transferable discharge permit scheme proposed in 1994, which is currently being implemented.

3.2 Location of the study

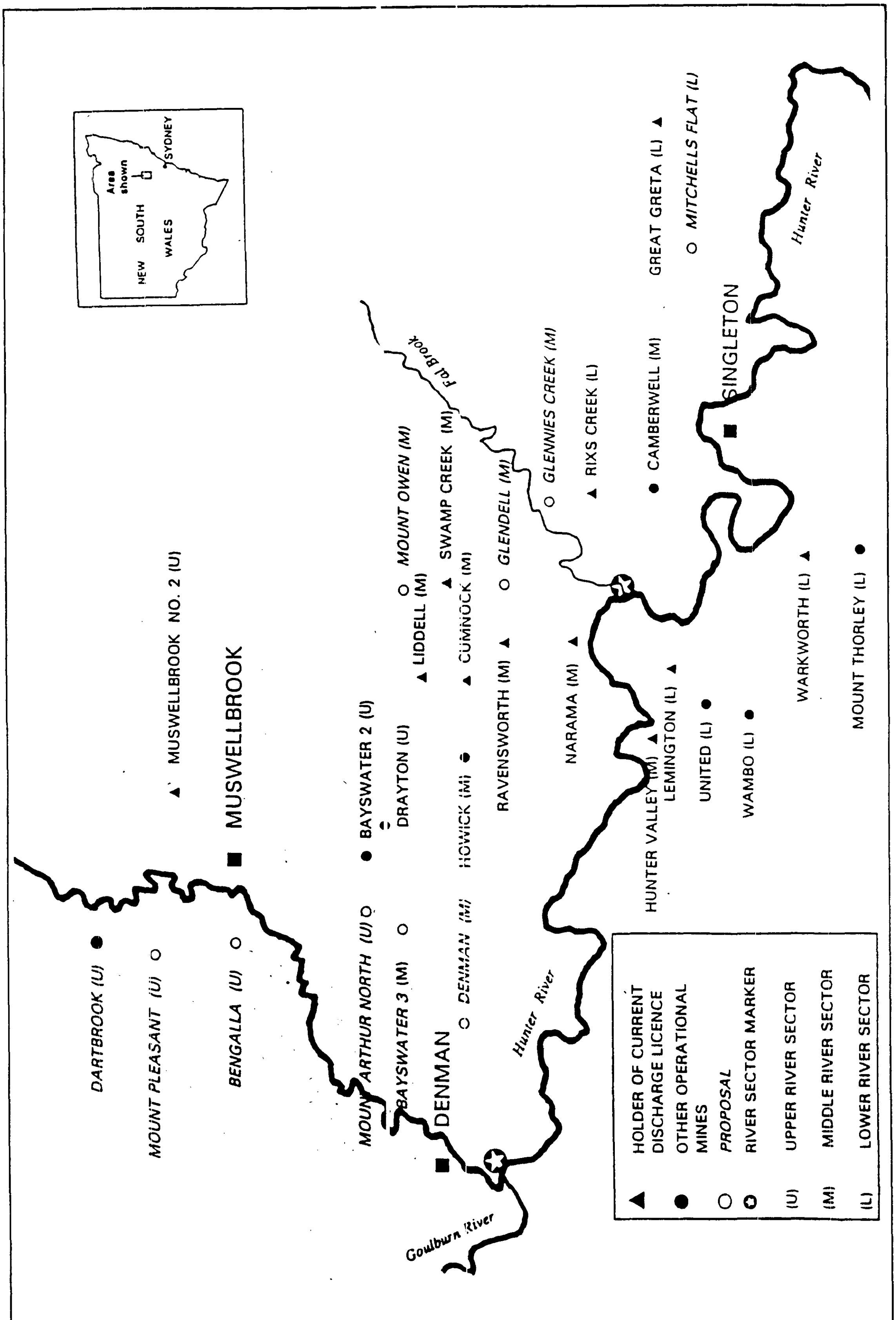
The Hunter Valley is located on the central coast of New South Wales and covers an area of about 22000 square kilometres. The Great dividing range forms the boundary to the north-west and west, while the southern boundary is a sandstone plateau. The city of Newcastle marks the mouth of the Hunter River, and is about 140 kilometres north of Sydney. The Upper Hunter Valley referred to in this study is upstream of Singleton, covering some 17000 square kilometres and includes the Goulburn River (see Figure 1.1) Coal mining activity is generally confined to the central area, between Singleton and Muswellbrook. The location of existing and proposed mines are shown in Figure 1.2. Mines that held discharge licences in 1994 are marked with a triangle.

The tax policy formulated in this study is applied to the mines in one reach of the river only. This reach is bounded by the Goulburn River junction and Singleton.



Source: EPA, 1994a

Figure 3.1 Location and boundary of the Hunter River system



Source: Adapted from EPA, 1994b

Figure 3.2 Location of the coal mines in the Upper Hunter Valley, NSW

3.3 Geology and the source of salt

One of the main factors governing salinity in the Hunter River and its tributaries is the geology of the catchment (PPK 1994). A large proportion of the catchment of the Hunter River contains saline sedimentary rocks. Consequently, surface water and underground drainage of the catchment contribute a natural "base load" of salinity to the river (DWR 1994).

The main groundwater aquifers occur in the coal seams. Under low flow conditions the river level is below the water table, and groundwater discharges into the river as baseflow. Prior to the construction of Glenbawn Dam, low flow conditions used to consist almost entirely of baseflow. Now, the low flow conditions tend to be entirely regulated with releases from the dams for irrigation purposes (AGC 1993). Consequently, the discharge control policy must be tailored to account for the low flow conditions where not only is the river assimilative capacity low, but the limited water is used for irrigation and must be of an acceptable water quality. Concentrations of salt are measured in units of electrical conductivity (EC), with high electrical conductivity readings indicating high concentrations of salt. A salinity reading of 700 EC (420 mg/L) is considered to represent an acceptable concentration of salt, because most agricultural crops, stock and municipal uses can tolerate salt at this level. Note that three units of measurement for salinity are commonly used, EC, µS/cm, and mg/L. EC and µS/cm are equivalent measures and can be used interchangeably, whereas the relationship between mg/L and EC is more complex. For the Hunter River, a suggested conversion ratio to convert EC to mg/L is 0.6 (EPA 1994a). This conversion ratio is appropriate for ranges of salinity up to 4000 mg/L.

Salinity in the Hunter River and its tributaries is wide ranging depending on the origin of the rocks forming the catchment and the aquifers which drain to the river. Some have good quality water with salinities less than 1000 mg/L while others, in the sediments of marine origin, with connate salt have extremely high salinities up to 17000 mg/L. Median salinities up to 7000 mg/L occur in the Greta coal sequences to the east of Singleton, and in the coal sequences to the east of Muswellbrook. These salinities can be contaminated by overlying and underlying marine strata (AGC 1984).

In general, the salinity of the river varies inversely with streamflow. Under high flow conditions, the salinity is generally lower, but as flow declines and baseflow becomes more important the water quality decreases (AGC 1992). River salinity also varies with distance downstream. As the river drains from Muswellbrook to Greta, the salts tend to accumulate, and salinity generally increases. Using salinity data from three key sites along the Hunter River over the past 15 years, AGC Woodward-Clyde (1992) observe that salinity readings are usually well below 420 mg/L upstream of Muswellbrook, except under extreme low-flow conditions. Downstream of Muswellbrook, salinity levels at Liddell, are greater than 420 mg/L 50 per cent of the time, and by the time the river reaches Greta, salinity levels over 600 mg/L EC are common.

3.4 Relative impact of human activities on salinity in the Hunter River

Salinity levels are also influenced by human activities, including farming, coal mining, power generation, industry and urbanisation. These activities increase the rate of mobilisation of salts from the soil profile to the river system. It is extremely difficult to find estimates of the relative contribution that activities have on the salt load in the river, however, one unsourced estimate quoted in EPA (1994a), reports that the natural salinity accounts for 80 per cent of the salt load. For the remaining 20 per cent it suggests that irrigation contributes more salt than the coal mines.

The relative impact of each activity will depend on the flow conditions of the river, as is shown in reports prepared for the New South Wales Coal Association by Croft and Associates (1983) and AGC Woodward-Clyde (1992). Croft and Associates (1983) estimate that the percentage increase in river salinity at Singleton as a consequence of the discharge of saline minewater ranges from 0.2 per cent during high river flows (the 10 percentile flow rate) to 34.4 per cent during low river flows (the 90 percentile flow rate). The report also estimates that if 3700 megalitres (ML) of saline minewater is discharged per annum, mining would contribute only 20 per cent of the salt in the river system. This appears to be a gross underestimate of the current situation in the Hunter. A later report by AGC Woodward-Clyde (1992), shows an expected mine discharge of saline water for 1995 to be 13990 ML per annum, with an average concentration of around 3100 mg/L and not the 1000 to 1500 mg/L used by Croft and Associates (1983).

Using these figures AGC Woodward-Clyde (1992) estimate that mines will raise the river salinity by 1.2 per cent during high flows and 67 per cent during low flows. This is quite a significant impact on the river, and with mining in the Hunter set to expand, this is likely to increase still further.

3.5 Coal mines and salinity

Minewater is often referred to as "clean water" or "dirty water". "Clean water" is water which has not been in contact with areas affected by mining, whereas "dirty water" has (AGC Woodward-Clyde 1992). "Clean water" is generally not subject to regulator control.

Three categories of "dirty water" are identified in AGC Woodward-Clyde (1992)

i) runoff from disturbed and rehabilitated areas, which usually only requires removal of suspended solids prior to release. This is done by collecting it in sedimentation dams which are allowed to overflow;

ii) mine water, incorporating ground water seepage and rainfall runoff collected in the pit or underground workings and which is typically saline; and

iii) water originating from industrial areas of mine sites.

There is a limited demand for water on a mine site. The major uses are in washing coal and for dust suppression. Dust suppression is necessary on both haul roads and stockpiles and in underground workings. Water is also lost through evaporation when large tailings dams are present. In many cases the volumes of dirty water generated on site are larger than the amount that can be used on site, and this presents an environmental problem. Hereafter, the excess "dirty water" will be referred to as minewater excess or total minewater.

Groundwater inflows to open cut coal mines is generally less than 1 ML/day and is associated with major rock fractures occurring along regional geological structures. For many pits the groundwater inflow is insignificant. Underground mines can produce up to 4 ML/day but generally the larger inflows occur due to fractured rock connecting the mine to surface storages or to the Hunter River.

It is important to note that the areas of disturbed land contributing the surface water component of dirty water are fixed as an inherent component of the mines' production capacity. This is not adjustable in short term changes in production level.

The salinity of the minewater depends on its source, but they can range from 2000 mg/L to 4250 mg/L. The mean salinity of discharge of all operating mines is 2870 mg/L. These values were obtained from survey results conducted by AGC Woodward-Clyde (1992).

The storage capacity for 9 mines as supplied by New South Wales Coal Association (AGC Woodward-Clyde 1992a) gives a total of 1041 ML from undisturbed land, 2131 ML from disturbed land, and 3207 ML from pit water. The nine mines are located in the main coal producing area targeted in this study.

The general capacity for additional storages appears to be limited. Reasons given by AGC Woodward-Clyde (1992) for this include:-

- many sites are relatively flat
- many sites have very little land available for the purpose of storing excess minewater,
- mine layout and plan are reasonably fixed;
- in some cases the original lease has already been mined and extensions and/or alternative developments are being considered.

3.6 Possible methods of avoiding discharge of mine waste water to the river

Salt abatement options in the Hunter River valley include desalination, piping the saline minewater to the ocean, evaporation and in-pit burial, and deep well injections. Croft and Associates (1983) reviewed each of these options for the coal mines of the Upper Hunter Valley and found desalination to be the best alternative a brief summary of their findings is included in the following.

3.6.1 Transport by pipeline

The construction of a pipeline to transport saline minewater from mines in the Upper Hunter Valley to the estuarine reaches of the river, was estimated to cost in the range of

$15 million to $25 million (1983 dollars). Costs depend to a large extent on the pipe diameter, with costs for the main trunkline and the necessary intake and booster pumps ranging from $9 million to $17 million. The remaining capital costs were expected to cover the cost of connecting subsidiary mines to the main trunkline. Even so, the largest pipeline considered was 300 mm in diameter, and this was expected to have an annual capacity of 3560 ML. In 1983, this may have been sufficient to service the existing mines, but would only accommodate half of the minewater excess produced currently. Operating costs would not vary with salt load, so a pipeline would have to consider a costing structure that charged mine operators for the volume of water pumped.

3.6.2 Deepwell injections

Deepwell injections dispose of saline minewater by injecting the water into aquifers under high pressure through boreholes ranging from 150 to 300mm in diameter. This method has been used to dispose of saline water where suitable permeable aquifers underlie a site, and where no unacceptable contamination of the aquifers would occur. In the Hunter, two main forms of aquifer include an unconfined aquifer contained in the river alluvium, which is linked directly to the river, and a confined aquifer in the Permian coal sequence. The first aquifer is unsuited to disposal of salt because the saline water would enter the river directly, with less control over accession rates than surface discharge. The second aquifer would contain the saline minewater, but was found unsuitable because of low permeabilities. In the confined aquifers occurring in the Permian coal seams, the low permeability means that an extremely high number of boreholes would be required (up to an estimated 7000 for an average year), together with extremely high pressure required to penetrate the aquifer. The cost of the method under these circumstances was considered too large to be investigated further. This method may, however, present an option for disposing of salt concentrates derived from either evaporation basins or brine concentrates from desalination plants, where the volume is considerably reduced.

3.6.3 Evaporation and in-pit burial

Evaporation rates in the Hunter are considered high enough to support the use of evaporating basins, however, the biggest obstacle to their use is the substantial land areas required. For a large opencut mine relying solely on evaporation ponds, 100 hectares or more of land for ponds and drying beds would be required. An alternative method of evaporation is considered where water is sprayed onto inactive parts of the pit and permitted to evaporate. Residual salt is then collected in the mine and buried as the pit advances. In pit spraying is generally not considered to be able to cope with all minewater excess, and if the final salt is buried too close to the surface it could hinder rehabilitation processes and remain a potential source of salt accessions to the river.

3.6.4 Desalination

There are a number of desalination methods available. These include, reverse osmosis, electrodialysis and ion exchange, distillation and freezing. Of all these methodologies, Croft and Associates (1983) concluded that reverse osmosis was the least-cost method of removing salt from brackish water like that in the Hunter coal mines. Briefly, reverse osmosis involves the movement of pure water molecules from brackish water through a semipermeable membrane into pure water. The semi-permeable membrane does not allow diffusion of the salt molecules. Osmosis is a natural process which occurs due to concentration differences. In order to reverse the osmosis a pressure needs to be exerted on the salt solution which exceeds the natural osmotic pressure. Minewater would require some pre-treatment to avoid fouling and precipitation by sediments and salts. The pH of the water used in the reverse osmosis plant must be monitored carefully and, in the Hunter, considerable expense is involved in lowering the relatively high pH.

The cost savings of reverse osmosis for brackish water show that capital costs were about 20 per cent of those for distillation. For desalination of seawater, however, there is little difference in capital costs because the pressure difference required in the reverse osmosis plant is so much higher due to the increased osmotic pressure that the higher salinity of seawater exerts, and which needs to be counteracted in the reverse osmosis plant. The operating and maintenance costs of reverse osmosis were also found to be considerably less (approximately 50 per cent) than alternative distillation methods. The fact that Pacific Power invested in a reverse osmosis desalination plant in 1985, at a capital cost of $15 million, lends support to the general finding that both operating and capital costs are lower for reverse osmosis plants used in desalinating water in the Hunter. The plant capacity allows removal of 80T of salt per day, approximately 30 ML per day when salinity is 2500mg/L. Desalination plants concentrate the levels of salt in the wastewater, such that highly saline brine concentrates are the biproduct of pure water production. The storage and disposal of this concentrated brine remains a problem from this treatment technique, and is a problem shared by the evaporation methods.

3.7 Current and future coal mines in the Upper Hunter Valley

Measuring the number of coal mines in the Upper Hunter is somewhat imprecise, since what one report counts as a coal mine in the Upper Hunter another seems to ignore. In general it seems fair to claim that there are currently 19 operating coal mines and approximately 11 of these need to discharge excess minewater periodically through the year (DWR 1994). While some of the more modern mines have "nil offsite discharge", this is not possible with all mines in the area.

In the future, mining in the Hunter is set to expand. The New South Wales Coal Industry profile identifies 17 operating mines and 10 proposed mines as at November 1994. The production of coal is projected to increase from 37.5 Mt of saleable coal in 1992-93 to roughly 46 Mtpa by 2000 (Coal Resources Development Committee 1994).

3.8 History of regulation of the discharge of mine wastewater to the Hunter River

Mines in the Hunter Valley who find it difficult to dispose of their excess water, are licenced by the EPA to dispose of their excess water either by irrigation or by controlled discharges to streams. The EPA's licensing powers do not force potential water polluters to hold a licence in order to operate (as is the case for industries with the potential to cause significant air or noise pollution). Although potential water polluters do not have to hold a licence, if they pollute waters without one they commit an offence under section 16(1) of the Clean Waters Act 1970. This allows the EPA to place conditions on the discharge to minimise its environmental impact (EPA 1994b).

