Evaluating Policy Options for Integrated and Cost Effective River Management in the Tidal Ouse

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Abstract

The River Ouse forms a significant part of the Humber river system, which drains about one fifth of the land area of England and provides the largest fresh water input to the North Sea from the UK. The tidal Ouse has suffered from a sag of dissolved oxygen (DO) during the last few decades, caused by effluent discharges from industries and Sewage Treatment Works (STWs). Poor water quality during the summer prevents the return of salmon, which is regarded by the Environment Agency (EA) as a key indicator of rivers' ecological health. The EA proposed to increase water quality in the Ouse by implementing more stringent environmental policies. This conventional management option, however, usually offers less flexibility in compliance and incurs excessive costs of pollution abatement to industries and STWs.

This thesis explores the potential to improve water management by adopting an integrated and cost effective river policy, which allows for variation in the assimilative capacity of river water. Various options to improve water quality are considered in a comprehensive framework for river policy. Reduction in both effluent discharges and water abstraction are considered together with choice of location for effluent discharge. Different instruments of environmental policy, tax-subsidy scheme (TSS) and tradable pollution permits (TPP) systems are compared with the command and control (CAC) approach. A hydrological model from the EA is combined with an economic model to identify the least cost solution for water quality management in the river system. This thesis provides a theoretical discussion of this problem in both static and dynamic settings. This framework is then applied to the empirical case of the tidal Ouse for particular water quality targets. To achieve the water quality target at least cost is a constrained optimisation problem, solved by computing software. The integrated river policy is able to achieve a significant improvement in water quality at lower cost than is currently incurred. This thesis also compares the different policy instruments for delivering this water quality improvement in the tidal Ouse.

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List of Abbreviations

APS	Ambient Permit System
BAT	Best Available Techniques
BATNEEC	Best Available Techniques Not Entailing Excessive Cost
BAU	Business As Usual
BOD_5	Five Day Biochemical Oxygen Demand
CAC	Command and Control
CAMS	Catchment Abstraction Management Strategies
CEH	Centre for Ecology & Hydrology
CIS	Common Implementation Strategy
CO_2	carbon dioxide
Defra	Department for Environment, Food and Rural Affairs
DIP	Dissolved Inorganic Phosphate
DO	Dissolved Oxygen
DO%	Dissolved Oxygen saturation
EA	Environment Agency
EC	European Commission
EPS	Effluent Permit System
ETP	Effluent Treatment Plant
EU	European Union
EWPCS	Estuarine Working Party Classification Scheme
FOCs	First Order Conditions
GAMS	General Algebraic Modelling System
GIS	Geographic Information System
GQA	General Quality Assessment
HMIP	Her Majesty's Inspectorate of Pollution
HOT	Humber-Ouse-Trent
IPC	Integrated Pollution Control
IPPC	Integrated Pollution Prevention and Control
LOIS	Land-Ocean Interaction Study
MBIs	Market-Based Instruments
MCA	Marginal Cost of Abatement

ML	Mega Litre
NERC	Natural Environment Research Council
NRA	National Rivers Authority
NH_3	ammonia
PIP	Particulate Inorganic Phosphate
PO	Pollution Offset
POC	Particulate Organic Carbon
PPC	Pollution Prevention Control
RACS(C)	Rivers Atmosphere, Estuaries and Coasts Study (Coasts)
RBD	River Basin District
RE	River Ecosystem
SS	Suspended Solids
SPM	Suspended Particulate Matter
STW	Sewage Treatment Work
TCM	Transfer Coefficient Matrix
TLCA	Tate & Lyle Citric Acid Ltd
TM	Turbidity Maximum
TPP	Tradable Pollution Permit
TSS	Tax-Subsidy Scheme
TWAL	Tradable Water Abstraction License
UKTAG	UK Technical Advisory Group
UWWTD	Urban Waste Water Treatment Directive
WFD	Water Framework Directive
WQM	Water Quality Monitoring
WQO	Water Quality Objectives
YW	Yorkshire Water

List of Mathematic Variables

A	Jacobian matrix
A_s	ambient water quality at WQM site s
$P_i(q_i, a_i)$	net benefit (profit) of firm
B_{sel}	aggregated BOD_5 inload to ETPs in Selby sources
$C_i(q_i, a_i)$	production or abatement cost of firm
E_s	aggregate impacts of industrial emissions to WQM site s
H_s	aggregate impacts of water abstractions to WQM site s
L	Lagrange Function
H	Hamiltonian Function
$\overline{Q}_s$	ambient water quality target at WQM site s
Q_s	water quality at any WQM site s
T_{ie}	sum of tax and subsidy for effluent discharge from site i
T_{ia}	sum of tax and subsidy for water abstraction from site i
p	exogenous product price
t	time
ρ	discounting factor
r	interest rate
j	sectors of activity (output, abatement and abstraction)
q_i	product output from site i
a_i	level of abatement activity at site i
e_i	emission level at site i
β_i	water abstraction level at site i
b_{is}	transfer coefficient of effluent from site i to WQM site s
d_{is}	transfer coefficient of abstraction from site i to WQM site s
ε_s	other environmental factors influencing the water quality
γ_s	variations not captured explicitly by this function.

λ_s	Lagrange Multiplier
t_{es}	tax or subsidy rate for effluent discharge at WQM site s
t_{as}	tax or subsidy rate for water abstraction at WQM site s
P_{es}	permit price for effluent discharge at WQM site s
P_{as}	permit price for water abstraction at WQM site s
P_{ei}	permit price for effluent discharge for pollution source at site i
P_{ai}	permit price for water abstraction for pollution source at site i
δ_i^j	depreciation rate of capital stock in sector j at site i
k_i^j	capital stock in sector j at site i
I_i^j	investment in sector j at site i
μ_i^j	co-state variables of Hamiltonian
η	eigenvalues (or characteristic roots) of the Jacobian matrix
X	distance of effluent discharge from the Trent Falls
SBOD	aggregated BOD5 discharged from the sources in Selby
Ouse	average river flow at the head of the river Ouse
Derw	average river flow at the confluence of the river Derwent
Sna	BOD5 discharged from the STW Snaith
Sand	BOD5 discharged from the STW Sandall
Tho	BOD5 discharged from the STW Thorne
$Cost_{ind}$	abatement cost of industry
$Cost_{STW}$	abatement cost of STWs
$Cost_{abs}$	cost of water abstraction
$Cost_{mov}$	cost of moving effluent discharge location

I DEDICATE THIS THESIS TO MY DEAR PARENTS

AND WIFE

YONGJU WANG AND XIAOFENG LEI

BINGYAN WANG

WITH LOVE

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First of all I would like say thanks to Professor Charles Perrings, one of my supervisors, for his encouragement during my MSc study, which gave me the courage and confidence to take the challenge of doing a PhD research, without which this thesis will not be here today. I also thank him for his suggestions and comments to my work during these years, which are always prompt (unless he is stuck somewhere without internet) and stimulating to my dull mind.

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Author's Declaration

Some results from Chapters 4 and 6 of this thesis have been presented at the 2006 European Summer School in Resource and Environmental Economics held in Venice, Italy from 25th June to 1st July 2006, organized by European Association of Environmental and Resources Economists (EAERE), the Fondazione Eni Enrico Mattei (FEEM), and the Venice International University (VIU). A similar draft of these results was also presented at the Third World Congress of Environmental and Resource Economists, held on 3rd to 7th July, 2006, at Kyoto Japan, organized by the Japanese Association of Environmental Economists (SEEPS), the Association of Environmental and Resource Economists (AERE), the European Association of Environmental and Resource Economists (EAERE), and the Latin American and Caribbean Association of Environmental and Resource Economists (ALEAR).

A draft of Chapter 3 of this thesis has been submitted to the journal of Science of the Total Environment, which is now under revision together with Professor Malcolm Cresser, Richard Freestone and Trevor Hardy and will be submit again very soon. A part of the Chapter 3 of this thesis was presented at the Internal Conference of River Basin Management 2005, held in Bologna Italy, from 6 - 8 September 2005, jointly organized by Wessex Institute of Technology, UK and University of Coimbra, Portugal.

With these exceptions, I declare that the work contained in this thesis is my own and has not been admitted from any other degree or award.

Chapter 1 Introduction

1.1 Introduction of Objectives

1.1.1 Review of issues in river policy

The tidal section of the Humber system forms a significant part of the Humber drainage basin, which is the largest catchment in England, draining one fifth of the land area of England alone (Edwards *et al.* 1997; Jarvie *et al.* 1997b). The sea spurn of the Humber makes the biggest freshwater contribution to the North Sea from the U.K, approximately 250 m^3/s (National Rivers Authority 1993). The tidal Ouse is an upper part of the tidal Humber system, stretching from Naburn to Trent Fall where it meets the tidal Trent, and includes four tributaries, the Wharfe, Derwent, Aire and Don. Water quality in the tidal Trent has been steadily improved over the last three decades (Edwards *et al.* 1997). However, the tidal Ouse remains one of the worst river reaches in the tidal section of the Humber system due to a number of factors. The poor water quality in the tidal Ouse has a significant negative impact on those of other river reaches in the Humber system due to its tidal nature. It also has negative impacts on the ecological systems supported by these rivers, on the economic activities related to water quality and ecological systems, and on various socio-economic attributes that are to be taken into account.

One of the obvious impacts of the poor water quality in the tidal Ouse during the warm summer months is the regular occurrence of dissolved oxygen (DO) sag, a common phenomenon of estuaries (Cashman *et al.* 1999). When the river flow is low, suspended sediments in the river move upstream and stay long enough around Selby to cause the observed DO sag. Because of the DO sag, the water quality in the tidal Ouse is too low to support the return of spawning salmon, which is regarded as an important indicator of ecological health of an estuarine river. The decline of the salmon stock in the Ouse system is influenced by a number of other factors external to the Ouse, which may include over-fishing off Greenland, commercial netting in estuaries, habitat loss, increasing sediment load and river morphology changes. However, the effluent discharges from the industries in Selby and sewage treatment works (STWs) along the river are regarded as one of the main reasons for the poor water quality and the decline of salmon, combined with the water abstraction in the catchment. They are believed to cause deterioration in the water quality, which was particularly highlighted during the dry summer of 1995, 1996 and 2003. The effluent from the industries and STWs affect the water quality by the effluents discharged into the river water. Five Day Biochemical Oxygen Demand (BOD_5) measures the amount of oxygen consumed by biochemical oxidation of pollutants in a five-day period (Standing Committee of Analysts 1989) and is regarded by the Environment Agency as an important indicator of water quality.

Some other natural factors also contribute to the decline of DO level in the tidal Ouse. Rainfall varies dramatically over space and time in the catchment region, with highest rainfall over 1600 mm p.a. in parts of the Pennines due to the prevailing wind and in the winter, and much less rainfall in the Southeast catchment and during dry summers (Law *et al.* 1997). The inland penetration of tides during low flow transports sediments upstream, while resuspension of sediments results in considerable DO consumption. The impact of the suspended sediments on water quality in the tidal Ouse has not been investigated in depth, but some estimates have been made, based on modelling on its DO consumption and transport (Freestone 2003; Tappin *et al.* 2003). The relatively high temperature in summer months, as well as the biomass of photosynthetic plankton also decreases the DO level. In addition, large quantities of river water are abstracted and transferred through its grid by Yorkshire Water to supply potable water for over 3.5 million people, and returned to the river system through sewage treatment works. One obvious effect on the water quality in the tidal Ouse related to water abstraction is the reduction of clean freshwater flows from northern rivers and rising volume of poor quality water returned from the industrial south tributaries (Edwards *et al.* 1997). The most severe DO sag in the summer persists in the upper reaches of the river between the Environment Agency (EA) water quality monitoring (WQM) sites at Selby and Long Drax.

The EA intends to improve river water quality by tightening discharge consents in Selby. A new system of pollution control is being implemented in order to restore water quality in the Ouse, which is driven by the EU Directive on Integrated Pollution Prevention and Control (IPPC). The essence of IPPC is that operators should choose the best option available to achieve an agreed level of protection of the environment taken as a whole. The Best Available Techniques (BAT) approach is typically modified by the declaration that the cost of applying techniques should not be excessive in relation to the environmental protection it provides. However, the IPPC Scheme requires BAT to be applied in the abatement of pollution while no clear definition of BAT is provided. A more rigorous way of addressing the issue of cost is to identify the most cost effective river policy for a given water quality target, by considering not only one but various factors affecting the water quality and their relative impacts.

Water abstraction has direct impacts on river water quality similar to effluent discharges, but is rarely considered in water quality regulation. Since river volume affects the assimilative capacity, it is apparent that water abstraction has adverse impacts on the river water quality, and the impacts are interdependent of the impacts of effluent discharged into the river body. The impacts of water abstraction on water resources is usually emphasized (Willis and Garrod 1998), but not as much in the perspective of water quality change. Therefore, it is necessary to include both industrial effluent and water abstraction in an integrated regulation system. To date, however, effluent discharge consents and water abstraction licenses have not taken into account either the variation in the assimilative capacity of river or the interdependence of these two activities.

Effluent discharge and water abstraction in the tidal Ouse and the Humber estuary are currently regulated by two different policies implemented by the EA. These are discharge consents for effluent discharge and the system of Tradable Water Abstraction License (TWAL) for water abstractions. For the purpose of improving water quality in the tidal Ouse, the effluent discharge consents to the Selby industries and major STWs along the river have been significantly modified during the last few years. Changes in effluent discharge consents for the STWs were recently tightened up by the European Urban Waste Water Treatment Directive (UWWTD) (Defra 2002a). Water abstraction in the Ouse and its tributaries has not been regulated for the purpose to address the DO sag while even more was abstracted during droughts such as 1995 and 1996 to guarantee the potable water supply.

Four plants in Selby were regulated by the improved effluent discharge consents: Tate & Lyle Citric Acid (TLCA), Greencore, Rigid Paper and BOCM. BOD_5 is an important indicator monitored in the effluent discharge consents, so are the concentrations of ammonia (NH_3), phosphorous and suspended solids (SS). The aggregate effluent discharge consents on BOD_5 from Selby as a whole have been reduced from some 30 tonnes per day to 3 tonnes per day over the last few decades. The four companies have continuously invested in effluent treatment plants at their home sites in order to comply with the changes in effluent discharge consents. Except BOCM, the other three industries are now applying similar anaerobic treatment to their effluent before disposal. BOCM has recently closed one of its production plants and shut off its direct BOD_5 discharge into the tidal Ouse. This decision was said purely based upon business and economic arguments, but there is a question mark over the influence of the consents upon the final decision. In an industrial town with a long history, these plants have been contributing to the local economy through direct and indirect means. The general recession in manufacturing industry made the industries in Selby more sensitive to regulation change and the consequent requirement for investment in their pollution abatement processes (Jarvie *et al.* 1997b; Cashman *et al.* 1999). Due to the lower average income in Selby compared with other towns in North Yorkshire and the general recession in manufacturing industry (Edwards *et al.* 1997; Jarvie *et al.* 1997b), the extra costs imposed by non-cost-effective regulations may have significant impact on the local economy and residents.

The STWs, as stated above, are currently responding to the revision of their effluent discharge consents and improving their sewage treatment capabilities to meet the domestic requirements of the UWWTD (HMSO 1994; Defra 2002a; HMSO 2003). Some major STWs have improved their effluent quality since 2000, while improvements in the remaining small-sized STWs are due by the end of

2005 (Defra 2002a). Water quality in the Don and Aire tributaries has also improved steadily due to continuous improvements in the STWs and industries' effort in these catchments (Edwards *et al.* 1997; Defra 2003a). Implementation of the UWWTD at the small STWs is expected to bring further improvements in water quality and reduce the DO sag problem in the tidal Ouse.

Unlike Southeast of England which now experiences severe drought in summer (Environment Agency 2006), the Ouse and Humber catchment is one of the few areas have additional water resource available in summer (Environment Agency 2001), but the water resource need to be well managed to ensure good quality. The major water abstraction from the water company and industries is regulated by TWAL in the tidal Ouse. An abstraction licence generally states how much water can be taken, from where, the way it is to be used and where to it is to be returned to the river. It usually takes the form of a fixed and constant amount for each year during the period licensed, regardless of the actual river flow volume. In a recent amendment, TWAL were suggested to be time limited and can only be renewed thereafter upon application (Defra 1999b). Water right trading is encouraged by the Environment Agency who expects to facilitate the trading process through Catchment Abstraction Management Strategies (CAMS) (Defra 1999b; Environment Agency 2002). However, at present there is hardly any transaction of trading (pers. comm. Trevor Spurgeon; Environment Agency).

The aim of the research in this thesis is to evaluate the cost effectiveness of water quality management and pollution control, taking into account effluent discharges to the river, water abstraction, and other properties of the river. Two objectives need to be achieved in order to produce meaningful results, (a) a review of the cost effectiveness of the current regulatory system of river policies for the Tidal Ouse; and (b) an evaluation of alternative regulation and instrument options for water quality control in the tidal Ouse. Investigation of the cost effectiveness of alternative regulation and instrument options will be carried out by comparing the cost incurred in achieving a given water quality target. It is anticipated that integrated river policy that takes into account both effluent discharges and water abstraction will offer considerable advantages for pollution control and water quality improvement regarding cost effectiveness. This research also aims to investigate the possibility of introducing alternative policy instruments for regulating the water qualities in the tidal Ouse and Humber, such as emission tax-subsidy or a Tradable Pollution Permit (TPP) system. Furthermore, the choice of capital investment for industry under specific environmental policy and target is to be determined by the dynamic analysis of investment equilibrium.

1.1.2 Motivation for the research

This research will focus on the cost effectiveness of effluent discharges and water abstraction in the tidal Ouse in relation to how the EA regulates the industrial effluents and water abstraction. There are several sources of inefficiency in the current regulatory system. Two of them are considered in this research: a disregard for variation in the assimilative capacity of river water, and the separation of regulations governing effluent discharges and water abstraction. The assimilative capacity, i.e. the ability of river water to self-purify after the discharge of pollution, depends on the volume of water in the river, flow velocity, surface area, temperature, and the micro plankton in the water. The three mains sources of DO recovery in the polluted river water are (a) oxygen in incoming effluents or tributary flows, (b) oxygen generated by photosynthesis and (c) oxygen from the re-aeration process. Because of this, assimilative capacity varies along the river and over time. Since the assimilative capacity determines the maximum load of pollution that the river could cope with for a given desired water quality, effluent discharge consents to the pollution sources for the desired water quality should also vary along the river and over time to avoid imposing excessive costs on the industries and to improve the cost effectiveness of pollution abatement. It is therefore necessary to include both effluent discharge and water abstraction into an integrated regulation system. However, in the tidal Ouse and Humber estuaries the fixed consents for effluent discharge and water abstraction do not yet take into account the variation in assimilative capacity of the river, nor of the interdependence between effluent discharges and water abstraction.

Because of the inefficiencies in the current regulatory system, excessive social costs will be imposed upon the industries involved and on the local economy, if improvement in river water quality is to be achieved by tightening fixed consents alone. The alternative pollution abatement options include moving the locations of effluent discharge and shifting the timing of discharges and reducing the water abstraction. Such management options could avoid imposing unnecessary costs in pollution abatement in industries and STWs for water quality control. Savings would accrue to the local economy if a more cost effective option could be implemented. Cost effectiveness is defined as the method of the least cost in achieving particular water quality improvement or pollution abatement. Cost efficiency, however, is another indicator used in cost analysis. Cost efficiency is achieved when the marginal benefit arising from water quality improvement is equal to the marginal cost of the pollution abatement that leads to this water quality improvement. Cost efficiency is not used in this research however, due to several reasons. The controversies and uncertainties surrounding the environmental valuation methods (Willig 1976; Diamond and Hausman 1994; Diamond 1996; Navrud and Pruckner 1997) associated with this quantification would lead to a less convincing result. The techniques of environmental valuation have been improved over the years and applied in many studies in the UK and worldwide (Pearce 1998; Gaterell *et al.* 1999a; 1999b; Bateman *et al.* 2002), even more than environmental issues (Hanley *et al.* 2003). But certain conditions are required for the relatively accurate estimation, which is unlikely to be satisfied in this research. Furthermore, environmental policy decisions usually reflect not only economic considerations such as cost efficiency, but also a wider set of political and ethical factors. Therefore it is less meaningful to investigate cost efficiency and argue for setting of environmental targets purely from an economic point of view. Rather it is more applicable to look at cost effectiveness in achieving the environmental target which has been determined by the authority from various considerations including economic cost.

Direct command instruments for enacting environmental policy, commonly referred to as "Command and Control (CAC)", have long been criticized by economists. They claim that CAC tends to ignore differences of marginal costs of pollution abatement among pollution sources, and differences in marginal damage between different locations. The CAC approach also provides little incentive for further pollution abatement and affords no flexibility in compliance. Economic instruments such as TPP and emission charges are advocated by economists for their advantages compared with CAC in terms of cost saving, providing dynamic incentives for abatement improvements and allowing flexibility in means of compliance. However, economic instruments have only been implemented to a limited extent in environmental policy during the last few decades and "apply with caution" has been the message following previous failures (O'Neil *et al.* 1983; Tietenberg 2006). This research therefore aims to investigate the feasibility of introducing such economic instruments in a regulatory system of river policy for the tidal Ouse, in order to avoid excessive costs in pollution abatement and to make the policy more effective both technically and economically.

The economic model to be developed in this research will explore the static and dynamic equilibria in pollution abatement and capital investment undertaken by polluters, under different policy instruments and targets. The results from the economic model of river policy should indicate the optimal investment path or choice for the polluters in a dynamic system in which capital investment and depreciation of a plant determine the pollution abatement capacity and are therefore of relevance to regulatory compliance. The analysis of differences among policy instruments should provide a useful argument for assessing the EA's regulatory decisions with regard to pollution sources.

1.2 Structure of the Thesis

The thesis tackles the problems raised above in the following ways. The second chapter starts with reviewing the background situation of the tidal Ouse, hydrological, geological, climatic and social economic conditions along the river which affect the water quality to some extent. Then it goes on to discuss the regulative system of river policy in the tidal Ouse and in general, comparison of different choices for environmental policies. Since research in this study involves application of the hydrodynamic model QUESTS1D, it also reviewed applications of the hydrological models in different river systems and for different purposes, with particular focus on the QUESTS1D model and another model used in the tidal Ouse before, ECoS3. Finally, some of the previous studies that pioneered in the combination of hydrological and economic models are reviewed.

The third chapter describes in detail the applications of QUESTS1D model in the tidal Ouse, with comparison to the results from ECoS3 model. The QUESTS1D models are used to assess the effectiveness of various options that could potentially improve water quality. The effectiveness is envisaged by the manipulated simulations in the QUESTS1D model representing various water management options. Among them, several effective ones are chosen to carry on with economic analysis in later chapters, as one part of the designed integrated cost effective river policy. Another important outcome from this chapter is the transfer coefficient matrix (TCM), which provides useful and convenient tools to the river policy maker.

Chapter 4 forms the theoretical backbone of this research, introducing the hydro-economic modelling framework in which water quality model is combined with an economic model, with the exogenous variables affecting them both at the same time. It represents the process of river policy determination in this research, balancing the different control variables, which are the various options of water quality management in this case, to achieve the water quality target, while offsetting the excessive costs in management. The theoretical analyses are carried out for both static and dynamic systems. The static analysis is a representation for short-term change with fixed abatement and abstraction capability, and the dynamic analysis tries to depict the long-term change with capability building up for abatement and abstraction. The purpose of the dynamic analysis is to advise the investment decisions to the pollution sources facing stringent water quality targets. The impacts of policy instruments on the pollution sources' behaviour of pollution are illustrated by comparative statics.

In Chapter 5, I introduce the source and methods to obtain the data for this research, as well as the methods of analyses for the research. This is followed by two chapters of empirical analysis, when the theoretical framework is applied with the obtained data for the tidal Ouse.

Chapter 6 indicates the solution of integrated cost effective river policy, based on the options of water quality management that are proved effective in Chapter 3, by solving the constrained static optimisation problem to meet the particular water quality target at the least total cost of river water quality management. Due to the constraints of UWWTD on STWs' abatement levels, several scenarios are designed to test the differences in the outcome. Suggestions towards the integrated cost effective river policy in a static system are made based on the optimisation solutions.

In Chapter 7, the dynamic model is used in the theoretical framework, fed with investment and capital stock data from the industries and STWs. The Chapter carries out similar scenario analyses to those in Chapter 6, but goes further to test the stability and convergence of the steady state equilibrium found by the dynamic analysis. The outcome of this Chapter is however constrained from implications to reality because of the insufficiency of data.

Chapter 8 continues with the outcomes from the static and dynamic analyses, discussing the policy instrument choices to deliver the optimal solution under static and dynamic analyses, had they been accurate enough to indicate the policy making in the future. Different policy instruments are evaluated against the criteria of instrument choices for environmental policy to make the recommendations to the policy implications of the optimal solutions from the analyses in the previous chapters.

The last chapter then reviews and concludes the outcome from this research, and discusses the successes and obstacles during the research and points out the future research needs.

Chapter 2 Background and Literature

2.1 The Humber River Basin and tidal Ouse

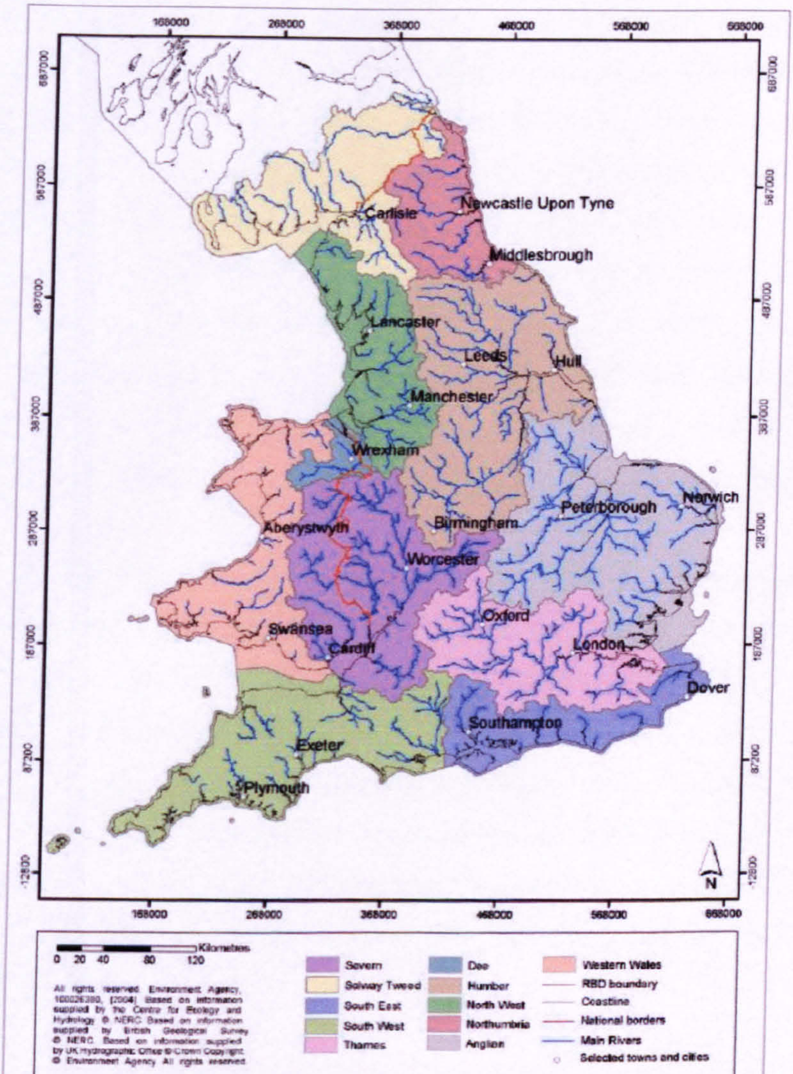

Figure 2.1 River Basin Districts in mainland UK

Source: Defra (2005a)

The Humber river basin is the largest river basin district (RBD) of the 11 England's RBDs shown in the Figure 2.1, covers an area of 26,109 km^2, from the north Yorkshire Moors to Birmingham, the Pennines to the North Sea and Stock on Trent to Rutland (Defra 2005b). This accounts for one fifth of the land area of England, which has a wide diversity of natural environment and land use. The rainfall in the catchment varies from less than 600 mm per annum in the Trent Falls to over 1600 mm per annum in the Pennines. Most of the precipitation happens in the west catchment due to the prevailing wind. Nearly 11 million people were reported living in the Humber catchment (Edwards *et al.* 1997), which is likely to grow marginally (by 0.1% per annum) in the future (Defra 2005b). The catchment drains through big cities such as Birmingham, Bradford, Derby, Hull, Leeds, Leicester, Nottingham, Sheffield and Stoke-on-Trent, but also drains the more natural areas of Pennines and north Yorkshire Moors. The major tributaries of the Humber RBD include the Trent, Ouse, Aire, Don, Derwent, Wharfe, Hull and Ancholme (Figure 2.2). The major industries in the Humber catchment include agriculture, food and drink, chemicals, iron and steel, non-ferrous metals, engineering, and electricity generation. Coal mining, which used to be one of the majors, has been greatly reduced in scale (Edwards *et al.* 1997). The Humber also has one of the largest port complexes in England.

The two largest tributaries are the Ouse and Trent. Although similar in the size of catchment that they drain, The Ouse and Trent catchments are distinctively different to each other (Edwards *et al.* 1997; Jarvie *et al.* 1997b). The river Trent mainly drains industrial area to the south and west of the district, and major populous cities within the district, Birmingham, Derby, Leicester, Nottingham, and Stoke-on-Trent. On the contrary, most tributaries bringing water to river Ouse have their sources in the Pennines, drains less populated agricultural areas (Jarvie *et al.* 1997b; Defra 2005b). The two large tributaries meet at the Trent Falls to form the Humber estuary.

In this research, we focus on the tidal section of these two large tributaries, particularly from the section of tidal Ouse to the sea Spurn because of the severe water quality issue. The tidal Ouse and tidal Trent start from their tidal limits at the weirs at Naburn and Gainsborough respectively. However, the tidal Ouse and Trent are quite different in their drainage networks. All of the principle tributaries join the Trent upstream of its tidal limit, while the Ouse has all its major tributaries joining downstream of the Naburn Weir, with much less flow from the non-tidal catchment (Edwards *et al.* 1997). The less flow from the beginning of the tidal Ouse makes it more vulnerable to water pollution.