Licences in the Hunter Valley were usually granted for one year, but in certain circumstances the EPA would issue short term licences. Special conditions could be attached to any licence, and applicants for licences have the right of appeal in the Land

and Environment Court over any decision relating to issue or conditions attached to the licence.

The situation at July 1994, was that 11 coal mines were licenced. The licences specified two conditions: (1) a maximum allowable level of conductivity in the river of 700 EC (420 mg/L) after the discharge, and (2) a maximum allowable increase of 40 EC (24 mg/L) in the conductivity of the river caused by the discharge (EPA 1994b). This form of discharge has been termed 'trickle' discharge.

The EPA is proposing to introduce a change to licensing where sources would generally be allowed to discharge at times when flows in the river are relatively high, and demands by and impacts on other users are relatively low, moreover, the change would allow transfer of discharge permits among themselves (subject to certain conditions)

The new features of the proposed scheme are summarised from EPA (1994b, p5), below.

- phasing-out of discharges under low-flow conditions by 31 December 1999.
- during high-flow conditions, there will be no limit on the allowable increase in conductivity caused by an individual discharge,
- each discharge source will be entitled to discharge a specified percentage of the total allowable salt load. This allowed discharge is termed proportional discharge credit. The total allowable salt load is the amount of salt that may be discharged collectively by all sources without exceeding the designated in-stream salinity levels at any point in time.
- a source can trade discharge credits with other sources.
- no new trickle licences will be issued
- above the Goulburn River junction, under low flow conditions, the new receiving water conductivity threshold would be 500 EC (300 mg/L) rather than the previous level of 700 EC (420 mg/L).

Three receptor points are proposed; the Goulburn River junction, the Fal Brook (Glennies Creek) junction, and Singleton. Water salinity objectives would be set at each receptor point, since each receptor defines the lower boundary to three associated river sectors. Trigger values at each receptor point must be satisfied for low-flow, high-flow and flood conditions to occur (Table 3.1).

Table 3.1 Trigger flows at each receptor for low-flow, high-flow and flood conditions

	Trigger streamflows (in megalitres per day (ML/day))		
	low-flow conditions	high-flow conditions	flood conditions
Goulburn R.	< 600	600-2000	> 2000
Fal Brook	<1800	1800-6000	> 6000
Singleton	<3000	3000-10000	>10000

Source:EPA, 1994b

Under high-flow conditions, sources would be allowed to discharge saline water so long as the conductivity in the river did not exceed 600 EC (360 mg/L) at the Goulburn River junction, 900 EC (540 mg/L) and the Fal Brook junction, or 900 EC (540 mg/L) at Singleton.

Provision is made not to give the benefits of increased water quality due to releases from the DWR water storages to permit holders. When high flows result from specific regulated flows, the licence/discharge entitlement are no longer valid. The sources who paid for the release from the reservoir are able to determine which sources can make use of the release and in what proportion. Under flood conditions all sources are permitted to discharge without limits on volume or salinity, unless otherwise directed by the District State Emergency Services controller.

Trades that result in the transfer of credits upstream could result in in-stream conductivity standards being exceeded. To prevent this, it is suggested that limits be established on the amount of credits that could be accumulated within the upper and middle sectors. Limits of 7.3 per cent of credits for the upper sector, and 53.7 per cent of credits in the middle sector have been proposed.

Other methods of reducing the discharge from mines which were considered by EPA include load based taxes and various staged discharge schemes. Load based taxes, are difficult to set efficiently to meet the environmental standard when the system is highly variable and stochastic. Once the tax rate is set, however, taxes are reasonably easy to implement and have the added attraction (for the regulating authority) of being able to generate revenue. Staged discharge schemes which aim to utilise the rivers natural assimilative capacity in combination with dilution flows released from Glenbawn Dam have also been considered. A trial was conducted early in 1993 by the Department of Water Resources for the Hunter Water Quality Task Group (DWR 1994), to investigate the operation of staged discharges It lasted for a period of one month and provided insights into the operation of a system of transferable discharge permits.

The proposed permit scheme outlined in this section is currently being introduced to the mines in the Hunter River system (G. Kaine, 1995, pers. comm.). Despite the preference for the permit scheme, the current study remains relevant for the insights it provides to setting taxes in dynamic stochastic environments and the implications for transferable discharge permit schemes.

3.9 Summary

The situation in the Hunter Valley consists of a salinity discharge problem from mines which, if left unchecked, would impact significantly on water users in the Hunter System. The current situation is already serious, with high salinity levels in the river and onsite mine storages at capacity. The outlook for the Hunter River is grim, with planned increases in coal production likely to cause river salinity levels to increase further.

Government intervention has been direct and regulatory in nature, with fines used to enforce a licence system. More sophisticated methods have been called for to control the saline discharges from mines in the Hunter. Currently a transferable discharge permit system is being introduced in the Hunter River. The taxes investigated in the following chapter serve to highlight the informational requirements and the likelihood of success for taxes in stochastic environments. The similarity in the cost-effectiveness of taxes and transferable discharge permits allow implications to be drawn from this research for use in the transferable permit scheme.

4. Research Methods

4.1 Introduction

This chapter has two major sub-sections. The first contains details of the development of the tax policy and the second presents the formulation of a stochastic dynamic simulation model. The tax policy is set along cost-effective policy guidelines, with an additional goal of reducing the tax costs. The model was developed using the software "STELLA II" (High Performance Systems 1994), and attempts to model the fluctuations in river assimilative capacity in order to set appropriate levels of tax which would achieve the environmental standard.

4.2 The development of a salt tax for saline minewater discharges to the Hunter River

The objectives of the tax formulated in this project are:

i) to meet the environmental standard and not violate it

ii) to utilise the assimilative capacity of the river where possible

iii) to achieve a cost-effective tax by using price control and never quantity control

iv) to reduce the tax costs

To efficiently set a pollution tax, the literature review in Chapter 2 has shown that the environmental standard together with the marginal abatement cost curve are essential. Additionally, the relationship between the pollutant measured in units of weight needs to be translated into a measure of pollution, that is, concentration. The pollution measure is dependent on the initial salt load and flow of the river (that is, the assimilative capacity) and the assimilative process itself - in this case dilution.

The paper by Jacobs and Casler (1979), which developed an effluent tax to reduce the payment by dischargers experienced under an efficiently set tax system, was critically reviewed in section 2.4.3. By avoiding the pitfalls displayed in the effluent tax of Jacobs and Casler, the tax policy developed here attempts to reduce the excess payment while at the same time not compromising the price control mechanism of taxes, which ensures their cost-effectiveness. It will be shown, however, that there are a number of assumptions and an extremely high requirement for information on the part of the regulating authority that cast serious doubt on the ability to formulate a fair and cost-effective tax.

There are four ways in which the tax developed here attempts to reduce the tax costs for dischargers. These features are introduced in section 4.5. In sections prior to this the MAC curve is examined, and a static flat tax discussed.

4.2.1 Environmental standard

The environmental standard has been chosen for this project on the existing EPA standard of 700 EC (420 mg/L). This level considered to be acceptable for human consumption, most irrigated crops, and the majority of instream uses, see AGC Woodward-Clyde (1992).

4.2.2 Marginal abatement cost curve for desalination using reverse osmosis

The marginal abatement cost curve used for the purpose of tax setting in this project, is based on the average variable cost of salt removal from the reverse osmosis plant at Pacific Power. A discussion of alternative salt abatement processes in Chapter 3 shows desalination to be the best option available for the specific characteristics of the Hunter River System. Although unwilling to give precise data, discussions with Jim Stewart (1995, pers. comm.) on the general shape of a likely marginal cost curve and an estimate of the average variable cost allowed a hypothetical marginal abatement cost curve to be drawn around a single point (see section 4.3. for further details). Since the curve is hypothetical, actual tax rates cannot be estimated. However, it does allow the tax to be generated showing the informational requirements needed for this tax, the volatility that can be expected, and the degree of success in meeting the standard that could be anticipated with this type of tax.

The marginal abatement cost curve used in this study is not given as a function of emission reduction in tonnes, as is the case for many of the figures in Chapter 2, but as a function of salinity to be removed, that is, salt removed per megalitre (T/ML). The rationale for this is presented in the following.

Typically, the marginal abatement cost curves used in the theoretical literature are shown as a function of emission reductions (T) (Pearce and Turner, 1992; Baumol and Oates, 1988). The typical curve is shown in Figure 2.2, where the marginal cost of abatement increases as the environment becomes cleaner, and more effort is required to remove additional units of pollutant. That is, as the concentration of pollutant decreases, the cost of removing additional units of the pollutant increases. Hence, when the level of emission reduction is shown in units of weight, the implication is that the marginal abatement cost curve is valid for a single volume of water. Where weight of salt is used as the emission, this translates into a separate marginal abatement cost curve for each volume of minewater available for treatment. A graphical demonstration is provided in Figure 4.1, where pollutant reduction, measured in tonnes of salt removed, is shown on the horizontal axis and cost is on the vertical axis. Assuming that all minewater has an initial concentration of 3000 mg/L, there will be a nest of marginal abatement curves for each volume of minewater which needs treatment. For example the marginal cost of removing one tonne of salt from 10 ML of minewater will be the same as removing two tonnes of salt from 20 ML of minewater. Thus points A and B in Figure 4.1 have the same marginal cost of abatement.

Each individual curve also carries an implied concentration which varies along its length. Thus, the concentration of A' is less than the concentration of A. However, when the costs of abatement are equal for salt removed from different volumes, the concentrations will be the same, since the marginal cost of removing salt is dependent on the concentration. Points A and B in Figure 4.1 not only have the same marginal cost of abatement, but also have the same concentration.

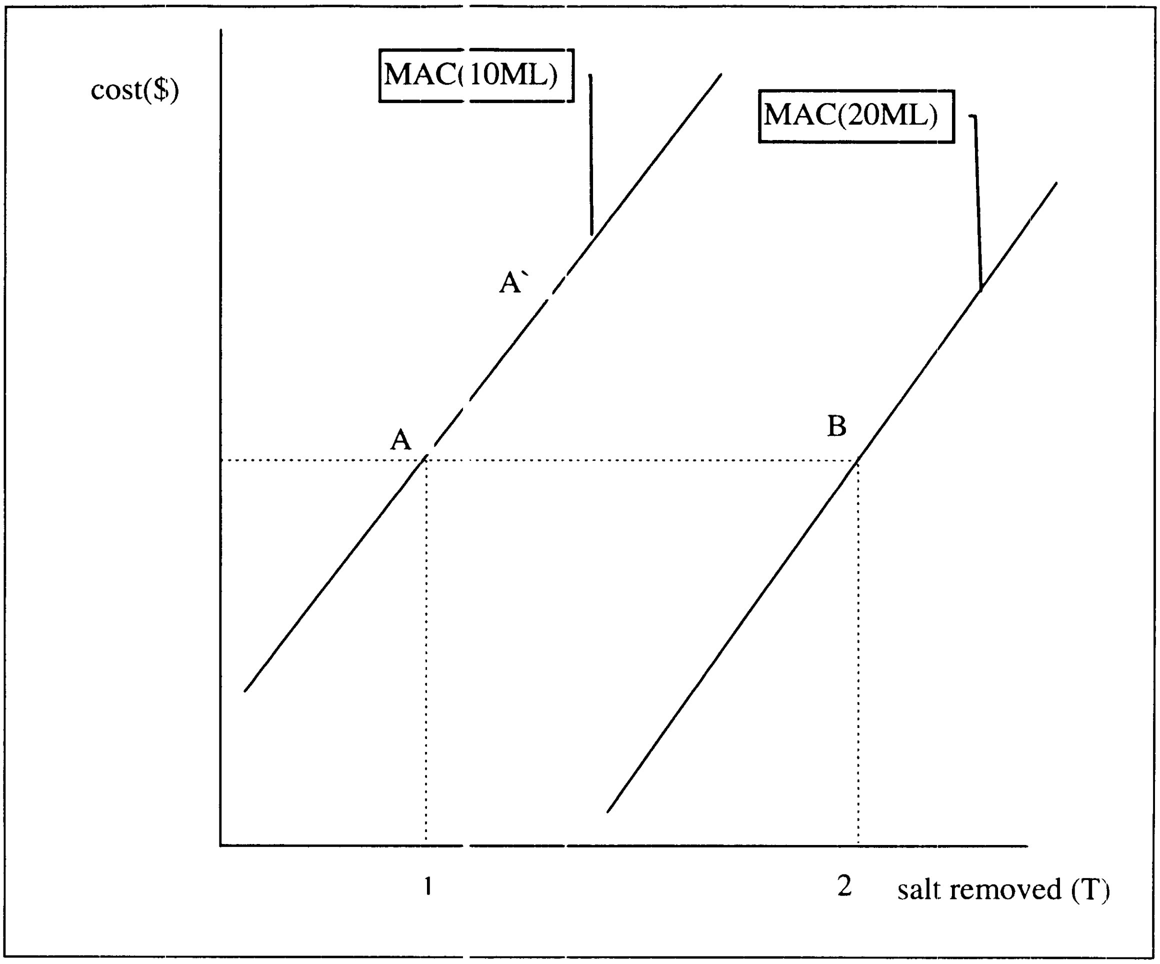


Figure 4.1 Marginal abatement costs as a function of salt removed

To simplify matters, the marginal cost of abatement can also be presented as a function of salinity. Thus the nest of curves necessary in Figure 4.1, can be shown with one curve, as in Figure 4.2.

Mines in the Hunter valley would have different volumes of minewater excess available for discharge from one period to the next. Given a marginal abatement cost curve of the type shown in Figure 4.1, the marginal abatement cost curve would vary each period with changes in the volume of minewater. To avoid this complexity, the marginal abatement cost curve for ML of water has been used to set the tax (that is a marginal abatement cost curve as a function of salinity as seen in Figure 4.2 has been used. Tax setting is described in section 4.3.

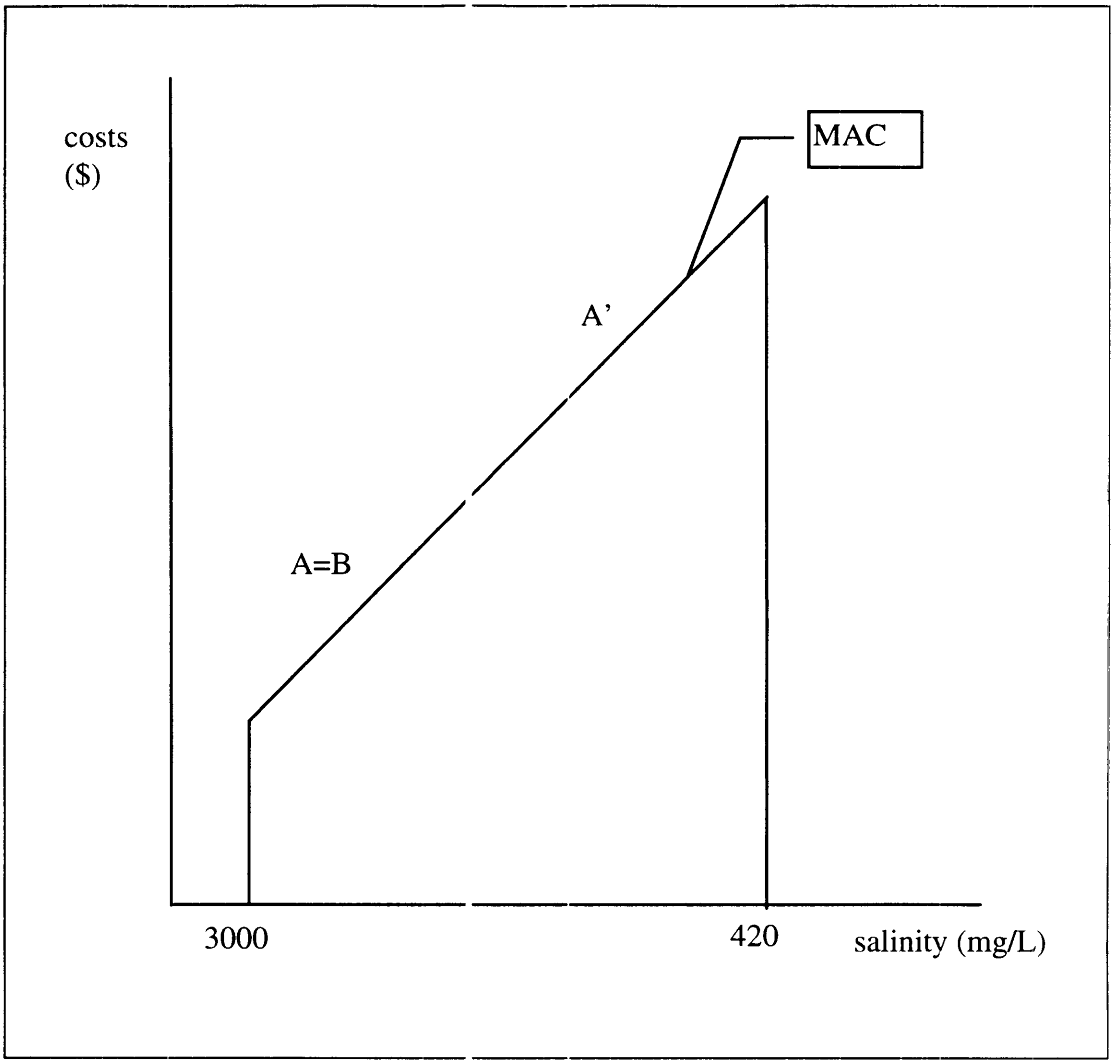


Figure 4.2 Marginal abatement costs as a function of salinity

Before proceeding further it is worth clarifying an assumption implicit in Figure 4.1, namely, that the initial salinity of the minewater is a constant, 3000 mg/L. This means that separate marginal abatement cost curves will be needed when different initial levels of minewater salinity occur. The likely curves would appear as shown in Figure 4.3, although the slope of the MAC curve will be the same because all curves are generated from the same marginal cost for treatment per unit of salinity reduced, the intercept on the cost axis will be different. Variation in minewater salinity may occur both within a single mine and between different mines.

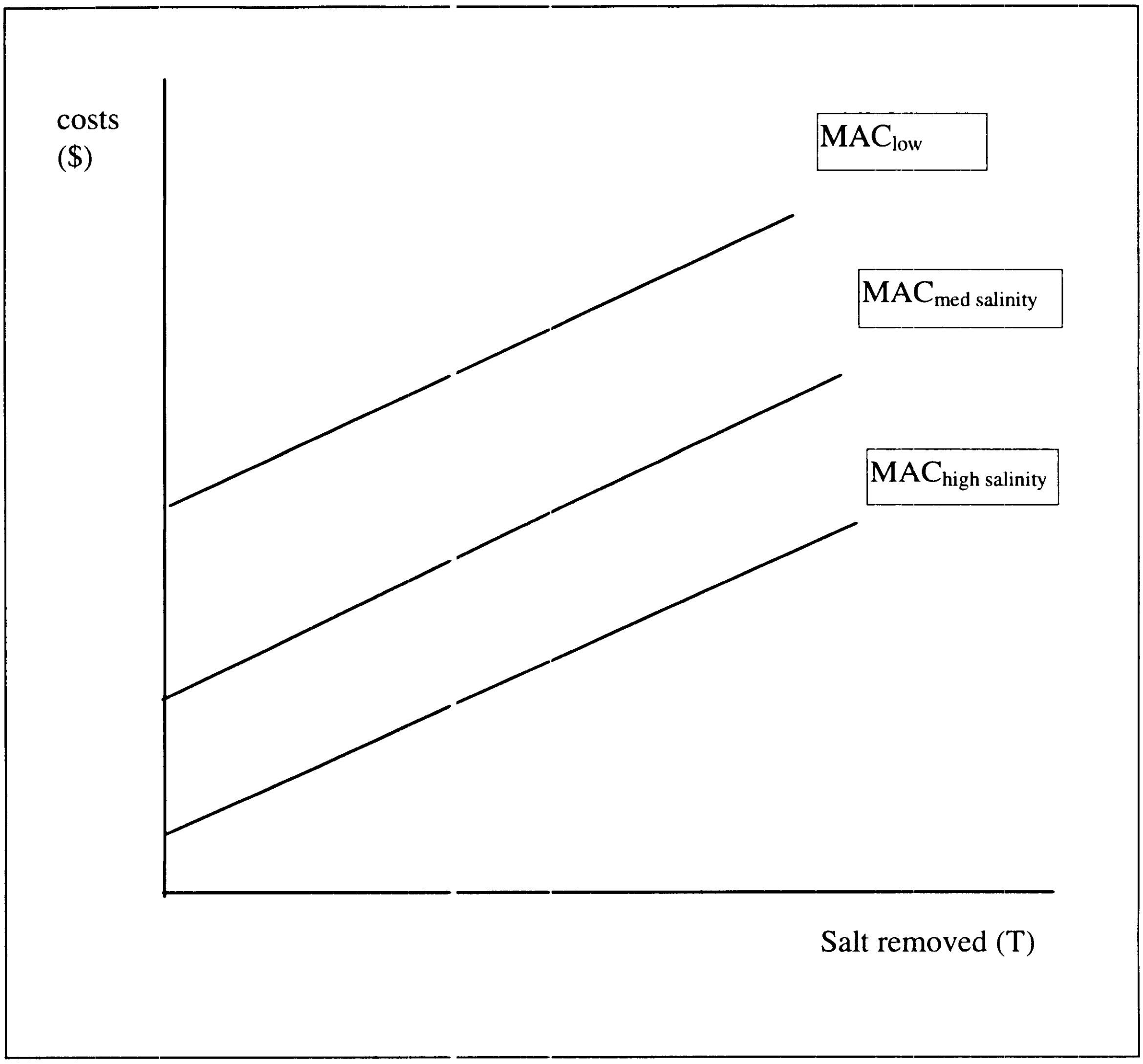


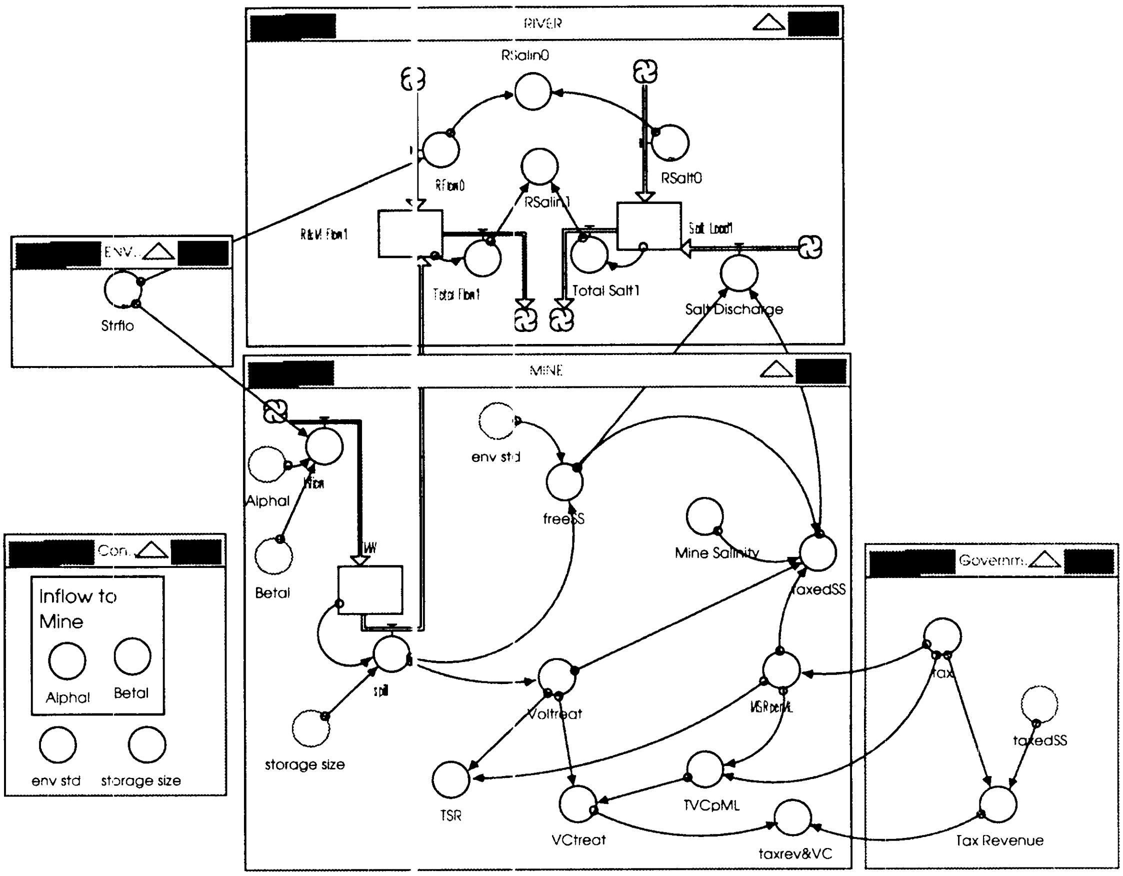
Figure 4.3 Marginal abatement cost curves for different initial minewater salinity

The salinity of minewater will vary throughout the year, depending on the ratio of groundwater to runoff from disturbed land in the storage. Thus, the salinity will be expected to vary with rainfall. Variation between mines will also occur depending on the area of disturbed surface which generates runoff relative to the groundwater inflow. The relative contribution of each component to the minewater will be a fixed component of the level of production (AGC Woodward-Clyde 1992). Inter-mine salinity will also vary on the basis of location, since the source of the groundwater salts and the origin of the sediments in the disturbed land vary with location, and are crucial factors in the salinity of the minewater. The general consensus, however, is that the impact of rainfall on salinity within a mine is considerably less than the between-mine variation, which independent on the fixed makeup of the mine site (and thus to the level of production) and its location.

The potential for the marginal abatement cost curves to vary between mine sites is important in its implication for cost-effectiveness, as shown in Figure 2.5. The theoretical cost-effectiveness of a tax relies on the assumption that individual dischargers are faced with different marginal costs of abatement, and are able to determine the optimal level of abatement and discharge which will vary between dischargers. If each curve on Figure 4.3 represents different mines, the amount of salt required to be removed will vary between the mines, and this is determined by the mines themselves, not the regulators. The cost-effectiveness of the tax developed in this study is not tested in the model because a single discharging mine has been modelled. However, by ensuring that the mine decisions are based on the tax rate and no quantity restrictions are placed on the discharger, then the tax formulated should be able to achieve a cost-efficient solution. Empirically testing for cost-effectiveness of the policy is beyond the scope of this study.

4.3 Setting the tax

The tax policy developed in this chapter uses a variable load based tax in an attempt to save costs through making better use of the assimilative capacity of the river, that is meeting objectives i) and ii). The tax policy is developed by beginning with a static flat tax and gradually refining it until the final tax policy is achieved. Figure 4.4 is included as a guide to the development of the model. It has four main sectors; environment, river, government and mine. The environment sector contains the streamflow data collected from the Department of Water Resources. The government sector sets the tax and calculates the tax revenue. The river sector calculates the river salinity upstream of the discharging mines, and the salinity downstream of the mines. The mine sector contains the state variable minewater, as well as the decision rules for discharge. Water volume (ML) and salt (T) are treated separately in the model.



RIVER SECTOR
RSalin0 (mg/L)is the salinity of the river prior to mine discharge
RSalin1 (mg/L)is the salinity of the river after mine discharge
RSalt0(T/wk) is the salt in the river prior to mine discharge
Salt load1 (T/wk) is the salt in the river after mine discharge
Total Salt1 (T/wk) = Salt load
RFlow0 (ML/wk) is the streamflow prior to mine discharge
R&M Flow1 (ML/wk) is the streamflow after mine discharge
Total Flow1(ML/wk) = R&M Flow
Salt Discharge (T/wk) is the salt discharged from the mine

GOVERNMENT
tax ($/T) is the tax rate
taxrevenue ($/wk) is the tax revenue collected per week

MINE SECTOR
Inflow (ML/wk) is the inflow the mine
Spill (ML/wk) is the overflow from the mine storage
voltreat (ML/wk) is the volume treated
VCtreat ($/wk) is given by the area under the MAC curve
MSRper ML (T/ML) is the salt removed per ML
TSR (T/wk) is the salt removed per week
taxed SS (T/wk) is the salt discharged to the river from the spill volume which is taxed
ZISspill (T/wk) is the salt discharged to the river from the spill volume which is not taxed
env. std (mg/L) is the environmental standard
taxrev&VC ($/wk) is the costs of treatment and tax

Figure 4.4 Diagram of model for static flat tax

The static flat tax consists of charges for every unit of waste discharged. It is set at the point where the marginal cost of abatement equals the tonnes of salt that need to be removed from the discharge water X as shown in Figure 4.5. This tax may achieve a cost-effective reduction in salinity of discharges of X (T/ML), however, this does not guarantee that the environmental standard in the river is met and not violated. Thus the flat tax would fail to satisfy all four of the objectives of the tax policy. It would not minimise violations of the environmental standard, (except when set high enough to force all salt to be removed prior to discharge); it would not utilise the assimilative capacity of the river, would not achieve the environmental standard cost-effectively, and it would also fail to reduce the costs to dischargers.

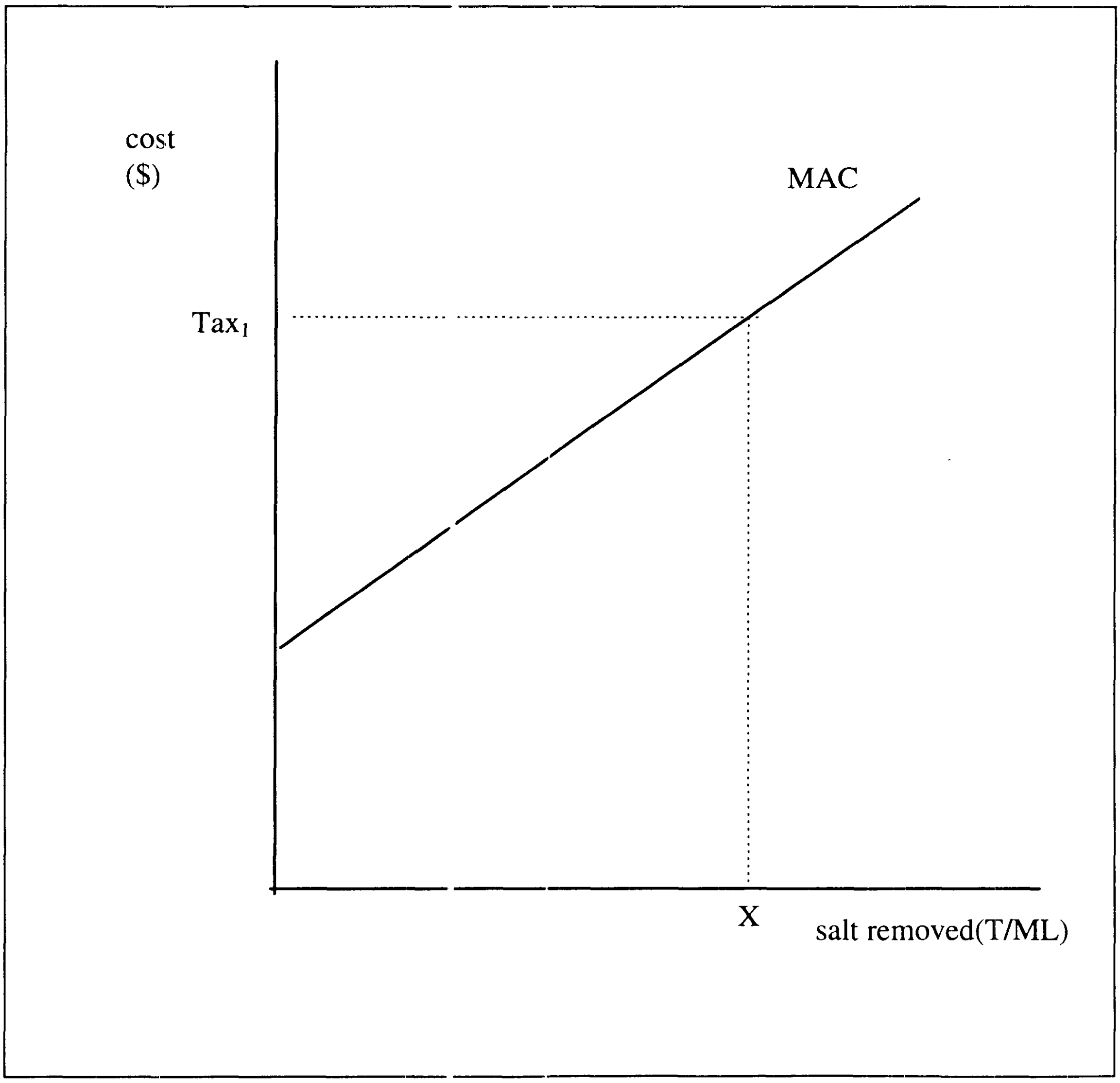


Figure 4.5 Setting a static flat tax

A variable tax rate would allow the first three objectives to be met. To meet the fourth objective a further three modifications to the tax policy are developed.

4.4 Setting a variable tax rate

In order to observe the environmental standard while at the same time to utilising as much of the assimilative capacity of the river as possible, the tax rate needs to be variable. The time step used will need to be able to capture the variability and unpredictability of the river. A time step of one week has been chosen as a compromise between the daily fluctuations in the assimilative capacity of the river and the task of resetting a tax.

The first step in setting a variable tax rate is is to determine the amount of salt which must be removed from the minewater such that the water released from the mine would not cause the salinity of the river to exceed 420 mg/L (700 EC). This quantity can then be used as X in the Figure 4.5. The variable tax is essentially the same as the flat tax except that it is altered every week. In order to determine the salt which must be removed the regulator requires the following information:

- salinity of the river
- flow of river
- salinity environmental standard
- contents of storage in every mine
- inflow in every mine

The first three quantities are used to estimate the tonnes of salt that the river can assimilate. The last two are used to set the tax rate.

4.5 Methods to reduce the costs paid by dischargers

The three modifications used to reduce the costs paid by dischargers are oulined below. First, although the tax is applied to the weight of salt discharged, any quantity of salt is allowed to be discharged to the river for free, providing it is released at a concentration equal to (or less than) 420 mg/L (that is, the environmental standard). Second, whenever the assimilative capacity of the river is high enough to assimilate all of the minewater excess available, no tax is charged on discharges. Third, to reduce the tax rate, but at the same time encourage storage of minewater which cannot be assimilated. Each of these suggestions is explained below.