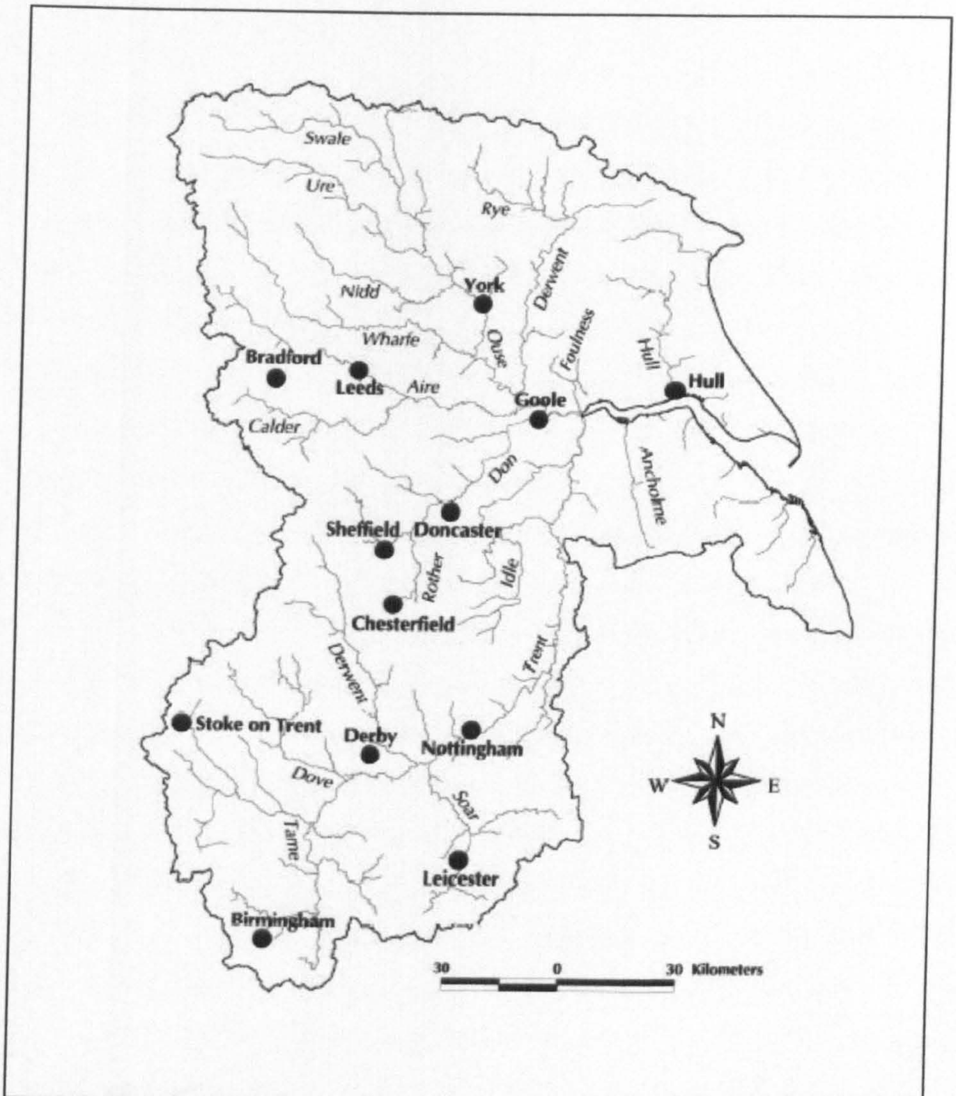

Figure 2.2 The Humber Catchment
Source: Oguchi *et al.* (2000)

The tidal Ouse is 61 km in length, and another 62 km from the Trent Falls to the Sea Spurn. The Sea Spurn of Humber has the largest fresh water source to the North Sea from the UK (250 m^3s^{-1}) and the second largest tidal range in the UK (7.2 m). However, most the impacts of water pollution remain inland. National Rivers Authority (1993) argued that "*This is several times greater than the seaward displacement due to the freshwater input during the tidal cycle. Thus* *effluents which discharge to the estuary are held there for a considerable length of time, being progressively diluted as they edge their way gradually to the North Sea. Thus residence period allows the full polluting effect of discharges to be exerted within the estuary".*

The Humber RBD has experienced severe water pollution since the mid nineteenth century (Sheail 1997) but significant improvements can be seen. The pollutants enter the river system through effluents from point sources and runoffs from the diffuse sources in rural and urban areas. The major point sources in the Humber RDB are from industries and STWs. All the major point sources are authorised by effluent consents for their discharges at specific place, there may also be consideration for the accidental discharges of harmful substances. In the Humber RBD, around 46% of the rivers length are at risk of point source pollution (Defra 2005b). The diffuse source arises from a wider variety of activities, among which agricultural farming is the most important one in the Humber RBD, especially for the tidal Ouse catchment which drains large farming lands in the rural area. Diffuse source pollution usually causes eutrophication by increasing the concentration of compounds of phosphorus and nitrogen. This will then lead to excessive growth of algae and other plants, which adversely affect the biodiversity and water quality. In the Humber RDB, 73.4% of the rivers length are at risk of diffuse pollution while another 18.8% are probably at risk (Defra 2005b). The impacts of diffuse and point sources pollution in the Humber RDB have drawn much attention of scientists, which has been intensively discussed from various aspects (Robson and Neal 1997; Tipping *et al.* 1997; House *et al.* 1997a; Jarvie *et al.* 1997a; House *et al.* 1997b; Jarvie *et al.* 1997c). Nevertheless, it has yet to attract more attention of economist to bring economic considerations for the water quality impacts and regulations. As a macro tidal estuary, the Humber and tidal Ouse have one more factor that significantly influences water quality: sediment. When the tide moves in, the sediments at the bottom of the river will be resuspended and moved upstream along the river, causing extra consumption of DO. Some pollutant may also be absorbed onto the sediments surface. In the Humber RDB particular, there are intensive flood defences to protect the flat flood-prone Ouse valley from extraordinary winter floods seen in 1982, 1991 and 1995. However, the flood defences also have the effect of trapping silts in the Ouse. This is then exaggerated by the long duration ebb that the Ouse relies on to keep open navigable channels. The Ouse is thought to be the river that has more sediments than any other river in Britain (Duckham 1967). Climate change may even exacerbate the problem (Cashman *et al.* 1999). Hence the behaviour and transport of resuspended sediments can be directly detrimental to the water quality of the tidal river system (Goodwin *et al.* 2003; Mitchell *et al.* 2003). However, measures on the impacts of resuspended sediments on water quality are still inadequate.

Water abstraction from both surface water and groundwater are common in the Humber RDB. The main purpose is to provide public water supplies and to serve industries and agriculture. The water company in this region, Yorkshire Water, is currently abstracting around 360 Mega Litre (Ml = 1 million Litre) of water from the tidal Ouse catchment every day, or 4.167 m^3s^{-1}. Most of the water abstracted is supplied for household use and ends in STWs. There is also abstraction at site of industries for production processes, among which some goes to the STWs as effluents. Through the drainage network in the Humber RDB, water resources are reallocated by abstraction and discharge from STWs. In the rivers Aire and Don, the effluents from STWs can consist as much as 50% of the flow in summer (Edwards *et al.* 1997). There is also significant abstraction at the Long Drax, around 20 km downstream of York, by the Drax power station, the UK's largest coal-fired power station. The water is abstracted as cooling water, returned at the same site, but with half of it lost in evaporation.

The traditional manufacturing sectors in the Humber RDB is declining, especially in the sectors coal mining, and iron and steel, though the overall economic activity is predicted to increase. Selby, from where the water quality starts to decline in the tidal Ouse and is likely to be affected most by regulations to improve water quality, is strongly dependent on the manufacturing, mining and construction industries. It has higher unemployment rate than most of the Humber RDB. Overall the Yorkshire and Humber region has the second lowest GDP per head in the UK (Cashman *et al.* 1999).

2.2 Regulative system of river policies in the tidal Ouse

The objective of water policy in England is to protect both public health and the environment by maintaining and improving the quality of water (Defra 2000a). Therefore it is a legal duty to prevent or reduce water pollution in the river, as well as a social and environmental responsibility. This objective has been continuously reinforced by both domestic law and European Commission (EC) directives. In England and Wales, the Department for Environment, Food and Rural Affairs (Defra) and the Environmental Agency (EA) was formed under the Environmental Act (1995), combining the National Rivers Authority (NRA), Her Majesty's Inspectorate of Pollution (HMIP), the former Waste Regulation Authorities and several smaller sections from the Department of the Environment. The aim of the EA is to provide high quality environmental protection and improvement. The EA seeks to achieve this by an emphasis on prevention, education and vigorous enforcement wherever possible.

The Environmental Protection Act 1990 established the statutory ground for a wide range of environmental protection purposes. It also introduced the concept of Integrated Pollution Control (IPC) to prevent pollutant emissions to the air, water and land. In 1991 the Water Act 1989 that controlled the pollution and supply of water was replaced by five separate Acts. The Water Resource Act 1991 replacing the corresponding section in the Water Act 1989, consolidated the previous legislation in respect of quality and quantity of water resources (Defra 2000a). It regulates the discharges to controlled waters, including rivers, groundwaters, lakes, estuaries and coast waters through a system of consents granted by the EA. The EA sets conditions of volume and concentration of particular substance enter in to the waters or imposes broader constraints to the nature of effluent. Each consent is made based on the water quality objectives set by the EA to the water the effluent is about to enter, as well as the relevant standards set out by the EC directives. The aim of this Act is to ensure the polluters pay the cost of the consequence of their discharge. The water abstraction from all sources is also prohibited by this legislation except under water abstraction licenses. The Water

Industry Act consolidated the regulations to the appointment of water and sewerage undertakers (the water service company), the conditions of appointment, water provision and sewerage services. No trade effluent could be disposed to sewerage undertaker unless trade effluent consent or permission from the sewerage undertaker is obtained. It is responsibility of the owner of effluent to ensure the effluent does not violate the permission and inform details of effluent to the sewerage undertaker. The sewerage undertaker is also able to set extra conditions to specific trade effluent depends on the nature of effluents. The Environment Act 1995 established the Environment Agency, and introduced measures to enhance protection of the environment, including further powers for the prevention and remediation of water pollution (Defra 2000a). Best Available Techniques Not Entailing Excessive Cost (BATNEEC) is required to be utilized in pollution prevention to minimize the pollution released to the environment.

The EC directives have been transposed and implemented in UK to ensure the standards of water quality protected as well as elsewhere in the member states of European Union (EU). The EC Surface Water Abstraction Directives (75/440/EEC) set the quality requirement for the surface water that serves as drinking water sources. The EC Bathing Water Directive (76/160/EEC) aims to ensure the protection of the health of swimmer and maintain the aesthetic value of bathing waters. The EC Freshwater Fish (78/659/EEC) and Shellfish Waters Directives (79/923/EEC), on the other hand, aim at protecting the health of freshwater fish and shellfish, designating the water in need of protection and the quality standards of those waters (Defra 2000a). Input of dangerous substances into the water is controlled under the EC Dangerous Substances Directives (76/464/EEC), together with the Water Resources Act 1991 to protect the water bodies and aquatic creatures. In England, the potential dangerous processes and substances are subjected to IPC. This is a regulatory system enforced by the EA in England and Wales, under Part I of the Environmental Protection Act 1990, to offer an integrated protection of the environment from release of these substances into water, air or land, or to reduce the emission to a minimum or harmless level using the BATNEEC. Throughout the UK, the IPC approach is being progressively replaced by a new Pollution Prevention Control (PPC) system to implement the EU's Directive on Integrated Pollution Prevention and Control

(IPPC - Directive 96/61/EC). The aim of IPPC is to achieve a high level of protection of the environment taken as a whole by, in particular, preventing or, where that is not practicable, reducing emissions into the air, water and land (Defra 2002b). Both IPC and PPC require the operator of plant installations and mobile plants to obtain a permit from the EA and comply with the conditions of the permit. They are similar in the respect that they requires an integrated approach to prevent or reduce the emission to the water, air and land in order to achieve a high level of protection for the environment as a whole, using BATNEEC. But PPC applies to a much wider range of activities. The IPPC process is being carried out sector by sector in UK from 2000 to 2007. Early implementations in the UK's industries have convinced it as a general principle and primary means to reduce the level of pollutant emissions into air, water or land, and to protect the environment as a whole.

River quality has been improved in many rivers through the improvements made to the outflows from Sewage Treatment Works (STWs) and industries under the EC Urban Waste Water Treatment Directive (91/271/EEC). UWWTD was agreed in 1991. It is one of a number of European Directives which protect both the water environment and use of water for drinking, recreation or industry (Defra 2002a). This European directive imposes requirements on the collection of sewage and standards for the disposal of sewage effluents. The main objective of the directive is to protect the environment from adverse effects of sewage effluents. Standard requirements are set according to the size of the discharge and the condition of receiving waters. The STWs in the Ouse catchment are of different sizes and are imposed by UWWTD for improvements under different time schedules.

The Water Framework Directive (WFD) (European Commission 2000) is the most substantial agreement on the water legislation of the EC so far. The directive is designed to integrate the way of water management in water bodies across Europe. The Directive takes account of all the different objectives for which the aquatic environment is protected (ecology, drinking water, health and particular habitats), and ensures that measures taken to achieve the objectives are co-ordinated properly (Defra 2002c). It requires all the water bodies in the member states to reach at least the "good status" by 2015. Through establishment of a river basin district structure, it aims at achieving long-term benefit on the water ecological health and sustainable management of water across Europe. Unlike previous EU Directives regulating on particular water issue or substance, WFD considers all the key issues in the water to decide whether it is of a "good status". It is for the first time a directive deals with the water issues on a water basin basis as a whole, taking into account inland and coastal waters, surface and groundwaters, and both water quantity and water quality to meet the objectives. Water needed for wetlands and to protect aquatic habitat for wildlife is also considered under WFD. The "joined-up" feature of WFD does not only imply the integration of managing different water bodies and considering both quantity and quality of water, but also to integrate the water management with other regulation or policy that are relevant to water environment, and to integrate the environmental and economic information in river policy decision (Defra 2005b). This need has been highlighted in the Defra report of "Directing the flow" and is regarded as priority of Defra and EA' responsibilities (Defra 2002c). Diffuse pollution sources, including the agriculture and urban runoffs are now recognised as prominent cause of water quality deterioration as clear evidence has shown that phosphorous, nitrogen, silt and other materials from farms are causing significant long-term degradation of rivers, lakes and groundwaters as well as harming the plants and animals that live in them. The WFD requires the member state to achieve the "good status" in a way balancing between economic, environment and social considerations. The benefit associated with the WFD was estimated to be around £560 million per annum (Defra and Welsh Assembly Government 2003). It required benefit to be delivered at the most cost effective without incurring disproportionate costs. Derogation of objective need to be applied subjected to approval by the Secretary of State if it entails disproportionate costs. The Directive therefore sets a framework that should provide substantial benefits for the long-term sustainable management of water. Public participation is called by WFD to ensure there is great public involvement to tailor the specific instruments of water management and sustainable water use in each member state. To pool the effort from each member state and ensure consistent understanding and implementation of WFD across Europe, a "Common Implementation Strategy" (CIS) has been established by member states and the European Commission to facilitate the exchange of best practice and experiences. Within the UK, Defra and EA established a UK Technical Advisory Group (UKTAG) to provide guidance and facilitate the implementation of WFD. As a key piece of European legislation, the long-term program of WFD would offer a major opportunity to improve the whole water environment and promote the sustainable use of water for the benefit of people and wildlife alike.

The Water Bill published on 20 February 2003 received Royal Assent on 20 November 2003, becoming the Water Act 2003 and published on 28 November 2003. As the latest legislation on water resources, the Water Act 2003 is in three parts, relating to water abstraction and impounding, regulation of water industries and other provisions. The first part reflects the need for changes in the system of water abstraction licenses. Three different types of water abstraction licenses are designated for various water use activities. The other two parts aim at improving the regulation system of water industries and boosting the opportunities for competition in water services.

2.3 Current river policy and management in the Ouse catchment

2.3.1 A Brief history of river policy and management in the Ouse catchment

Sheail (1997) reviewed the relevant river-management bodies of Yorkshire, North England, illustrating how the pace and direction of watercourse and catchment management are influenced by the preoccupations, aspirations and knowledge of policy makers and engineers. Yorkshire engineer, Malcolm McCulloch Paterson, described to the Parliament in 1896 two ways of pollutions in river: Positive Pollution occurred when pollutant was added into the river while Negative Pollution referred to abstraction of natural clean water (Sheail 1997). In 1894, a River Board replaced the previous joint committee, taking over power to mitigate pollution as an independent entity, although almost every regulation was opposed by local authorities and the mill owners. The Board acted initially via prosecution and policing, and then expanded its role to offering guidance and approval to local councils and traders. A River Ouse (Yorkshire) Catchment Board was appointed in 1922 by West Riding County Council for the purpose of land drainage management. Under the River Board Act of 1949, the responsibilities of the West Riding of Yorkshire River Board, Catchment Board and Yorkshire Fishery Board were brought together under a Yorkshire Ouse River Board. The River Board was replaced by the Yorkshire Ouse and Hull River Authority in 1965, and the Yorkshire Water Authority was appointed in 1974, which had the longest length of class 1 river (unpolluted water) and second longest length of class 4 river (heavily polluted) in England. Within the context of Yorkshire Rivers, the Environment Agency in Leeds acts as the environmental authority to design regulations and river policies aimed at achieving sustainable use of river and estuarine resources.

2.3.2 Water target and effluent discharge consents

Effluent discharge consents are authorized by the EA, which usually prescribe the maximum concentration of specific pollutants, effluent flow and other aspects such as pH and temperature. This is broadly utilized, not only in UK but also over the world, as the main instrument to control the point sources of pollution discharged into water bodies. The EA implements local regulations and European directives to set various water quality targets for inland streams, ground waters, lakes, estuaries and coast waters, and set up the corresponding effluent discharge consents of pollution sources in order to achieve the target. Water Quality Objectives (WQO) is one of the defined water targets in order to protect identified uses of surface water and there are associated water quality standards. In the tidal Ouse, different objectives are set by the EA for different sections of the river, according to the nature of river, pollution sources and their abatement abilities, and the designated function of river water. The current WQOs were classification regulated under the Surface Waters River Ecosystem (RE) Regulations 1994. Selby, Drax and Boothferry Bridge are currently designated to the objective of RE4, which specifies a target of 10 percentile (%ile) DO saturation to be higher than 50%. Cawood and Naburn are subject to higher target of RE3 and RE2, with 10%ile DO saturation no less than 60% and 70% respectively (Boorman 2003b). The annual assessments of general river water quality are undertaken by the EA and reported as the General Quality Assessment (GQA). This is designed to provide a consistent assessment of the state of water quality and enable comparisons to be made between different years and places. The GQA addresses four aspects: chemical, biological, nutrient and aesthetic grading. In 2005, 72% of the rivers were of good biological quality, compared with 69% in 2000. Between 1990 and 2005, 31% of rivers improved in biological quality. As for chemical quality of the river, 68% of rivers were of good chemical quality in 2005, same as 2000, but overall 41% of rivers improved between 1990 and 2005. This is largely due to the large investment into river water quality by the industries, e.g. water companies spent over £6 billion on improving inland waters. The EA declared more than 70% of the rivers in the Yorkshire and Humber in 1995, including the rivers Ouse, Aire, Calder and Humber have their chemical water quality classified as "very good", "good" or "fairly good". The percentage for biological water quality was 71%.

For each river system, the EA allocates the effluent discharge consents to the pollution sources, mostly point sources, according to the WQO of the particular river, the previous GQA records and the nature of effluent from each source. The effluent discharge consents are usually fixed value of concentration of pollutants and flow rate, in terms of either daily means or maxima, not allowing for variation. For most of the point sources, the outflow of effluent is a mixture rather than one particular pollutant. Therefore, an effluent consent is usually subject to a set of different pollutants, which is similar within an industry. The effluent consent also prescribed the location of discharge, receiving water, the monitoring process and other aspects of retaining the compliant effluent discharges. In order to recover the regulatory cost as well as motivate reduction of effluent discharge, a charge for effluent discharge is applied to everyone who holds discharge consent under the Water Resources Act or groundwater authorisation under the Groundwater Regulations 1998. Currently the standard application charge is £772 and the annual charge financial factor is £596, which will be multiplied by other relevant factors to determine the total effluent charge for a particular consent holder. The total effluent charges are expected to account for £64m in year 2005/06 (Environment Agency 2005).

2.3.3 The Tradable Water Abstraction License system

In England and Wales, the EA is responsible for managing the water resources in rivers, wetlands, lakes, underground water and reservoirs. Any water abstraction more than 20 cubic meters per day from either surface or underground sources will need a license. The water abstraction license granted by the EA is supposed to control the level of water abstraction and protect the water resources. Same as effluent discharges, there is a water abstraction charge to each water abstraction license holder in order to recover the administrative cost of the EA. Currently, the water abstraction charge is £10.03/1000m^3 in Yorkshire, which is the lowest in England and Wales. A Defra report (Defra 2000b) concluded that increasing the water abstraction charge is not able to provide strong enough incentives for reducing water abstraction, either to satisfy particular water management objective, or to reduce the cost of environment damage from water abstraction.

The water abstraction license has been in place since the 1960s. After more than 30 years, the problem associated with the license system has become apparent, such as over-licensing, licences issued in perpetuity and the lack of flexibility. After a Water Summit held by the Government with water companies, the EA and key stakeholders, the license system is currently under review. The Government has proposed some changes in the water abstraction license system in 1999, relevant to the time limit of the licenses. Many of the changes require new legislation, which has been incorporated into the Water Act 2003. The new time-limited tradable water abstraction licensing system is to be managed through the Catchment Abstraction Management Strategies (CAMS) process, which commenced in 2001. CAMS is constructed at a local level, to make more information publicly available and to determine the balance of need between water abstraction and aquatic environment through consultation within the locally interested parties (Environment Agency 2002). The Rivers Ouse and Humber system has been divided into six CAMS areas to determine the water abstraction level in each of them. However, the issue of over-licensing and lack of flexibility still exist, although revocation or variation for the so-called "sleeper" licences", under which there has been little or no actual abstraction for several years, has been required by the Government report "Taking Water Responsibly" (Defra 1999b). The new time-limited licenses under the CAMS have a long review cycle of 12 years, which is still slow to variation and uncertainties.

The water abstraction licenses issued by the EA give water rights to the license holder. The trade of water rights from one party to another for profit is allowed by tradable abstraction licenses. This economic instrument is supposed to generate efficient cost savings in the water abstraction and provide greater flexibility to accommodate varying demands (Defra 2000b). It is also expected to realize the true economic value of water contained in the abstraction license. However, no significant water right transfer has ever taken place through the tradable licenses (Defra 1998a). The Water Act 2003 has incorporated some changes to provide more facilities to remove barriers to trading. The objective of tradable water abstraction licenses is, in principle, to establish effective means to achieving the optimal distribution of water resources within and between different sectors of use and thus contribute to a sustainable development.

2.4 Review of other environmental policy instrument

The choice of environmental instruments has been controversial for decades. The commonly used means of regulation through design and performance standards are mainly statutory instruments per se, taking forms such as prohibition of processes or products; technology specification; discharge standards and permits; emission caps and harvest limits. Most criticism of the statutory instruments is based upon the fact that the fixed standards and consents ignore the difference of marginal costs of pollution abatement among various pollution sources and the different marginal damage caused by pollution at various places, resulting in inefficiency in social welfare along with environmental degradation. On the other hand, Market-Based Instruments (MBIs), mostly referred to emission charge and TPP system, are preferred by economists not only because they can provide considerable efficiency gains over arbitrary standards, but also because that they provide more flexibility in compliance to regulation and continuous dynamic incentive for pollution control (Oates and Strassmann 1984; Cowan 1998; Hanley *et al.* 1998; Perman *et al.* 1999).

2.4.1 Emission charge

An emission charge is a fee, collected by the government, levied on each unit of pollutant emitted into the air or water (Tietenberg 2006). Emission charges induce the firms to reduce their pollution because there are substantial costs of the pollution emitted by firms. A firm assumed to be profit seeking, will then reduce their pollution to the point where its incremental cost of abatement equal to the emission charge they must otherwise pay (Hanley *et al.* 1997). An effective emission charge will be set such that it leads to the emission reduction at the desired level to the regulator. Emission charge can always ensure the cost effectiveness for the required pollution reduction target although the target may not be cost efficient.

The emission charge has its advantages and limitations.

1. An obvious barrier of emission charge application is that neither abatement cost nor private benefit of the firm are known to regulators. The information asymmetry then prevents the regulators from establishing appropriate charge rate at the first try. An iterate trial-and-error process to find the appropriate charge rate which initiated from an arbitrary charge is then inevitable. In addition, the uncertainty involved requires the "appropriate charge rate" to vary all the time.
2. One important advantage of emission charge is that it will stimulate the development and acceptance of cheaper and cleaner pollution abatement technology (Turner *et al.* 1994; Hanley *et al.* 1997; Tietenberg 2006).
3. As emission tax is a tax levied on a public bad rather than public good, there are little or no distortion impacts on the economy. Some economists have

suggested that the revenue from emission tax could be "recycled" to alleviate the distortion taxes in the economy. This may generate benefit more than that associated with environmental protection. Pearce (1991) suggested the use of environmental taxes to reduce distortion taxes in a revenue neutral way could result in two benefits, not only environmental protection but also release of distortion taxes. This is also called the double dividend of emission charge/tax. But Xepapadeas (1997) discussed that with the appealing claim for double dividend, theoretical and empirical research does not seem to support the strong double dividend hypothesis; and the environmental benefit of emission taxes still remain crucial in justifying their introduction.

Hahn (1989) and Cowan (1998) investigated the charge systems in different countries, France, Germany, the Netherlands and the United States. In France, the system is primarily designed to raise revenue for investment in sewage treatment and other projects for water pollution control. This charge is based on the load estimated by regional water agency. They provide a significant source of revenue of water quality improvement. In Germany, the system is similar to the French one. The charge is used to cover administrative expenses for water quality management and to subsidize projects that improve water quality. Uniform charges system across the country is based on the expected value of concentration and varies with industry types and municipalities. Although lacking of data, water quality seems improved by the effluent charge system. There was a large increase in abatement investment after the introduction of the charges. The Netherlands has one of the oldest and best administered charge systems, with one of the highest charge rate levied on the effluent stream. The charges are set to finance sewage treatment costs and have steadily risen over time. Like those in France and Germany, water quality is managed by both permits and effluent charges. The large polluters are monitored for actual levels while households and small polluters pay flat-rate charges. As a result, effluent discharge declined by 90% over 15 years. The US has relatively moderate effluent charges compared to Europe. The primary purpose of the charges in the US is to raise revenue in order to help the treatment plants that are heavily subsidized by the federal government. The environmental and economic impacts of the effluent charge are apparently small due to their small size and limited application of the revenue.

The effluent charge practices in Europe and US appear convincing to the advantages announced before, especially in the Netherlands. The gradual and steady rise of effluent charges is regarded as one of the keys to the success. However, in all these cases the effluent taxes are cooperating with existing direct regulations.

2.4.2 Tradable pollution permit system

TPPs were first proposed by Crocker in 1966 and Dales in 1968. Rather than increasing prices through a tax to reduce demand, TPPs set a maximum level of pollution or resource depletion that will be allowed through certain amount of permits. These permits may be issued in two ways, grandfathering, or auctioning. The permits are freely transferable. Assuming all the firms are costs minimizing, and the permits market is competitive, the overall cost of achieving the environmental target will be minimized. This virtue allows government to meet its policy objective while allowing greater flexibility in how to achieve the target (Tietenberg 2006). Although Zylicz (2003) argued that this is not valid when fixed costs are large, or with non-convex cost function. An obvious advantage of TPP over effluent charge is that no information is needed for the abatement cost.

The TPP instrument is more favoured in the United States, applied both to water and air pollution. The success of the emission trading scheme under the 1970 Clean Air Act of US have been reviewed by Hahn (1989) and Tietenberg (1990), and they found a substantial cost saving of over $10 billion. However, TPP applications in water pollution control always have very small market and thus are not very successful. Several important conditions for TPP system are discussed below:

1. In general, transaction costs are ubiquitous in economics thus it is impossible for the trade of pollution permits to avoid it. Therefore, the trade equilibrium does not equilibrate marginal abatement costs among pollution sources, but the sum of marginal control costs and marginal transaction costs (Stavins 1995). Application of TPP system in Fox River actually failed due to high transaction costs in the form of administrative

requirements which essentially eliminated potential gains from trade (Hahn and Hester 1989).

2. TPP may also result in hot spots generated by trade. There is always a possibility that trade will concentrate discharges in some places where high control costs exist. Thus, the ambient standards in that area are very likely to be violated. In the case of water pollution control or for other non-uniformly mixed pollutants, the location of polluters matters in the trade process. It is therefore imperative for the environmental authority to differentiate among polluters by their locations to achieve cost effectiveness (Baumol and Oates 1988), and different transfer coefficients of polluters on various monitoring sites are helpful (Zylicz 2003).

It is argued that agricultural pollution control is best accomplished using voluntary 'Best Management Practices', and that quantitative discharge limits and economic incentives are impractical (Young and Karkoski 2000). But as a new direction of TPP scheme, experiments have been carried out for pollution permit trade between point and non-point pollution sources (Jarvie and Solomon 1998).

2.4.3 Comparing emission charge scheme and TPP system

The idea of using effluent charge for least-cost management of effluent was introduced decades ago (Johnson 1967), so was the TPP system (Montgomery 1972). Generally, under ideal conditions, emission charge schemes and TPP system are symmetric to each other (Pezzy 1992). However, where the control cost is not known with certainty, the two instruments differ. Weitzman (1974) indicated that when information on costs and benefits is imperfect, which instrument is likely to lead to larger bias from optimal equilibrium depends on the statistical characteristics of these uncertainties. Nevertheless, Shrestha (1998) argues that the statistical characteristics of uncertainties only prefer the effluent charge scheme when the predetermined standard is excessively stringent; for all the other situations, a TPP system is superior to an effluent charge scheme. The choice between a TPP system and an emission charge scheme has been discussed for a long time. In general, the TPP system is more preferred in the US while the tax scheme is more popular in Europe. Baumol and Oates (1988) suggested some of the possible reasons why one is more preferable to the other one under different circumstances.

The first and the major advantage of a TPP system, is the TPP system can reduce the uncertainty and adjustment costs involved in attaining a required environmental quality standard. The permit amount issued by the environmental authority always guarantees the required pollution reduction, yet on the other hand the emission charge scheme cannot guarantee the desired response from the pollution sources, hence the authority have to alter the rate from time to time. The adjustment to achieve the optimal rate in emission charge scheme is very costly or even unavailable. Similarly, the emission charge scheme also needs alterations in the charge rate due to economic growth and inflation. Therefore, it is very difficult to determine the optimal charge rate. In both situations, the market forces will automatically adjust the price of permits, so there is no need for imposed adjustment and no increase in pollution.

The second reason lies in the financial burden incurred because of the costs imposed by the emission charges. Although the emission charge scheme will reduce the total costs of pollution control, it imposes a financial burden to the plants. Therefore, it is unfavourable to the pollution sources. On the other hand, if the pollution permits in a TPP system could be distributed free, through so-called "grandfathering" determined by the current level of existing plants, it can effectively eliminate the adverse effects on the plants from the extra financial burden that the emission charge scheme would otherwise impose. Although the "Polluter Pays Principle" requests that no one else but the polluter should bear the costs of pollution abatement, the plants will lose their competitiveness if the other plants in the industry were not charged at the same rate or were not charged at all, particularly at an international scale. For this reason, TPP system is preferable to the emission charge scheme by the managers of plant and some government officials seek aiming at improving economic competitiveness.

There are also some arguments favouring the emission charge scheme, one of which involves the saving in the transaction costs. If the pollution permits are not distributed optimally in the first place, a number of transfers of pollution permit need to take place in order to achieve the least cost solution. There will be costs incurred during the search and bargain activities relevant to the trade, usually known as the transaction costs. When the transaction cost is too high, there would not be enough transfer occurrences even though it could reduce the costs of pollution control. Stavins (1995) argued that in the presence of transaction costs, the equilibrium of pollution permits trade is no longer independent from the initial allocation of permits, but dependent on it. In the water pollution problem, the common situation is that there are usually only few pollution sources along the river, as in the Forth Estuary (Hanley *et al.* 1998) and along the tidal Ouse in this research. Therefore, it is possible that a single plant could dominate trading for both buying and selling, which is also impeditive to the trading of permit. Furthermore, considering the fact that a plant usually discharges a mixture of effluent containing various pollutants, the uniqueness of the plant's effluent discharges makes it more likely to be dominant in the market and harder to find appropriate traders. Strategic behaviour by traders is evident in the Fox River example (O'Neil *et al.* 1983).

Unlike the TPP system for air pollution control, such as carbon dioxide (CO_2) emission as prescribed in the Kyoto Protocol, the TPP system to control water pollution is dealing with non-uniformly fixed pollutants, which need to take into account the location of pollution when trade is undertaken. Pollution from the upstream sources will affect the water quality downstream while the pollution from the downstream sources usually only has small impacts on the upstream. Therefore, impact from one unit pollution emission on the river water and environment depends on the location of discharge and other natural factors, so should be the price of permitting the emission discharge. This will add more difficulties to the trade of permit to being efficient, without generating the "hot-spot" of pollution where the cost from pollution is undervalued. A pollutant dispersion model approved by the environment authority has to be obtained before the trade is undertaken. This kind of model is usually expensive and may raise critical questions, which ultimately lead to the transactions being denied. Even in the successful permit trading scheme controlling the air pollution in the US, few trades requiring this modelling for non-uniformly mixed pollutants have been actually consummated (Tietenberg 1990).

2.4.4 Command and Control and Market-based Instruments: empirical evidences in UK

Despite the glorification from economists of the advantages of MBI over the CAC approach, most environmental regulators, and even many pollution dischargers retain their preference to the less cost-effective environmental standards in environmental management and pollution control. One of the most obvious reasons is that the CAC standards such as effluent consents or technology standards are more easily managed and implemented by the regulator, and more easily understood and provide clearer abatement targets for the regulated pollution sources than MBI. Other reasons have been argued by many researchers in an extensive set of literature (Stavins 1995; Hanley *et al.* 1997; Helm 1998; Russell 2001; Bell 2003; Zylicz 2003).

Most economists agree that MBIs are more efficient, both in cost saving and dynamic incentive over CAC approach (Oates and Strassmann 1984; Baumol and Oates 1988; Hanley *et al.* 1997; Cowan 1998; Perman *et al.* 1999). But the context of uncertainty over both pollution-related environmental damage and costs benefits estimates, the risks of significant environmental hazards, and the nature of some projects involving significant fixed costs and little marginal costs could all shift the favour to the direct regulatory standards as a "better" approach against MBI, which is consistent with the precautionary principle and environmental ethic (Turner *et al.* 1994; Zylicz 2003).

The unsuccessful cases, for example the Fox river discussed by O'Neil et al.(1983), which neither achieved the desired level of pollution abatement nor provided incentive for pollution abatement, indicated that the application of MBI approaches needs to be implemented with great caution. In addition, like the cases of water pollution control through emission charges in Europe, the MBI schemes are quite frequently complementary with existing regulatory standards.

2.4.5 Policy instruments in the perspective of the UK Government

The UK Defra has considered (Defra 1998b; Defra 2000b) the possible movement away from the localised CAC approach to a river policy system utilizing economic instruments such as emission charges and TPP system, in order to improve the cost effectiveness of control for both pollution emission and water abstraction. It has been agreed by the Government that applying an emission charge scheme and a TPP system could improve both water quality targets and the efficiency of abatement efforts. However, before the introduction of these economic instruments into river policy, the Government is still to commission consultancy work over a series of issues about the practicability of these instruments in the regulation system of UK, and to estimate the possible effects of them to the domestic, commercial and industrial water users.

Emission charge schemes in water pollution control could be introduced into the current framework of price regulation of the water and sewage companies, but has to make sure the charge is not to be passed in full to the consumers. The appropriate level and the structure of the charge are also critical issues under consultation. The Government is also aware of the interaction of the emission charges with the regulation of abstractions in this report, but with no further details. Trading system of pollution permit would induce more difficulties in introduction, as there is no current provision in water or other legislation for the operation of tradable permit (Defra 1998b). New legislation may be needed in order to justify the trading process. As discussed before about the general issues for TPP system, the Government is fully concerned for the inherited difficulties of the TPP system even various degrees of success have been seen in the US. It is by no means easy to guarantee sufficient existing and prospective dischargers to trade, nor are there big enough differences among their marginal costs to make the trade profitable after transaction cost, and the flexibility of increasing or decreasing abatement capacity facing stepped cost functions or sunk cost of facility. Creation of "hot-spot" through trade is also acknowledged by the Government. As declared by Defra in this report, within the context of the 'no deterioration' principle and a policy of localised quality objectives, there is little scope left for the manipulation of TPP system. Discussions are raised for the details of the possible introduction of emission trading scheme in pollution control, the possible roles the EA and the Government can play, and the structure of the scheme.

Defra (2000b) has concluded that raising the current water abstraction charges in various regions for incentive purpose, either to achieve a certain reduction of water abstraction (by 15 or 30%) or to mitigate the social damage, is not able to change the behaviour of major water abstraction licence holders, i.e. the water supply companies. Therefore the small benefit from water abstraction reduction together with a significant level of revenue from charges, imply a significant distributional effects, which would obstruct the charge scheme unless distortion of the incentive can be avoided. The trading possibility of water abstraction license has been presented from the beginning, unlike the effluent discharge consents, so there is no legislative gap to the trading of water rights. Defra believes that with more facility provided to the water rights trading in the Water Act 2003, the creation of markets in abstraction authorisations should be feasible in a number of catchments, with efficiency and environmental gains in both short- and long-terms realised through trade of water abstraction licenses.

2.4.6 Integrated river policy for static and dynamic efficiency

So far, most of the policies of river management only focus on the effect on effluent discharges to the river, or just the impact of water abstraction, and most of the river policy researches were investigated using static analysis only. This research aims to investigating integrated river policy taking account of both effluent discharges and water abstraction, whose impact on river quality are interdependent on each other. The integration of the policy can only be derived from a comprehensive understanding of the influences from effluent discharges and water abstraction, and the nature of river quality dynamics. It will therefore lead to more cost effective solution of river management. This research also initiates in exploring the dynamic equilibrium of water quality control within relevant activities. Although the TPP system is able to accommodate the effects on the permit prices of economic growth and inflation automatically through market forces, it is not applicable to the dynamic growth in individual plant. A dynamic analysis for the equilibrium of pollution control, and the associated capital and investment decision would shed a light into the river management over time rather than sticking at a stationary point.

2.5 In-stream water quality models and their applications

There has been a long history of using quantitative techniques to assess the impacts of pollutants on river water quality, in terms of DO particularly, in river systems (Cox 2003b). Up to now, most of the hydrological models used in this context are constructed based on the concept of mass balance of constituents first used by the Royal Commission of 1912. Streeter and Phelps derived the classic equations for simulating DO and BOD in rivers in 1925 (Cox 2003b), which have formed the basis of many present sophisticated computer-based water quality models. With the increased use of computer technology in hydrology in the last two decades, more advanced hydrological models have been developed and improved for better simulations of the dynamics in the in-steam water system (Bikangaga and Nassehi 1995; Lewis *et al.* 1997; Cox 2003b; Deflandre *et al.* 2006) .

The large number of existing models is also partly because most studies of water quality and hydrology in rivers are more or less specific to a particular situation. Therefore, often the result is 'local' water quality models suitable only for use where the models are derived. In a review of currently available water quality models by Cox (2003a; 2003b), several popular water quality models for in-stream and river processes are evaluated on the basis of availability for providing simulations of DO in lowland rivers. The conclusion shows all of them contain assumptions and limitations on the interpretation of model simulation outputs. Because water quality models are widely used by environmental authorities to assist operations such as consents setting and evaluating potential effects of future planning, they are often driven by environmental legislations and water regulations in various countries. Therefore, it is not surprising that although few are widely used by institutions, water quality models are often specific to one country, one institution or even one river catchment.

The water quality models used to predicting both static and dynamic change in water qualities in the rivers of UK include QUASAR (and the descendant QUESTOR) and MIKE-11. QUASAR and QUESTOR models were both developed by the Centre for Ecology & Hydrology (CEH) UK and have been extensively used by the EA as a planning tool and also during the LOIS project to simulate the dynamic change in river quality in the Yorkshire Ouse river system (Lewis *et al.* 1997; Whitehead *et al.* 1997; Eatherall *et al.* 1998). Both of the models represent a river system by a one-dimensional network of reaches. One-dimensional models assume complete mixture of the water in the river's cross-section area at any point along the river; hence, the concentrations of water-borne substances are distributed in one-dimension along the river length. QUESTOR is a software framework to support in-stream water quality modelling at CEH Wallingford. It was developed from QUASAR to support increasingly demanding model applications. An example of utilizing QUESTOR in water quality simulation, together with detailed data preparation, calibration and validation processes, can be found in the Defra report of Economic Instruments for Water Pollution Discharge (Defra 1999a). In this report, Defra examines advantages and disadvantages of a charge on water pollution if introduced in the UK. QUESTOR is utilised to simulate the influence of changes in emission rate on water quality. The models have relatively small data requirement, which simplify the calibration and reduce the run-time. However, the models cannot simulate a river system with back flows or loops, which is a common phenomenon in a tidal river system. Boorman (2003a; 2003b) carried out an extensive and consistent exercise in river modelling, through simulating several constituents including Dissolved Oxygen (DO), in the six catchments draining along the Ouse system to the Humber estuary. In most of the simulations, the model provided very accurate quantitative assessment. However, as the author mentioned, the lack of data prescribing many of the point pollution sources and abstractions is regrettable especially in terms of assessing in-stream conditions during the summer months. The simulation results of the QUESTOR model were used as inputs to model the downstream estuary after the tidal limit with the ECoS3 HOT model (Tappin *et al.* 2002).

MIKE-11 was developed by the Danish Hydraulic Institute and was marketed in a suite of software in the UK and Europe. MIKE-11 is an engineering software package for the simulation of flows, water quality and sediment transport in estuaries, rivers, irrigation systems, channels and other water bodies. It is a dynamic one-dimensional modelling tool for the detailed design, management and operation of both tidal and freshwater rivers. MIKE-11 is an advanced model of flow and water quality simulations (Cox 2003b). However, the large amount of data required limits running the model in many cases at a high enough level of complexity to generate accurate simulations. So is the problem for the process of calibration. Another issue of MIKE-11 is its lack of stochastic component, which makes it unfavourable to practices in relation to UK legislation, where the regulations are based on probabilities of water quality achievement. On the contrary, McIntyre and Wheater (2004) introduced a WaterRAT model based on Monte Carlo simulation to identify the significant uncertainties and to evaluate the degree to which the decision generates risks. But the model itself has much simpler form than MIKE-11 so it is not as accurate in water quality simulations. Hanley et al. (1998) utilized the MIKE-11 model as part of an environmental-economic modelling exercise aimed at quantifying the potential cost savings from a Tradable Pollution Permit (TPP) system in the Forth Estuary, Scotland. The MIKE-11 water quality model was combined with step-wise integer and linear programming models representing firms' abatement costs. They also mentioned the difficulty regarding data availability and introduced a term of certainty equivalency based on Chebychev's Inequality to include the stochastic property into the results.

2.6 Reviews of ECoS3 and QUESTS1D model

Most of the models except MIKE-11 that Cox (2003b) evaluated are only suitable for the simulation of water quality, particularly of DO, in freshwater systems. In a river system such as the tidal Ouse that has diurnal tidal movement, the nature of water quality and flow is more complicated than can be satisfactorily simulated by a generalized model. Two models have been successfully utilized in simulating flow and water quality in the tidal Ouse: ECoS3 software and the QUESTS model. ECoS is acronym of Estuarine Contaminant Simulator, a quick, flexible framework that is utilized in many estuarine systems. Developed by the Plymouth Marine Laboratory, it aims to estimate the distribution of contaminants in a theoretical estuary (Pham *et al.* 1997). Tappin *et al.* (2003) constructed a Humber-Ouse-Trent (HOT) model with the ECoS3 software for extensive research on the fluxes and transformation of suspended particles, carbon and nitrogen in the Humber estuary system. The estuary quality model QUESTS was developed by WRc and widely used by the EA for consents setting and as a planning tool. The QUESTS model was used by Cashman et al. (1999) to evaluate the efficiency of the mechanism for the allocation of consents between pollution sources in the tidal Ouse.

2.6.1 Previous research by ECoS3 Software and HOT model

ECoS has been utilized by many researchers (Pham *et al.* 1997; Liu *et al.* 1998; Harris and Gorley 2003; Punt *et al.* 2003; Tappin *et al.* 2003) aiming at simulating the distribution of turbidity, salinity, particles, carbon, nitrogen and metal contaminants in estuaries. Most of the researches have obtained satisfactory simulations, although in some instances it is hindered by lack of sufficient data (Tappin *et al.* 2002; Punt *et al.* 2003), suitable reaction coefficients (Liu *et al.* 1998) or other constraints.

Applications of ECoS software or models built within ECoS range widely from the Tamar estuary in southwest England to the Tweed estuary in northeast England and as well as the Gironde estuary in France. Among them, the Humber estuary has been intensively investigated and simulated by utilizing ECoS software on suspended particles, carbon and nitrogen (Tappin *et al.* 2003). In order to do so, a site-specific model was built within the ECoS3 software to represent the complicated biological, chemical and physical transactions taking place in the estuary. The HOT model takes into account a wide range of processes linked with a dynamic mathematical model into a coherent system (Harris and Gorley 1998b).

Although in theory ECoS3 can represent biogeochemical transformation systems of indefinite complexity in one and two dimensional advection-dispersion contexts which may be branched and layered (Gorley and Harris 1998), the models built within the software to simulate the constituents in river water are usually tidal averaged and are one-dimensional. The HOT model components treat the estuary in terms of concentrations, process- and transport- averaged across its cross-section, i.e. it is a one-dimensional model as it only represents the water flows and the advection and dispersion of substances along the axis of the estuary, assuming the river is instantaneously and completely mixed across its width and depth (Harris and Gorley 1998b). This is applicable when the estuary is well mixed, showing negligible variation across a section. Furthermore, the HOT model is a tide-averaged model, as the requirement to simulate the estuary over seasonal time scales precluded the explicit representation of the tide in the model. Any direct effect of the ebb and flow of the tide on constituents observed has been ignored. Variations directly due to the tidal oscillation are treated as error, and the indirect effects of the tides are modelled as dispersion (Tappin *et al.* 2003). The reason for using a tide-averaged model stems from the fact that the volume of water in many estuaries does not depend on the flow of river water into the estuaries from headwater sources, but largely on the ebb and the flow of tide (Harris and Gorley 1998a). Averaging over the tide would generate effectively constant volume of water in a particular estuary thus making it possible to calculate the water velocity in the estuary through cubature, in which the velocity at any point along the river is determined by the given volume and cross-sectional area at the point.

The ability of ECoS3 software is not restricted only to tide-averaged model as in HOT; it is also able to construct models with tidal variability. In a model constructed by ECoS3 of solute transport in the Tweed Estuary, the estuarine model consisted of a one-dimensional hydrodynamic scheme with tidally variable channel cross-sectional area (Punt *et al.* 2003). ECoS3 software is a general framework for modelling hierarchical spatial systems (Harris and Gorley 2003). Its modular structure and template models allow it to be rapidly adapted for new or highly specific modelling needs. Tappin (2002) has integrated the HOT model with the QUESTOR model by using the results from the latter to feed into the HOT model as input data for the tidal Ouse at its tidal limit at Naburn. This flexibility of the ECoS structure makes it ideal for modelling complex systems including the dynamics in an estuary, and spatial and temporal distributions of physical, chemical and biological processes. It has been suggested to be a suitable tool for environmental management, adaptable to the needs of industry and regulators (Punt *et al.* 2003).

2.6.2 Previous research through the WRc QUESTS1D

Like the ECoS3 HOT model, the QUESTS1D model is a one-dimensional representation of the tidal river system stretching from the tidal limits of the Ouse, Wharfe, Aire, Don and Trent to the sea spurn. The QUESTS1D model has not been as widely applied to various estuaries as ECoS3 software. Nevertheless, this does not affect its reliability as most of the current water quality models are localized to specific rivers. In fact, the QUESTS1D mode has been intensively utilized in the Humber system, by the EA for the consents setting and regulation planning purpose.

The QUESTS1D model has been available for simulating the river water quality in Humber system since 1994. Some recent previous works undertaken after the validation in 1999 are used as the predictive tool focusing on implications on river water quality of both water abstractions in tidal Ouse and reductions in Selby effluent discharges.

Freestone (2001) used the QUEST1D model to estimate the magnitude of the effects on DO concentration distribution of changes in water abstraction regime, i.e. abstracting the same amount of water from elsewhere rather than current locations. The effects of improving the inputs from rivers Aire and Don, the two biggest tributaries of the tidal Ouse, and effects from effluent discharges at Selby on the DO concentration distribution were also investigated. The simulation was based on the flow conditions of 1996 and 1997, a very dry year and an average rainfall year respectively. The simulated results illustrated significant changes in water abstraction locations and in effluent loads of Selby industries. The effluent loads had much bigger impact than the former factor, particularly in the drier year of 1996. Among these three possible changes, only elimination of effluent discharge could guarantee the DO target alone of no less than 30% saturation at 5%ile, no matter dry or wet year. The input improvements in Aire and Don had very marginal effects on the DO sag between Selby and Long Drax, and caused relatively more noticeable improvements downstream of their confluences. This is a confirmed finding of previous researchers that the effluent discharges from Selby industries are a significant contributing factor in the DO depletion in the river Ouse between Selby and Long Drax.

Similar research carried out later by Freestone (2003) to some extent quantified the effects of these three changes. Changing existing water abstraction location from the Ouse and Derwent to the Aire and Don, where the water would otherwise be transferred to through the water supply and sewage system, would have improved the value of minimal DO saturation between Selby and Drax from 5% to 13% in 1996 and from 20% to 28% in 1997. Elimination of discharges from Selby could have improved the minimal DO saturation in the same area from 5% to 39% in 1996 and from 20% to 40% in 1997. With existing water abstraction locations and the improved inputs from Aire and Don, elimination of discharge load from Selby resulted in an equally good water quality with a minimum of 41% DO saturation between Selby and Drax. The oxygen demand of sediment, which is not available to be explicitly estimated in the QUESTS1D model, was related to the difference between minimal DO saturation at Naburn of 67%, and that between Selby and Drax under the best situation of 41%, roughly 26% (Freestone 2003).

Longer time series of simulations were made in 2002 for the river conditions between 1996 and 2000 (Freestone 2002). The simulated results were assessed upon the Rivers Ecosystem (RE) Objectives to investigate the effects of effluent discharge reductions in Selby's industries. The research concluded that even under very stringent discharge consents of no greater than 100 tonnes of BOD/year for all the four plants in Selby, it could just guarantee meeting the RE4 target at Selby and Drax. It also indicated that fresh water flows are an extremely important factor for the DO concentration in the tidal Ouse, which is unfortunately beyond our control.

The most intensive simulations for the effects of alternative pollution control management on the DO concentrations in the Ouse were joint research by the EA and the Environment Department in the University of York. It considered the following alternative management actions (Cashman *et al.* 1999):

- A change in the location of discharge of Selby industry effluents to the Ouse/Humber system.
- A change in the timing of discharge of Selby industry effluents with respect to local high water.
- A change in the timing of discharge of Selby industry effluents over the year.
- A change in the net level of Yorkshire Water abstractions from the Ouse and Derwent.
- A change in the level of discharges from the Selby industries to meet the Environment Agency's targets.

The effects of these management actions were examined against compliance to RE Objectives, the General Quality Assessment (GQA) scheme, EA's DO standards, and the composite score of Estuarine Working Party Classification Scheme. The results concluded that, among all the actions, moving the location of effluent discharges from Selby to Boothferry, about 20 km further downstream, would generate a remarkable benefit to the DO saturation in the tidal Ouse. Return of the abstracted water in either the Derwent or the Ouse has marginal effect on DO saturation. Return in the Derwent has better effects as the water in Derwent is well-aerated and low in oxygen demand, while water return in the Ouse only slightly improves the situation due to the poorer input water quality. Storing effluent to discharge in winter instead of in summer has barely noticeable impacts, depending on the proportion stored and discharge location. Similarly, discharging on the ebb tide hardly makes any difference. Reduction of effluent discharges in Selby would significantly improve the water quality as expected, but at a much higher price.

2.7 Previous research on hydrological and economic impacts of river policy

There is little previous research trying to combine the water quality issues with economic analysis. O'Shea (2002) offered a general discussion of the difficulties and possible approaches to combine these two aspects in reducing water pollution. One of the conclusions reached at the end is "*Each case must be decided on its own merit*". She also pointed out that the water pollution issues "*call for close cooperation between scientists and economists*". Same as a later research by Zylicz (2003), cost effective management in the river to reducing water pollution are perused, followed by discussions regarding the proper policy instruments, particularly market-based instruments to deliver the management.

There is also little research trying to add dynamic features in water quality and river policy analyses, despite the fact that the pollution issue in river water usually has dynamic features related to changes in effluent discharge, abatement measures, and runoff from land use or interactions within the water body. When water pollution reduction is combined with economic analysis as suggested above, the dynamic feature widely associated with economic activities calls for more consideration of not just one-off solutions under the current situations, but for solutions that can evolve with time and can accommodate changes within the system.

Combining water resources management with economic analysis has attracted more attention than water quality management, partly because the uses of water resource as public and industrial water supply or irrigation in agriculture are more closely related to and easier to quantify their economic cost and benefit than water quality. Rosegrant et al. (2000) introduced an integrated economic-hydrologic water modelling at the Maipo river basin in Chile. A holistic hydrological model at river basin scale was combined with economic analysis for farmers' agricultural activities and water use efficiency. Water rights trading were introduced to allow water resource to go to higher valued agriculture. Economic profits were found by the optimisation. However, water quality was not concerned in this research.

Brouwer *et al.* (2005) and Dellink (2000) discussed the prospects of evaluating total cost of economy for pollution abatement through general equilibrium model. The work of Brouwer *et al.* (2005) was based on scenario analysis in a top-down approach within which various abatement levels were assumed. This approach depicts well the direct and indirect costs of pollution abatement targets, but the options of pollution abatement were inflexible, ignored the significant local differences among river systems, which is not in a "*case by case*" style. Dellink's research (2000) is similar but using optimisation analysis rather than scenarios. He also focused on the dynamic interaction between economy and environment and found that the dynamic specification is highly relevant to the results. As a top-down approach, these studies concentrated on the GDP impacts of implementing pollution reduction requirements such as that required by the WFD.

On the contrary, this research is taking a bottom-up approach, looking for specific abatement options and focus on the impacts within the catchment, especially those along the river. Indirect costs are not considered in this research due to data limitation. A well-calibrated water quality model, which simulates the changes of water quality instead of just focusing on abatement levels, is combined with specific case-based economic analysis in this research. This offers more flexibility and more realistic optimisation to the water quality management.

Although it is rare, there are some studies in the UK taking a similar bottom-up approach to achieve the cost effectiveness in water quality management, back from 1979 by Rowley *et al.* at the Tees Estuary (Rowley *et al.* 1979). The Tees Estuary was "grossly polluted" in 1970 so that it was not able to support fishing livelihoods from Stockton to the mouth of the estuary. Rowley et al. (1979) investigated the possibility of utilizing an emission charge instrument rather than regulatory consents to control the pollution and achieve satisfactory water quality. Nine major industrial pollution sources were included in an economic model in which a least-cost solution in the pollution control can be found combined with transfer coefficients from a water quality model. An *appropriate* charge rate can also be determined for a particular water quality target.

Another one is at the Forth Estuary in Scotland (Hanley and Moffatt 1993; Hanley *et al.* 1998). The most significant problem in the Forth Estuary, similar to the tidal Ouse, is the seasonal DO sag associated with low flow and high temperature conditions, which prevents the salmon from returning after migration. The industrial sources account for 87% of the total BOD loading. Economic model was developed for the estuary to minimize the control costs subject to the environmental constraints, alongside a model of water quality. The research also tried to explore the potential of using a tradable permits system to improve the water quality at a lower cost than that under a CAC approach. In the paper in 1993, Hanley and Moffatt (1993) showed that the flexible regulation was closest to the least cost solution although it could not provide the continuing incentive of reducing pollution in the efficient manner as TPP and emission charges. Hanley (1998) conducted the analysis for both the Emission Permits System (EPS) and the Ambient Permits System (APS). The research emphasized the importance of heterogeneity among the WQM sites in the estuary. The author also pointed out that significant impact of resuspended bottom sediment on the DO distribution can be expected in estuary, from both past and current anthropogenic activities.

Chapter 3 River Water Quality Model

3.1 Introduction

Using quantitative techniques to assess the impacts of pollution on river water quality has a long history, but only recently it has been enormously improved with the assistance of computer technologies such as Geographic Information System (GIS) and much stronger calculation capability. But the underlying concept of most current water quality models is still the same as before, based on the concept of mass balance of constituents first used by the Royal Commission of 1912. Depending on various processes to simulate the river system and different levels of detail information, there are many different water quality models. Most of the current water quality models are constructed with different forms of computer programs. However, the river system is always complicated and difficult to predict. After years of continuous development, assessing the impacts on water quality of diffusing pollution remains one of the major difficulties. As pointed out by Cox (2003b), the simulating abilities of many of the water quality models are still sensitive to the river system they are applied to.

For the reason above, I selected two water quality models in this research that were applied to the tidal Ouse before. The ECoS3 HOT model is good at simulating the fluxes and transformation of pollutants in the river (Tappin *et al.* 2003), while QUESTS1D model is used by EA to assist the design of effluent discharges, which is therefore more focused on the DO issue in the tidal Ouse. Both of them have been successfully applied to various simulations in the tidal Ouse before, but the research later found the ECoS3 HOT model is not well equipped to simulate the location of DO sag in the tidal Ouse despite its success in simulating other major pollutants. The simulation results from QUESTS1D model fit better to the observations. It has also been continuously improved with calibrations and validations because of its special role in EA's decision making in discharge consents.

This chapter comprises 4 sections. Section 1 is the overview of the simulation results from the hydrodynamic models and describes how this chapter is structured. Section 2 describes the development of the ECoS3 HOT model and its previous applications in the estuaries of the UK and Europe. The results of previous applications are introduced and evaluated to determine the possibility of utilising it to model water quality of the Ouse/Humber estuaries. The discrepancy between observed data and simulation results is described in this section, and potential reasons for the model's inadaptability to this research are discussed. Section 3 introduces the QUESTS1D river water model, which is applied to the Ouse and Humber estuaries by the EA for setting consents and as predictive tool for the implementation of potential pollution control options. The QUESTS1D model treats DO saturation and concentration as one of the key determinants in the river water as well as in the model processing. Several studies on the Ouse and Humber estuaries have utilised the QUESTS1D model. The details of these research and their results are introduced. Section 4 analyses the simulation results from the QUESTS1D river water model. Diverse management options are designed to improve the water quality in the tidal Ouse and to tackle the DO sag between Selby and Drax during the summer months. The effectiveness of each control option for improving water quality and tackling the DO sag is evaluated based on the DO profile and the composite score. As these management options are not exclusive, a combination of some of them might be the best practice. The analyses of simulation results also lead to the construction of a transfer coefficients matrix, which would be a critical parameter of the economic model constructed in the next chapter.

3.2 ECoS3 HOT Model

3.2.1 Description of the HOT model

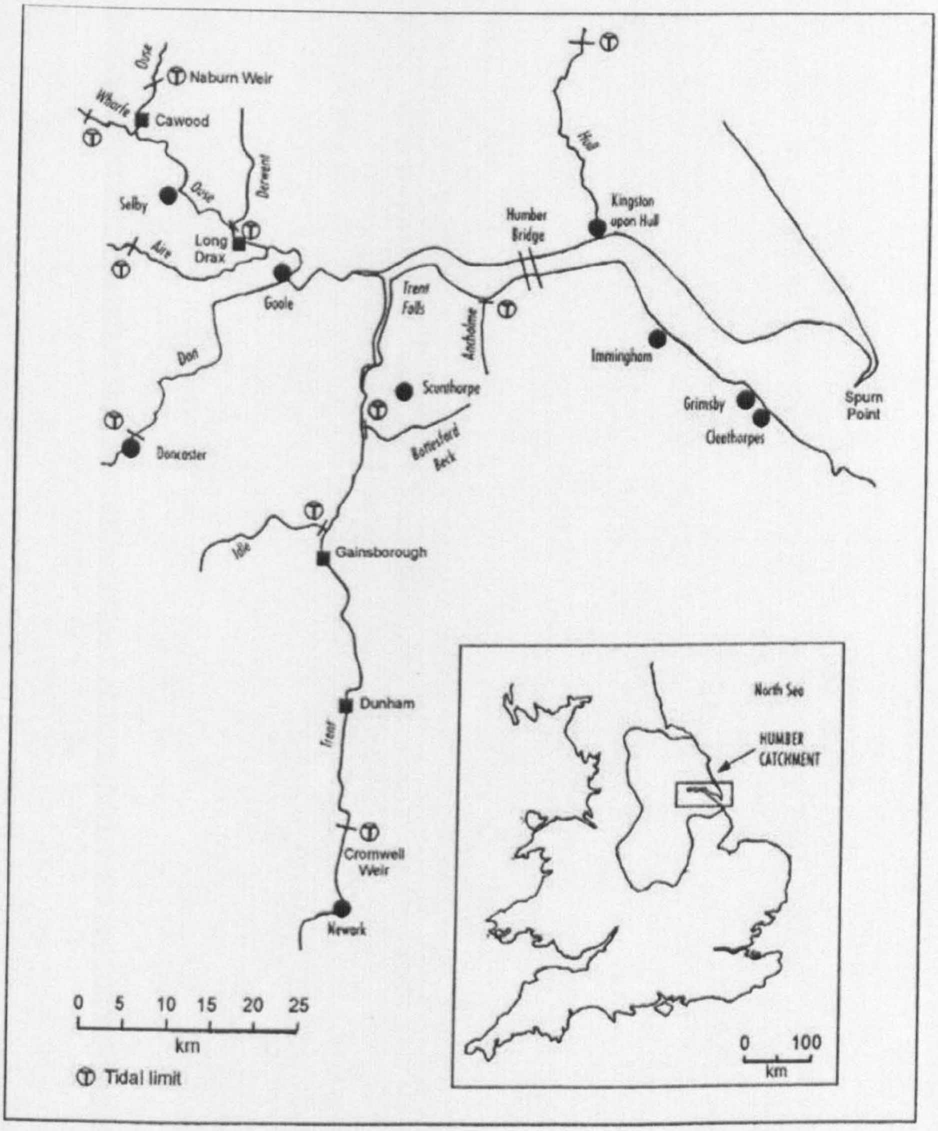

Figure 3.1 The tidal section of Humber, Trent and Ouse system and main tributaries
Source: Tappin *et al.* (2003) redrawn from (National Rivers Authority 1993)

Figure 3.1 shows the structure of the tidal section in Ouse, Trent and Humber system and their main tributaries. The estuaries of the Ouse, Trent and Humber are divided as follows: The Humber estuary extends from the confluence of the Ouse and the Trent, at Trent Falls, to Spurn Point (a distance of 63 km); the Ouse and Trent estuary extend from their tidal reaches, Naburn Weir and Cromwell Weir respectively, to their confluence, distances of 61 and 85 km respectively. Inputs from the Wharfe, Derwent, Aire and Don are located at their confluence as point inputs, while regressions are used to reduce the problem of missing data to an acceptable extent (Tappin *et al.* 2003). All the estuaries are divided into 500-meter segments along their longitudinal axes, and the simulated values of determinants in each segment represent the averages over the cross-section.