The free discharge of the first 420 mg/L is justified on the grounds that if salt was discharged to the river at this concentration it would not adversely effect the salinity of the river. In instances where initial river salinity is less than 420 mg/L, then releases of minewater at 420 mg/L may increase river salinity slightly but they would not cause river salinity to exceed the threshold. If initial river salinity were greater than 420 mg/L then minewater released at 420 mg/L may even lower the river salinity, although never to the level of the standard. Mine dischargers should therefore not be taxed on this salt which does not adversely affect river salinity.

In the model, dischargers will be allowed to release 0.420 T/ML of salt at no cost, mine discharge at higher salinity levels will attract the tax on each unit of salt exceeding this level. This portion of free salt discharge will be termed ***zero impact salt***.

When the zero impact salt is added to the quantity of salt the stream can assimilate, the quantity of salt that needs to be removed from the minewater in order for the environmental standard to be met is reduced to **X'** (Figure 4.6), causing the rate of tax to be lowered from Tax_1 to Tax_2.

A further reduction in tax and treatment costs is possible whenever the assimilative capacity of the river is high enough to accommodate 100 per cent of the minewater available for discharge. Reducing the tax to zero ensures that all dischargers will release all minewater present in their on-site storage reservoirs. Salt discharged under these terms will be termed ***free assimilated salt***.

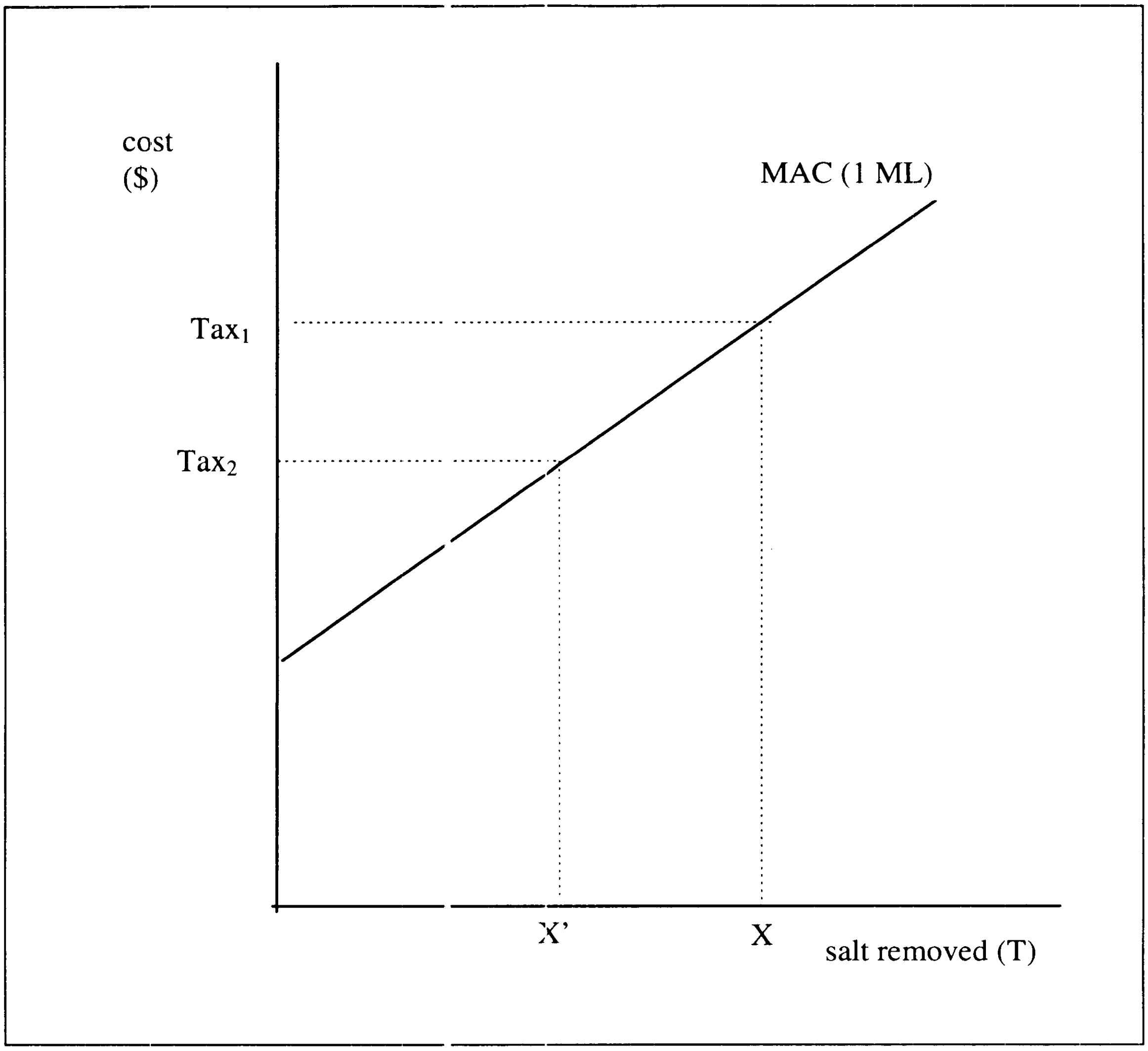


Figure 4.6 A tax with free discharge of zero impact salt

In order to maintain the model within manageable limits the following assumptions were made:

i) there is the same concentration of minewater in every mine

ii) mine operators can react within the week to discharge and treat appropriatelevels of salinity

iii) mine operators have no expectations of future tax rates

iv) mine operators are cost minimisers

v) minewater storages have no operational costs associated with them and the use of the storage is free

Onsite minewater storages influence the variable tax rate as discussed below.

Because of assumptions iii) and v) the decision rules will be based solely on information available at the start of the week. When the river is able to assimilate some, but not all of the minewater available for discharge, a tax would encourage the mine to store its wastewater and not to discharge, since the storage is free and discharge or treatment of the water attracts a fee. Thus, the river assimilative capacity moves downstream unused. This all or nothing approach will also cause the tax rate to rise as the mine stores more water.

Once the storage is full, however, minewater will overflow or spill from the storage. This water will be termed *spill.* Once a spill occurs, the option to store no longer exists and the decision becomes how much salt should be removed before discharging water to the river. A tax set on the basis of the potential for all of the mine wastewater to be discharged would be high and cause more salt to be removed from the spill than is necessary. Figure 4.7 shows how a tax set on the basis of stored plus spill minewater, would result in X'_R (T/ML) of salt being removed from the spill water. However, because the volume of spill is less than the total minewater, the treatment level does not need to be this high. Instead, the treatment level should be X'_S. This means that X'_R-X'_S has been removed unnecessarily with an associated cost of A+B. Area A is the surplus treatment cost, and area B is the surplus tax paid.

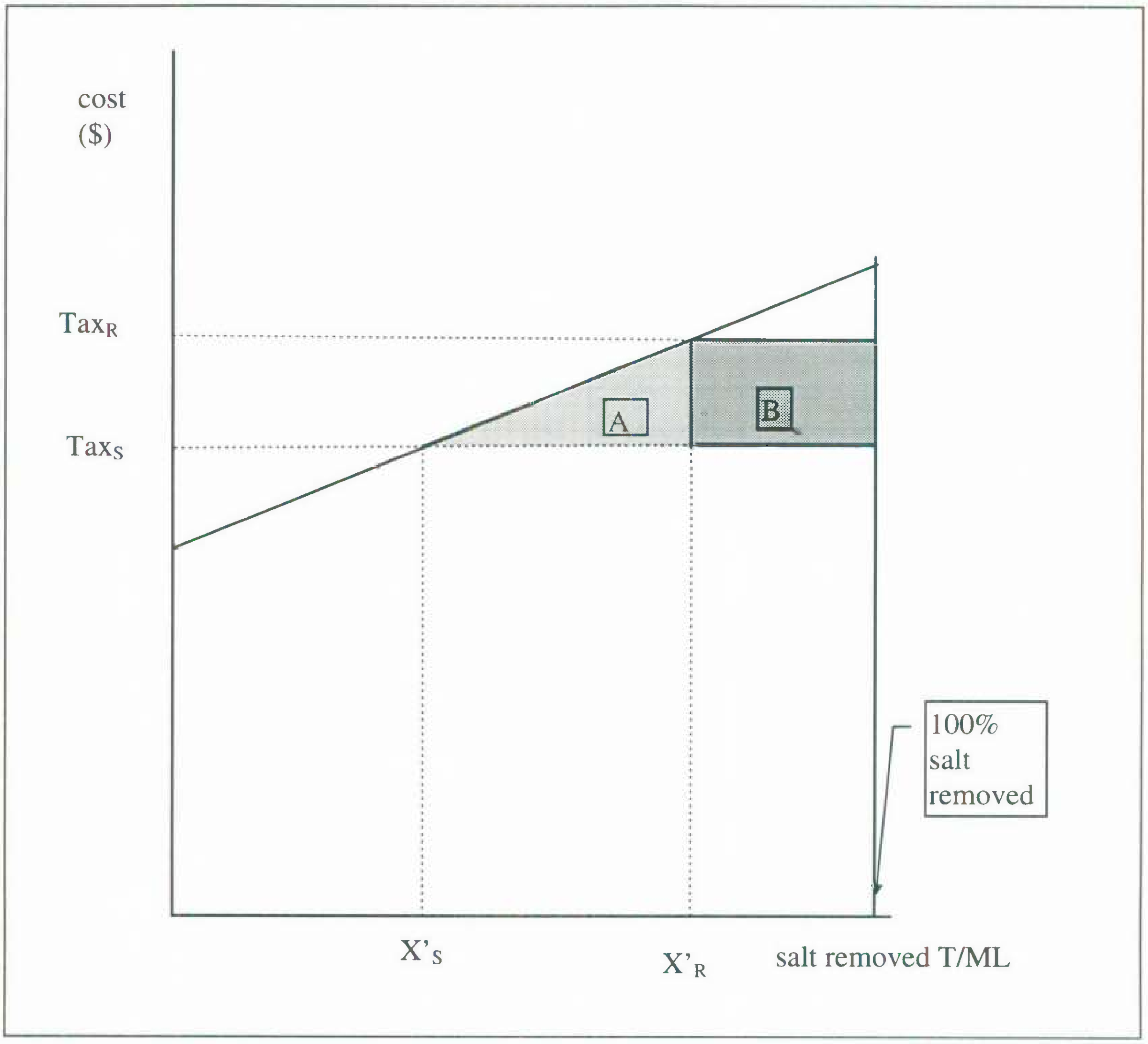


Figure 4.7 The impact of storages on tax rate

If the entire spill could be assimilated but not all the stored minewater, then there would be no need to remove salt from the spill and the level of tax would correspond to NTR on Figure 4.8. Taxes paid under this scenario would equal the area under the tax line (NTR) since paying the tax will always be cheaper than the cost of treatment. Theoretically the level of the tax could be reduced to zero, if there were some way of preventing discharge of all storage contents. Instead, a reduced tax rate, though still a positive one, could be used to reduce the tax burden. For example, a tax rate of half the NTR is used in Figure 4.8 ($NTR_{0.5}$). Since taxes paid are equal to the area under the tax rate, reducing the tax rate saves costs equal to area C.

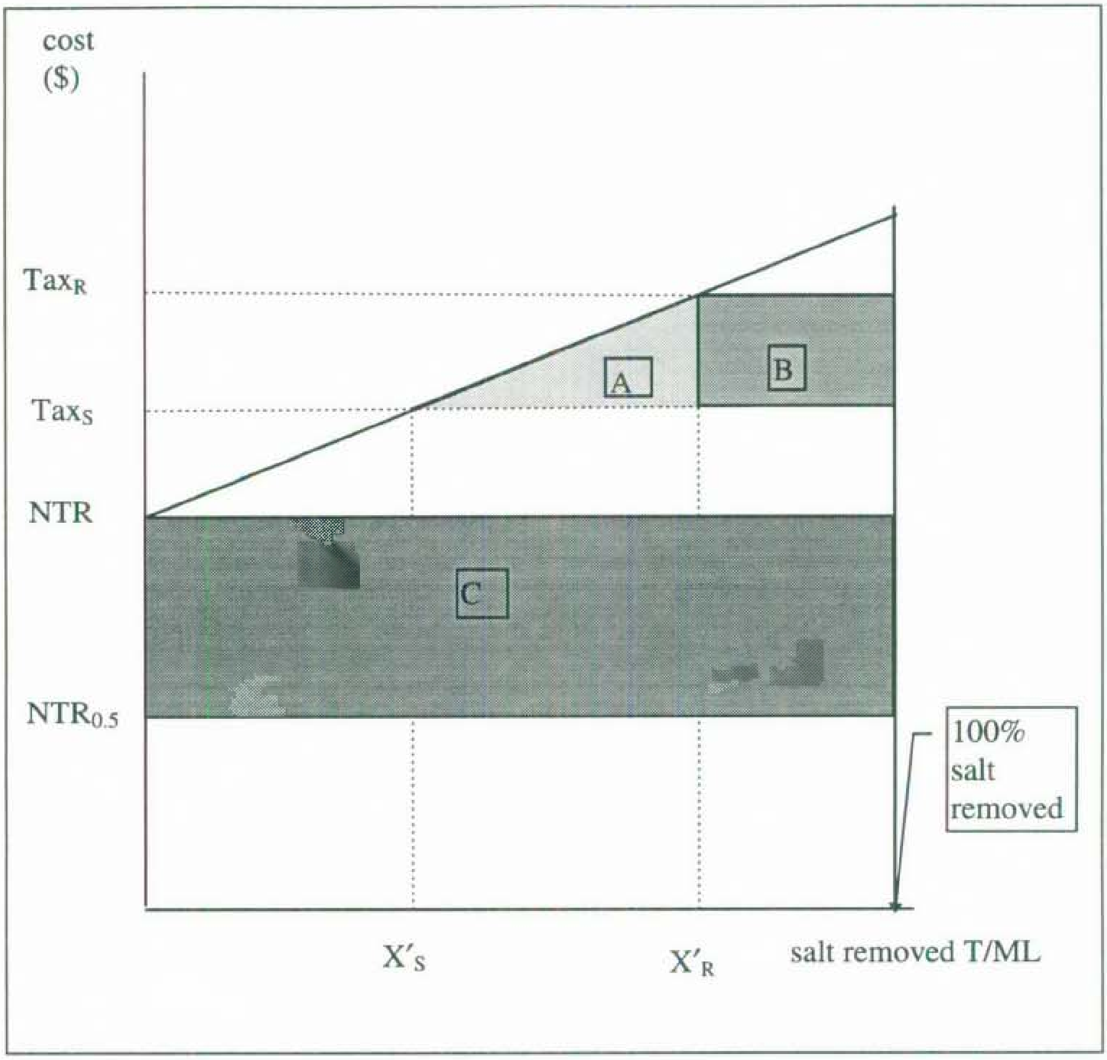


Figure 4.8 Reducing the taxes paid by dischargers

Thus it is proposed that, once the storage is full, the regulator sets the tax on the basis of the volume and salinity of the spill. The exact amount of this tax rate is arbitrary, it must be greater than zero and less than the marginal cost of treatment at 3000mg/L, but it is meant to deter the mine operator from discharging from the storage. When discharge of the spill to the river could only partly be assimilated by the river, the tax must be set so that the salinity of the spill is reduced to the acceptable salinity level.

The tax policy outlined below attempts to include each of the characteristics which have been discussed.

4.5.1 Suggested tax policy

i) Zero impact salt discharge

Discharge from the mine will only attract a charge on the quantity of salt which exceeds 420 mg/L. If a spill exists when the assimilative capacity of the river is zero, water discharges will still be allowed free of charge at a salinity level equal to the environmental standard.

ii) Total free discharge

Free discharge will only occur when the quantity of salt available for release can be totally assimilated by the river. The mine will respond to this by releasing all the salt that is held on site as ***free assimilated salt***.

iii) Nominal tax rate

The nominal tax rate (NTR) applies only when a spill exists and the salt present in the spill can be assimilated in the river, but untreated discharge of the total minewater available would cause the river salinity to exceed the threshold.

The nominal tax rate serves as a deterrent to discharging when the total mine salt available for discharge would cause the river salinity to exceed the threshold. The nominal tax rate encourages storage of minewater onsite for the reasons outlined in the previous section. The NTR will only produce tax revenue once a spill exists, if the quantity of salt in the spill does not exceed the assimilative capacity of the river.

iv) Tax rate equal to the marginal cost of abatement at the environmental standard

This tax rate is invoked when some of the spill can be assimilated by the river, but not all of it. In this situation the tax rate will signal to the discharger that they need to abate that portion of the spill for which the marginal cost of abatement is less than the tax and discharge the remaining quantity.

4.6 The model

A computer simulation model has been designed to show how the tax regime described above might operate in the Hunter River system. The stochastic dynamic simulation model uses historic streamflow as a guide to the likely fluctuations that could be expected in the Hunter River, and provides an insight into the degree of variation a tax might need in order to maintain the environmental standard. The model was developed using the software ' STELLA II".

The following sub sections give a description of the model. The programming used is included in Appendix A.