Dynamic representations given by this model include water, salt, suspended particulate matter (SPM), suspended phytoplankton, detrital particulate organic carbon (POC), nitrate, nitrite, ammonium, dissolved inorganic phosphate (DIP), particulate inorganic phosphate (PIP), temperature, dissolved oxygen (DO) and other determinants. The model also simulates the dynamics of POC, ammonium and DO in the bed sediments in a simple way. More details of the structure of the HOT model and its rationale have been intensively introduced (Harris and Gorley 1998a; Harris and Gorley 1998b; Harris and Gorley 2003). The HOT model calculates the velocity of the water and solutes by cubature, with aggregate inputs of river, tributaries and discharge sources and constructed mid-tide mean cross-sectional area at any point. Water abstraction currently is not taken into account in the model but could be included as an output of water. For the three reaches, the river Ouse, river Trent and river Humber, any effects of evaporation, precipitation, abstraction or diffuse inputs (from drainage, groundwater) on the volume of water are ignored (Tappin *et al.* 2003).

A description of the transfers and cycling of the constituents simulated by HOT model is given as Figure 3.2. Detailed description and mathematical representations of the transfers between theses constituents can be found in Tappin *et al.* (2003). Data on water flow and constituents concentrations from the estuaries and their tributaries are provided by the Natural Environment Research Council (NERC) Land-Ocean Interaction Study (LOIS) rivers programme. River inputs from the Ouse and the Trent are located at their tidal limits while the inputs from the main tributaries are located at their confluence with the Tidal Ouse, although, except for Derwent they all have long distance of tidal reach beyond the confluence with the tidal Ouse (National Rivers Authority 1993). For unavailable data, the model uses regression to reduce the difficulties of missing of earlier data to an operable extent, based on assumption of flow-concentration relationships (Tappin *et al.* 2003). The HOT model includes sixteen point sources of industrial discharges and sewage effluents in the Humber, five in the Ouse and four in the Trent. Two out of four industries in Selby are included in the model, Harmann & Reimer (now Tate & Lyle Citric Acid) and Hazelwood (now Greencore). These inputs are regarded as minor relative to river inputs, with the exception of ammonium discharge in the Humber and the Ouse (Tappin *et al.* 2003).

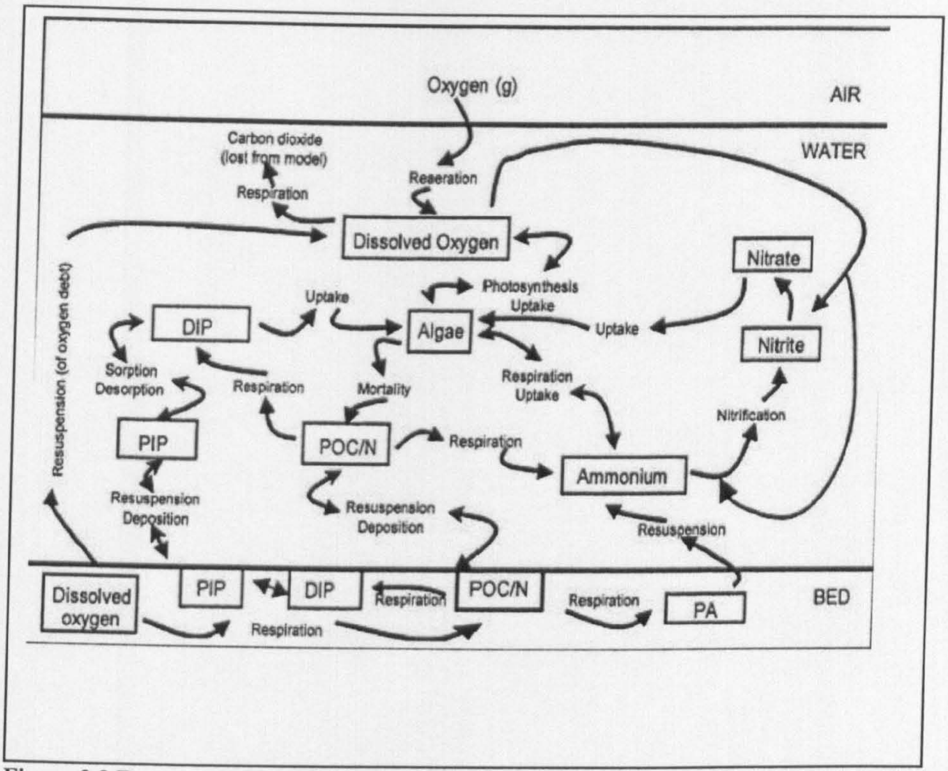

Figure 3.2 Reservoir and transfers of carbon, nitrogen and phosphorus in the model
Source: Tappin *et al.* (2003).

The HOT model was calibrated on the Rivers Atmosphere, Estuaries and Coasts Study (Coasts) (RACS(C)) database for the Ouse and Humber, for carbon, nitrogen and suspended particles utilising a Marquardt minimisation procedure. The part of model for the Trent is not calibrated due to lack of data, but the model incorporates calibrated parameters generated from the Ouse and the Humber to the

Trent model. Harris (2003) provides a detailed process for calibration illustrated by the simulation of the surface salinity profile of Humber estuary.

The axial concentration along the river of SPM, POC, nitrate, nitrite and ammonium in the tidal Ouse and the Humber estuaries are simulated with the HOT model and are compared with the measured concentrations used for model calibration. The simulated concentrations and distributions have also been tested against independent data from the 27 surveys of the Yorkshire Ouse and Humber estuaries during 1994-1996, the LOIS database and the EA measurements. The fits between model results and these data are reasonably good, but with some exceptions.

3.2.2 Simulation of DO distribution

Because of the previous successful experience in the Humber estuaries, ECoS3 HOT model was first applied in this research to analyse the DO saturation (DO%) distribution over the tidal Ouse-Humber estuaries, especially in an attempt to represent the DO sag during the summer months in the Tidal Ouse around Selby and Long Drax. The flexibility of model structure in ECoS3 makes it feasible to change the timing and locations of discharge effluents and include water abstraction as an output. Therefore it should be capable to analyse the effectiveness of alternative pollution abatement options in this research.

However, the simulated DO distribution in the summer along the Tidal Ouse does not fit the DO sag observed in the river. Instead of having significant DO sag around the river length between Selby and Long Drax and having gradually improved DO% as approaching to the Trent Falls, the simulated result in most of the time indicates a smooth DO curve monotonically declining along the river, having the minimal DO concentration around the Trent Falls. Figure 3.3(a) and 3.3(b) indicates the significant discrepancy between model simulation result and observed DO concentration data in two different years.

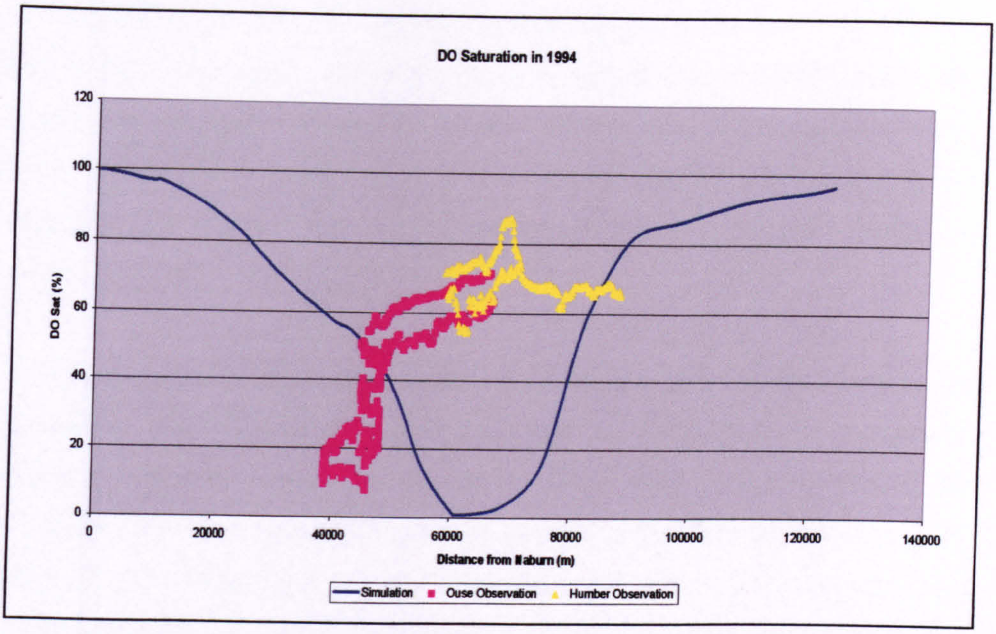

Figure 3.3(a)

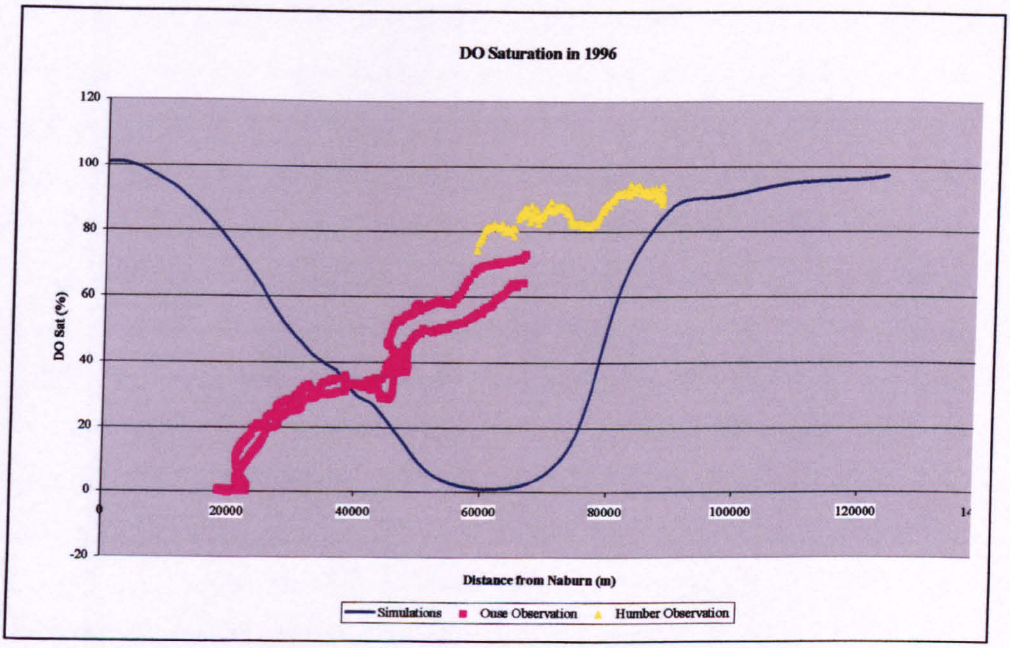

Figure 3.3(b)

Figure 3.3 (a) The ECos3 simulation and observed DO distribution along the river from Naburn to Sea Spurn in summer in 1994 and (b) in 1996. The X-axis is the distance from Naburn Weir, tidal limit of Ouse till the sea Spurn. The dots are observations from LOIS dataset.

The HOT model was not calibrated against DO data when constructed. This is due to the time pressure on the project within which the HOT model was constructed to analyse the transformation and transport of suspended particles, carbon and nitrogen. After looking into the model in detail, I concluded several reasons might be responsible for the discrepancy between simulated results and observed results:

The first is that the biology/chemistry of DO dynamics is not well described in the model. The relationship between the POC mineralisation and ammonium nitrification was not included in the model. There might be competition for the DO in the river water between these two transactions according to their respective kinetics. On the other hand, the rate of nitrification has been found dependent on the concentration of suspended sediment in the water, and this relationship was included in the ECoS3 modelling based on a linear representation derived by Owens (1986). Xia *et al.* (2004) argued that the nitrification rate would increase non-linearly with the increment of suspended solid rather than following the linear relationship applied in the model. But both studies agreed that the nitrification rate would increase with suspended particles in the river water. In the HOT model, POC concentration was assumed to be a constant percentage of SPM in the river water. Therefore the maximum SPM concentration, usually in the turbidity maximum (TM) zone, always accompanies the DO sag or is just slightly upstream. This was also confirmed by the changes in the locations of DO sags when shifting the TM zone. From the simulations, the model suggested the DO consumption from nitrification and POC mineralisation to be the most significant sources of DO consumption in the river, particularly the POC from resuspended bed-exchanged sediment. The DO consumption from the transactions in the sediment was also suggested to be as significant (Hanley *et al.* 1998; Parr and Mason 2004), but it cannot be successfully separated from other sources. However, no matter where the TM zone is located, the DO sag in the simulation always persists until the confluence with the river Trent, without showing a recovery stage which is observed in the lower river Ouse. So the problem seems to lie in the simulation of DO recovery in the river water, which will be discussed later.

There are also some transformations that are not included in the modelling, such as denitrification. Denitrification is the process through which nitrate is reduced to nitrogen or nitrous oxide when the DO concentration in the water is very low. However this process is treated as an insignificant process in the river Humber in HOT model and would have less significant impacts on the DO distribution. Likewise, nitrification is also DO concentration dependent. The minimum DO concentration for nitrification ranges from 0.6-0.7 mg/l (Forster 1974) to 2 mg/l (Clabaugh 2001; Ho *et al.* 2002) under different pH and temperature conditions. The simulation from HOT model was to some extent improved after a threshold concentration of 2 mg/l DO was introduced into the nitrification process.

As for the issue of DO recovery in the river, the major sources of DO in the polluted river include the following, (a) oxygen in incoming or tributary flow, (b) oxygen generated by photosynthesis and (c) oxygen from the reaeration process (Cox 2003a). Because of the muddy nature in the tidal Ouse, photosynthesis from the plankton plants is negligible compared to the other sources. The oxygen reaeration process in the model is represented by the exchange rate between water surface and atmosphere. This process was defined as exchange in the HOT model in order to allow both directions of oxygen dispersion. The direction of exchange is determined by the oxygen concentration between the surfaces while the dynamics of this process were related to the variable of clearance time, which is determined by the river depth and piston velocity of oxygen. The air exchange rate also depends on wind speed (Kremer *et al.* 2003), but it is not represented in the model. Contrast to 2 to 5 m/d with normal water temperature (Broecker and Peng 1983), the piston velocity in the model has to be lowered by several orders of magnitude in order to have realistic simulation in the river. However, modification of exchange rate across the air-water surface would affect simulation of DO% along the whole tidal Ouse, but not be able to correct the wrong location of DO sag predicted by the model. The oxygen from the incoming tributaries maybe an important source in the river Ouse, especially considering the fact that the recovery of DO sag observed in the river Ouse starts around its confluence with the river Aire. The river Don, which is another relatively clean river with large volume, joins the river Ouse 10 km further downstream. But, considering the fact that data of the HOT model is provided through the comprehensive surveys in LOIS programme, there is little possibility that the lack of improvements contributed by inputs the rivers Aire and Don is inherited from false data.

As stated above, these tributaries are not treated as rivers in the HOT model but as point sources. This characteristic implies another important reason for the discrepancy between simulation and observation. Not being treated as tributaries means all the bio-chemical transformation and sediment transport that would otherwise happen in the tributary rivers are ignored, so are the consequences for the flux to the river Ouse. These processes then would be assumed to happen in the river Ouse rather than in the tributaries themselves. Because the tributaries' inputs are data from gauge stations along them, the longer the distance from the gauge station to their confluences with river Ouse, the larger the error introduced in the simulation results. There would be more effects in the summer when the salt wedge extends far up the river Ouse and into these tributaries. Restoring the tributaries in the model to what they were in reality is feasible, but demands sufficient hydrological data to construct the river reach in the model and to represent the transformations within the new river reaches.

This section has discussed several potential reasons why the HOT modelling generates discrepancy between simulations and observed results, but they are by no means all the potential reasons, nor are they as significant to the discrepancy observed. But to quantify their relative contributions in the discrepancy would be a time consuming job, especially when all the parameters in the HOT model have already been calibrated against observations for some other constituents relevant to the DO% through various physical, chemical and biological transformations of species.

3.3 QUESTS1D model

3.3.1 Structure of QUESTS1D model

The original objective of WRc QUESTS1D water quality model was to provide calibrated, time dependent, one-dimensional water quality model of the Humber estuary to National Rivers Authority (NRA) in the Anglian, Seven Trent and Yorkshire Regions (Slade and Morgan 1993a). It was to aid the establishment of discharge consents in the river system comprising the Humber, Ouse, Trent, Aire, Don and Wharfe estuaries. This model is still used by the EA for this purpose. In addition, the model also provides a predictive tool for the impact of pollution loads and scope for evaluating potential pollution control options.

The current QUESTS1D model is a one-dimensional representation of the tidal river system stretching from tidal limits of the Ouse, Wharfe, Aire, Don and Trent to the sea spurn. The QUESTS1D modelling system is made up of several linked programs. The rough data from routine survey and monitoring need to be processed in the data preparation stage to generate the daily data file through statistical programs, SHARE, SYNTH and COLLATE. SHARE is to run under the Test Data Facility (Ellis *et al.* 1992; Clark and Ellis 1993; Slade and Morgan 1993a) to generate description of input data and the latter two are to generate an auxiliary file containing time series of daily input loads (Slade and Morgan 1993a).

The total length of the tidal river system represented by the QUESTS1D model is around 313 km, divided into 282 cells in total. Each cell represents about 1 km length along the river from the tidal limits of the Ouse and Trent at Naburn and Dunham respectively, to some 4.5 km offshore from the sea spurn , with distances of 62.5 km in the Ouse, 84.8 km in the Trent and 62.2 km from their confluence at Trent Falls downstream towards the sea. The tributaries of the Ouse, except river Derwent, are treated as river rather than point sources as they are in ECoS3 HOT model. The rivers Wharfe, Aire and Don in QUESTS1D model stretch from their confluences with Ouse to their tidal limits at Tadcaster, Beal and North Bridge, distances of 69.9, 48.9 and 45.7 km respectively. The river Derwent for the Ouse and river Hull for the Humber are treated as point source due to their low volume and short tidal sections.

For a prototype of the model, the hydrodynamic component and water quality component were simulated by two separate models, which are combined into the QUESTS1D system. The hydrodynamic model is to simulate the tidal movements for a range of tidal conditions and the effect of density gradients on tidal flow, to predict variables such as river level and water velocity. The water quality model, based on the conservation of mass, is compatible to the hydrodynamic model, using the results from hydrodynamic model to incorporate the processes of advection, diffusion, decay and interactions between the determinants in the river water. Concentrations of DO, BOD_5, suspended solids, ammonia, phosphate and metals in each cell are simulated in the water quality model as well as temperature and salinity. As with the ECoS3 HOT model, sediments are also modelled in the QUESTS1D model, with a three interactive layered system. It is made up from a water layer, in which solids are suspended (SS), an upper sediments layer (SED) and a more compact lower bed layer (BED) (Slade and Morgan 1993a). Finally, a statistical shell enveloping these two models was developed to generate statistical output on the modelled water quality results.

At the time of construction, there were approximately 216 discharges to the system, with 52 major inputs, and six additional sites were included in the model in 1993 (Slade and Morgan 1993a). The QUESTS1D model utilized in this research has 56 inputs excluding the returned abstraction from Drax station.

3.3.2 Calibrations and validation of QUESTS1D

The Hydrodynamic model was calibrated against information from a neap tide between 15th and 19th May 1978 and from a spring tide between 19th and 23rd June 1978. Calibration for the Don and Wharfe was using data from October 1967 and March 1968 respectively due to insufficient data in the previous stated period

(Slade and Morgan 1993b). By adjusting the bed friction coefficients along the estuary, satisfactory fit was achieved against the observed levels.

Secondly, the water quality model, which is developed building upon the results of hydrodynamic model, was calibrated against data from a previous water modelling study (Humber Estuary Committee 1982). This dataset comes from two intensive surveys during May and June 1978 for the pollution loads inputs and water quality within the estuary.

The calibration of both models resulted in satisfactory simulation of DO against the observed data. However, as result of the calibration procedure, some questions concerning the modelled processes are raised for further investigation, including:

- The interaction of chlorophyll and suspended solids and its effects on DO.
- The low levels of suspended solids predicted in the upper Ouse.
- The high levels of BOD_5 predicted on the estuary.
- The over-prediction of temperature in the lower Humber.

A further modification was carried out in 1994 after the model was checked against continuous monitoring data from 1992, which aimed to improve the prediction of DO in the lower Ouse. It was found that good agreement of DO in the lower Ouse could be achieved as long as the simulated bed sediment is sufficient to provide a realistic level of suspended solid throughout the run-time. This modification, therefore envisaged the significant impact of suspended solids on the DO concentration, which is also illustrated in the ECoS3 HOT model and other research (Hanley *et al.* 1998; Parr and Mason 2004). Several conclusions were drawn about the relationships between suspended solids, DO, and oxygen demand in the river (Cashman *et al.* 1999), as summarised below:

- Suspended sediment (organic) contributes significantly to total oxygen demand.
- A mobile 'pool' of sediment moves upstream under low freshwater flows and high tidal range conditions which the model did not predict.

- Net upstream transport of sediment under low flow high tidal range condition is not predicted because sediment which is transported upstream during a flood tide is immediately transported downstream on the ebb tide.
- It should be possible to improve the predictive capability of the model by improving understanding of sediment composition and transport and by adjusting model parameters which control sediment transport (critical erosion and deposition shear stress, settling velocity).

These conclusions are also implicitly supported by the HOT model, especially the first one. As for the second conclusion, this attribute of sediment 'pool' was enabled by an upstream-moving TM zone in the HOT model. This is because the frictional dissipation of tidal energy tends to produce a shorter, more rapid flood and a slower ebb, exacerbated by the effects of density-driven vertical circulation, resulting in the upstream movement of fine sediment and the TM zone (Harris and Gorley 1998a). But towards to the limit of salt-water intrusion, this trend would be balanced by steady fresh water outflow towards the sea (Harris and Gorley 1998b), which is the basis of the third conclusion. The fourth conclusion suggests a required improvement for the HOT model as well.

Although further works on the QUESTS1D model is required to improve the way it handles sediment dynamics and oxygen demand, validation against continuous monitoring data for spring and summer in 1995 and 1996 produced satisfactory result. The pattern of DO sag was reasonably predicted in the model, being only slightly optimistic. The most recent validation of the model predictions of DO and salinity against continuous monitoring data was undertaken using the data of 1999. The results of this validation displayed good agreement between the predicted and observed data (Freestone 2001). Thus the QUESTS1D model was considered suitable for river water quality simulation in the Humber system, and consequently for the purpose of evaluating potential pollution controls relating to effluent discharge consents.

3.4 Modelling Results of the QUESTS model

3.4.1 Measures of water quality

In previous research, several water quality measurements were utilized to evaluate the simulated results. During the utilizations, only the pertinent constituents were assessed, despite some of the measurements requiring a set of major constituents to meet the compliance to environmental regulation. DO saturation is a key indicator of water quality and of the health of aqueous habitats and the environment (Freestone 2002). In addition, DO is also the key constraint on many transformations of constituents in the water and thus significantly influences the concentrations of other constituents in the river. Since the DO sag during the summer between Selby and Drax is the most serious issue, DO saturation, BOD_5 and NH_3 in the Ouse were taken as the key indicators of any effects from alternative management actions.

RE Objectives and the GQA are both based on the last three years routine sampling data, comparing with the 95%ile value of the summary statistics from either observations or simulated results against the designated class limits. The RE classes range from RE1 (good quality) to RE5 (bad) and GQA grades range from A (very good) to F (bad). Both of the measurements have a wider range of assessed determinants than just three key ones mentioned above. They are not considered in previous research, nor are they in this one. Further details of these RE and GQA measurements are available from the EA.

The EA adopted a DO minimum standard of 30% saturation in the tidal Ouse, mainly to allow the return of Salmon during the summer, specifically for the juvenile salmonids. This standard only applies to saline reaches in the Ouse and Humber system, which is basically from the confluence of the rivers Ouse and Trent. However, it is also used to assess the compliance at each cell in the model, including both saline and fresh water reaches for 1%ile or 2%ile values. The number of cells, which fail to comply with this standard of no less than 30% DO saturation, is reported for each simulation run.

The composite score of the Estuarine Working Party Classification Scheme (EWPCS) provides a convenient way to quantify the effects on DO saturation from various controlling options. It is calculated as below: Only the DO saturation as a water quality indicator is considered in the classification; a value range from 0 to 10 will be assigned to each cell depending on its 5%ile DO saturation level. Specifically, the composite score calculated as follows in this research: 10 for the cell with above 60% DO saturation at 5%ile, 6 for above 40%, 5 for above 30%, 4 for above 20%, 3 for above 10% and 0 for below 10% (Environment Agency 1998). The composite score of each river reach is represented by the summation of the values in each cell within the river, therefore it reflects dissolved oxygen levels throughout the estuary over the twelve months period of a model run (Cashman *et al.* 1999). The changes in the composite scores among simulation runs illustrate the difference of their impacts on the DO saturation throughout the estuary, particularly in the area suffering from DO sag.

3.4.2 Scenario design and data processing

Similar scenarios of alternative pollution control options have been evaluated under QUEST1D model as in previous research, but more intensive experiments were carried out for those options of high sensitivity to the distribution of DO saturation in the tidal Ouse. The evaluated alternative options include:

- Changes in the locations of effluent discharge from the four major plants in Selby
- Changes in the timing of effluent discharge over the year, for example no discharge in summer or only discharge in high-flowed winter.
- Return of water abstracted by Yorkshire Water beyond the tidal limit of River Ouse and Derwent.
- Change in the timing of effluent discharges during the day with respect to local high water, i.e. from continuous discharges to discharges in six hours during the ebb only.
- Change in the load of effluent discharges from the Selby industries.

The research also simulated the water abstraction at the Drax station, in order to quantify its impact on the DO saturation in tidal Ouse.

Seven alternative locations of effluent discharges were arbitrarily chosen for the analysis, located along the tidal Ouse at 61km, 51km, 35km, 25km, 22km, 13km, and 1km upstream of the confluence of the river Ouse and Trent at the Trent Falls. The original effluent discharge locations for the four plants at Selby are around 41km upstream of confluence. Discharge point other than the original location is reached by building pipes to transfer the effluents. This scenario is to evaluate the location effects of effluent discharges on the DO saturation along the tidal Ouse and to provide information on the optimal discharge location.

The alternative discharging scenarios over the year are without discharge in summer and only discharge in winter. In the former scenario, there is no effluent discharge from the four plants in Selby in the tidal Ouse during June, July and August while the effluent discharges in December, January and February doubled. In the "only discharge in winter" scenario, all the effluent discharges from the Selby plants are discharged during December, January and February and there is no discharge in the rest of the year. Since the DO sag usually happens in the low-flowed summer, this scenario is expected to see to what extent the problem could be avoided by shifting the discharge patterns over the year. However, to undertake this option, the plants need to build vast waste storage facility to store the effluents.

The impact of returning the significant water abstraction from Ouse and Derwent are examined, either return abstracted water in any one of them or both. Since the abstracted water is to supply potable water to the population in the tidal Ouse catchment, the water would be otherwise abstracted from elsewhere in the Ouse system or from other systems if it was not abstracted from Derwent and Ouse. The abstracted water will be returned to the river system through the STWs. Therefore returning abstracted water is not a feasible option unless the potable water could be attained from elsewhere.

The alternative to discharge continuously during each day was to store the effluent and to discharge only for six hours in the day during the local high water. The local high water at Selby was considered four hours later than that at the sea spurn. This scenario is to evaluate how much the local high water during the ebb could alleviate the DO sag issue. The changes in the level of effluent discharges try to evaluate the consequent impacts on DO profile under different proportion of effluent discharge from Selby, in order to evaluate their marginal effects on DO distribution along the tidal Ouse and Humber.

Since the dataset for effluent discharge was not fully complete, in order to evaluate the DO saturation with the improved discharge consents in Selby after 2000, two separate datasets were constructed for 2001 and 2002. One is at the same level of effluent discharges as previous years; another is updated according to the "future" effluent discharge consents, which is supposed to be implemented by the EA since 2000.

The data for simulation ranges from 1995 to 2004. However, 1998, 2003 and 2004 are not simulated in the QUESTS1D model due to insufficient or incomplete dataset. Some other data are also missing in the dataset from the EA as follows:

1. The data of salinity for all years are missing. Salinity was defined as the weight of dissolved inorganic compounds in grams in 1 kg of seawater, after all the bromide and iodide is converted to chloride, and all carbonates converted to oxides. This could be calculated from chlorinity of the flow following the Knudsen equation: $S‰ = 0.030 + 1.8050 \times \mathrm{Cl}\,(\mathrm{g\,Cl/l}) \times 1/\mathrm{P}$, where P is the density of seawater at that chlorinity. Since the river and tributaries upstream of the confluence of River Ouse and Trent are regarded as fresh water, P is the same as the fresh water density 1000g/l.
2. Data for total phosphorus is not available for all the years. It is suggested that orthophosphorus, which is available, is about 80% of total phosphorus for all the inputs including riverhead water along the river Ouse. Therefore, in this research total phosphorus data is calculated according to orthophosphorus.

3. Detailed data of effluent discharges from BOCM and Tate & Lyle Citric Acid (TLCA) are missing. This data adequacy is solved as follows. For TLCA, effluent discharge before 2000 were assumed at the same level as that in 1996 and 1997, which was described in the report of Cashman *et al.* (1999); effluent discharges since 2000 were assumed same as the Environmental Agency's "future" consents described in their report. This is reasonable because since 2000 TLCA has managed to reduce its effluent just below the "future" consents although the consents were not brought into force at that time. For BOCM the effluent discharge in the simulation were assumed to be the same level as that in1996 and 1997 described by Cashman *et al.* (1999), the possible improvement after 2000 were incorporated in the simulation with "future" consents for all the four plants.

Through the analyses, the research was able to estimate the Transfer Coefficient Matrix (TCM) for the Ouse system from different discharge locations to the EA's water quality WQM sites, by comparing the reduction in pollutants between discharge points and the WQM sites. The TCM can provide a quick reference for the distribution of assimilative capacity of the river water, indicating how much pollution in the river could be degraded through the assimilation process. More details of the applications of TCM in the regulative system of river policy will be found in later chapters.

3.4.3 Results of Analysis

All the simulations are based on simulation for one calendar year. As RE and GQA classifications published by the EA are based on three successive years, therefore RE and GQA are not used as main indicators in the analyses of simulation results. The results are evaluated and ranked upon the composite score of EWPCS, and illustrated by the DO saturation profile along the tidal Ouse.

Figure 3.4 displays the flow in the river system from 1995 to 2003 and Figure 3.5 shows the simulated base-run DO saturation for each year based on existing situation without scenario manipulations. The effluent and boundary dataset for the simulation has been discussed before.

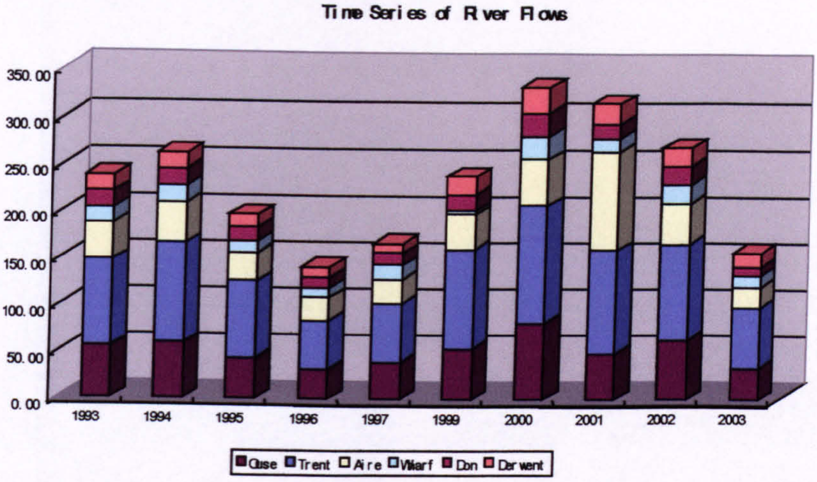

Figure 3.4 Time series of river flows in the Ouse, Trent, Humber and their tributaries

The vertical axis of Figure 3.4 is m^3/s and all the flow from tributaries and rivers are added together so the height of each column means the average of total flow of river system in that year. In Figure 3.5, the DO saturation are depicted at the 5%ile value, this is because the RE and GQA are classified on 95% of compliance and the composite score of EWPCS are also calculated based on 5% DO saturation profile over the estuaries. The value of the x-axis is the cell number along the tidal Ouse from Naburn downstream to sea spurn, from cell 158 at the Naburn Weir to cell 282 at the end of river Humber, one kilometre for each cell. The vertical lines in between represent the locations of EA's WQM sites for water quality, named Naburn, Cawood, Selby, Long Drax, Boothferry Bridge and Blacktoft. The DO saturation profile for 2001 and 2002 were simulated twice, first is based on the previous effluent discharge consents for the plants in Selby and the second is based on the EA's "future" effluent discharge consents. It shows that the river flows are of particular importance to the DO saturation profile in the river system. The worst DO sag appeared in 1995 and 1996, while 1996 is a very dry year with drought in summer. The DO profiles of other years are increasing as the total flow of river system increases. 1995 and 1997 has similar averaged water flows and tributary inputs, but the DO% of these two years are significantly different. This is because 1995 has very high flow in winter and much less water during the summer months (Tappin *et al.* 2003) while the flow are more evenly distributed during 1997, therefore the river system suffered from DO sag in the 1995 although the averaged flow did not indicate the insufficient flow in summer.

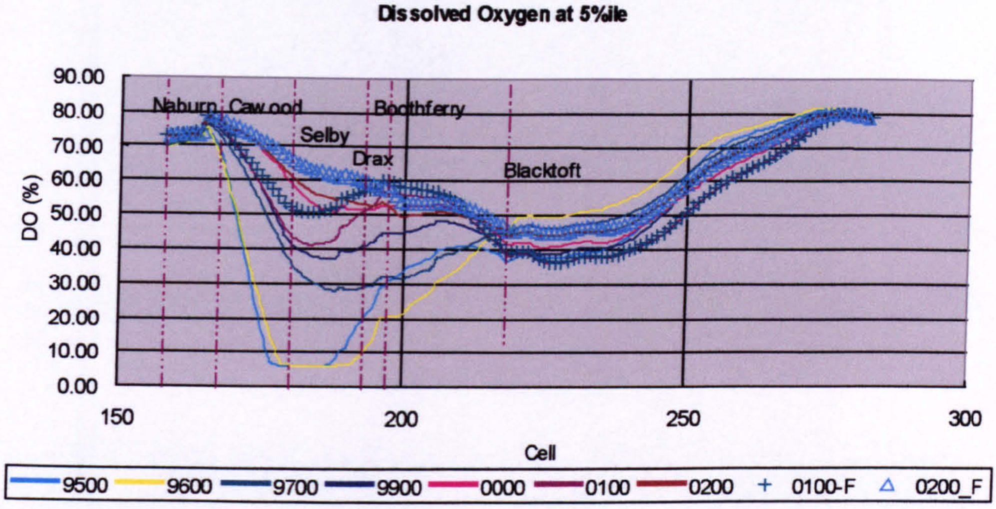

Figure 3.5 The Base-run DO saturation for different years

3.4.3.a Location Effects

The direct expression of the location effects of the effluent discharge can be seen from Figure 3.6, which indicates the distribution of DO saturation in the tidal Ouse when the effluents from the four major plants in Selby are discharged at various locations along the river.

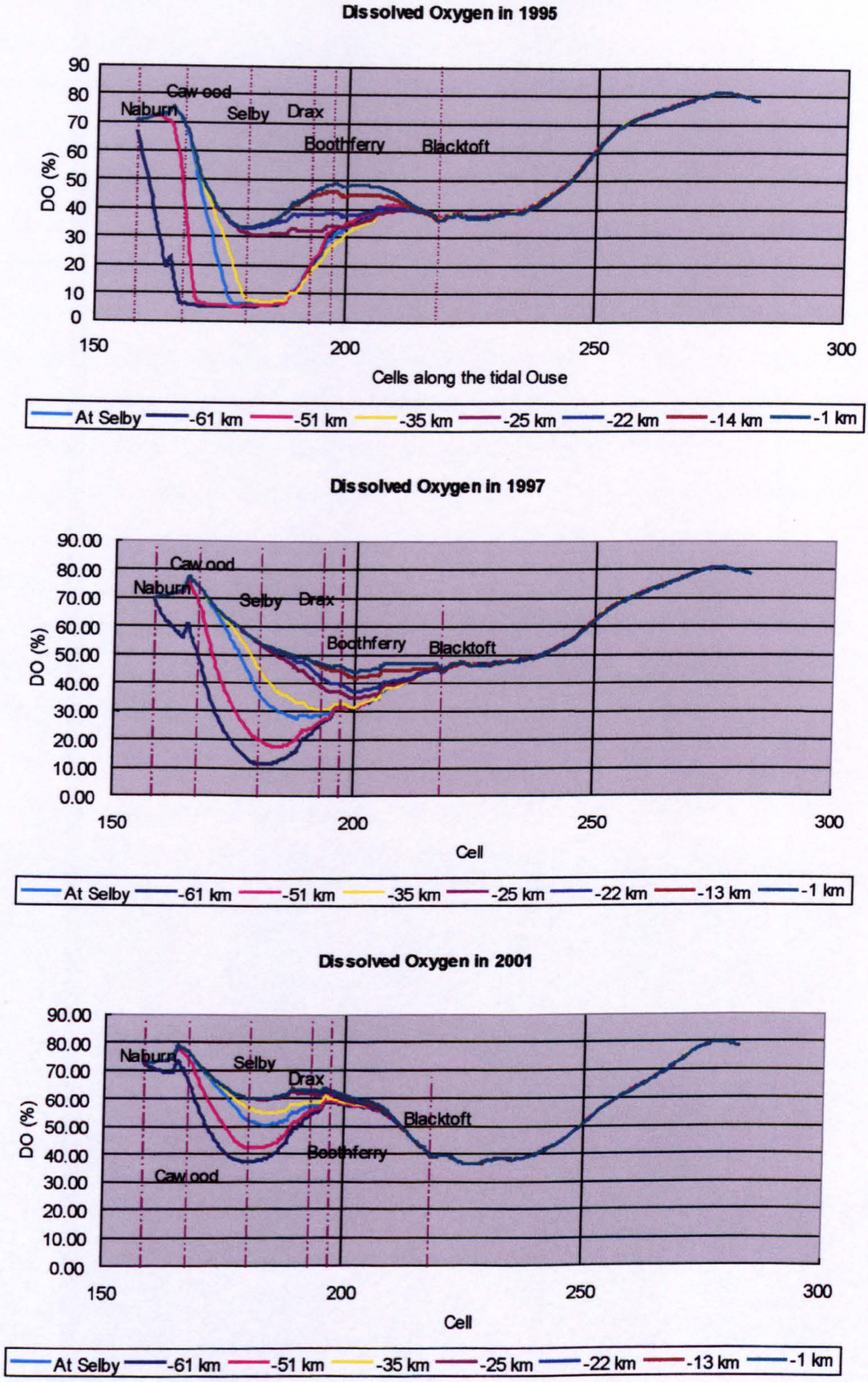

Figure 3.6 5%ile DO profile at various effluent discharge locations in 1995, 1997 and 2001

It is obvious to see the trend of water quality improvement as the location of effluent discharges moves downstream towards the confluence of Ouse and Trent. In 1995, there was a significant DO sag between Selby and Drax when effluents were discharged anywhere beyond Selby. However, the minimum of 30% DO saturation that is required by the EA could be marginally achieved when the effluents were discharged from somewhere 25 km upstream from the confluence, with a leap from the poor water quality. If the effluent discharges were moved even furtherer downstream, the significant DO sag appeared in 1995 could be effectively dispelled from tidal Ouse. In 1997 and 2001, there is no obvious leap as the discharge location moves downstream, partly because of the better water quality in these two years. As expected, the location effects are greater in the year of poor water quality than those with better water quality, and are decreasing as water quality improves. Generally, their effects depend on location within the same year. The improvement in DO saturation is always increasing fast in the region from 41 to 22 km upstream of the Trent Falls. After this region, the improvement is getting slower towards the seaward direction, and negligible after the Boothferry Bridge.

Table 3.1: The EWPCS scores and rate of improvements at various discharge location

Overall Estuaries EWPCS Score of different discharge locations in three years

Point	Distance	1995	1997	2001	Improvement per km		
					1995	1997	2001
A	-61	2061	2219	2355			
B	-51	2130	2261	2373	6.90	4.20	1.80
C	-41	2162	2298	2397	3.20	3.70	2.40
D	-35	2171	2321	2413	1.50	3.83	2.67
E	-25	2245	2338	2473	7.40	1.70	6.00
F	-22	2257	2351	2481	4.00	4.33	2.67
G	-13	2281	2364	2493	2.67	1.44	1.33
H	-1	2285	2368	2501	0.33	0.33	0.67

Ouse/Humber EWPCS Score of different discharge locations in three years

Point	Distance	1995	1997	2001	Improvement of moving downstream		
					1995	1997	2001
A	-61	694	825	865			
B	-51	753	866	883	5.90	4.10	1.80
C	-41	791	903	899	3.80	3.70	1.60
D	-35	801	926	915	1.67	3.83	2.67
E	-25	869	940	975	6.80	1.40	6.00
F	-22	876	949	983	2.33	3.00	2.67
G	-13	894	958	991	2.00	1.00	0.89
H	-1	894	958	995	0.00	0.00	0.33

The EWPCS scores for the overall estuaries and Ouse/Humber estuary under each discharge location were indicated in the Table 3.1. The negative sign in the second column simply means the effluent is discharged upstream of the Trent Falls. The monotonic water quality improvements as effluent discharges moves downstream from point A to H are represented by the increasing composite scores, both for the Ouse/Humber estuaries alone and for all the estuaries including Ouse, Trent, Humber and their tributaries. The last three columns to the right indicate how fast the DO saturation is improving along the river length, displayed by the increase of composite score for each kilometre the location of effluent discharge moves downstream.

For 1995, the high rates of improving take place when discharge location moves from D to E. The improvement rate between point A and B is also high, but discharging in both of the points would lead to even worse pollution in the tidal Ouse than current, so they are not considered as an acceptable option. This applies to 1997 and 2001 as well. The highest rate of improvement in 1997 is at different region to 1995, from point C to D and E to F, whereas for 2001 the area with most improvement for every kilometre is same as in 1995. Therefore, indicated as the highlighted area, the discharge locations between point C and F have the highest rate of improvement in DO saturation for each kilometre moving downstream. Point C is the existing discharge location in Selby area, and point F is at Boothferry, which is one of the six EA's WQM sites along the tidal Ouse. The difference of improvement rates is attributed to various factors, including geographical structure, hydrological dynamics, phytoplankton composition, ambient water quality and tributary inputs. They are as a whole considered as the assimilative ability of the river water in degrading the pollutants inside the water. For the location where the assimilative ability is low, the same amount of effluent discharge leads to larger impact on DO saturation than the location with high assimilative ability. The diminishing rate of improvement along the river therefore indicates an increasing assimilative ability in the river water. Hence, the highest rate of change in water quality actually indicates the location least resistant to the effluent discharges.

3.4.3.b Changes in the Timing of Effluent Discharges

Two different scenarios are designed to evaluate the effects of changes in the timing of discharges on the DO saturation in tidal Ouse. Since the significant DO sag happens most during the summer when water flow is low, shifting the effluent discharges from summer to winter is expected to alleviate the DO sag issue in summer. The first scenario is to store the effluents in June, July and August and double the effluent discharges during December, January and February, which requires the plants to store at least 25% of their annual effluents. The second one evaluated the effects when effluent discharges are only allowed in winter, i.e. December, January and February and to store up to 75% of the plants' annual effluents. Some of the simulated results are illustrated by Figure 3.7 (a)-(d).

Figure 3.7 (a)-(d) illustrate the effects on the DO saturation profile of shifting discharge timing over the year in 1995, 1996, 1997 and 2002. Significant improvement could be found in 1995 and 1996 when severe DO sag were present during the summer. Year 1997 has moderate improvement by shifting the effluent discharges into winter while the improvement in 2002 is almost negligible. As stated above, due to the exceptional water flows distribution over 1995, shifting discharge into winter has greater impacts on DO saturation than in 1996. Storage of 25% annual effluents would be able to elevate the DO sag barely above the 30% minimum prescribed by the EA in 1995, but this scenario does not lead to the same improvement in 1996. As for 1997, 25% storage still led to a significant improvement, which eliminated the DO sag between Selby and Drax and elevated the DO saturation above 40%, even with original water quality that was much better than 1995 and 1996. The shifting of effluent discharges could hardly make any change in the DO saturation profile in 2002 when the future effluent discharge consents were implemented, because of the high summer flow in 2002 and the stricter effluent discharge consents. For all these four years, another 50% annual effluent storage in the scenario of "discharge only in the winter" could only result in small proportion of DO

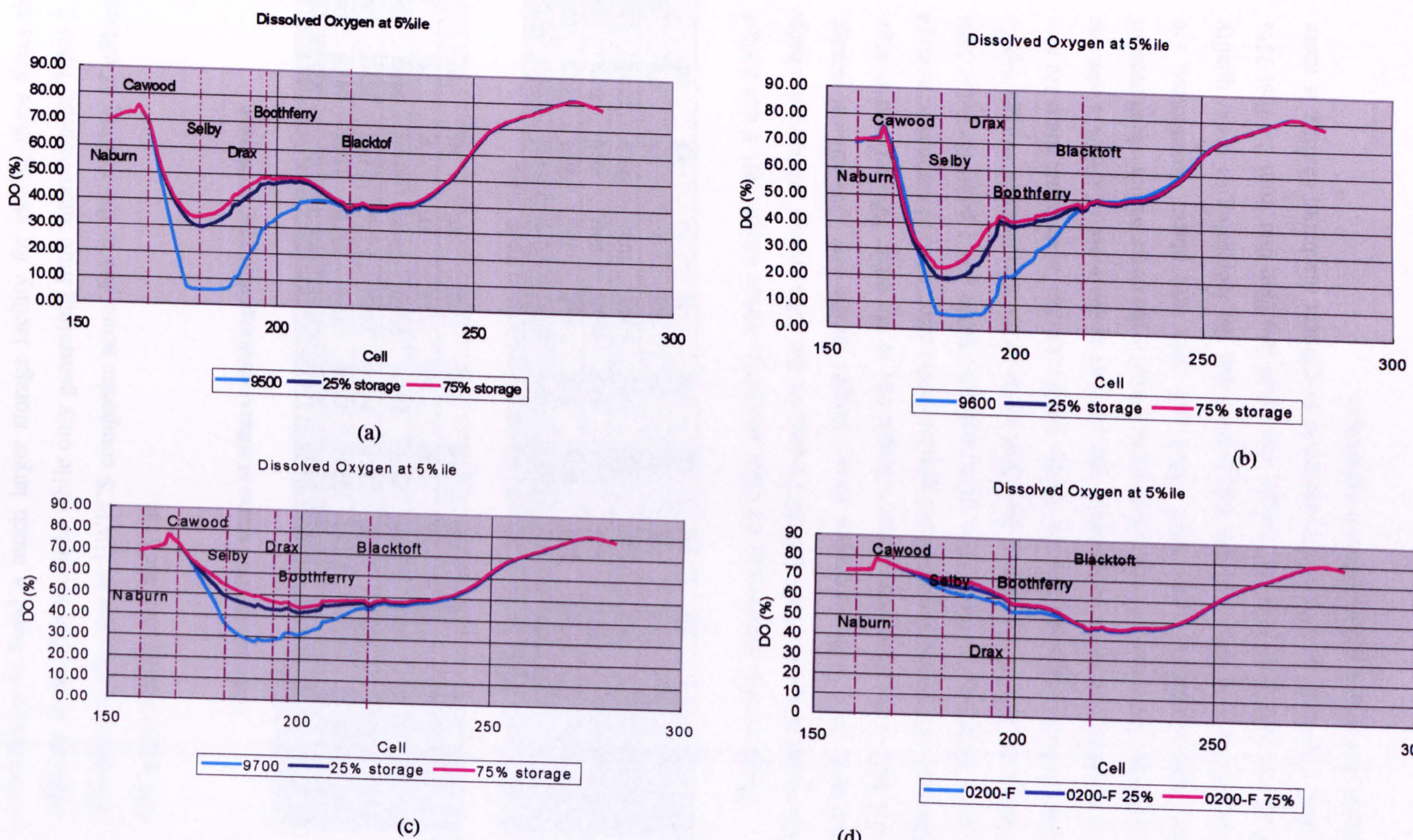

Figure 3.7 Effects of changing discharge timing over the year

saturation improvement compared with the first scenario. Therefore, it is probably unnecessary to build a much larger storage facility for storing three times the effluents in the first scenario while only generating little improvement. Table 3.2 indicates the changes in EWPCS composite score due to the shifting of effluent discharges timing over the year.

Table 3.2: EWPCS scores of effluent discharge shifting over the year

Overall Estuaries

Scenario	1995	1996	1997	1999	2000	2001	2001-F	2002	2002-F
Continuous	**2162**	**2173**	**2298**	**2350**	**2404**	**2376**	**2397**	**2436**	**2488**
25% Storage	**2285**	**2288**	**2360**	**2402**	**2552**	**2458**	**2470**	**2512**	**2540**
75% Storage	**2299**	**2308**	**2376**	**2463**	**2560**	**2509**	**2509**	**2540**	**2544**
SC 1	**123**	**115**	**62**	**52**	**148**	**82**	**73**	**76**	**52**
SC 2	**14**	**20**	**16**	**61**	**8**	**51**	**39**	**28**	**4**

Ouse/Humber NWC Score of different discharge locations in three years

Scenario	1995	1996	1997	1999	2000	2001	2001-F	2002	2002-F
Continuous	**791**	**804**	**903**	**920**	**954**	**895**	**899**	**970**	**1010**
25% Storage	**892**	**890**	**950**	**948**	**1042**	**952**	**964**	**1022**	**1038**
75% Storage	**905**	**906**	**962**	**993**	**1050**	**1003**	**1003**	**1038**	**1038**
SC 1	**101**	**86**	**47**	**28**	**88**	**57**	**65**	**52**	**28**
SC 2	**13**	**16**	**12**	**45**	**8**	**51**	**39**	**16**	**0**

There are two simulations for 2001 and 2002, within which 2001-F and 2002-F are simulations of 2001 and 2002 based on the EA's 'future' effluent discharge consents. SC1 is the composite score changes for the first 25% effluent storage, and SC2 is the composite score changes due to the other 50% effluent storage. Because of the predetermined designation in the calculation of composite score (4 points difference between 60% DO% and 40% DO% and 1 point for every 10% change in DO% below 40%), the effect of the extra 50% effluent storage was to some extent exaggerated by the higher weight. On the other hand, there are also some changes without being taken into account as the score of EWPCS remains unchanged between 40% and below 60%, which is the range within which most of the improvement in years 1997, 1999 and 2001 take place. Nonetheless, the quantification of water quality still proved that the increasing of water quality from the 50% extra effluent storage was much less than that from the first 25% effluent storage, and the improvement from effluent discharge shifting is more apparent with less summer flow in tidal Ouse.

3.4.3.c Return of Water Abstraction

Due to insufficiency of abstraction data in Ouse and Derwent, this scenario was only evaluated for the simulations of 1996 and 1997. The water abstraction in river Ouse and Derwent are both taking place beyond their tidal limits, therefore the return of water are represented by the changes in the headwater of river Ouse and Derwent.

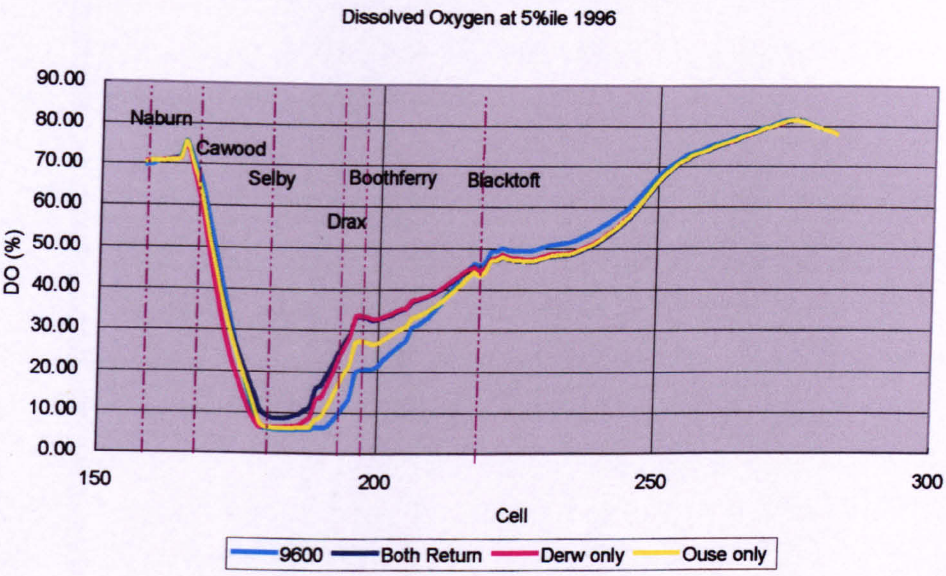

Figure 3.8 (a) Effects of abstraction return in 1996

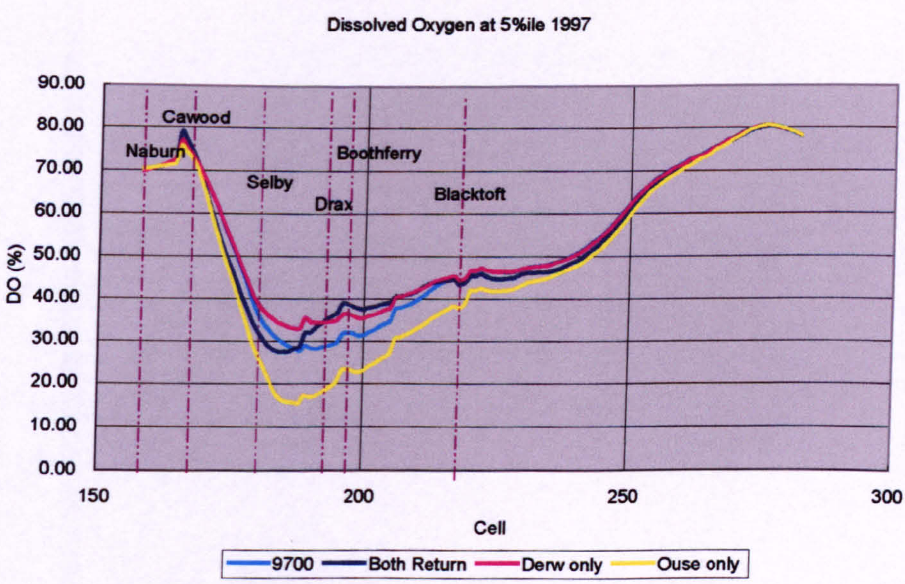

Figure 3.8 (b) Effects of abstraction return in 1997

Figure 3.8 (a) and (b) indicate the effects of returning abstracted water on the DO saturation profile. Three difference return scenarios are examined in each year: returning the abstraction in either Ouse or Derwent only or returning abstraction in both rivers. As expected, the return of abstracted water under 1996 flow conditions leads to slight improvements in all the scenarios. The return of water abstraction in Derwent results in more improvement than water abstraction in Ouse because of its better water quality and higher volume abstracted. The improvement of water abstraction return becomes significant from confluence of Derwent towards to the confluence of Trent. Similar simulation result was also discussed in Cashman et al.(1999).

However, the return of abstraction water resulted in some unexpected result for 1997. Unlike the improvement in water quality from the return of abstracted water in river Derwent, the return in river Ouse knocked down the water quality by a remarkable extent. Therefore, the return of abstraction in both rivers has less improvement than returning in river Derwent alone. A possible reason for this phenomenon is that because of the poor water quality of Ouse, the returned water abstraction adds pollutants back into the system as well, whose effects on the water quality overshadows the impacts on water quality from increased flow. More analyses are needed when abstraction data of other years is available.

The returned water would generally improve the water quality over the whole river length, depending on the fresh flows and effluent discharge conditions at the time. Comparing with returning water abstraction in river Ouse, abstraction return in river Derwent has more improvement on the water quality downstream of its confluence than upstream, from some 30 km upstream of Trent fall. It also has more effects than return in Ouse due to its better water quality. However, the DO saturation improvement from the option of water abstraction return is still limited comparing with the two options above, therefore it cannot be considered as an effective option alone.

3.4.3.d Discharge Effluents with the Ebb

The first attempt of discharge during the local high water is to take advantage of the ebb tide so that it can bring the effluent discharge downstream quicker than other time of the day. There is a four-hour lag between the local high water and the high water at the sea spurn. In this scenario, the daily effluent is discharged within six hours during the ebb tide instead of continuously discharged over the whole day. However, the simulation results show no difference from the daily discharge shifting. The reason could be that, due to the tidal nature of Ouse and Humber, the river flow is held up in the river system for a long time before it passes through tidal Ouse. Therefore, the effluents from the plant would be completely mixed with river water and stay long enough to have their impacts on DO saturation regardless the timing of discharge during the day.

3.4.3.e Changes in Effluent Loads from Sources

Three years are chosen to evaluate the effects of the effluent discharges from the four major plants in Selby. Year 1995 is regarded as one of the years with most severe DO sag, mainly due to its exceptional flows distribution over the year and drought in summer. Year 1997 has similar average of water flow as 1995 but more evenly distributed over the year, which is regarded as a year with moderate flow. Year 2001 has the second highest flow from 1993 to 2003, with the more stringent effluent discharge consents, resulting in much better water quality in tidal Ouse above 50% DO saturation at 5%ile during the summer months. The three different years were expected to represent the effects of effluent discharge on DO saturation under the worst, moderate and very good water quality conditions. Their respective effects on the DO saturation profile are illustrated by Figure 3.9 (a) to (c).

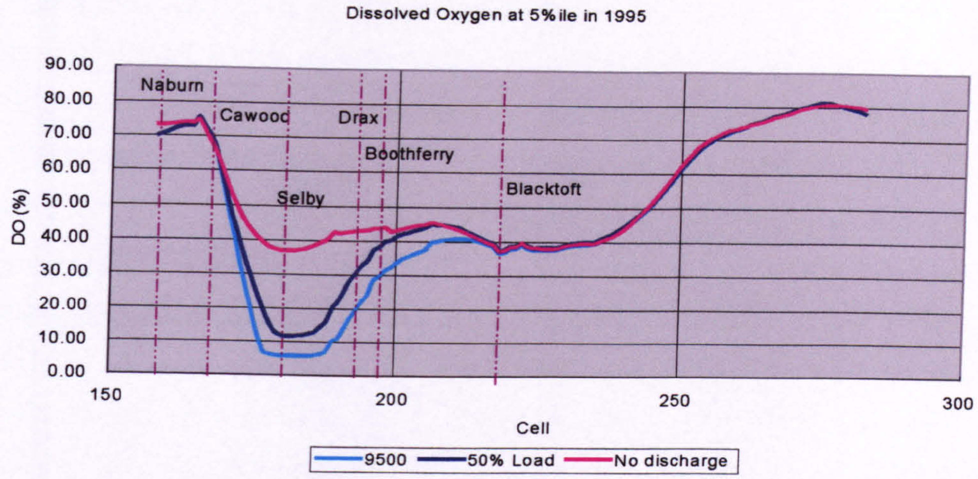

Figure 3.9 (a) Effects of effluent discharges in 1995

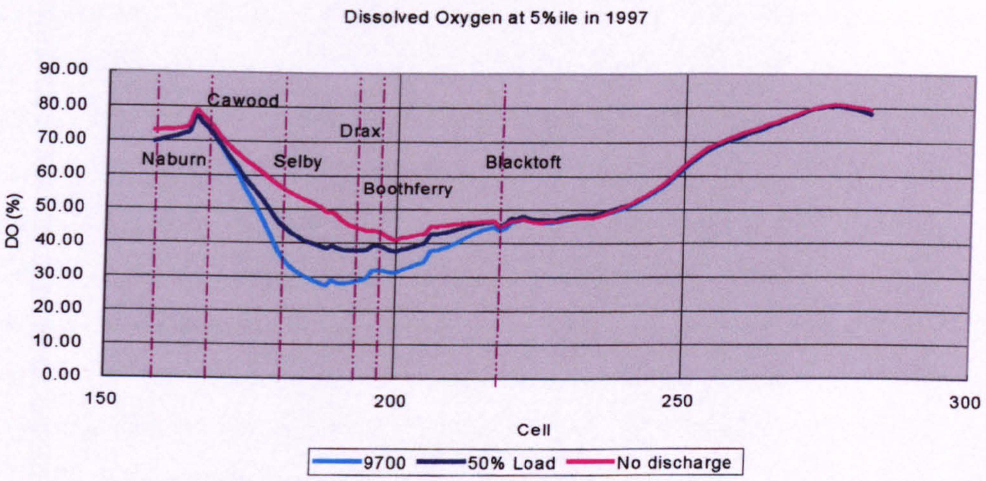

Figure 3.9 (b) Effects of effluent discharges in 1997

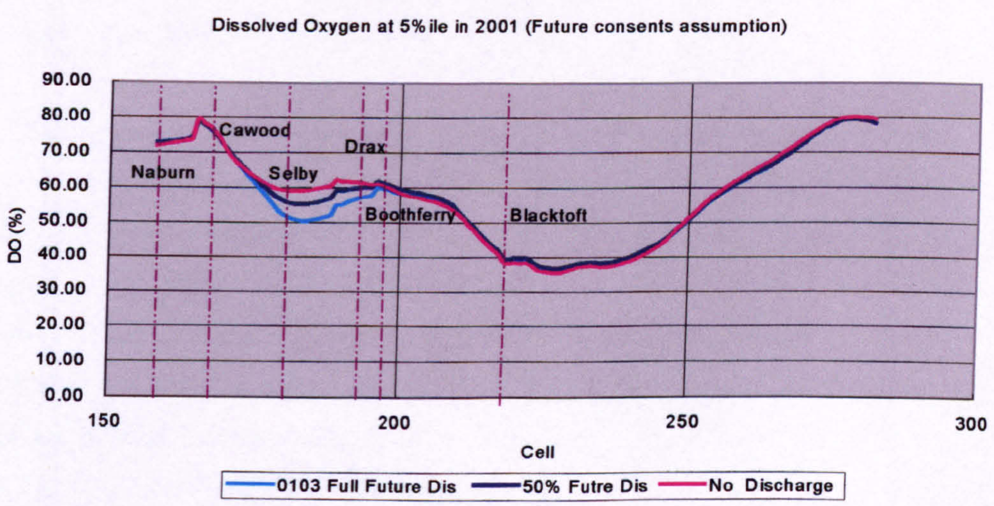

Figure 3.9 (c) Effects of effluent discharges in 2001

The DO saturation profiles generated indicate significant improvement in 1995 and 1997, and only slight improvement in 2001. However, up to 50% of effluent reduction from Selby only resulted in small proportion of improvement in 1995, as 50% effluents from Selby were already strong enough to suppress the DO saturation down to just above 10% saturation during the exceptional dry summer. When there is no effluent discharge from Selby plants, it would allow the DO saturation to be elevated to just below 40% around Selby and Blacktoft. The first decline of DO saturation might be due to the effluent from STWs whereas the latter one was probably dominated by the inputs from river Trent. The large difference between the 50% and 100% effluent reductions reveals a rapidly declining curve of the DO saturation with increase in effluent discharges. On the other hand, the result also illustrated that in a year like 1995, the reduction of effluent discharge should not be used as an effective option to tackle the DO sag problem. The situation in 1997 was slightly different due to the higher flow in summer. The improvements from the first and second of 50% reduction are similar. The first 50% reduction of effluent discharge could lead to more than 10% increase in DO saturation towards 40% at some particular positions, therefore gives more credibility to effluent reduction as an option to increase DO saturation in 1997. For the situation in 2001, the improvement in DO saturation from reduction of effluent is very limited. With above 50% DO saturation along most of the length of river Ouse, even 100% of effluents reduction could not bring much difference in the DO saturation. This reflects the other sources of DO% consumptions, such as suspended solids with tide.