4.6.1 Assumptions

For simplicity, mines have been modelled as a single discharging firm facing a single marginal abatement cost curve for treatment of the excess minewater. The following assumptions were made in the model:

- The regulating authority has the following information:
    - knowledge of the environmental standard
    - knowledge of the aggregate MAC curve
    - knowledge of the total mine water excess for all mines and the average salinity level
    - knowledge of the storage capacity at each mine site
    - knowledge of the assimilative capacity of the river each week
    - the weekly discharges from mines
- the marginal abatement cost curve faced by the discharging firm is the same as that used in tax setting (that is, regulators have perfect information regarding cost curves)
- the storage represents a sunk cost, and does not add to the marginal cost of abatement to the mine
- initial salinity of the mine waste water excess is constant at 3000 mg/L
- dischargers do not attempt to predict future tax rates, that is, their optimising behaviour is not dynamic

- dischargers respond only to the tax rate in time period t, and to minewater and salt levels in the same time period.

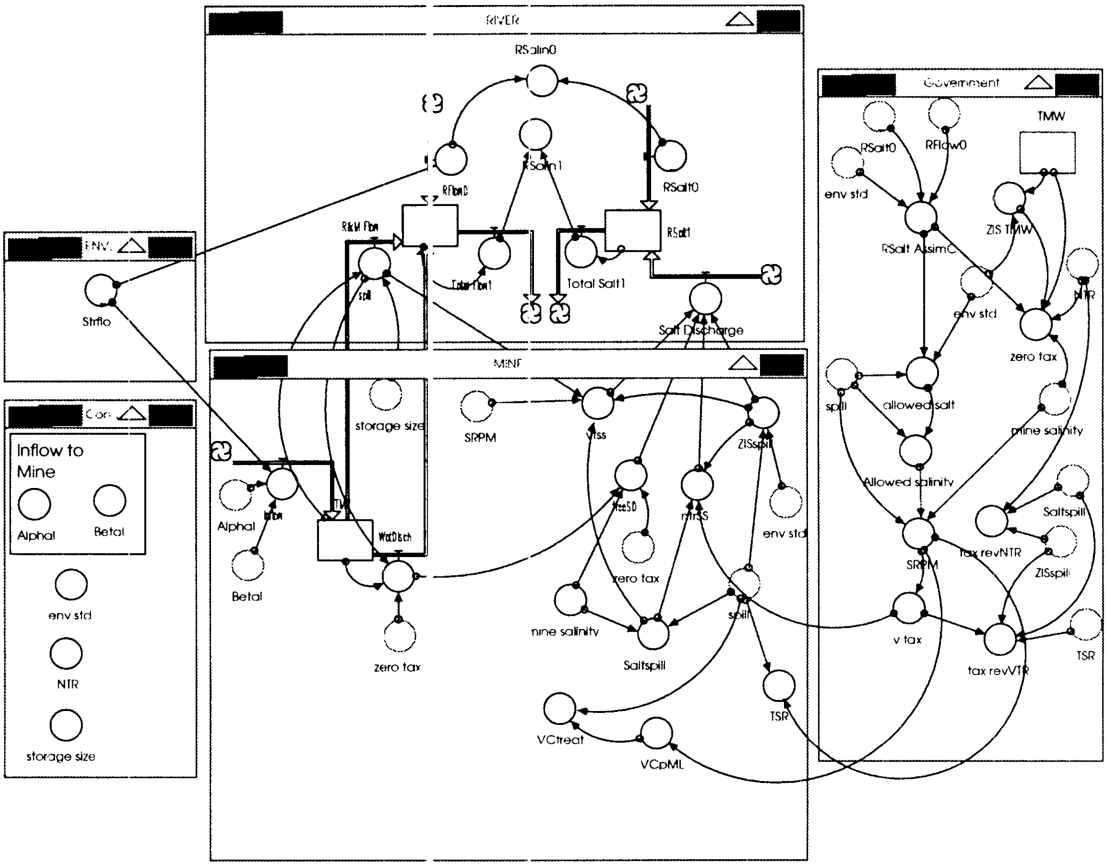
4.6.2 Simulation of streamflow and assimilative capacity

The assimilative capacity of the river is the salt load that the river would be able to assimilate without causing the salinity of the river to exceed the environmental standard. It is calculated as the difference between the environmental standard and the salinity of the river upstream of the discharging mines. The salt load of the river and the flow are measured, in the period t-1, at a gauging station upstream of the discharging mine. It is assumed that this measurement would be indicative of the salt load and flow of the river at the discharge site in the following period (t). The assimilative capacity of the river (RSalt AssimC) is defined as:

$$\text{RSalt AssimC} = (\text{env. std}*\text{RFlow}/1000) - \text{RSalt1} \tag{4.1}$$

where RFlow0 is river flow upstream of the discharging mines (t-1)(ML/week) and RSalt0 is the salt (T/week) present in river at upstream station (t-1). RFlow0 (ML/week) is the sum of observed mean daily streamflow data from two DWR stream gauging stations, and Rsalt0 (T) is the amount of salt in the river, these two variables are determined exogenously as explained in Chapter 5.

Figure 4.9 shows the development of the model from the flat tax presented earlier in Figure 4.4 to the model which allows for each of the cost saving options suggested in section 4.2. In the government subsector, the assimilative capacity of the river is included in the setting of the tax.



Additional variables to those in Figure 4.4
GOVERNMENT SECTOR
NTR ($/T) is the nominal tax rate
zero tax this is the tax rate which applies when all of the minewater can be assimilated by the river
ZIS TMW (T/wk) is the zero impact salt present in the TMW
ZISspill (T/wk) is the zero impact salt present in the spill
allowed salt (T/wk) is the salt which the river can assimilate it includes the zero impact salt
allowed salinity (mg/L) is the allowed salt divided by the volume of discharge
SRPM (T/ML) is the salt removed per megalitre
v. tax ($/T) is the variable tax rate per tonne of salt discharged
tax revNTR ($/wk) is the tax revenue due to the NTR (this rate only applies when the salt present in the spill can be asimilated by the river)
tax revVTR ($/wk) is the tax revenue when the variable tax rate applies

MINE SECTOR
salt spill (T/wk) is salt present in the spill
vtss (T/wk) is the salt discharged in the salt spill which is taxed at the variable tax rate (v. tax)
freeSD (T/wk) is the salt discharged for free when the total minewater is able to be assimilated by the river
ntrSS (T/wk) is the salt discharged as salt spill when NTR applies
WatDisch (ML/wk) is the water discharged from the mine when all of the minewater can be assimilated by the river

RIVER and ENVIRONMENT SECTORS are the same as for Figure 4.4

Figure 4.9 Diagram of model for the minimum variable tax

4.6.3 Calculation of the marginal abatement cost curve

The model uses a marginal abatement cost curve which is a function of allowed salinity. It is presented in the form of a MAC for the removal of salt from one megalitre of minewater (initial salinity of 3000mg/L). This simplifies the number of abatement cost curves needed in the model, as illustrated in section 4.2.2.

The MAC curve used in the model is linear with a positive slope, showing that the cost of removing additional units of sal from one megalitre requires an increasing amount of effort. It has a positive intercept of 100 because the cost of removing the first unit of salt from the minewater at an initial concentration of 3000 mg/L is assumed to be \$100/T. The equation of the MAC curve is:

$$\text{MAC}(\$/\text{T}) = 100 + 15(\text{SRPM}) \tag{4.2}$$

where SRPM is the salt removed per megalitre (T)

$$\text{SRPM (T/ML)} = (\text{mine salinity - allowed salinity})/1000 \tag{4.3}$$

where mine salinity is the salinity of the minewater. This is a constant (3000 mg/L).

Allowed salinity is the maximum salinity that the discharge can contain, such that when it is released it will not cause the river salinity to violate the environmental standard. In terms of *allowed salt* this is the amount of salt the river is able to assimilate (RSalt AssimC) plus the salt load that the discharge would contribute if it were released at 420 mg/L (that is, the zero impact salt). The allowed salinity is simply the weight of *allowed salt* divided by the volume of the discharge.

Since the variable tax rate only applies to spills discharged when all of the spill is not able to be assimilated by the river the tax rate is set using only the quantities of salt and volume from the spill. The zero impact salt used in setting the variable tax rate is given the abbreviation ZISspill (see equation 4.4). The For more details on setting the tax rate see section 4.6.4. Equation 4.5 shows the allowed salinity

$$\text{ZISspill(T/ML)} = 420*\text{spill}/1000 \tag{4.4}$$

$$\text{allowed salinity(mg/L)} = (\text{RSalt AssimC} + \text{ZISspill})/\text{spill}*1000 \quad (4.5)$$

The MAC equation (4.2) was set arbitrarily, based on a single average variable cost of \$120/T of salt removed. This cost was provided by Pacific Power (J. Stuart pers. comm., 1995), as an indication of the order of magnitude of costs they experience in their desalination plant. To turn a single cost into a MAC curve involved the assumptions that this cost corresponds to the midpoint salinity, and a range of \$40 to cover the range from 420 mg/L to 3000 mg/L salinity. The MAC curve used in the model causes the tax and treatment costs to be highly speculative. For this reason the sensitivity of the tax and treatment costs to changes in the marginal cost curve will be tested later.

4.6.4 Calculation of the tax

In section 4.5.1 the tax was broken into four distinct parts; free discharge of *zero impact salt*, total free discharge of *assimilated salt*, nominal tax rate (NTR), and tax rate equal to the MAC at the environmental standard. These are included in the model in the following way.

i) Zero impact salt discharge

The release of salt at a salinity of 0.420 T/ML is permitted at no cost. Mine discharge at higher salinity levels will attract the tax on each unit of salt exceeding this level. This portion of free salt discharge could be simply refunded to the dischargers after paying tax on the full salt load of their discharge. However, this would ignore the reduction in tax rate that should occur, which in turn would encourage higher discharges of salt to the river. To include this in the tax setting, the allowed salinity is calculated by equation 4.5.

ii) free discharge

The model imposes a *zero tax* when the total minewater stored (TMW) on the minesite can be assimilated by the river. The model calculates the amount of salt that the river can assimilate if the entire volume of minewater is discharged,

$$\text{allowed salt(T/wk)} = \text{RSalt AssimC} + \text{ZIS TMW} \quad (4.6)$$

where ZIS TMW is the zero impact salt for total minewater, and is calculated as

$$\text{ZIS TMW(T/wk)} = \text{environmental standard*TMW/1000} \qquad (4.7)$$

If equation 4.6 is greater than or equal to the total mine salt stored (that is, TMW*mine salinity/1000), then a zero tax is applied, allowing dischargers to release the contents of their mine storage for free. However, if equation 4.3 is less than the total mine salt then a nominal tax rate (NTR) is applied, whereby any discharge to the river is taxed at this rate. When NTR is applied, the response of the mine will be to store whenever there is sufficient storage capacity to do so.

iii) Nominal tax rate

The nominal tax rate (NTR) serves to deter discharging when the total mine salt available for discharge would cause the river salinity to exceed the threshold. Since the nominal tax rate causes the mine operator to store minewater whenever storage space permits, it will only produce tax revenue when a spill exists, and the salt load of the spill does not exceed the assimilative capacity of the river. The nominal tax rate need only be greater than zero, and less than the marginal cost of abatement for the first unit of salt removed from one megalitre of minewater (initial concentration of 3000 mg/L).

The relative impact of the nominal tax rate on total tax revenue is tested in the model using rates of $100/T and $50/T.

iv) Tax rate equal to the MAC at the environmental standard

The tax is set equal to the marginal cost of treatment for the allowed salinity of the spill. Equation 4.8 shows the tax rate

$$\text{v. tax (\$/T)} = 100+15\text{*SRPM} \qquad (4.8)$$

where SRPM is the salt which needs to be removed from the spill to meet the allowed salinity which will not cause the river to exceed the threshold. It is assumed that the regulator would know the assimilative capacity of the river, and the volume and initial salinity of the spill. From this information the regulating authority knows the appropriate amount of salt to remove from the spill (SRPM) in order for the river to assimilate the mine spill discharge.

Note that dischargers are allowed to discharge any quantity of salt to the river providing that its concentration does not exceed the salinity of the environmental standard, (that is, zero impact salt). The model assumes however, that this salt will only be discharged if one of two conditions exist:

1. that a spill from onsite storages occurs or
2. that the assimilative capacity of the river can cope with the total salt in the storage

In all other cases it would be cheaper to store the water.

4.6.5 Simulation of mine discharge

The mine operator faces the tax set for the week, a storage with a limited capacity, and a marginal abatement cost curve. The decision rule is:

if v. tax=0, discharge all minewater without removing any salt

if 0<v. tax<=NTR, discharge spill water, without any treatment

if v. tax >NTR, discharge spill water with (tax-100)/15 tonnes of salt removed per megalitre.

4.6.6 Feedback from the mine to storage

Total minewater (TMW) is a state variable which keeps account of inflows and outflows to the mine. This volume is calculated at the start of each week, by summing the water carried over in the storage from the previous week and the likely amount to be generated in the current week, from inflow to the mine. Any discharges and spills from the mine are subtracted from this value to give the TMW at the end of the period, and carried over to the start of the next period. Note that during period t, the TMW can exceed the storage capacity if a spill occurs during the week.

4.6.7 River salinity prior to and following discharge of minewater to the river

The salinity of the river prior to discharge from the mines is assumed to be given by:

$$\text{RSalin0(mg/L)} = (\text{RSalt0} * 1000/\text{RFlow0}) \quad (4.9)$$

and the salinity of the river after mine discharge is

$$\text{RSalin1(mg/L)} = (\text{Total Salt1} * 1000/\text{Total Flow1}) \quad (4.10)$$

Total Flow1 and Total Salt1 include the discharge from the mine. RSalin1 can be compared to the environmental standard to measure the success of the policy in meeting the environmental standard.

4.6.8 Costs paid by dischargers

4.6.8.1 Treatment costs

Calculation of the treatment cost involves integrating the marginal abatement cost curve for 1 ML of water, bounded by zero and the salt removed from 1 ML of minewater. The area under the curve is also given by;

$$\text{VCpML(\$/ML)} = 100\text{SRPM} + 7.5\text{SRPM}^2 \quad (4.11)$$

$$\text{VCtreat(\$/wk)} = \text{VCpML} * \text{spill} \quad (4.12)$$

4.6.8.2 Taxes paid

Taxes are collected whenever spill water is discharged which exceeds the salinity level of the standard. When the NTR applies, the tax will be given by equation (4.13)

$$\text{tax revNTR(\$/wk)} = \text{NTR} * \text{salt spill} - \text{ZISspill} \quad (4.13)$$

When the variable tax rate (VTR) applies, the tax revenue will be given by equation 4.14

$$\text{tax revVTR(\$/wk)} = \text{tax} * (\text{salt spill} - \text{TSR} - \text{ZISspill}) \quad (4.14)$$

where TSR is the total salt removed per week from the minewater prior to discharge to the river.

4.7 Summary

This chapter showed the formulation of a salt tax designed to control saline minewater discharges in order that the environmental standard is met. The tax suggested, uses four methods for reducing the costs of both treatment and tax. The simulation model used to simulate the tax was also defined. Chapter 5 presents the data used in the model and the results are reported in Chapter 6.

5. Data analysis and model calibration

5.1 Introduction

This chapter provides a description of the data used in the model, including the source and quality of data. Data inputs to the model consist of river salt (T/week), streamflow (ML/week), saline inflows to the mine, onsite storage capacity, and marginal cost of abatement. Each of these data requirements, with the exception of the marginal abatement cost curve are reported in this chapter. The marginal cost of abatement has been addressed in Chapter 4. Any manipulation, modifications or analysis which have been made to data prior to inclusion in the model are described in this chapter.

5.2 Streamflow and conductivity(salinity) data

5.2.1 Availability of data

The choice of stream monitoring sites was based on four criteria;

- that continuous streamflow and conductivity monitoring were available
- that records of streamflow were available for a reasonable length of time
- that the monitoring sites were located upstream of the major discharging mine sites
- that major tributaries located above these mine sites were included

Only two monitoring stations satisfied all of these criteria. Muswellbrook (station no. 210002) and Sandy Hollow (station no. 210031) located on the Goulburn River.

Streamflow data were obtained from the Department of Water Resources for the stream gauging stations 210002 and 210031 for the years 1980 to 1995. The stream gauging network of the Hunter is reasonably comprehensive, however, prior to 1980 records were not collected regularly, and in general are not available on computerised data bases. From 1980 a number of key gauging stations began daily monitoring of streamflow conditions. The data are good to high quality. Missing records account for a small proportion of the data.

Conductivity data for the period 1992-1995 were available for sites which corresponded to the above stream gauging stations. The data are also of good to high quality, but the limited availability of this data has meant that conductivity data needed to be generated, in order for the model to simulate river assimilative capacity over a significant period of time (15 years). Section 5.2.5 presents the methodology used in generating this conductivity data.

5.2.2 Aggregation of data to a weekly time step

Data were aggregated to a weekly time step as a compromise between having a reasonably fine grid, reflecting the fluctuations in river assimilative capacity, and a workable period for regulators to set a tax and for dischargers to respond to the altered tax. The aggregation of data smoothed some of the extreme swings in the river flow and conductivity readings, but the flow salinity relationship was relatively unaltered.