More scenarios have been examined by QUEST1D model to evaluate impacts on river water quality from various effluent discharge levels. Effluent discharge levels that were simulated within these three years range from 0% to 150% of original levels. Their effects on the water quality, in forms of EWPSC composite score, are listed in Table 3.3 as below. The first three columns are the composite scores of the estuaries under various effluent discharge levels. The next three columns are the corresponding changes in composite score compared with the score of original effluent load. The largest increase of composite score by effluent reduction was found in 1995, followed by 1997 and 2001 because there are larger impacts on the DO% in the years like 1995 with less assimilative ability.

Increasing the effluents of 2001 by 50% only results in 4 points decrease in the composite score, reflects a very promising assimilative capacity of the river water in rainfall-rich years as 2001.

Table 3.3: Effects on river water quality of various effluent levels

Overall Estuaries NWC Score of different discharge loads in three years

Load	1995	1997	2001	1995	1997	2001
0%	**2288**	**2368**	**2463**	**126**	**70**	**66**
50%	**2248**	**2340**	**2434**	**86**	**42**	**37**
90%	**2176**	**2310**	**2405**	**14**	**12**	**8**
100%	**2162**	**2298**	**2397**	**0**	**0**	**0**
110%	**2154**	**2295**	**2389**	**-8**	**-3**	**-8**
150%	**2116**	**2263**	**2380**	**-46**	**-35**	**-17**

Ouse/Humber NWC Score of different discharge loads in three years

Load	1995	1997	2001	1995	1997	2001
0%	**903**	**970**	**962**	**112**	**67**	**63**
50%	**856**	**931**	**932**	**65**	**28**	**33**
90%	**800**	**915**	**907**	**9**	**12**	**8**
100%	**791**	**903**	**899**	**0**	**0**	**0**
110%	**784**	**901**	**899**	**-7**	**-2**	**0**
150%	**754**	**874**	**895**	**-37**	**-29**	**-4**

3.4.3.f Transfer Coefficients Matrix (TCM)

Combination of the analyses of location effects and effluent discharge levels sheds a light into the transfer coefficients matrix, which gives parameters to the following chapters of this research. TCM is a matrix of transfer coefficients within which each transfer coefficient indicates how much the concentration of pollutant has changed between two points along the river. Since the reduction in the pollutant concentration is mainly the result of assimilation processes in the river water, the TCM depicts the assimilative ability of the river water between any two points along the river. The TCM calculated in this research is based on BOD_5 concentration in effluent discharges from Selby industries. Therefore, it is a TCM of BOD_5 concentration. The details of the calculation are as follows. Various effluent discharge levels from the four plants in Selby are simulated, range from no effluent discharge to 150% of existing effluent discharge load, discharged at the eight discharge locations A to H discussed above for the fresh flow conditions of 1995, 1997 and 2001. The changes of effluent load and location result in different concentration of BOD_5 at the WQM sites along the river. Comparing the BOD_5 concentration at the location of discharge and change in BOD_5 concentration at any other location along the river would tell how much BOD_5 is degraded through the assimilation processes between these two points. Tables 3.4 to 3.6 indicate the resulteing TCM of BOD_5 in the three different years.

Table 3.4: Transfer Coefficients Matrix for BOD5 discharge in 1995

Transfer Coefficient Matrix of 1995 (dry year with high discharge)							
Point	Distance	Naburn	Cawood	Selby	Long Drax	Boothferry	Blacktoft
A	61	0.17	0.57	0.39	0.11	0.07	0.02
B	51	0.00	0.53	0.83	0.32	0.21	0.05
C	41	0.00	0.01	0.97	0.53	0.35	0.07
D	35	0.00	0.00	0.62	0.82	0.63	0.08
E	25	0.00	0.00	0.06	0.74	0.89	0.22
F	22	0.00	0.00	0.01	0.46	0.73	0.32
G	13	0.00	0.00	0.00	0.09	0.26	0.62
H	1	0.00	0.00	0.00	0.07	0.07	0.78

Table 3.5: Transfer Coefficients Matrix for BOD5 discharge in 1997

Transfer Coefficient Matrix of 1997 (wet year with high discharge)							
Point	Distance	Naburn	Cawood	Selby	Long Drax	Boothferry	Blacktoft
A	61	0.12	0.59	0.40	0.19	0.13	0.03
B	51	0.00	0.45	0.93	0.45	0.30	0.07
C	41	0.00	0.00	0.96	0.66	0.44	0.11
D	35	0.00	0.00	0.43	0.84	0.59	0.14
E	25	0.00	0.00	0.02	0.72	0.89	0.27
F	22	0.00	0.00	0.00	0.47	0.72	0.37
G	13	0.00	0.00	0.00	0.10	0.30	0.67
H	1	0.00	0.00	0.00	0.00	0.02	0.80

Table 3.6: Transfer Coefficients Matrix for BOD5 discharge in 2001

Transfer Coefficient Matrix of 2001 (wet year with EA's 'future' discharge)							
Point	Distance	Naburn	Cawood	Selby	Long Drax	Boothferry	Blacktoft
A	61	0.12	0.60	0.42	0.18	0.09	0.04
B	51	0.00	0.43	0.84	0.35	0.21	0.07
C	41	0.00	0.01	0.97	0.52	0.31	0.10
D	35	0.00	0.00	0.36	0.77	0.42	0.12
E	25	0.00	0.00	0.00	0.67	0.79	0.25
F	22	0.00	0.00	0.00	0.29	0.53	0.50
G	13	0.00	0.00	0.00	0.00	0.29	0.62
H	1	0.00	0.00	0.00	0.00	0.00	0.81

The column of distance indicated the distance of discharge locations upstream from the Trent Falls. The tables are constructed based on changes in the mean of BOD_5 concentration rather than 5%ile value as for DO. This is because the TCM is expected to provide reference under most of the circumstances, telling the most possible impacts of effluent discharges at one location on the water qualities at the WQM sites, rather than that in extreme conditions.

In the tables above, each value in the cell indicates how much the BOD_5 concentration would change at the water quality WQM sites if there were one unit increase in BOD_5 concentration at the effluent discharge locations. Because of the definition, the value of transfer coefficient ranges from zero to one, where one means the variation of BOD_5 concentration is the same between these two points and zero means the BOD_5 contribution from effluent discharge has no effects on the other point. Tables 3.4 to 3.6 give examples of three different flow and effluent load conditions as stated: 1995 has low flower (in summer) and higher effluent discharges; 1997 has high flow and high effluent discharges while 2001 has high flow and lower effluent discharges. The water quality WQM site at Selby is just 1 km downstream of the discharge location C at Selby therefore it has very high transfer coefficient to the discharge from point C. Point F is also very close to WQM site at Boothferry, only 1 km upstream of the discharge location. The flow-inverse dispersion may not be very strong and the dilution effects from river Aire which joins the tidal Ouse less than 2 km upstream, so the transfer coefficient from point F to Boothferry is only about 0.72 and 0.73 in 1995 and 1997, and as low as 0.53 in the year of 2001. When the transfer coefficient is zero, as some of cells show, means the changes in the BOD_5 concentration at the discharge location has no effect on the concentration at the WQM site. This usually applies to the WQM sites that are further upstream of the discharge locations.

Despite the three years with different combination of flow and effluent discharge, the value of transfer coefficient does not vary very much. Although the assimilative capacity of river water would fluctuate over the years depending on many factors, the relatively stable value of TCM suggests that it is probably dominated by the kinetic of assimilative processes, geographical structure, water surface area, tributary positions and other factors that are generally consistent over time. It is also possible reason that because that the 5%ile data used to derive the TCM are mostly happened in summer therefore the seasonal variation was minimized. Because of the relatively stable values, an averaged TCM of these three matrixes would be more convenient for the EA or other river management authority, or they can also choose the TCM of particular flow and effluent combination depending on the situation at the time.

3.5 Concluding Remarks

The analyses of the simulation results from QUESTS1D river water model provided a useful tool to assess the effectiveness of several alternative water management options aiming at improving the water quality, particularly to tackle the DO sag issue during the summer months between Selby and Long Drax in the tidal Ouse. The industrial plants at Selby discharge substantial effluent loads into the river Ouse, which are thought to be primarily responsible to the deterioration of water quality and appearance of DO sag. The results of simulation proved that the effluent discharge at Selby does relate to the DO sag downstream of its disposal, especially during the year with less flow in summer such as 1995 and 1996. However, as a tidal river system with the largest catchment in England, there are also remarkable contributions to the water quality issue from resuspended sediments and its landward transport with tide, as well as the inputs from STWs and diffuse pollution draining from agricultural farms and highly populous areas. The effluent discharges from the industries does not account for all the pollution in the tidal Ouse. The sediments move up the river system during the low flow period and remain around Selby long enough to cause the observed DO sag in summer (Cashman *et al.* 1999), the effluent discharge from Selby exacerbates the situation when the flow is low but should not be regarded as the only reason. Therefore, reduction of the effluent discharges in the Selby plants may not always be an effective option to tackle the DO sag problem, though it is regarded as the only option by the EA under most circumstances.

The choice of locations for effluent discharges from the Selby industries could dramatically change its effects on the DO saturation in tidal Ouse. Moving the discharge location downstream along the river would monotonically increase the water quality over the whole length of tidal Ouse because the water quality downstream of Trent fall is dominated by the water from the river Trent. Its effectiveness applies in both dry and wet years. Shifting the effluent discharges from summer into winter also leads to significant improvement on the DO sag issue, since the worst DO saturation usually happens in warm and less-flowed summer. In most situations, 25% annual effluent storage would be enough to meet the EA's target of 30% minimum DO saturation at 5%ile, except for the very dry year of 1996. The water abstraction return at river Ouse and Derwent has only marginal effects on the DO saturation, too little to tackle the issue of severe DO sag in the summer months alone, particularly during the low flow years.

Discharge on the ebb only is not able to improve the water quality in either dry or wet years, due to the long clearance time for river flow in the tidal Ouse and the landward invasion of tide. There is also water abstraction and effluent input from Drax Station. It abstracts the river water at the rate 2 m^3/s for cooling process and returns half of the volume back to the river at the same position, with 0.75°C elevated. Cashman *et al.*(1999) revealed that the temperature difference between effluent and river water has negligible impact on the water quality. The results of simulation from QUESTS1D in this research show no impact due to the loss of water, only marginal influence to the water qualities in the dry summers in 1995 and 1996.

Therefore, the changes in effluent discharge loads, discharge locations and discharge timing over the year deserve more consideration as effective management options to improve the water quality and tackle the DO sag issue. They are not, however, exclusive to each other, therefore the best option of river water management could be a combination of them all or variations on them. The combination of the analyses for two of them also produces TCM as important reference to the river management. The TCM would be a quite useful tool to the river policy design that will be discussed in the later chapters.

Chapter 4 Economics of River Policy

4.1 Introduction

4.1.1 Review of regulative system of river policy in the tidal Ouse

Although the river quality has been improved significantly in the Ouse system over these years, it still suffers from the DO sag in the summer months, especially downstream of Selby industrial effluent discharges. As a result DO levels in some parts of the river and at some times of the year are too low to support salmon, which is regarded as a key indicator of the river's ecological health. The decline of salmon stock in the Ouse system is due to a number of sources, which may include over-fishing around Greenland, commercial netting in estuaries, habitat loss, increasing sediment load and river morphology changes, etc. But among those, pollution discharge and water abstractions have significant influences on dissolved oxygen. In the Ouse system, the most serious oxygen sag happens downstream of Selby, usually between the water sampling points at Selby and Long Drax. The EA is considering to improve the river water quality by tightening the discharge consents in Selby. A new regulation system of pollution control is being implemented in order to restore water quality in the Ouse system, which is driven by the EU Directive on Integrated Pollution Prevention and Control (IPPC).

"The basic purpose of the IPPC regime is to introduce a more integrated approach to controlling pollution from industrial sources. It aims to achieve a high level of protection of the environment taken as a whole by, in particular, preventing or, where that is not practicable, reducing emissions into the air, water and land. The main way of doing that is by determining and enforcing permit conditions based on Best Available Techniques (BAT)." (Defra 2002b). The essence of IPPC is that operators should choose the best option available to achieve an agreed level of protection of the environment taken as a whole. The BAT approach is typically modified by the qualification that the cost of applying techniques should not be excessive in relation to the environmental protection they provide. The environmental benefits of the IPPC target stem from the reduction in effluent emissions of BOD into River Ouse. The Environment Agency aims to improve the DO levels in River Ouse through the IPPC, not only for the return of salmon but also for assuring river water quality for various purposes including recreation, angling, agriculture, industrial abstraction and amenity value etc. It tries to achieve this purpose by tightening the consents on discharge from industrial sources in Selby area and employing the IPPC Scheme. However, the IPPC Scheme requires BAT to be applied in the abatement of pollution while there is no one specific definition of BAT provided. The BAT is varying among each plant depending on its cost and benefit conditions. As the prerequisite of BAT is that the application of abatement technique without incurring excessive cost, an alternative way of addressing the issue of cost is to identify the most cost effective method for achieving a given target of water quality, which is applicable to both pollution abatement techniques and pollution control policies.

The current regulation system controlling effluent discharge and water abstraction in Tidal Ouse and Humber estuaries consist of two different policies implemented by the EA, the discharge consents for effluent discharges and the system of tradable Water Abstraction Licenses for water abstractions respectively. The consents for effluent discharges are usually fixed amounts over the year, although some of them allow certain extent of violation over the year such as the BOD discharge consent for Tate & Lyle Citric Acid (TLCA) in Selby and big Sewage Treatment Works (STW). An abstraction licence generally states how much water could be taken, from where, the way to be used and where to return water to river. Since a recent amendment, the Tradable Water Abstraction License became time limited and can only be renewed upon new application. However, the amount of water abstraction granted by license is given on the annual base that allows the abstractor to take water from river any time of the year, no matter what the river condition is.

4.1.2 The Structure of the Chapter

This chapter consists of four sections as follows. The second section discusses the static analysis of environment policy to control pollution in the tidal Ouse. A tax-subsidy scheme (TSS) and tradable pollution permits (TPP) system are compared with a direct command approach using a static equilibrium analysis to evaluate the ability of each policy instrument in achieving the least cost solution. The difficulties might be encountered in design and practice are also discussed. The third section introduces a dynamic analysis of the same policy instruments. Here we discuss differences between the three policy instruments in their ability to achieve the dynamic equilibrium in a dynamic system. The differences are illustrated. The convergence and stability properties of the steady state equilibrium, determined by the capital stocks and investment choices of the firm are analysed. Analysis shows that the steady state equilibrium is a saddle point, and thus only one trajectory of the system will eventually converge to the steady state. Comparative statics was carried out in the fourth section to identify the impacts of environmental policy instruments in the dynamic system. Policy adjustments are necessary to achieve the prescribed environmental target. The last section provides a summary of the results from the preceding analyses and discusses the potential implications of the research for environmental policy and river water management.

4.2 Static Analysis of Environmental Policy

According to economic theory, environmental policy makers can correct for market failure in environmental issues, in a full information competitive context, by using various environmental policy instruments which internalise external social damages. Environmental policy instruments can be divided into two categories: economic instruments, which are also known as market-based instruments, and direct regulation, also known as "Command and Control" (Xepapadeas 1997). However, the prerequisites for the efficient environmental policy are not easily achieved. Information on production and emissions is usually incomplete, while industrial managers are generally not willing to share this information with regulatory authorities. The prerequisite for fully competitive markets presents another difficulty in reality.

As stated above, water policies in England and Wales dealing with effluent discharge and water abstraction can be categorized into the two instrument types. Discharge consents involve direct regulation only, whilst Water Abstraction Licenses combine regulative instruments with at least some potential for license trading. Water abstraction carries similar consequences on river water quality as emissions due to the impact of water volume on the quality and assimilative capacity of river water. Currently, neither abstraction licenses nor discharge consents take the timing of river flow into account. Since the river flow has an impact on assimilative capacity and consequently on river water quality, it might be necessary to consider the effect of time-varying consents and licenses to cope with changes in the volumes, and velocities of river flows, water temperatures, tidal influences, geographic factors and so on.

4.2.1 The general model of cost efficiency of pollution abatement

Consider a typical pollution externality produced by several plants in a market. The plants are competitive with each other. The plants produce a homogenous output q_i and during production generate emissions e_i to the whole market. With exogenously determined prices for the industry's output and for the inputs to pollution abatement, the firm's profit and emission functions can be defined as below:

$$P_i(q_i, a_i) = pq_i - C_i(q_i, a_i) - \mathrm{T}(e_i) \qquad \ldots(4.1),$$

$$e_i = Z_i(q_i, a_i) \qquad \ldots(4.2),$$

where P_i and C_i are the net benefit (profit) and production cost of the firm, q_i is product output from site i facing an exogenous price p, a_i denotes the level of abatement activity at site i and $\mathrm{T}(e_i)$ reflects private emission-related costs incurred at site i. Emission e_i is a function represented by Z_i of output production and the level of abatement. The emission-related costs are usually attributed to the existence of environmental policy (Xepapadeas 1997). The environmental policy instruments implemented by an economically rational environmental authority, either regulatory approaches or economic instruments, would seek to achieve the level of environmental quality where the marginal cost of damage from pollution equalled to the marginal cost of pollution abatement. At this point, social welfare would be maximized. Private plants however, would minimize their costs of pollution abatement, because they seek to maximize profit, by equating the private marginal cost of abatement and the marginal benefit of abatement. Thus, in order to achieve the social optimum, full information about the cost and emission functions of plants is essential.

4.2.2 Cost effectiveness analysis with water abstraction

4.2.2.a Ambient water quality with effects of water abstraction

Although regulation of effluent discharges can reduce the pollution in a receiving water body, ambient water quality in the river is the subject of regulatory concern here. So the effects of effluent on ambient water quality should be evaluated in order to achieve the desired water quality level at specific WQM sites. For the tidal Ouse estuary, there are five WQM sites downstream from Naburn Weir to Boothferry Bridge before its confluence with the tidal Trent at Trent Falls.

Since discharges from industry and abstraction by water companies both have adverse impacts on river water quality, an integrated river management strategy for both consents and water abstraction would appear to be more appropriate than dealing with these two issues separately. Water quality in a river such as the tidal Ouse is influenced by several factors including the tributaries water qualities, industrial emissions, water abstractions by water companies and also the indigenous river properties such as volume, velocity and micro plankton activities.

Suppose that water quality at any WQM site s in the tidal estuary can be described by a function which takes the form $Q_s = f_s(A_s, E_s, H_s, \varepsilon_s) + \gamma_s$, and the ambient water quality target at site s is $\overline{Q}_s$, then the water quality at the WQM sites must satisfy $Q_s \geq \overline{Q}_s, \forall s$. The variables in the water quality function are given below.

A_s is ambient water quality including the inputs from other tributaries at WQM site s;

E_s is aggregate impacts of industrial emissions to WQM site s;

H_s is aggregate impacts of water abstractions to WQM site s;

ε_s is a vector of other environmental factors that will influence the water quality, including velocity, volume, river flow and tide etc. There might be various influencing variables in different water quality models;

and γ_s are the variations which are not captured explicitly by this function.

When the locations of polluters and abstractors matter, it is not appropriate to simply sum up emissions and abstractions from all the sources. Instead, transfer coefficients are applied to evaluate and aggregate their impacts on the water quality at various water quality sites from various sources. For simplicity, it is common to assume that the sources contribute linearly to the aggregate emissions or abstractions on the water quality at WQM site s (Zylicz 2003). Thus we have

$$E_s = b_{1s}e_1 + b_{2s}e_2 + \cdots\cdots + b_{ks}e_k = \sum_{i=1}^{k} e_{is} = \sum_{i=1}^{k} b_{is}e_i \qquad \ldots(4.3),$$

where b_{is} is the transfer coefficient of impact of one unit pollution discharge (in the case of this research is BOD_5) from the pollution at site i on the water quality at WQM site s, and e_i is the pollution discharged at site i. Similarly for water abstraction, there is

$$H_s = d_{1s}\beta_1 + d_{2s}\beta_2 + \cdots\cdots + d_{ks}\beta_k = \sum_{i=1}^{k} \beta_{is} = \sum_{i=1}^{k} d_{is}\beta_i \qquad \ldots(4.4),$$

where d_{is} is the transfer coefficient of impact of one unit water abstraction from the abstraction site i on the river water quality at WQM site s, and β_i is the amount of water abstracted at site i.

4.2.2.b Cost effectiveness of environmental policy

Suppose the cost function of each firm at site i, either an industrial plant or a water company, takes the form of $C_i(q_i, a_i, \beta_i)$ where q_i and a_i are the industrial output and abatement level at site i respectively, and β_i is the amount of water abstraction at site i. It is assumed that given any combination of q_i and a_i, the industrial effluent discharged to the river from site i, e_i can be determined. Currently there is no plant in the tidal Ouse catchment discharging effluent to river and abstracting substantial amount of water at the same time, while some are using ground water for their production process. But in order to provide a perspective on future development, this economic model allows a firm at site i to have e_i and β_i at the same time. For a pure effluent discharger, $\beta_i = 0$ and $e_i = 0$ for a pure water abstractor.

The river quality management objective of the regulator will be achieved by a cost allocation of effluent abatement and water abstraction among the different plants in the catchment, which is given by the solution of the following constrained optimisation problem:

Minimize $\sum_i C_i(q_i, a_i, \beta_i)$

Subject to: $Q_s = f_s(A_s, E_s, H_s, \varepsilon_s) + \gamma_s \geq \overline{Q}_s$, for all the $s = 1,2\ldots r$ ...(4.5)

The aggregate cost burden on the plants is minimised, subject to achieving the water quality target at each WQM sites. The Lagrange Equation is:

$$L = \sum_i C_i(q_i, a_i, \beta_i) + \sum_s \lambda_s \cdot (\overline{Q}_s - f_s(A_s, E_s, H_s, \varepsilon_s) - \gamma_s) \qquad \ldots(4.6)$$

λ_s here is the shadow price (Lagrange Multiplier) for water quality. It forms an essential part of the optimal solution and reports the marginal impact of water quality constraint at binding point. Under the assumption of convexity of the cost functions, the Kuhn-Tucker conditions to this optimisation problem imply (Simon and Blume 1994):

$$\frac{\partial C_i(\cdot)}{\partial q_i} - \sum_s \lambda_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s} \cdot \frac{\partial e_i}{\partial q_i} \geq 0; \qquad \ldots(4.7)$$

$$\left[\frac{\partial C_i(\cdot)}{\partial q_i} - \sum_s \lambda_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s} \cdot \frac{\partial e_i}{\partial q_i}\right] \cdot q_i = 0, \forall i; \qquad \ldots(4.8)$$

$$\frac{\partial C_i(\cdot)}{\partial a_i} - \sum_s \lambda_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s} \cdot \frac{\partial e_i}{\partial a_i} \geq 0; \qquad \ldots(4.9)$$

$$\left[\frac{\partial C_i(\cdot)}{\partial a_i} - \sum_s \lambda_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s} \cdot \frac{\partial e_i}{\partial a_i}\right] \cdot a_i = 0, \forall i; \qquad \ldots(4.10)$$

$$\frac{\partial C_i(\cdot)}{\partial \beta_i} - \sum_s \lambda_s d_{is} \frac{\partial f_s(\cdot)}{\partial H_s} \geq 0; \qquad \ldots(4.11)$$

$$\left[\frac{\partial C_i(\cdot)}{\partial \beta_i} - \sum_s \lambda_s d_{is} \frac{\partial f_s(\cdot)}{\partial H_s}\right] \cdot \beta_i = 0, \forall i; \qquad \ldots(4.12)$$

$$\overline{Q}_s - f_s(A_s, E_s, H_s, \varepsilon_s) - \gamma_s \leq 0; \qquad \ldots(4.13)$$

$$\left(\overline{Q}_s - f_s(A_s, E_s, H_s, \varepsilon_s) - \gamma_s\right) \cdot \lambda_s = 0, \forall s; \qquad \ldots(4.14)$$

$$q_i, a_i, \beta_i, \lambda_s \geq 0, \forall i, \forall s.$$

From equations (4.8), (4.10), (4.12) and (4.14), at least one of the two factors in each of the products has to be zero. Recalling the economic meaning of $q_i, a_i, \beta_i, \lambda_s$, they are all positive unless the plant is shut down or water qualities at some WQM sites are of no interest to the regulator. Thus, assuming the water quality constraint is binding at each WQM site and no plant shut down, $q_i, a_i, \beta_i, \lambda_s > 0$, these first order conditions (FOCs) for the optimal solution become:

$$\frac{\partial C_i(\cdot)}{\partial q_i^*} = \sum_s \lambda_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s} \cdot \frac{\partial e_i}{\partial q_i^*} \qquad \ldots(4.15);$$

$$\frac{\partial C_i(\cdot)}{\partial a_i^*} = \sum_s \lambda_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s} \cdot \frac{\partial e_i}{\partial a_i^*} \qquad \ldots(4.16);$$

$$\frac{\partial C_i(\cdot)}{\partial \beta_i^*} = \sum_s \lambda_s d_{is} \frac{\partial f_s(\cdot)}{\partial H_s} \qquad \ldots(4.17);$$

$$f_s(A_s, E_s, H_s, \varepsilon_s) + \gamma_s = \overline{Q}_s \qquad \ldots(4.18).$$

The function $\sum_i C_i(q_i^*, a_i^*, \beta_i^*)$ thus represents the minimised total abatement costs under the optimal combination of output level, effluent abatement and water abstraction from each plant and the optimal allocation of those factors among the plants. Equation (4.18) indicates a binding condition regarding the water quality target.

The economic implications of above equations could be explained as, for example of equation (4.17), the cost effective allocation of water abstractions is where the marginal abstraction cost $\frac{\partial C_i(\cdot)}{\partial \beta_i^*}$ from a particular source is equal to the sum of "impacts" of that abstraction on all WQM sites, which is measured by a linear combination of the products of the Lagrange multipliers λ_s and marginal effect of abstraction from site i on water quality at each WQM site, weighted by transfer coefficient d_{is}.

In a large catchment, when cost effectiveness could be achieved and the water quality target is binding at each WQM site simultaneously, it could be shown that the following condition would be satisfied, $\lambda_i = \lambda_j = \cdots = \lambda_s = \lambda$ for all the WQM sites in the catchment, if the water quality constraint at each WQM site is independent of each other. The reason for this equation is that: at the cost effective allocation, if there were any two WQM sites whose water quality targets were achieved at different marginal cost, unless the basic requirements of river water quality are violated, the total abatement costs $\sum_i C_i(q_i^*, a_i^*, \beta_i^*)$ could always be reduced further by increasing water quality at the cheaper site and decreasing it at the more expensive site. However, in a river system, the water qualities at each WQM site are usually closely related, as will be shown in the detailed case study of the tidal Ouse in the later chapters.

Thus the equations (4.15)-(4.17) imply that

$$\frac{\partial C_i(\cdot)/\partial q_i^*}{\sum_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s} \cdot \frac{\partial e_i}{\partial q_i^*}} = \frac{\partial C_i(\cdot)/\partial a_i^*}{\sum_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s} \cdot \frac{\partial e_i}{\partial a_i^*}} = \frac{\partial C_i(\cdot)/\partial \beta_i^*}{\sum_s d_{is} \frac{\partial f_s(\cdot)}{\partial H_s}} = \lambda \qquad \text{...(4.19).}$$

Equation (4.19) has clear economic meaning: the optimal cost efficient allocation of effluent abatement and water abstraction across the whole catchment requires the ratio between the marginal private cost of output level, effluent abatement and water abstraction at site i ($\partial C_i(\cdot)/\partial q_i^*$, $\partial C_i(\cdot)/\partial a_i^*$ *and* $\partial C_i(\cdot)/\partial \beta_i^*$,and the marginal impacts of each activity at site i on the river water qualities at all the WQM sites (the three denominators of Eq. 4.19) , is the same for each choice variable. The ratio is equal to the marginal abatement cost, or shadow price, of the ambient water quality at the level $\overline{Q}_s$ at WQM site s.

However, in order to determine the allocation of effluent emissions and water abstraction among the plants, the marginal cost of ambient water quality, or shadow price of ambient water quality, must be appropriately determined in order to link the marginal damage cost of pollution on the community (which is equivalent to the marginal benefit to the community from pollution abatement), and the marginal effect from either effluent emissions and water abstractions on water quality at the WQM sites. Because of the controversies surrounding the accuracy and eligibility of environmental benefit valuation, it is very difficult to establish a shadow price to equate the marginal cost of pollution abatement to the marginal damage cause by pollution, or to the marginal benefit for the community derived from pollution abatement.

Apart from this, it is also easy to prove that economic analysis is usually not the main driver behind current environment policy in reality. Economic considerations do contribute to the environmental policy making process. But other influential factors such as political acceptability, legitimatisation process, equity, social preference and international obligations also have their say in the process. These factors will be discussed in later chapters.

As a second best position in the absence of the elusive shadow price of water quality, for any prescribed environmental target, cost effectiveness is achieved if the environmental target is met at least cost to society. By switching the focus from cost efficiency to cost effectiveness, value of shadow price is no longer required for the optimal allocation of emissions and abstractions. However, equation (4.15), (4.16), (4.17) and (4.18) will still apply for cost effectiveness. In fact, *any* constant value of λ will be sufficient to achieve cost effectiveness in the allocation of emission and abstraction among the plants. This is represented by the equation below, where ψ could be any constant value without violating the environmental standards.

$$\frac{\partial C_i(\cdot)/\partial q_i^*}{\sum_s b_{is}\frac{\partial f_s(\cdot)}{\partial E_s}\cdot\frac{\partial e_i}{\partial q_i^*}}=\frac{\partial C_i(\cdot)/\partial a_i^*}{\sum_s b_{is}\frac{\partial f_s(\cdot)}{\partial E_s}\cdot\frac{\partial e_i}{\partial a_i^*}}=\frac{\partial C_i(\cdot)/\partial \beta_i^*}{\sum_s d_{is}\frac{\partial f_s(\cdot)}{\partial H_s}}=\psi \qquad \ldots(4.20)$$

4.2.3 Policy instruments

According to the analysis above, the cost efficient or cost effective [1] equilibrium allocation of effluent abatement and water abstraction under a regulation scheme for river water quality will satisfy the constraints below:

$$\begin{cases}\frac{\partial C_i(\cdot)}{\partial q_i^*}=\lambda\cdot\sum_s b_{is}\frac{\partial f_s(\cdot)}{\partial E_s}\cdot\frac{\partial e_i}{\partial q_i^*} & \cdots\cdots(4.21)\\ \frac{\partial C_i(\cdot)}{\partial a_i^*}=\lambda\cdot\sum_s b_{is}\frac{\partial f_s(\cdot)}{\partial E_s}\cdot\frac{\partial e_i}{\partial a_i^*} & \cdots\cdots(4.22)\\ \frac{\partial C_i(\cdot)}{\partial \beta_i^*}=\lambda\cdot\sum_s b_{is}\frac{\partial f_s(\cdot)}{\partial H_s} & \cdots\cdots\cdots\cdots(4.23)\\ f_s(A_s,E_s,H_s,\varepsilon_s)+\gamma_s=\overline{Q}_s & \cdots\cdots(4.18)\end{cases}$$

for all the i and all the site s that is water quality is binding .