5.2.3 Review and description of data

Streamflow at Muswellbrook

Streamflow at Muswellbrook is influenced by releases from Glenbawn dam, a major water storage dam. The streamflow used in the model does not treat the these releases differently to the observed streamflow data. The majority of flows tend to range from 1000 to 4000 ML/week, with flows in excess of 10000 ML/week occurring occasionally. Streamflow readings show a weekly flow rate, generally less than 6000 ML and an average of roughly 3000 ML. Figure 5.1 shows streamflow at Muswellbrook for the period February 1992 to March 1995. (This is the period for which the continuous conductivity data exists). This period reflects a fairly dry spell when examined in the context of the 15 year period presented in Figure 5.2. Note that the scale for the 3 year period goes to 16000 ML/week whereas the scale for the 15 year period has a maximum value of 140000 ML/week

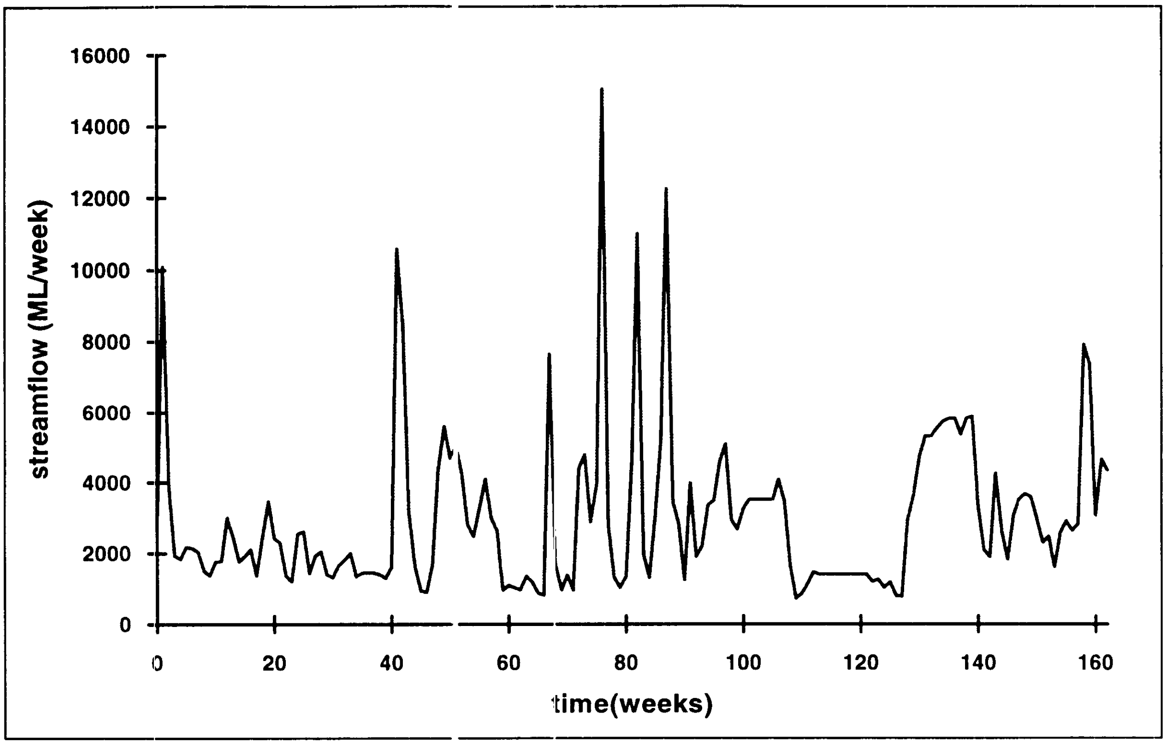


Figure 5.1 Streamflow at Muswellbrook 1992-1995

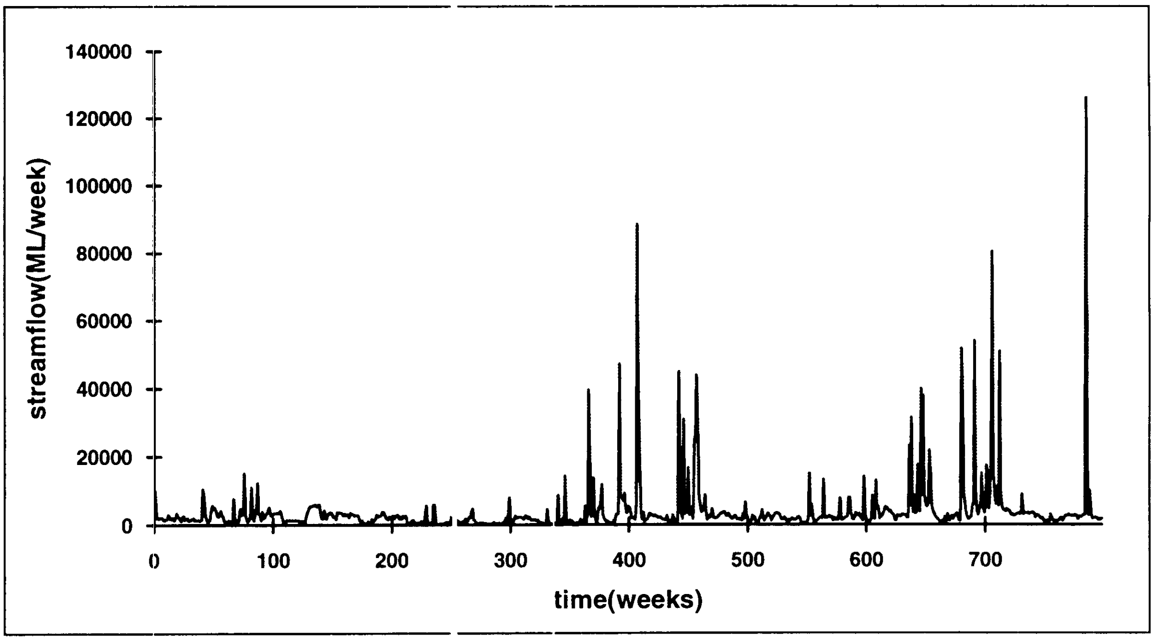


Figure 5.2 Streamflow at Muswellbrook 1980-1995

Streamflow at Sandy Hollow

The dry period 1992-1995 is also reflected in the streamflow recordings at Sandy Hollow (Figure 5.3). The majority of streamflow readings are below 2000 ML with many of these being only 1000 ML per week or less. This is approximately half of the flow recorded for the Muswellbrook site. With the exception of several high flow spikes, flow rate at Sandy Hollow seems to fluctuate less from one week to the next than at Muswellbrook. The spike shown in Figure 5.3 which exceeds the scale, is a single data point of approximately 130000 ML. Figure 5.4 shows this high flow event in the context of some other high flows which have occurred in the Goulburn River on occasion. The magnitude of the flow relative to other flows is increased due to the summation of the high daily streamflows which occur during a flood, to get the weekly streamflow.

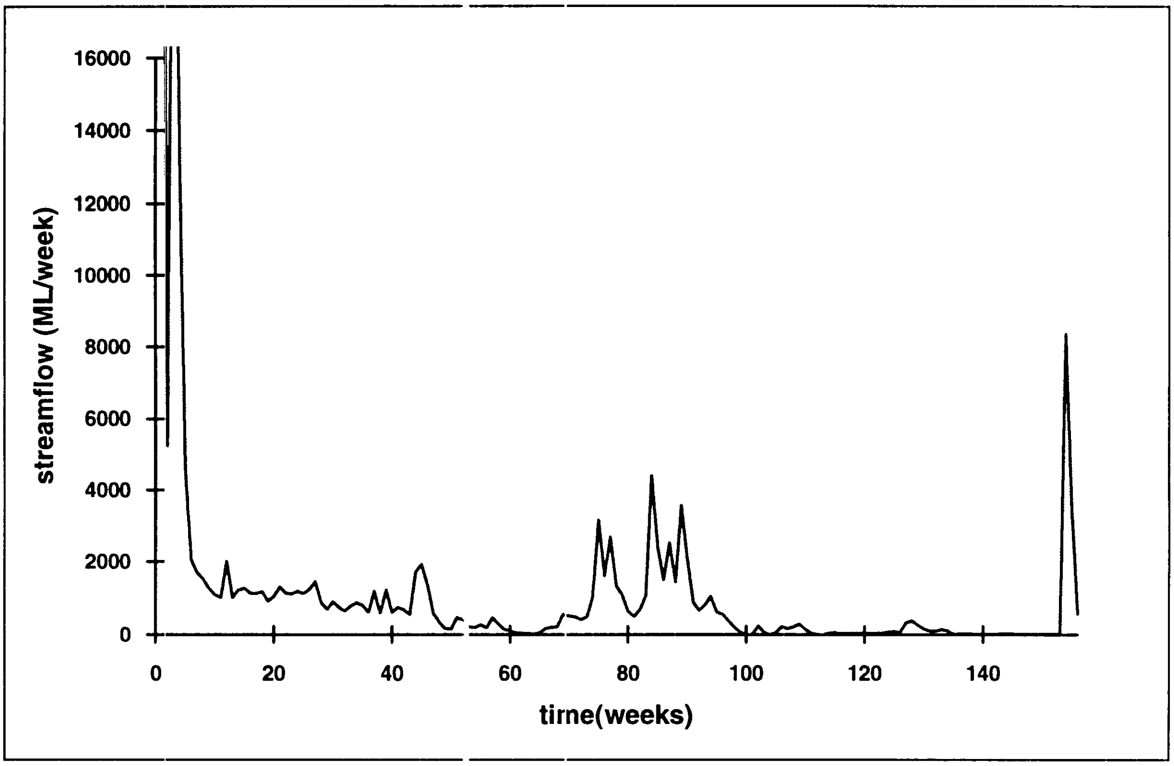


Figure 5.3 Streamflow at Sandy Hollow (Goulburn R.) 1992-95

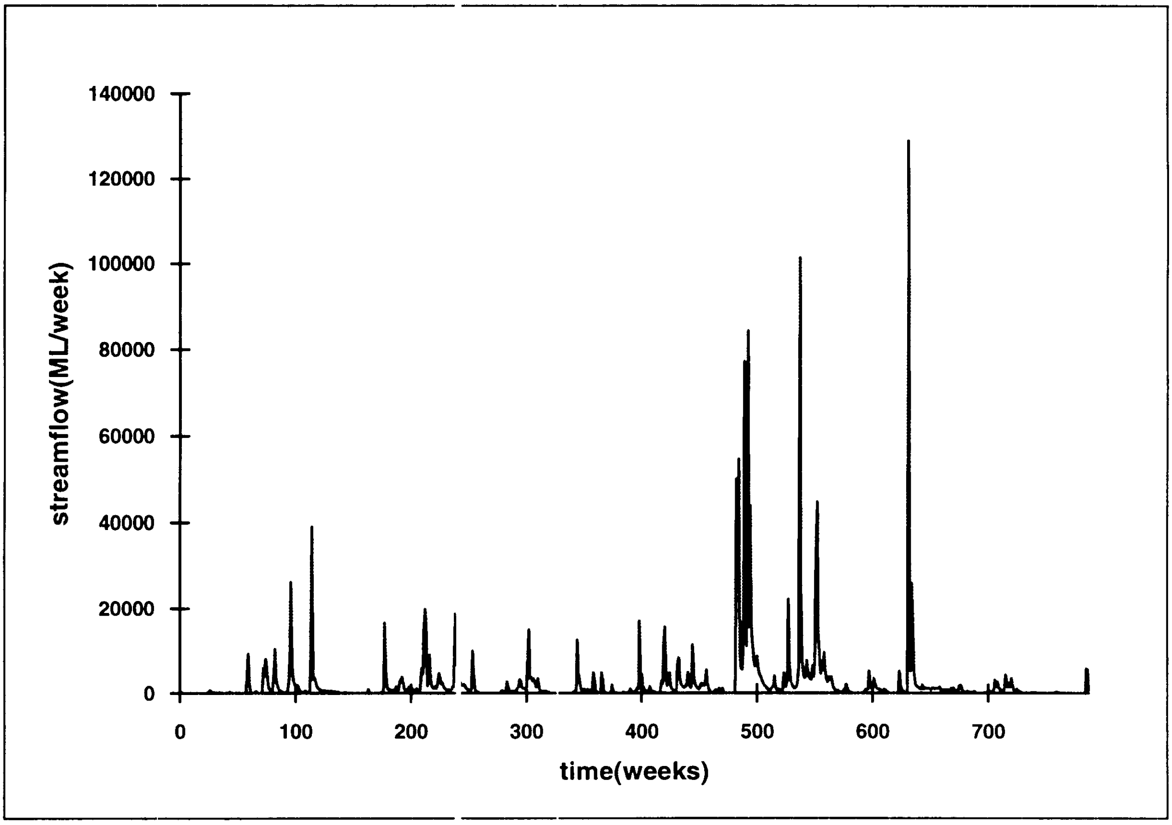


Figure 5.4 Streamflow at Sandy Hollow 1980-1995

Conductivity

Late in 1991, continuous conductivity and water level recorders were installed at five of the gauging sites in the Hunter River Basin, namely Muswellbrook, Sandy Hollow, Liddell, Wollombi and Greta. The recordings came on line early in 1992, and the first two of these locations are used in this project.

Conductivity at Muswellbrook

Conductivity readings supplied by the Department of Water Resources for the period, February 20, 1992 to April 4 1995 at Muswellbrook are mostly below the environmental standard, of 420 mg/L, as shown in Figure 5.5.

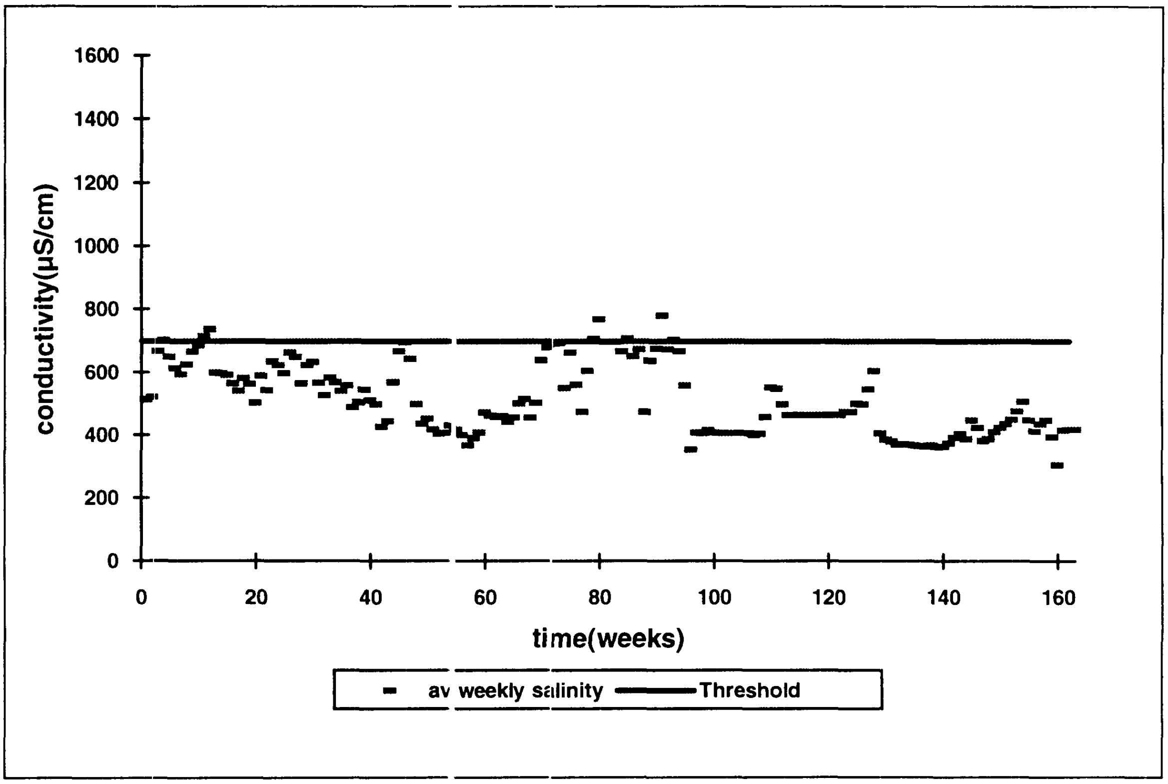


Figure 5.5 Conductivity at Muswellbrook 1992-1995

Conductivity at Sandy Hollow on the Goulburn River

Figure 5.6 shows readings from the recording station at Sandy Hollow are considerably higher than those for Muswellbrook, with the majority of readings above 420 mg/L (700 µS/cm). Sandy Hollow is located low in a catchment which contains highly saline Permian rocks. The higher salinity readings are supported in the report by AGC Woodward-Clyde (1992), who note that this site is characterised by high variability in conductivity readings which range from 180 mg/L (300 µS/cm) to 1200 mg/L (2000 µS/cm).

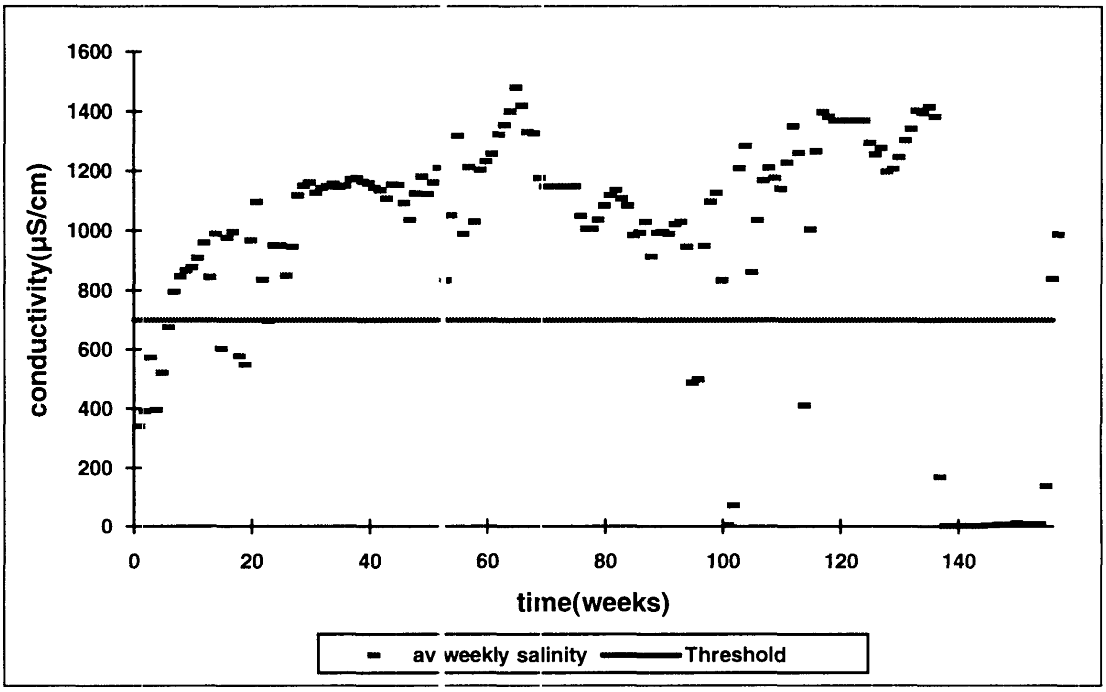


Figure 5.6 Conductivity at Sandy Hollow (Goulburn R.) 1992-1995

5.2.4 Flow and conductivity relationships

Muswellbrook

Generally speaking there is an inverse relationship between conductivity and streamflow, this can be seen in Figure 5.7, and is also supported in the negative coefficients and t-ratios for flow reported in the data analysis performed later in this section. A visual inspection of Figure 5.7 shows some degree of heteroskedasticity, where conductivity displays greater variability at high flows. In general, however, both the inverse relationship and heteroskedasticity appears weak for the Muswellbrook site.

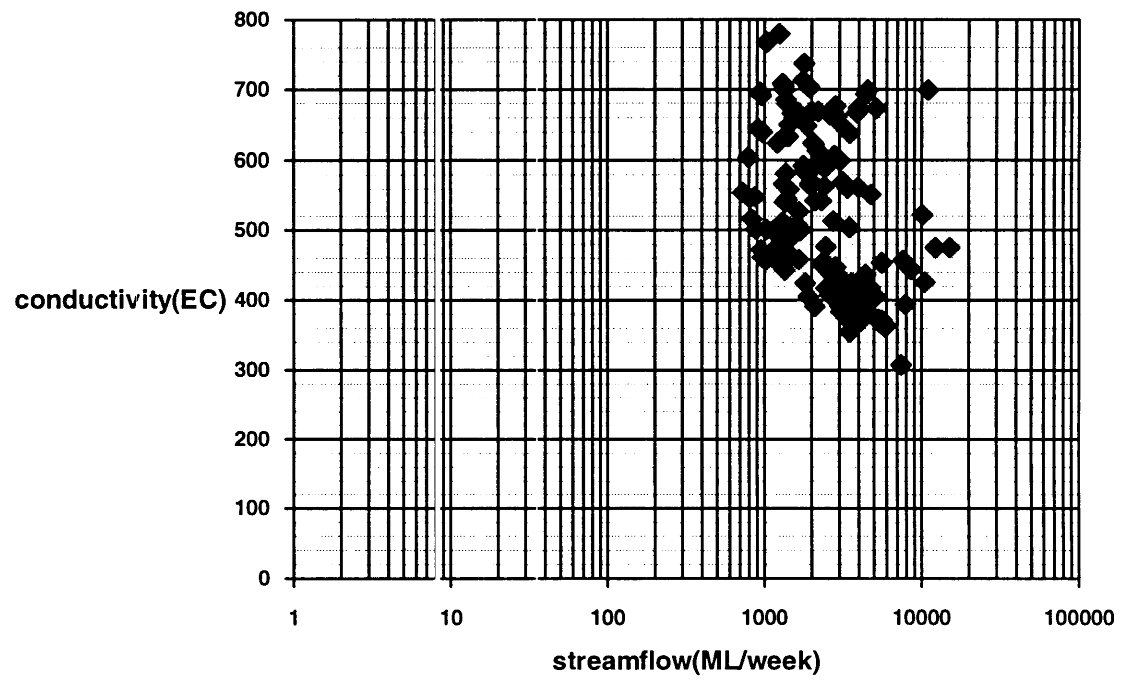


Figure 5.7 Streamflow-conductivity relationship at Muswellbrook (1992-1995)

Sandy Hollow

The inverse relationship between streamflow and conductivity appears for the main part to be stronger at Sandy Hollow, than at Muswellbrook, see Figure 5.8. At Sandy Hollow, large variation in conductivity recordings are shown at low flows, and the small number of conductivity readings available for high flows makes it impossible to judge the presence of heteroskedasticity.

Continuous conductivity data for the period, 1980-1991 was unavailable. Generating this data was necessary in order to give greater insight into the assimilative capacity of the Hunter River, over a longer time frame, than the two years for which data was available.

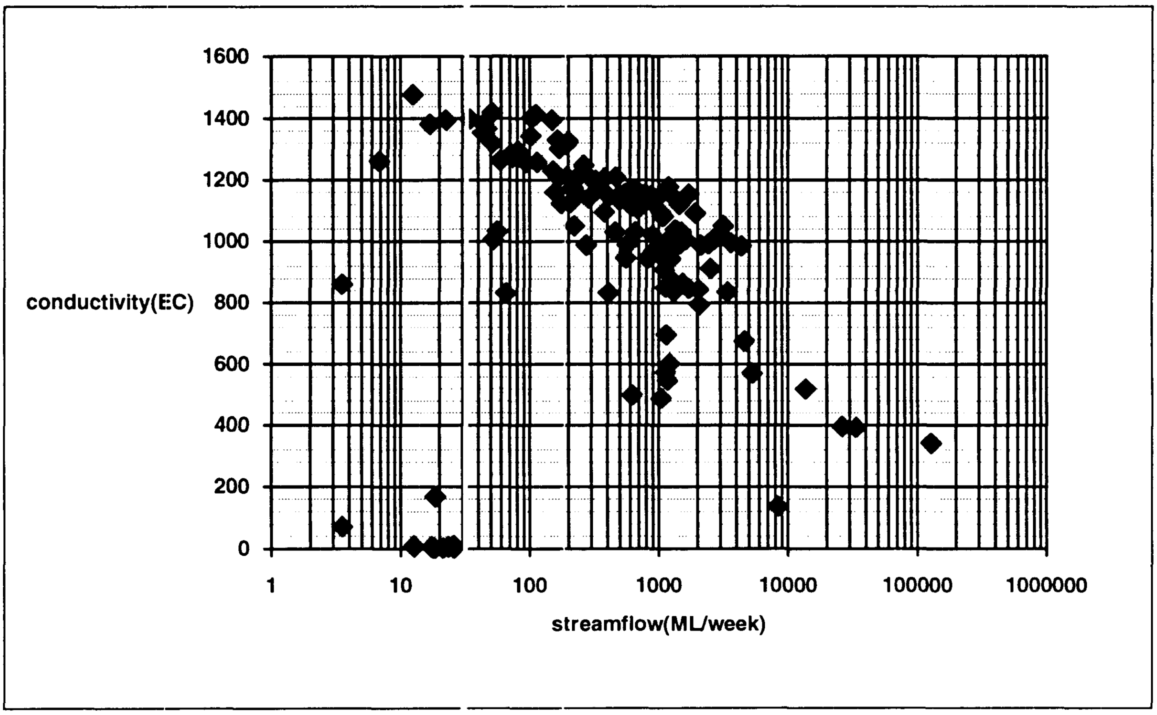


Figure 5.8 Streamflow-conductivity relationship at Sandy Hollow (1992-1995)

The infinite distributed lag model described in the following section was chosen over other lagged models and simple linear regression. Sampling procedures using appropriate frequency distributions are not used because it is difficult to capture the sequential nature of streamflows and the associated seasonality.

5.2.5 Generating conductivity data for the period 1980-1991

The method used for obtaining conductivity values used in the tax model is outlined as follows. Daily conductivity readings (1992-1995) were aggregated to weekly conductivity values using a weighted average (that is, using mean daily flow to weight the relative contributions to the weekly conductivity value). Attempts were then made to model the inverse relationship between flow and conductivity observed in Figures 5.7 and 5.8. This was done by firstly considering a simple linear regression model of flow upon conductivity. The simple regressions of conductivity on streamflow explained no more than 16 per cent of the variation in conductivities. Therefore, a range of finite- and infinite-distributed lag models were fitted to the data so as to consider the possibility that past streamflows may help explain more of the variation in conductivities. The following infinite distributed lag model was selected.

$$con_t = \beta_0 + \beta_1 flow_t + \beta_2 con_{t-1} + e_t, \qquad t=1,2,...,T, \qquad (5.1)$$

where con_t is the conductivity in week t;

$flow_t$ is the river flow in week t;

the β_i are unknown parameters;

and e_t is an error term.

The structure of this model is the same as that of the adaptive expectations model (Griffiths, Hill and Judge 1993). Ordinary least squares (OLS) estimates of this equation are presented in Table 5. .

Table 5.1 Infinite distributed lag model - OLS results

	Muswellbrook 210002	Sandy Hollow 210031
β_0	143.33	202.23
β_1	-0.011817	-0.003349
	(-5.7991)	(-0.58481)
β_2	0.78286	0.79244
	(19.[illegible]60)	(16.363)
std error	54.853	252.22
R-squared	0.7511	0.6493
Durbins HStat	2.9221	-1.7913

T-ratios shown in brackets

The R-squared values are considerably higher than those obtained from the simple linear regressions. These indicate that the estimated models explain 75 per cent and 65 per cent of variation in conductivities at Muswellbrook and Sandy Hollow, respectively.

The t-ratio for flow is significant (at the 5 per cent level) for Muswellbrook but not for Sandy Hollow. Both have negative signs, as expected for the inverse relationship between flow and conductivity. The t-ratios for lagged conductivity are large and significant for both stations.

The regression coefficients for lagged conductivity are positive and between 0 and 1 in value as one would expect for the adaptive expectations model (see Griffiths et al., 1993). It is noted that the Durbins H Statistic for Muswellbrook is significant. This indicates that autocorrelation could be a problem for this data set. This problem was not considered relevant for the present analysis as the objective is to generate a realistic data series for conductivity rather than to determine the 'true' model structure.

Model 5.1 was used to generate conductivity values for each of the stations. Conductivities were generated for each week for each gauging station using equations 5.2 and 5.3 below.

$$con002_t = 143.83-0.011817(flo002_t) + 0.78286(con002_{t-1}) + nor(54.863) \qquad (5.2)$$

$$con031_t = 202.23 - 0.003349(flo031_t) + 0.79244(con031_{t-1}) + nor(252.22) \qquad (5.3)$$

The coefficient values in these equations are taken from the regression results in Table 5.1. Equation 5.2 gave the salinity values for station number 210002 (Muswellbrook), and equation 5.3 for station number 210031 (Sandy Hollow). The random error term was added to reflect the uncertainty and unpredictability found in the stream The estimated standard errors were used to sample from a normal distribution with zero mean via the SHAZAM command nor(standard deviation(SD)). For Muswellbrook, for instance, the error term was sampled from a normal distribution with mean 0 and SD 54.863.

The values obtained from equations 5.2 and 5.3 were aggregated into single river flow values and river salt values. River salt was given by the weighted sum of weekly conductivity and flow values.

$$\text{River Salt (T/week)} = (\text{flo002} \times \text{con002} + \text{flo031} \times \text{con031}) \times 0.6/1000 \quad (5.4)$$

Figure 5.9 shows the generated salinity values and streamflow values used in the tax model for the three year period 1980-82.

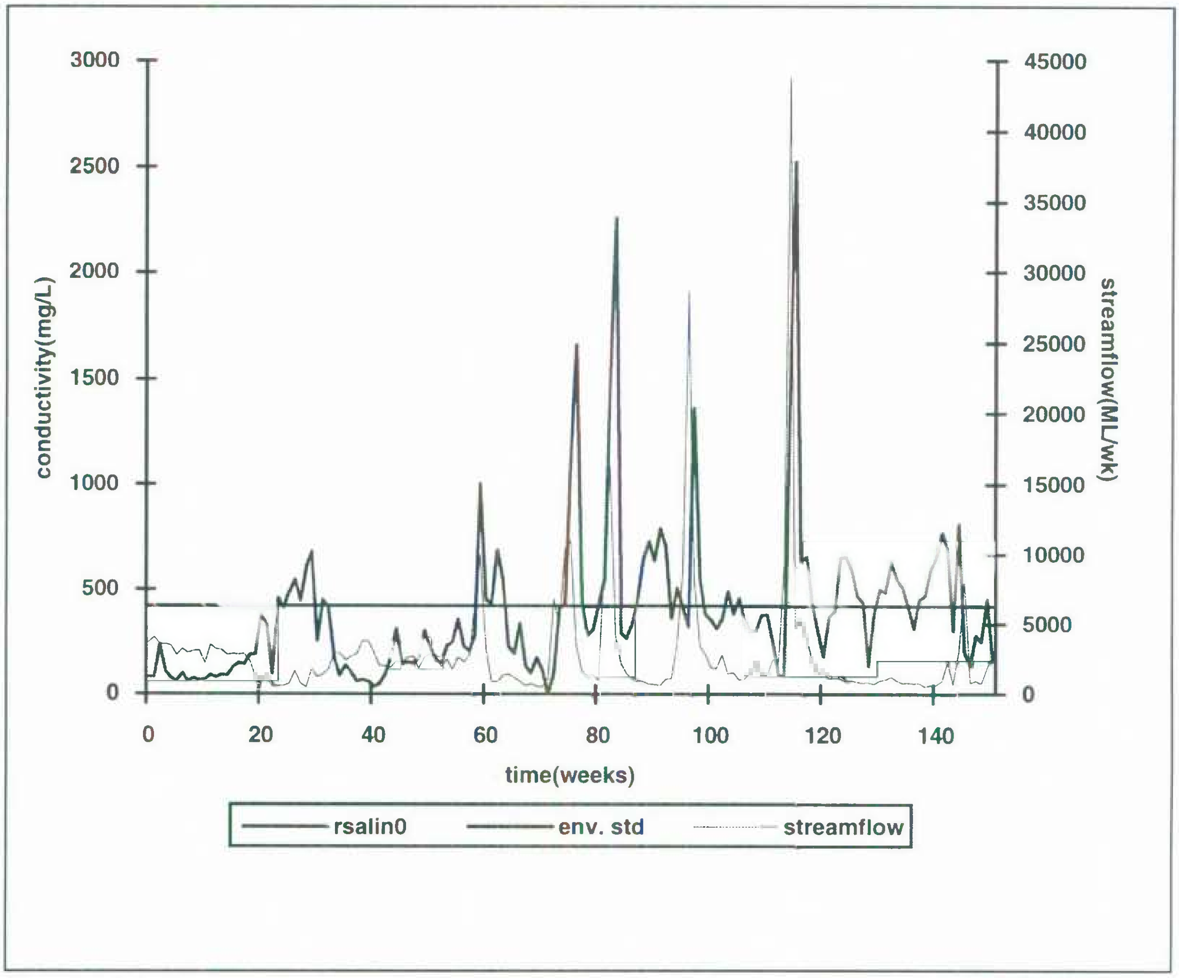


Figure 5.9 Generated river salinity and observed streamflow data (1980-1983)

Figure 5.10 shows the complete data set used in the model for the 15 year period, 1980-1994. The generated conductivity shows several very high spikes of much greater magnitude than would be expected. Once the river salinity exceeds the environmental standard the magnitude of the salinity is of no consequence in the simulation model. Once the river salinity exceeds the threshold assimilative capacity is no longer positive and so the tax rate ensures that discharge is zero. The number of data points which exceed the environmental standard are of more importance, and this appears to be reasonably consistent with other studies (see AGC Woodward-Clyde 1992; PPK 1994).