Several policy instruments are available to regulators in order to achieve the optimum level of pollution reduction. Discharge consents directly regulated by the EA as a Command and Control (CAC) approach, whilst market-based instruments (MBIs) include the pollution tax scheme and the tradable pollution permit (TPP) system. The MBI options are usually thought superior to the CAC approach because they are cost effective *per se*, and could achieve cost effectiveness for *any* level of pollution reduction, even not at the cost efficient level. However, the choice among policy instruments is not straight forward between the tax scheme and TPP system, nor is it between CAC and MBI approaches. Details of instrument choices will be discussed in a later section.

[1] When λ takes the value of the shadow price of ambient water quality, this becomes a cost efficient allocation.

4.2.3.a CAC Approach

The CAC takes the form of statutory standards or consents that pollution dischargers must not violate, otherwise heavy penalties, fines and other forms will be applied. There are several forms of this instrument. Two different consents focusing on different targets are discussed below:

1. Discharge Consents

Discharge consents for industrial effluent, such as the effluent discharges from Selby sources into the Ouse, specify the emission limit for the plant. This is similar to water abstraction as the regulator specifies the amount of water that can be abstracted by each water company. Supposing $\bar{e}_i$ and $\bar{\beta}_i$ are the maximum emission and water abstraction allowed at site i respectively, the problem faced by a cost minimising plant is then to:

Minimize $C_i(e_i, \beta_i)$,

subject to $e_i \le \bar{e}_i, \beta_i \le \bar{\beta}_i$ where $e_i = Z_i(q_i, \alpha_i)$.

The Lagrange function is $L = C_i(e_i, \alpha_i) + \lambda_{ie}(\bar{e}_i - e_i) + \lambda_{ia}(\bar{\beta}_i - \beta_i)$, where λ_{ie} and λ_{ia} can be interpreted as the shadow price of the emission and abstraction limits respectively. These shadow prices indicate the impacts on the abatement cost of changes in the stringency of the emission and abstraction standard.

Solving the maximization problem by finding the Kuhn-Tucker conditions, it is not difficult to show that the optimal solution for the firm is to discharge and abstract up to the allowed limits in a wide range of cases. So the limits chosen by the policy regulator, the Environment Agency in the case of the River Ouse, are particularly influential over the river quality.

In general, effluent consents set by regulators are not cost effective because they tend to be applied uniformly across polluters. Cost effectiveness in pollution control requires the polluter with lower costs of pollution control to abate more, and those with higher costs to abate less. Therefore, uniform effluent consents across polluters and over time are unlikely to be cost effective unless they are set at the exact level where the marginal cost of pollution abatement is equal to the marginal benefit of pollution control.

If the effluent consents are set higher than the optimal level, i.e. when the polluters have to abate more than the socially optimal level, the polluters with higher costs of abatement have to reduce their pollution more, to the level where marginal cost of abatement exceeds the marginal benefit from pollution control. This situation is not cost effective because social welfare is decreased by the extra cost of pollution abatement. If the effluent consents are set too low, however, the polluters with lower abatement costs will have no incentive to abate further to the optimal level as that would incur extra costs of pollution abatement.

In this thesis, giving the consideration to changes in both location and timing of discharge will lead to the conclusion that cost effectiveness requires polluters to abate more where and when assimilative capacity of the river is lower, and to abate less where and when assimilative capacity is higher. Since the marginal benefits from emission reduction are actually the marginal damages avoided by pollution elimination, they vary with changes in the assimilative capacity in the river. If consents are insensitive to variations in assimilative capacity in the river, the marginal social benefits of emission reductions could not be equalised. However, to have the discharge consents varying with assimilative capacity are simply impractical in reality.

2. Ambient Quality Consents

As alternative, the environmental authority could design the direct control regulations within an ambient water quality system which aims to achieve water quality that matches the required target, rather than focusing directly on the effluent emissions and water abstractions of polluters. In an ambient water quality system, E_s and H_s, as discussed before, are the aggregate effects of emissions and abstractions on the water quality at the WQM site s from all the plants along the river, which are defined by equation (4.3) and (4.4). Therefore the problem of cost minimisation is:

Minimize $C_i(e_i, \beta_i)$,

subject to $Q_s = f_s(A_s, E_s, H_s, \varepsilon_s) + \gamma_s \geq \overline{Q}_s$ for all the $s = 1,2\ldots r$.

$\overline{Q}_s$ is the regulated ambient water quality level required by the EA.

The Kuhn-Tucker conditions defining the minimum cost are:

$$\frac{\partial C_i(\cdot)}{\partial e_i^*} - \lambda^* \cdot \sum_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s} \geq 0, \text{ with equality if } e_i^* > 0 \qquad \ldots(4.24);$$

$$\frac{\partial C_i(\cdot)}{\partial \beta_i^*} - \lambda^* \cdot \sum_s d_{is} \frac{\partial f_s(\cdot)}{\partial H_s} \geq 0, \text{with equality if } \beta_i^* > 0 \qquad \ldots(4.25);$$

$$\overline{Q}_s - f_s(A_s, E_s, H_s, \varepsilon_s) - \gamma_s \geq 0, \text{ with equality if } \lambda^* > 0 \qquad \ldots(4.26)$$

for all the i and s.

Since e_i^*, β_i^* and λ^* are positive values in most situations, if λ^* is assigned the value of shadow price, at which the value of $\lambda^* \cdot \sum_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s}$ and $\lambda^* \cdot \sum_s b_{is} \frac{\partial f_s(\cdot)}{\partial H_s}$ are equal to the marginal damage to the community from pollution, this environmental consents of effluent emissions and water abstractions would be cost efficient. However, as we discussed before, it is very difficult to determine the proper value of λ^*, so is to estimate the marginal damage from pollution deterioration.

Without knowing the value of λ^*, cost effectiveness could still be achieved as long as $\dfrac{\partial C_i(\cdot)/\partial e_i^*}{\sum_s b_{is} \frac{\partial f_s(\cdot)}{\partial E_s}} = \dfrac{\partial C_i(\cdot)/\partial \beta_i^*}{\sum_s d_{is} \frac{\partial f_s(\cdot)}{\partial H_s}} = \psi$. When this is the case, regulation in the effluent emissions and water abstractions would be cost effective, provided that the desired environment quality was not violated. Only when the constant value is equal to the true shadow price of environment quality, regulations will be cost efficient.

If the marginal effect of aggregate effluents and abstractions on water quality, $\frac{\partial f_s(\cdot)}{\partial E_s}$ and $\frac{\partial f_s(\cdot)}{\partial H_s}$, could be assumed fixed at each WQM site, the environment agency would then be able to determine the cost-effective consents for effluent emission and water abstraction for each plant at any site i when the plant's cost function and their transfer coefficients were known. However, the marginal effect of aggregate effluents and abstractions on water quality at the WQM sites is subject to seasonal variation associated with changes in assimilative ability in the river water. Also the information asymmetry between the plants and environmental authority presents problem for the design of cost effective consents. The cost function of polluters is usually unknown to the regulator and the polluters are reluctant to share this information. When there are many polluters involved, finding out the individual cost function of each polluter and designing individual consents for each of them would require an impractical level of effort by the environmental authority. Therefore direct controls which aim to achieve cost effectiveness in the river management are not usually feasible in practice. The onerous calculation involved and the requirement of iterative amendment due to seasonal change, new technology and new entry are unacceptable the unacceptability both to environmental authority and political process.

4.2.3.b Pollution tax-subsidy scheme (TSS)

An emission charge or pollution tax is a fee, collected by the government, levied on each unit of pollutant emitted into the air or water (Tietenberg 2006). Pollution taxes induce plants to reduce their pollution because of the substantial costs of the pollution emission. Assuming they are profit seeking, they will reduce their pollution to the point where the incremental cost of control is equal to the emission charge that they must otherwise pay (Hanley *et al.* 1997). An effective pollution tax will be set such that the emission reduction is what is desired by the regulator. When the emission charge is set at the level of marginal social damage of pollution emissions, emissions will be reduced to the level where the marginal damage of pollution to society is equal to the marginal abatement cost, provided there is a convex abatement cost function. Such a pollution tax scheme would induce the optimal level of pollution abatement with cost efficiency, and is usually called a "Pigouvian Tax" (Mankiw 2001; Tietenberg 2006).

A successful pollution tax on the effluent discharges and water abstractions should motivate industries to mitigate the impacts of their emission and abstraction on the river water quality. However, here I will analyse a bit different pollution TSS. Pezzy (1992) suggested a combination system of charges and subsidies could be applied as an integrated scheme, mentioned that pure pollution taxes usually encounter objections because of their political unacceptability and the financial burden imposed on industry. In this scheme, polluters pay the amount of $T_e = t_e \cdot (e_i - e_i^0)$, where t_e is the tax rate set by regulator; e_i^0 is a targeted emission right which is initially granted as a property right to each existing firm for discharge at site i. The value of e_i^0 may vary from firm to firm, but not over time.

Similarly, when considering water abstraction, each water company would be required to pay the amount $T_a = t_a \cdot (\beta_i - \beta_i^0)$, where t_a and β_i^0 are the counterparts of the TSS for water abstractions.

If the firm has less than its targeted level effluent or abstraction ($e_i < e_i^0$ or $\beta_i < \beta_i^0$), it will receive corresponding *subsidy* from the authority. When e_i^0 and β_i^0 are set to zero, this scheme reduces to a pure Pigouvian pollution charges system. Since the TSS is a form of MBIs it is inherently cost effective. As usual, different tax rates will produce different cost-effective allocations of effluent discharge and water abstraction, although only one will truly be cost efficient. However, e_i^0, t_e and β_i^0, t_a are not necessarily set by the regulator at efficient levels, and this can be influenced by many factors representing their own interests. Therefore this scheme may not be revenue-neutral for the regulator (Pezzy 1992).

The regulator's problem is now to set an appropriate tax system through which the plant will bear same marginal cost of effluent and abstraction control, which is equal to the marginal damage cost of pollution. However, in contrast to what discussed above, a set of tax and subsidy rates different at each WQM sites are set, because of the different transfer coefficients for emission and abstractions from various sources.

Let T_{ie} and T_{ia} be the sum of tax and subsidy for effluent emission and water abstraction at all the WQM sites. Therefore the cost function at each cost minimising plant is to:

Minimizing $C_i(e_i, \beta_i) + T_{ie} + T_{ia}$,

where $T_{ie} = \sum_s \left(e_i - e_i^0\right) \cdot b_{is} \cdot t_{es}$ and $T_{ia} = \sum_s \left(\beta_i - \beta_i^0\right) \cdot d_{is} \cdot t_{as}$.

The FOCs are

$$\frac{\partial C_i(\cdot)}{\partial e_i^*} + \sum_s b_{is} \cdot t_{es} = 0 \qquad \ldots(4.27),$$

$$\frac{\partial C_i(\cdot)}{\partial \beta_i^*} + \sum_s d_{is} t_{as} = 0 \qquad \ldots(4.28).$$

Comparing Eqs (4.27) and (4.28) with Eqs (4.24) and (4.25), it can be seen that $t_{es} = -\lambda \cdot \frac{\partial f_s(\cdot)}{\partial E_s}$ and $t_{as} = -\lambda \cdot \frac{\partial f_s(\cdot)}{\partial H_s}$, where the negative sign means that the direction of tax and subsidy revenue is opposite to the effect of emissions and abstractions on the water quality. Under the TSS scheme, the resulting river water quality at all the WQM sites will be same as the river water quality achieved with cost efficiency in section 4.2.3.a, therefore the tax and subsidy scheme would be cost efficient. This is categorized by Pezzy (1992) as short-run efficiency. Taking opportunity cost into account, long-run efficiency could also be achieved under proper entry-exit rules for the industry (Pezzy 1992). It should be noted that this TSS takes into account not only the effects of discharge and abstraction locations, but also the effects of discharge and abstraction *timings*. The effects of timing changes can be manifested through the factors of $\frac{\partial f_s(\cdot)}{\partial E_s}$ and $\frac{\partial f_s(\cdot)}{\partial H_s}$. The charge rates should also be allowed to vary over time accordingly.

However, the difficulty in estimating the value of λ also applies to the tax and subsidy scheme. Without knowing the value of λ, it is not possible to achieve the cost efficiency though the tax and subsidy scheme. But assume that λ takes the same value at all the WQM sites, and the environment quality is not violated, the tax and subsidy scheme would retain its cost effectiveness. The tax and subsidy rates, as discussed previously, will be influenced by many other factors to reflect different preferences or interests. At the equilibrium, the tax and subsidy rate will equate the marginal cost of emission abatement and abstraction reduction among all the polluters at each WQM site, but these costs are not necessarily equal to the marginal damage from pollution deterioration to the community.

4.2.3.c Tradable Water Abstraction Licenses and TPP Scheme

Tradable emission permits represent a system of tradable property rights for the management of environmental pollution. Originated by Crocker (1966) and Dales (1968), they have gained much popularity recently in environmental economics. An ideal tradable emission permits system involves:

- A decision regarding the total quantity of pollution is to be allowed. If an efficient system is to be attained, the total quantity of emission permits issued (measured in units of pollution) should be equal to the efficient level of pollution.
- A rule that ensures that any firm is only allowed to produce pollution up to the quantity of emission permits it possesses. Any emission beyond that level is subjected to a prohibitively expensive fine or other penalty.
- A choice by the control authority over how the total quantity of emission permits is to be initially allocated.
- A guarantee that emission permits can be freely traded between plants at whichever prices are agreed for that trade.

Different versions of tradable pollution permit systems have been proposed and established worldwide. Three of them are described here: the ambient permit system (APS), the emission permit system (EPS) and the pollution offset (PO).

1. Ambient Permit system (APS)

In an APS, the environmental authority determines the amount of permits issued to the polluters based on the effects of their emission on ambient water quality at the WQM sites. In order to take into account the differences of spatial characteristics among polluters, transfer coefficients are utilized to facilitate permit trading at each WQM site. The trade of permits at a WQM site is not to be carried out on a one-for-one basis, but a rate relevant to the ratio of the polluters' transfer coefficients at the WQM site. Thus a separate permit market needs to be established at each WQM site, and the polluters are required to produce a "portfolio" of pollution permits for all the WQM sites they affect, depending on the transfer coefficients between sources and the WQM sites. In an APS, the environmental authority is responsible for specifying the transfer coefficients matrix for all the plants at each of the WQM sites through which the trading ratio among any plants at any WQM site is established and adhered to during the trade.

Supposing $\sum_i \sum_s e_{is}^0$ is the total quantity of emission permits issued by the environmental authority and e_{is}^0 is the initial permits initially allocated to site i through either auction or "grandfathering" at WQM site s. No TTP system is in place for the tidal Ouse and Humber catchment. However, tradable license of abstraction are implemented in water abstraction management. Following the notation for pollution permits, let $\sum_i \sum_s \beta_{is}^0$ and β_{is}^0 denote the initial total quantity of abstraction permits and the permits for water abstraction for company i at site s in the catchment, then net demands for emission and abstraction permits at site i are $\sum_s (e_{is} - e_{is}^0)$ and $\sum_s (\beta_{is} - \beta_{is}^0)$.

Since plants are allowed to trade their permits in markets, equilibrium permit prices for effluent discharge and water abstraction will be established at each WQM sites. The cost minimizing plant will then face the problem:

Minimize $C_i(e_i, \beta_i) + \sum_s [P_{es} \cdot (e_{is} - e_{is}^0) + P_{as} \cdot (\beta_{is} - \beta_{is}^0)]$, where P_{es} and P_{as} are the equilibrium permit prices for effluent discharge and water abstraction respectively in the permit market established at each WQM site s. These prices may vary from site to site and over time.

Knowing that $e_{is} = b_{is} e_i$ and $\beta_{is} = d_{is} \beta_i$, and assuming that neither of them is zero or negative, the FOCs of the cost minimising solution for effluent discharges and water abstractions are:

$$\frac{\partial C_i(\cdot)}{\partial e_i^*} + \sum_s b_{is} P_{es}^* = 0 \qquad \ldots(4.29);$$

$$\frac{\partial C_i(\cdot)}{\partial \beta_i^*} + \sum_s d_{is} P_{as}^* = 0 \qquad \ldots(4.30).$$

Thus the optimal equilibrium prices for effluents and abstractions that clear the permit market at site s are

$$P_{es}^* = -\lambda \frac{\partial f_s(\cdot)}{\partial E_s} \qquad \ldots(4.31);$$

$$P_{as}^* = -\lambda \frac{\partial f_s(\cdot)}{\partial H_s} \qquad \ldots(4.32).$$

As we discussed previously, the value of λ reflects the preference of the environment authority. If the total quantity of permits is chosen optimally, i.e. at the level where the marginal damage function intersects the demand for permits, then the social optimum is achieved. Or, putting this another way, when the environment authority makes λ equal to the value of the shadow price of water quality, then the TPP system will be cost efficient in the delivery of social welfare. If the total amount of pollution permits is determined by the influence of some other factors, λ might be chosen at other value rather than the shadow cost of water quality. In this case, then permits trading will ensure that the target will be met at least social cost (Xepapadeas 1997), thus the cost-effectiveness of the TPP system will remain in achieving the prescribed water quality target.

APS is a TPP system in which separate permits regulate the impact of emission and abstraction form various individuals on water qualities at separate WQM site, where the permits are exchanged at rates governed by the transfer coefficients. Each WQM site s is called an ambient permit system. The APS appears to offer a very simple regulatory process to the environmental authority. Only the proper amount of pollution permits at each WQM site need to be determined and distributed, either through auction or a "grandfathering" process in which the permits are granted to the current existing polluters in proportion to their current emission levels. However, this system would be extremely cumbersome from the viewpoint of polluters. Each polluter will have to hold a "portfolio" of permits at each WQM site that it might affect, for both effluent emission and water abstraction if necessary. There will be one market at each WQM site. When there are many markets involved in the trading process if a polluter who affects these WQM sites wishes to increase his pollution through purchasing corresponding pollution permits, the transaction cost will be so high that it might actually prevent the trade from occurring. The second deficiency of APS would be the possibility of generating "hot spot" of pollution through the trading of pollution permits, since locations which tend to generate more pollution by purchasing more permits are usually the locations which are more difficult to abate pollution and more sensitive to damage of pollution. In the case of this research, although the assimilative capacity of river water would change seasonally, it is not very feasible to set the effluent emission permits and water abstraction licences specific to different time periods during the trade process.

2. Emission Permit system (EPS)

Some of the difficulties and complexities associated with the APS could be reduced to some extent by using an emission permit system (EPS). However, an EPS does not have the least cost property that APS can offer, so it is not a cost effective TPP system. In an EPS the environmental authority will divide the whole region into several zones, within each zone the pollution sources are allowed to trade the pollution permits on a one-for-one basis and ignore the spatial differences among their locations. This system could facilitate trading among the polluters and avoid the "hot spot" problems (Xepapadeas 1997). Unlike APS, EPS could greatly simplify life for polluters (Baumol and Oates 1988), because of the simple exchange rate between polluters within the zone. But when the dispersion characteristics of the pollutions from different polluters, i.e. the transfer coefficients in our case, are very different, the EPS could deviate far from the least cost allocation of pollution abatement. As a result of the simpler trade process for potential traders, the environment authority then have to bear the burden of calculating the amount of pollution permits at each WQM site, and then readjusts it iteratively as times goes by in order to achieve the solution nearest to the least-cost allocation.

3. The pollution-offset (PO) system

A system which combines APS and EPS is the pollution offset (PO) system. A PO system is able to circumvent the problems associated with APS and EPS systems (Baumol and Oates 1988). In a PO system, the pollution permits are defined in terms of emissions, as in the EPS, while the trade of the permits is not on a one-for one basis, but instead are undertaken at ratios that depend on the contribution of their pollution to the ambient water quality at the WQM sites.

The PO system inherits the least-cost property from the APS system, because the ratio of permits trade will eventually lead to a cost effectiveness allocation of permits. This property does not depend on the initial allocation of permits as in APS, any initial allocation will be led to the cost effective equilibrium by the market force (Baumol and Oates 1988; Xepapadeas 1997). There is also no heavy burden to the environmental authority to solve the minimization problem of the polluters in each zone as it would be required under the EPS. Under a PO system, the environment authority needs to establish the transfer coefficients matrix of all the polluters, $[b_{is}]$, $[d_{is}]$, and the effects of effluent discharges and water abstractions on the ambient water quality, $\frac{\partial f_s(\cdot)}{\partial E_s}$ and $\frac{\partial f_s(\cdot)}{\partial H_s}$.

Unlike the APS, the polluters could trade their emission directly with other polluters in a PO system, therefore there is no need to trade the pollution permits in a multitude of separate permits markets at each WQM site, thus the high transaction costs associated with the APS could be avoided. The only constraint on the operation of a PO system is that the trade process must not generate a violation of the prescribed water quality at any WQM site. Therefore, the PO system offers a promising approach to the design of a TPP system, although there may be cases in which an EPS is preferred because of its simplicity in trade. A "non-degradable offset" PO system, requires that the total emissions in the zone must not increase after any trade. This "non-degradable offset" PO system has been argued to be relatively more cost effective than a range of other TPP systems (Atkinson and Tietenberg 1984).

4. The link between APS and PO system

In a PO system, the trade ratio of pollution permits is based on their effects on the water quality at the WQM site, which determined by the aggregated pollution level at each WQM site. The value of the trade ratio between any two effluent emission at sites i, j, $\varphi_{i,j}^{e}$, should be determined based on the effects of emissions from the two sources on water quality at the binding monitoring site, or based on the permits prices at each site and their transfer coefficients to the WQM sites.

$$\varphi_{i,j}^{e}=\frac{\sum_{s}\frac{\partial f_s(\cdot)}{\partial e_i^*}}{\sum_{s}\frac{\partial f_s(\cdot)}{\partial e_j^*}}=\frac{\sum_{s}\frac{\partial f_s(\cdot)}{\partial E_s}\cdot\frac{\partial E_s}{\partial e_i^*}}{\sum_{s}\frac{\partial f_s(\cdot)}{\partial E_s}\cdot\frac{\partial E_s}{\partial e_j^*}}=\frac{\sum_{s}\frac{P_{es}^*}{-\lambda}\cdot b_{is}}{\sum_{s}\frac{P_{es}^*}{-\lambda}\cdot b_{js}}=\frac{\sum_{s}P_{es}^*\cdot b_{is}}{\sum_{s}P_{es}^*\cdot b_{js}} \quad \ldots(4.33),$$

where e_i^* and e_j^* here denote the optimal level of emission form each site and P_{es}^* denotes the market clear permit price at each WQM site.

Because the equilibrium price of pollution permits at each WQM site is known to all the potential traders once the market clears, and also the transfer coefficient matrix for all the potential traders is authorized and published by the environmental authority, then the trade ratio $\varphi_{i,j}^{e}$ between any two polluters can easily be found and used in the trading process. In this way, the pollution level at each WQM site remains unchanged after the trade of pollution emissions whilst the total cost of pollution abatement is reduced. Thus the desired ambient water quality is achieved at the least cost through the TPP system. In the same way, the trade ratio between water abstraction at site i,j, $\varphi^{a}_{i,j}$, could be found out, which is equal to

$$\varphi^{a}_{i,j}=\frac{\sum_s\frac{\partial f_s(\cdot)}{\partial\beta_i^*}}{\sum_s\frac{\partial f_s(\cdot)}{\partial\beta_j^*}}=\frac{\sum_s\frac{\partial f_s(\cdot)}{\partial H_s}\cdot\frac{\partial H_s}{\partial\beta_i^*}}{\sum_s\frac{\partial f_s(\cdot)}{\partial H_s}\cdot\frac{\partial H_s}{\partial\beta_j^*}}=\frac{\sum_s\frac{P^*_{as}}{-\lambda}\cdot d_{is}}{\sum_s\frac{P^*_{as}}{-\lambda}\cdot d_{js}}=\frac{\sum_s P^*_{as}\cdot d_{is}}{\sum_s P^*_{as}\cdot d_{js}} \qquad \ldots(4.34).$$

More importantly, the trading ratio could also be expanded to cover the trade between permits for effluent emission and permits for water abstractions, an opportunity which has not been realized so far. According to the criteria that the trading ratio should be based on the effects on the aggregated pollution level at the WQM site[2], the value of the ratio for trading between emissions and abstractions at any two different sites is[3]

$$\varphi^{e,a}_{i,j}=\frac{\sum_s\frac{\partial f_s(\cdot)}{\partial e_i^*}}{\sum_s\frac{\partial f_s(\cdot)}{\partial\beta_j^*}}=\frac{\sum_s\frac{\partial f_s(\cdot)}{\partial E_s}\cdot\frac{\partial E_s}{\partial e_i^*}}{\sum_s\frac{\partial f_s(\cdot)}{\partial H_s}\cdot\frac{\partial H_s}{\partial\beta_j^*}}=\frac{\sum_s\frac{P^*_{es}}{-\lambda}\cdot b_{is}}{\sum_s\frac{P^*_{as}}{-\lambda}\cdot d_{js}}=\frac{\sum_s P^*_{es}\cdot b_{is}}{\sum_s P^*_{as}\cdot d_{js}} \qquad \ldots(4.35).$$

4.3 Dynamic Analysis of Environmental Policy

In a dynamic analysis, product output, effluent abatement and water abstraction from a plant are considered to be dynamic functions of capital stock. This assumes that the costs of labour are negligible compared with capital costs, or that they could be taken into account as the operational costs. The capital stock of firm depends on its investment through time, which is an exogenous choice variable in the model, and depends on the depreciation of the current stock. The dynamic analysis aims to find out the optimal investment path for a given plant under different environmental policy instruments, and to evaluate the feasibility of optimal investment schemes under different policies.

[2] As discussed before, water abstraction is regarded as another type of pollution besides the effluent emissions in this research. More detailed discussion could be found in Sheail (1997).
[3] This also the opportunity of trade in the same plant between effluent emission permits and water abstraction permits, which is the case of emission trade within one plant, described by Tietenberg (1990).

4.3.1 Dynamic problem with discrete time

The production output, effluent abatement and water abstraction of a firm in discrete time are functions of the firm's capital stock at time t, i.e. $q_{i,t} = q_i(k^q_{i,t})$, $a_{i,t} = a_i(k^a_{i,t})$ and $\beta_{i,t} = \beta_i(k^\beta_{i,t})$, respectively. $k^q_{i,t}$, $k^a_{i,t}$ and $k^\beta_{i,t}$ here denote the capital stock which firm i has available for be used in these activities.

In the economic problems, the time horizon of dynamic problems is usually allowed to approach infinity. This is not because the firm or environmental authority has to consider to adopt a policy of sustainable development for indefinite time, but that even the time horizon of planning is to stop at some point of time, the remaining stocks still have to be valued along the horizon by discounting what they could produce in the future (Aronsson *et al.* 2004), unless they will have no value after the planning period, which is not usually the case. Therefore, the environmental authority will only plan for a period but take into account the present-value of remaining stock after the management, such as a optimisation problem to maximise $\sum_{t=0}^{T} \rho^t \cdot F(t) + V(T)$, where $F(t)$ is the production function at time t, ρ is discounting factor, and $V(T)$ is the present-value of remaining stock after the management period. Since $V(T)$ is equal to the value produced in the future after the management, $V(T) = \sum_{t=T}^{\infty} \rho^t \cdot F(t)$.

Therefore the problem becomes,

$$\sum_{t=0}^{T} \rho^t \cdot F(t) + V(T) = \sum_{t=0}^{T} \rho^t \cdot F(t) + \sum_{t=T}^{\infty} \rho^t \cdot F(t) = \sum_{t=0}^{\infty} \rho^t \cdot F(t),$$

same as an optimisation problem over a finite horizon will end up the same as the problem with an infinite horizon.

In our case, the optimisation seeks to minimize the aggregate costs of achieving the desired level of water quality. The environmental authority is unlikely to adopt a perspective with infinite time horizon for its policy making; nonetheless, the water quality is still to be kept at least on the required level after the finite time horizon. Therefore the aggregate costs of keeping the water quality after the finite regulation time horizon, is still to be taken account in the dynamic optimisation, so the minimization of total abatement cost is also using an infinite time horizon.

For the dynamic optimisation in this research, the capital stocks in the firm are the state variables; investment levels to the capital stocks are the control variables which determine the level of state variables to achieve the optimisation. The capital is constrained to the three elements, output production, effluent abatement and water abstraction, independent of each other. The water quality target is to be met at each WQM site at each time period. The water quality target is the constraint to the dynamic optimisation, represented by Eq (4.38). The dynamic of capital is represented by Eq (4.37) where the capital stock in the next period for each element is determined by the current capital stock for each element, on-going investment in the element and the depreciation of current capital stock.

The objective of the environmental authority, which wants to achieve the desired environmental target at the least abatement cost, is therefore

$$\mathrm{Min}\sum_{t=0}^{\infty}\sum_{i}\rho^{t}\cdot C_i(q_{i,t},a_{i,t},\beta_{i,t},I_{i,t}^{j}) \qquad \ldots(4.36),$$

$$\text{s.t.}\quad k_{i,t+1}^{j}-k_{i,t}^{j}=I_{i,t}^{j}-\delta_i^{j}k_{i,t}^{j},\quad j=q,a,\beta \qquad \ldots(4.37);$$

$$Q_s=f_s(A_s,E_{s,t},H_{s,t},\varepsilon_s)+\gamma_s\geq\overline{Q}_s, \qquad \ldots(4.38);$$

$k_{i,0}^{j}$ is given.

$\rho=(1+r)^{-1}$ is the discount factor, with r equals to the interest rate. δ_i^j represents the depreciation rates for each element of capital stock.

The current-value Hamiltonian of this dynamic problem is

$$\hat{H}=\sum_i\left[C_i(q_{i,t},a_{i,t},\beta_{i,t},I_{i,t}^{j})+\sum_j\rho\mu_{i,t+1}^{j}(I_{i,t}^{j}-\delta_i^{j}\cdot k_{i,t}^{j})\right] \qquad \ldots(4.39);$$

The water quality target is to be satisfied at each WQM site and at any time. The constraint is then captured by the Lagrange Equation below:

$$L = \hat{H} + \sum_{s} \lambda_{s,t} \cdot (\overline{Q}_s - f_s(A_s, E_{s,t}, H_{s,t}, \varepsilon_s) - \gamma_s) \qquad \ldots(4.40).$$

$\mu^{j}_{i,t+1}$ is called co-state variable or dynamic Lagrange multiplier. This could be interpreted as shadow price or value of one unit extra capital stock at time t to the firm's costs. The $\lambda_{s,t}$, this indicate the consequences which a marginal change in the water quality constraints at site s in time t would carry for the minimised cost of water quality compliance. The Lagrange multipliers $\lambda_{s,t}$, which were static variables before, are now dynamic variables, i.e. their values can change through time. This feature is necessary because the environmental quality constraint must be satisfied at every time slot t in the planning period.