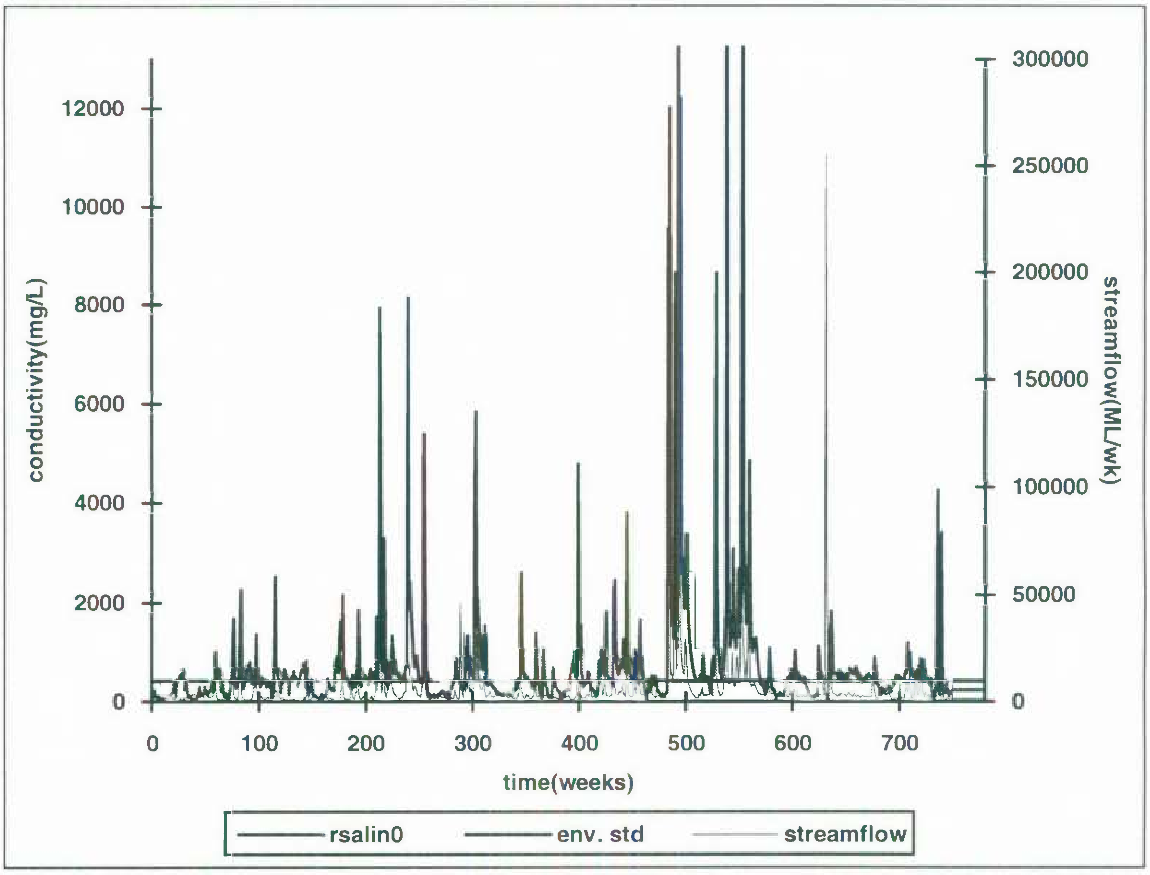


Figure 5.10 Generated river salinity and observed streamflow data (1980-1994)

5.2.6 Quality of data

Streamflow

Measurement of river flow prior to 1980 was not part of the daily recording procedure. Streamflow is measured using standard procedures, whereby river height is measured in metres and converted into megalitres of flow using a calibration curve. The calibration where the streambed is in a state of perpetual change through erosion and deposition processes, such as would occur on the floodplain (for example Singleton), the streamflow measurements may be less reliable. For the two sites used in this project, the streambed is relatively unchanging and the streamflow measurements are reliable.

Conductivity

Conductivity records used in this project were obtained from continuous conductivity recordings only. Prior to the installation of this equipment at selected sites, conductivity measurements were undertaken for the main purpose of warning irrigators of salinity spikes. These recordings were less consistent than the current system, and measured irregularly, up to weekly, during the irrigation season, and less often at other times of the year. Since time series is of importance in simulating the assimilative capacity of the river in this model, only the more accurate and reliable data has been used, despite the short time span it covers.

The lack of continuous records measuring salinity at the gauging stations for the period prior to 1992, presented the problem that only an extremely short snapshot in time could be looked at in the model. This would mean that historic flow and conductivity sequences in the river could not be reasonably measured. The time frame covered, as discussed earlier, covered only periods of relatively low flow. This limitation on data is quite serious because it provides a skewed data set, biased toward low flow and relatively high salinity readings. This bias to low flows has two main consequences. First, if the observed data set was to be used for the short period that it covers, the low assimilative capacity of the river would cause the results of the model to be extremely conservative, with high tax rates forcing low discharges and high treatment costs. To try to reflect a more varied range of streamflows, the model could utilise the 15 years of streamflow available. To do this, however, the conductivity values for the same period would need to be estimated. A range of econometric methods are available for this type of predictive work, however the biased data set lowers the accuracy for predicting conductivities associated with high streamflows. This poor predictive capacity at high streamflows is the second consequence of this skewed data set. Despite these shortcomings the extra information that the model can generate from covering the shortcomings the extra information that the model can generate from covering the greater period of time, makes the generation of conductivity values, as presented in section 5.2.5, worthwhile.

Overall the quality of data is good, however, the amount of missing data is significant for both conductivity and streamflow measurements. Figures 5.11-5.14 show the periods of missing data (represented by blank bars) for both variables, for each of the gauging stations for the period 1992-1995.

Missing streamflow data was replaced by averaging the streamflow before and after the missing data. The average of these two values was then used for all missing values in that period. This method was used for all missing streamflow and conductivity data.

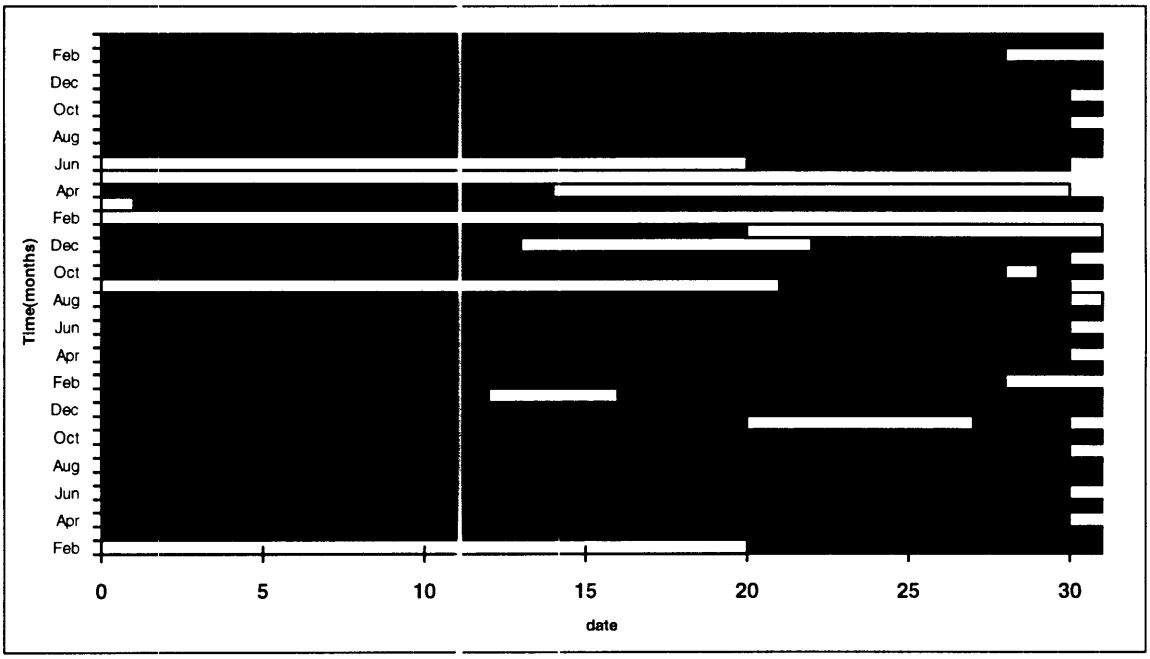


Figure 5.11 Missing data (blank bars) from conductivity recordings for Muswellbrook (1992-95)

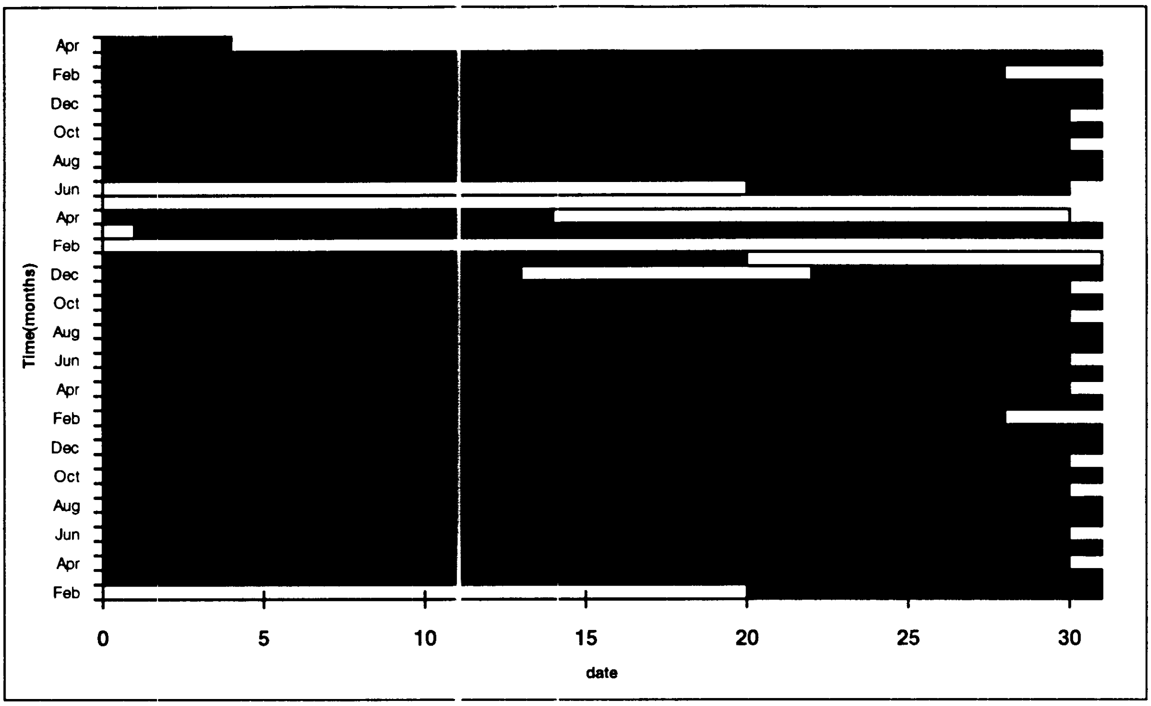


Figure 5.12 Missing streamflow (blank bars) data for Muswellbrook (1992-1995)

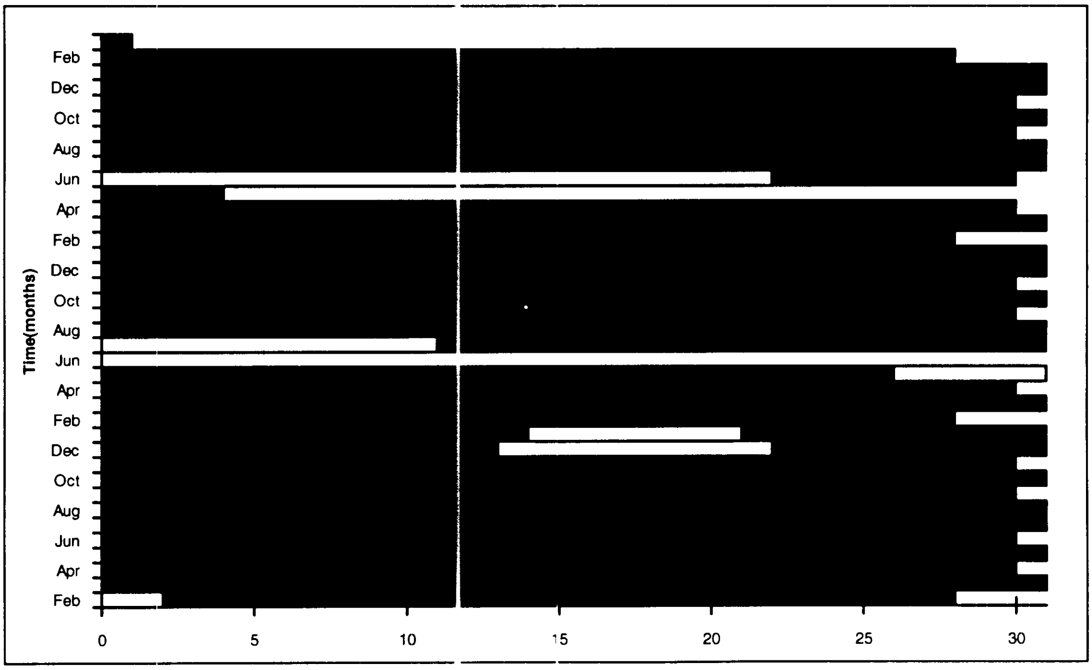


Figure 5.13 Missing data (blank bars) from Sandy Hollow conductivity data (1992-95)

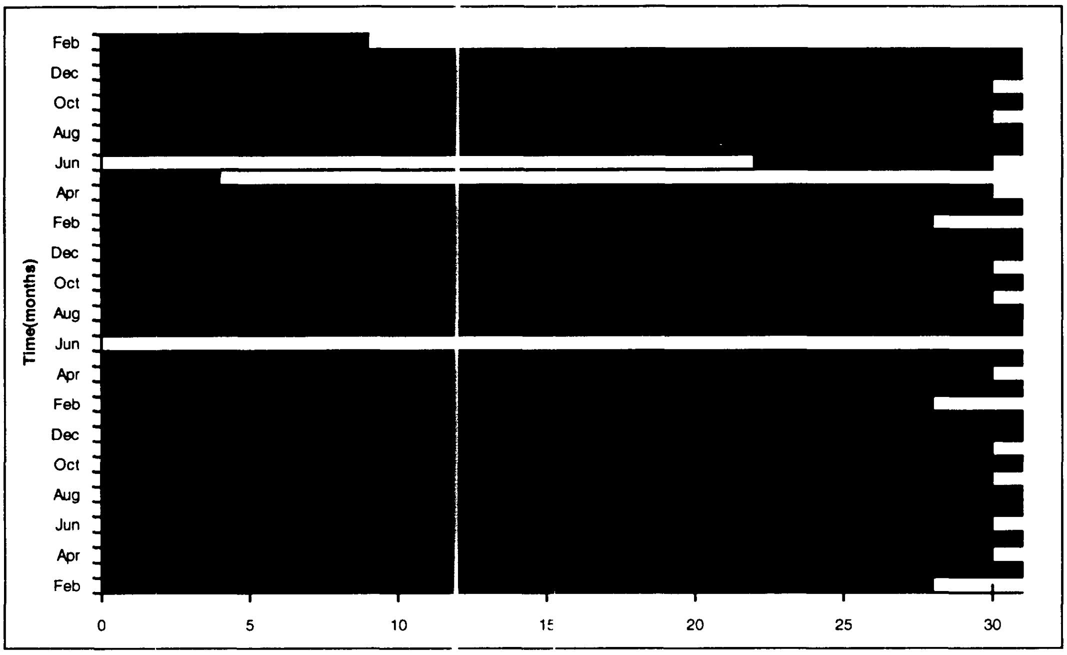


Figure 5.14 Missing streamflow records (blank bars) for Sandy Hollow (1992-1995)

5.3 Mine inflow data

The inflow data used in the model was estimated algebraically from tables presented in AGC Woodward-Clyde (1992). It was assumed that mine inflow is a function of streamflow described by the equation:

$$\text{Inflow} = \alpha_I \times \text{StrFlo}^{\beta_I} \tag{5.5}$$

where $\alpha_I = 20.94$

$\beta_I = 0.31$

Equation 5.5 provides annual mine inflows to the mine that are similar to the annual minewater excesses presented in the draft AGC report, which canvassed 9 of the then 11 mine operators for their estimates of minewater excess.

Limitations of mine inflow data

Mine inflow is not only a function of streamflow, it is also a function of runoff from the mine site and of groundwater seepage. The model currently simulates a single discharging firm, so the variations in mine inflow were not accounted for. The individual inflow to mine sites would vary with location in the catchment, rainfall distribution and the type of catchment surface. The simulation of these factors is out of the scope of this dissertation; however, the aggregate inflow to mines can be indirectly captured by streamflow, since this variable is also a function of rainfall and runoff. Under most circumstances aggregate inflow can be expected to vary as described above; however, during uncommon events such as floods, the structure of the function may change. No attempt was made to estimate an equation for these events.

5.4 Storage capacity

The size of on site storage reservoirs in the model was assumed to be 6000 ML. This is the sum of mine storages for the 9 mines who responded to the survey conducted in the AGC Woodward-Clyde (1992a) report. As such, this would represent the lower bound on storage size for the single discharger modelled here. With the entry of new mining sites the construction of additional storages may be possible, however, for the mines currently operating it was assumed that there was no possibility to increase the size of storages given the geological and physical constraints of the mine sites.

5.5 Summary

This chapter presented an analysis of the data used to calibrate the model. The methods used to generate conductivity and mine inflow were described and the quality of each data set was discussed. Overall, the data used in the model provides a reasonably accurate picture of the fluctuations in river flow and salinity levels which can be expected in the Hunter system. The results from model runs using the input data described here are presented in the next chapter.