The FOCs of the minimum cost solution to this dynamic problem are (Barro and Sala-i-martin 1999) as below, where variables with asterisk represent their values at the dynamic equilibrium:

$$I^{j*}_{i,t} \cdot \left[\frac{\partial C^*_i(\cdot)}{\partial I^j_{i,t}} + \rho \mu^j_{i,t+1} \right] = 0,\ I^{j*}_{i,t} \geq 0 \qquad \ldots(4.41);$$

$$\lambda_{s,t} \cdot [\overline{Q}_s - f_s(A_s, E_{s,t}, H_{s,t}, \varepsilon_s) - \gamma_s] = 0,\ \ \lambda_{s,t} \geq 0 \qquad \ldots(4.42);$$

$$\rho \mu^j_{i,t+1} - \mu^j_{i,t} = -\frac{\partial L^*}{\partial k^j_{i,t}} \qquad \ldots(4.43);$$

$$\frac{\partial L^*}{\partial \mu^j_{i,t}} = 0 \Rightarrow k^j_{i,t+1} - k^j_{i,t} = I^j_{i,t} - \delta^j_i k^j_{i,t} \qquad \ldots(4.44);$$

and the transversality condition

$$\lim_{t \to \infty} \rho^t \cdot \mu^j_{i,t} \cdot k^j_{i,t} = 0 \qquad \ldots(4.45).$$

From Eq (4.40), we have:

$$\mu^j_{i,t} - \rho \mu^j_{i,t+1}(1 - \delta^j_i) = \frac{\partial C^*_i(\cdot)}{\partial k^j_{i,t}} - \sum_{s} \lambda_{s,t} \cdot \frac{\partial f^*_s(\cdot)}{\partial k^j_{i,t}} \qquad \ldots(4.46).$$

At the steady-state equilibrium, theses is

$k_i^j = k_{i,t}^j = k_{i,t+1}^j$;

$I_i^j = I_{i,t}^j = I_{i,t+1}^j$;

$\mu_i^j = \mu_{i,t}^j = \mu_{i,t+1}^j$;

Therefore Eq (4.44) implies that at the steady state equilibrium $I_i^j = \delta_i^j k_i^j$, which says that at the steady state equilibrium, the investment undertaken by the firm in every year should equal the depreciation incurred in the already deployed capital stock. At the steady state equilibrium Eq (4.46) reduces to the function below after substituting $\rho = (1+r)^{-1}$,

$$\rho \cdot r = \left(\frac{\partial C_i^*(\cdot)}{\partial k_i^j} - \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^j}\right) / \mu_i^j - \rho \cdot \delta_i^j \qquad \ldots(4.47).$$

This equation implies some fundamental economic interpretation. The left hand side is the product of the interest rate and the discount factor. The interest rate could be regarded as the average rate of return from other areas of economy for one unit capital investment. So the left hand side equals to the discounted average rate of return from investing capital in other areas of the economy. The right hand side consists of two parts. The first one, similar to the static analysis, gives the average rate of return to net cost of water quality management, which is equal to the marginal effect of an extra unit of capital stock, either in output production, effluent abatement or water abstraction, on the firm's current-value individual cost, net of the marginal effects generated by that extra capital expenditure on the ambient environmental quality, then weighted by the shadow prices of the extra capital. The shadow price of capital is the amount by which the current-value cost at dynamic optimum $C_i^*(q_{i,t}, a_{i,t}, \beta_{i,t}, I_{i,t}^j)$ will increase if the capital k_i^j were to increase by a small amount (Hoy *et al.* 2001). The second part is the discounted depreciation rate of the capital stock that is invested in the plant. The right hand side is then the overall average rate of return on the capital investment in the plant, taking account of environmental costs in water quality. Therefore, at the steady state equilibrium, the capital investment should deliver the same average rate of return in both internal (i.e. in the firm) and external (i.e. in other area of economy) investment decisions.

4.3.2 Dynamic problem with continuous time

In a continuous time setting, the product output, effluent abatement and water abstraction are assumed as functions of current capital stock, which are themselves dynamic functions of time dependent on previous investment decisions and on-going depreciation.

$$q_i(t) = q_i(k_i^q(t))$$

$$a_i(t) = a_i(k_i^a(t))$$

$$\beta_i(t) = \beta_i(k_i^\beta(t))$$

The objective of the environmental authority to achieve the environmental target at the minimal cost becomes:

$$\text{Min } \int_0^\infty e^{-rt} \cdot \sum_i C_i(q_i, a_i, \beta_i, I_i^j) \qquad \ldots(4.48)$$

$$\text{s.t. } \dot{k}_i^j = I_i^j(t) - \delta_i^j k_i^j(t), \quad j = q, a, \beta, \qquad \ldots(4.49);$$

$$Q_s = f_s(A_s, E_s(t), H_s(t), \varepsilon_s) + \gamma_s \geq \overline{Q}_s, \qquad \ldots(4.50);$$

$k_i^j(0)$ is given.

The current-value Hamiltonian for this continuous time dynamic problem is

$$\hat{H} = \sum_i \left[C_i(q_i(t), a_i(t), \beta_i(t), I_i^j(t)) + \sum_j \mu_i^j(t) \cdot (I_i^j(t) - \delta_i^j \cdot k_i^j(t)) \right] \qquad \ldots(4.51)$$

The corresponding augmented Lagrange Function is

$$L = \hat{H} + \sum_s \lambda_s(t) \cdot (\overline{Q}_s - f_s(A_s, E_s(t), H_s(t), \varepsilon_s) - \gamma_s) \qquad \ldots(4.52).$$

The co-state variables $\mu_i^j(t)$ and static Lagrange multiplier $\lambda_s(t)$ have similar economic meaning to those described in the discrete time setting, only now they are functions of the continuous variable t rather than discrete time variables.

The FOCs for the minimum cost solution are

$$I_i^j \cdot \left(\frac{\partial C_i^*(\cdot)}{\partial I_i^j} + \mu_i^j \right) = 0, \quad I_i^j \geq 0 \qquad \ldots(4.53);$$

$$\lambda_s \cdot \left[\overline{Q}_s - f_s^*(A_s, E_s(t), H_s(t), \varepsilon_s) - \gamma_s \right] = 0, \quad \lambda_s \geq 0 \qquad \ldots(4.54);$$

$$r\mu_i^j - \dot{\mu}_i^j = \frac{\partial L^*}{\partial k_i^j} \Rightarrow \dot{\mu}_i^j = (r + \delta_i^j)\mu_i^j - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} + \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^j} \qquad \ldots(4.55);$$

$$\frac{\partial L^*}{\partial \mu_i^j} = 0 \Rightarrow \dot{k}_i^j = I_i^{j*} - \delta_i^j k_i^{j*} \qquad \ldots(4.56);$$

and the transversality condition

$$\lim_{t \to \infty} e^{-rt} \cdot \mu_i^j(t) \cdot k_i^j(t) = 0 \qquad \ldots(4.57).$$

From Eq (4.53)

$$\mu_i^{j*} = -\frac{\partial C_i^*(\cdot)}{\partial I_i^j} \qquad \ldots(4.58).$$

For simplicity in writing, from now on, I use $C_i^{'}(\cdot)$ to represent the first order partial derivative of cost function against investment I_i^j, $C_i^{''}(\cdot)$ to represent the second order partial derivative against I_i^j, and so on. Variables with asterisk indicate that they are at the level of steady state equilibrium value. From Eq. (4.58), we have

$$\frac{\partial I_i^j}{\partial \mu_i^j} = \frac{1}{\partial \mu_i^j / \partial I_i^j} = -C_i^{''}(\cdot)^{-1}, \quad \forall j. \qquad \ldots(4.59),$$

therefore,

$$\dot{\mu}_i^j = \frac{\partial \mu_i^j}{\partial t} = -\frac{\partial C_i^{'}(\cdot)}{\partial t} = -\frac{\partial C_i^{'}(\cdot)}{\partial I_i^j} \cdot \frac{\partial I_i^j}{\partial t} = -\partial C_i^{''}(\cdot) \cdot \dot{I}_i^j \qquad \ldots(4.60).$$

Substituting Eqs (4.60) into Eq (4.55) gives

$$\dot{I}_i^j = -\frac{1}{C_i^{''}(\cdot)} \cdot [(r + \delta_i^j)\mu_i^{j*} - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} + \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^j}] \qquad \ldots(4.61).$$

Eq (4.61) and Eq (4.56) could form another pair of Hamiltonian dynamic equations regarding the control variable of investment $\dot{I}_i^j$, and state variable, capital stocks $\dot{k}_i^j$, which have more practical meaning than $\dot{\mu}_i^j$ and $\dot{k}_i^j$

described by Eq (4.55) and Eq (4.56), because in reality I_i^j is much easier to be seen μ_i^j.

In the new system, the steady state long-run equilibrium is defined as the point at $(I_i^{j*}, k_i^{j*}), \forall i, \forall j$, where there is $\dot{I}_i^j = \dot{k}_i^j = 0, \forall i, \forall j$.

Then from Eq (4.56) and (4.61),

$$I_i^{j*} = \delta_i^j \cdot k_i^{j*} \qquad \ldots(4.62)$$

$$\frac{1}{\mu_i^{j*}} \cdot \left(\frac{\partial C_i^*(\cdot)}{\partial k_i^j} - \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^j}\right) - \delta_i^j = r \qquad \ldots(4.63)$$

Eqs (4.62) and (4.63) have similar economic interpretations to their counterparts in the discrete-time dynamic problem. Eq (4.62) says that at the long-run steady state equilibrium, the investment rate of the firm, in all the three elements of capital stocks, should be equal to the depreciation rate of each element of capital stock so that the capital stock remains constant at levels which comply with the environmental regulations. Eq (4.63) expresses the same meaning as Eq (4.47) but in a continuous time setting, stating that under optimal investment management, the internal rate of return of increasing capital stock on the current-value cost at dynamic optimum should equate the external rate of return on capital invested elsewhere in the economy.

4.3.3 The Convergence and Stability Properties of the Steady State Equilibrium

According to the analysis above, the solution to the dynamic cost minimisation problem could be found using the FOCs of Hamiltonian equilibrium. The steady state equilibrium could be found out by setting the motion of the co-state, state and control variables of the dynamic system to zero. In our case these are the variables μ_i^j, k_i^j and I_i^j respectively. However, knowing the steady sate equilibrium alone is of little use without discussing the convergence and stability properties of the dynamic system. An equilibrium point which only exists in principle, but which cannot be approached or which is such that the slightest disturbance could produce divergence away from it – an unstable equilibrium point – is obviously not very relevant from an economic point of view (Gandolfo 1997).

The following analysis, investigates the convergence and stability properties of the dynamic system of water pollution control under investment decisions based on the continuous time series. Analysis for the solution of the dynamic problem in discrete time could be illustrated similarly. In the following analyses of convergence and stability, the capital stock and investment are assumed having independent impacts on the total cost, i.e. $\frac{\partial^2 C_i(\cdot)}{\partial k_i^j \partial I_i^j} = 0$.

Stability and convergence are typically investigated using a phase plane approach in which two relevant variables form the two axes of the phase plane. Depending on the variables and the steady state equilibrium we would like to investigate, there are two Hamiltonian dynamic system formed by the co-state, state and control variables, namely k_i^j, μ_i^j and k_i^j, I_i^j, $j = q, a, \beta$. The quantitative analyses are carried out in each system as follows.

4.3.3.a Dynamic System in terms of $\dot{k}_i^j$, $\dot{\mu}_i^j$

From discussion above, there are

$$\dot{\mu}_i^j = (r + \delta_i^j)\mu_i^j - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} + \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^j} = F(\mu_i^j, k_i^j) \qquad \ldots(4.55),$$

$$\dot{k}_i^j = I_i^{j*} - \delta_i^j k_i^{j*} \qquad \ldots(4.56).$$

From Eq(4.58), $\mu_i^{j*} = -\frac{\partial C_i^*(\cdot)}{\partial I_i^j} = -C_i'(\cdot) = g^{-1}(I_i^{j*})$ where g^{-1} represents the reverse function of function g, so we have $I_i^{j*} = g(\mu_i^{j*})$. Therefore, Eq(4.56) could be rewritten as a function of μ_i^j, k_i^j,

$$\dot{k}_i^j = I_i^{j*} - \delta_i^j k_i^{j*} = g(\mu_i^{j*}) - \delta_i^j k_i^{j*} = G(\mu_i^j, k_i^j) \qquad \ldots(4.64).$$

Due to the nonlinearity of the dynamic system defined by $\dot{k}_i^j$, $\dot{\mu}_i^j$, the global stability of this system cannot be investigated. Here we use the linearization method[4] to analyse the local stability of the steady state equilibrium. As this system is autonomous, the following linearised system in the neighbourhood of the steady state is a good approximation to the original non-linear system formed by Eq (4.55) and (4.64) around the steady state equilibrium (Gandolfo 1997).

The linearization method states that for $\dot{x}(t) = f(x(t)), f: R^n \to R^n$, if x^* is an equilibrium, then $\dot{x}(t) = A(x(t) - x^*), A = \left[\dfrac{\partial f_i(x^*)}{\partial x_j}\right], i, j = 1,2,\ldots n$, where A is the Jacobian matrix of the system evaluated at the equilibrium point. If the equilibrium point in the linear approximation is globally stable, then it is also locally stable at the original non-linear system. The converse is not necessarily true (Xepapadeas 1997).

For the original non-linear system formed by Eqs (4.55) and (4.64),

$$A \equiv \begin{bmatrix} \dfrac{\partial F(\cdot)}{\partial \mu_i^j}, & \dfrac{\partial F(\cdot)}{\partial k_i^j} \\ \dfrac{\partial G(\cdot)}{\partial \mu_i^j}, & \dfrac{\partial G(\cdot)}{\partial k_i^j} \end{bmatrix}_{\left(\substack{\mu_i^j = \mu_i^{j*} \\ k_i^j = k_i^{j*}}\right)} = \begin{bmatrix} a_{11} & a_{12} \\ a_{21} & a_{22} \end{bmatrix}$$, where it can be proved that

$$a_{11} = r + \delta_i^j > 0,$$

$$a_{12} = \sum_s \lambda_s \cdot \frac{\partial^2 f_s^*(\cdot)}{\partial k_i^{j2}} - \frac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j2}},$$

$$a_{21} = \frac{\partial I_i^j}{\partial \mu_i^j} = \frac{1}{\partial \mu_i^j / \partial I_i^j} = -C_i''(I_i^{j*})^{-1},$$

$$a_{22} = -\delta_i^j < 0,$$

$$j = q, a, \beta.$$

[4] For the details of linearization method, see Appendix 1

If the determinant of Jacobian matrix A, denoted as $\det A = a_{11}a_{22} - a_{12}a_{21}$, is nonzero, the qualitative behaviour of the trajectories of the non-linear system in the neighbourhood of its steady state point (k_i^{j*}, μ_i^{j*}) is the same as that of the trajectories of the homogeneous system before, which is linearised from the non-linear system (Xepapadeas 1997). The *sign of* $\det A$ implies the stability properties of the dynamic system, which depends on signs of $\frac{\partial^2 f_s^*(\cdot)}{\partial k_i^{j^2}}$, $\frac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j^2}}$ and $C_i''(I_i^{j*})^{-1}$.

Recall the economic meaning of k_i^j and I_i^j, the signs of the partial derivatives of costs and water quality with respect to these variables determine the sign of $\det A$. Since it is more expensive to accelerate the increase in capital stock, $C_i'(I_i^j) > 0, C_i''(I_i^j) \geq 0$. From the relationship stated in Eq (4.56), it is also reasonable to assume that $\frac{\partial C_i(\cdot)}{\partial k_i^j} > 0, \frac{\partial^2 C_i(\cdot)}{\partial k_i^{j^2}} \geq 0$ [5] (i.e. cost increases at an increasing rate as capital is accumulated for output production, effluent abatement and water abstraction). For the partial derivatives of water quality with respect to the different elements of capital stock, it is obvious to see that $\frac{\partial f_s(\cdot)}{\partial k_i^q} < 0, \frac{\partial f_s(\cdot)}{\partial k_i^\beta} < 0$ and $\frac{\partial f_s(\cdot)}{\partial k_i^a} > 0$ (i.e. water quality reduces as production and abstraction increase, and increases as abatement increases). Due to the widely existing increasing marginal damage of water pollution, we can assume that $\frac{\partial f_s^2(\cdot)}{\partial k_i^{q^2}} \leq 0, \frac{\partial f_s^2(\cdot)}{\partial k_i^{\beta^2}} \leq 0$. On the other hand, the effect of abatement on pollution effluent is either constant or diminishing in most of the situations, so $\frac{\partial f_s^2(\cdot)}{\partial k_i^{a^2}} \leq 0$.

[5] This may not always be true in reality. An exceptional case in reality could be found in Hanley et al. (1998), in which the abatement of pollution in a particularly large firm has decreasing marginal cost, i.e. $\frac{\partial^2 C_i(\cdot)}{\partial k_i^{a^2}} < 0$.

From the discussion and assumptions above, we now have enough information to determine the sign of $\det A$. For each element of capital stock and its corresponding investment element j we have:

$$a_{12} = \sum_s \lambda_s \cdot \frac{\partial^2 f_s^*(\cdot)}{\partial k_i^{j2}} - \frac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j2}} \le 0,$$

$$a_{21} = -C_i^{"}(I_i^{j*})^{-1} \ge 0.$$

Therefore $\det A = a_{11}a_{22} - a_{12}a_{21} < 0, \forall j$. Thus, $\det A$ must be negative. Because of the negative sign of the $\det A$, the eigenvalues of the Jacobian matrix A are of opposite sign. Therefore the steady state equilibrium (k_i^{j*}, μ_i^{j*}) of the non-linear dynamic system constructed by $\dot{k}_i^j$ and $\dot{\mu}_i^j$ is a saddle point, and the trajectories in the (k_i^j, μ_i^j) phase plane display the property of a saddle point, at least locally around the equilibrium.

We can also utilize the phase plane analysis to help us analyse the stability and consequence properties of the non-linear dynamic system, defined in terms of $\dot{k}_i^j$ and $\dot{I}_i^j$.

4.3.3.b Dynamic System in terms of $\dot{k}_i^j$, $\dot{I}_i^j$

Because the co-state variable μ_i^j, which was considered in section 4.3.3.a is not easy to evaluate and control in reality, a more practical dynamic system is constructed by the state variable k_i^j and control variable I_i^j as below:

$$\dot{I}_i^j = -\frac{1}{C_i^{"}(\cdot)} \cdot [(r+\delta_i^j)\mu_i^j - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} + \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^j}] \qquad \ldots(4.61)$$

$$\dot{k}_i^j = I_i^{j*} - \delta_i^j k_i^{j*} = F(k_i^j, I_i^j) \qquad \ldots(4.56).$$

From Eq (4.58), $\mu_i^{j*} = -C_i^{'}(\cdot)$, so Eq (4.61) becomes

$$\dot{I}_i^j = -\frac{1}{C_i^{"}(\cdot)} \cdot [-(r+\delta_i^j) \cdot C_i^{'}(\cdot) - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} + \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^j}] = G(k_i^j, I_i^j) \qquad \ldots(4.65).$$

After linearization, the non-linear dynamic system constructed by Eqs (4.65) and (4.56), yields the Jacobian matrix

$$A \equiv \begin{bmatrix} \frac{\partial F(\cdot)}{\partial k_i^j}, & \frac{\partial F(\cdot)}{\partial I_i^j} \\ \frac{\partial G(\cdot)}{\partial k_i^j}, & \frac{\partial G(\cdot)}{\partial I_i^j} \end{bmatrix}_{\left(\substack{k_i^j = k_i^{j*} \\ I_i^j = I_i^{j*}}\right)} = \begin{bmatrix} a_{11}, & a_{12} \\ a_{21}, & a_{22} \end{bmatrix}, \text{ where}$$

$$a_{11} = -\delta_i^j < 0,$$

$$a_{12} = 1,$$

$$a_{21} = -C_i''(\cdot)^{-1} \cdot (\sum_s \lambda_s \cdot \frac{\partial^2 f_s^*(\cdot)}{\partial k_i^{j2}} - \frac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j2}}),$$

$a_{22} = r + \delta_i^j > 0$[6],

$$j = q, a, \beta.$$

As the signs of the partial derivatives in a_{21} are determined before, it is not difficult to see that $\det A = a_{11}a_{22} - a_{12}a_{21} < 0, \quad \forall j$. Therefore the steady state equilibrium (k_i^{j*}, I_i^{j*}) of the new non-linear dynamic system is also a saddle point equilibrium for each element of capital stock and investment. This is the same conclusion as in terms of (k_i^{j*}, μ_i^{j*}), as these two dynamic equilibria are actually same in both systems, only the viewpoint differs.

4.3.3.c Qualitative analysis: phase plane

Since many dynamic systems of non-linear differential equations cannot be solved analytically, the qualitative properties of their solutions can sometimes be described and examined by using a graphic device, the phase plane (Léonard and Long 1992).

Taking the non-linear dynamic system in $\dot{k}_i^j$, $\dot{I}_i^j$ as an example, the intersection of the lines $\dot{k}_i^j = 0$ and $\dot{I}_i^j = 0$, if it exists, would be the saddle point steady state equilibrium (k_i^{j*}, I_i^{j*}). These two lines are called isoclines,

[6] For the detailed derivation of a_{22}, see Appendix 2

which are the loci of the points satisfying $\dot{k}_i^j = 0$ and $\dot{I}_i^j = 0$ respectively. The coordinate system of (k_i^j, I_i^j) is called the phase plane of the system. In the phase plane (also sometimes called the plane of the states) (k_i^j, I_i^j), the slope of the isoclines are $-(F_k / F_I)$ and $-(G_k / G_I)$ respectively (Gandolfo 1997), defined by Eqs (4.65) and (4.56), where subscript represents partial derivatives.

Therefore the slopes for the isoclines are

$$\dot{k}_i^j = 0 \Rightarrow \frac{\partial I_i^j}{\partial k_i^j} = -(F_k / F_I) = \delta_i^j > 0 ;$$

$$\dot{I}_i^j = 0 \Rightarrow \frac{\partial I_i^j}{\partial k_i^j} = -(G_k / G_I) = -\frac{\sum_s \lambda_s \cdot \frac{\partial^2 f_s^*(\cdot)}{\partial k_i^{j2}} - \frac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j2}}}{-C_i''(\cdot)^{-1} \cdot (r + \delta_i^j)} \le 0 , j = q, a, \beta .$$

When $\frac{\partial^2 f_s^*(\cdot)}{\partial k_i^{j2}} \le 0$, $\frac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j2}} \ge 0$ and $C_i''(\cdot) \ge 0$ have continuous values, and $\frac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j2}}$ increases faster than does $C_i''(\cdot)$ with k_i^j, the function describing the slope of $\dot{I}_i^j = 0$ implies that

$$\lim_{k_i^j \to 0} \left(-\frac{\sum_s \lambda_s \cdot \frac{\partial^2 f_s^*(\cdot)}{\partial k_i^{j2}} - \frac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j2}}}{-C_i''(\cdot)^{-1} \cdot (r + \delta_i^j)} \right) = 0 ;$$

$$\lim_{k_i^j \to k_{i\,\max}^j} \left(-\frac{\sum_s \lambda_s \cdot \frac{\partial^2 f_s^*(\cdot)}{\partial k_i^{j2}} - \frac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j2}}}{-C_i''(\cdot)^{-1} \cdot (r + \delta_i^j)} \right) = -\infty ,$$ where $k_{i\,\max}^j$ is the maximum value k_i^j could take. Considering the increasing marginal damage to the water quality resulting from effluent discharge and water abstraction, and the diminishing marginal effects of pollution abatement, the existence of the maximized k_i^j is expectable, even though we cannot have the accurate value of k_i^j without knowing the exact functions. The above properties of $\dot{I}_i^j = 0$ guarantee an interior solution for the steady state equilibrium.

The next step in constructing the phase plane is to determine the sign of $\dot{k}_i^j$, $\dot{I}_i^j$ in the regions separated by their isoclines. Since $\frac{\partial \dot{I}_i^j}{\partial I_i^j} = r + \delta_i^j > 0$, this means that holding k_i^j constant, an increase (decrease) in I_i^j will result in an increase (decrease) in $\dot{I}_i^j$. $\dot{I}_i^j$ is thus positive above the isocline $\dot{I}_i^j = 0$ and negative below it (Hoy *et al.* 2001). Similarly, $\frac{\partial \dot{k}_i^j}{\partial k_i^j} = -\delta_i^j < 0$, meaning that $\dot{k}_i^j$ is negative for points in the phase plane to the right of isocline $\dot{k}_i^j = 0$ and positive for the points to the left.

Now we have sufficient information to construct phase plane for the non-linear system of non-linear equations defined by $\dot{k}_i^j$ and $\dot{I}_i^j$. The result is shown in Figure 4.1.

The two isoclines divide the phase plane into four different regions and the directions of the trajectories in each region are indicated by the arrows. Since the phase plane is the set containing all the possible trajectories in the system, for any combination of (k_i^j, I_i^j), the system would move along trajectories in the directions specified as time increases. Therefore, knowing the directions of the trajectories, the convergence properties for any initial combination of (k_i^j, I_i^j) can be found out. Since we have proved that the Jacobian matrix A has a negative determinant, the dynamic system above is unstable with a saddle point equilibrium, (k_i^{j*}, I_i^{j*}). A unique property of saddle point equilibrium is that there is only one trajectory in the phase plane which would converge to the steady state equilibrium, while all others only diverge away from it (Hoy *et al.* 2001). The two lines, s and r, determined by the eigenvectors of the Jacobian matrix A respectively, are the asymptotes to all the remaining trajectories (Gandolfo 1997). From the phase plane it can be seen that only point in the line s will eventually converge to the steady state equilibrium (k_i^{j*}, I_i^{j*}) while the all points elsewhere on the phase plane will ultimately diverge away. Therefore line s is the only trajectory which converges to the equilibrium, which is called the stable arm while line r the unstable arm. Because functions of cost and water quality are not specified in this model, trajectories will not necessarily be exactly the same as that indicated in the phase plane above. Accurate estimation of the dynamic movement of (k_i^j, I_i^j), and the location of the stable arm of saddle equilibrium relies on the accurate specification of the cost and water quality functions. Analytical and numerical methods for identifying the stable arm of a saddle-point equilibrium with specified functions are discussed by Shone (Shone 2002). In the economic theory however, the form of a cost function is often not specified, only with its qualitative properties given. Therefore the phase plane provides a useful tool for qualitative analysis of stability and convergence properties, but is not very helpful for finding equilibrium solutions, or stable approaches to them, unless the constituent functions of the dynamic system are known in considerable details.

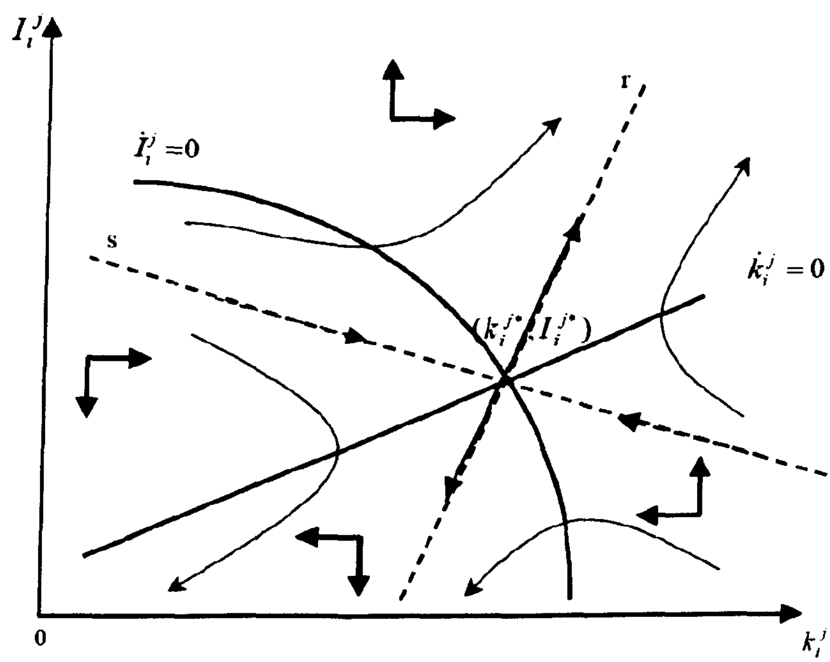

Figure 4.1 the saddle point equilibrium of steady state

4.3.4 TSS and TPP system for Dynamic Solutions

4.3.4.a Pollution TSS

As demonstrated in the static analysis of TSS, a firm needs to pay (receive) an aggregate emission and water abstraction tax (subsidy) at all the sites along the river for its effluent discharge and water abstraction activities. The level of payments (subsidies) depends on whether they discharge or abstract more than the right initially allowed to them by the environment authority.

The amount of tax (subsidy) for firm i which discharges effluent and abstracts water at site i are $T_{ie} + T_{ia} = \sum_{s}\left(e_i - e_i^0\right)\cdot b_{is}\cdot t_{es} + \sum_{s}\left(\beta_i - \beta_i^0\right)\cdot d_{is}\cdot t_{as}$, where t_{es} and t_{as} are the tax rates for the unit effect of pollution at all the sites s influenced by the effluent and abstraction. Thus the objective of a cost minimising firm under the TSS in a dynamic setting of continuous time is to

$$\text{Min} \int_0^{\infty} e^{-rt}\cdot\left[C_i(q_i, a_i, \beta_i, I_i^j) + \sum_{s}\left(e_i - e_i^0\right)\cdot b_{is}\cdot t_{es} + \sum_{s}\left(\beta_i - \beta_i^0\right)\cdot d_{is}\cdot t_{as}\right]$$

s.t. Eq (4.49), $e(t) \geq 0, \beta(t) \geq 0$ and $k_i^j(0)$ is given. Different value of j represents the three elements of capital stock, production, effluent abatement and water abstraction.

The current-value Hamiltonian for this problem is

$$\hat{H} = C_i(q_i, a_i, \beta_i, I_i^j) + \sum_{s}\left(e_i - e_i^0\right)\cdot b_{is}\cdot t_{es} + \sum_{s}\left(\beta_i - \beta_i^0\right)\cdot d_{is}\cdot t_{as} + \sum_{j}\mu_i^j\cdot(I_i^j - \delta_i^j\cdot k_i^j)$$

The following FOCs are then implied for the cost minimising solution:

$$\frac{\partial \hat{H}}{\partial I_i^j} = \frac{\partial C_i^*(\cdot)}{\partial I_i^j} + \mu_i^j = 0 \qquad \ldots(4.66),$$

$$\dot{\mu}_i^j = (r + \delta_i^j)\mu_i^j - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} - \sum_{s}\left(t_{es}\cdot b_{is}\cdot\frac{\partial e_i^*}{\partial k_i^j} + t_{as}\cdot d_{is}\cdot\frac{\partial \beta_i^*}{\partial k_i^j}\right) \qquad \ldots(4.67);$$

$$\dot{k}_i^j = I_i^{j*} - \delta_i^j k_i^{j*} \qquad \ldots(4.56);$$

and the transversality condition is:

$$\lim_{t\to\infty} e^{-rt} \cdot \mu_i^j(t) \cdot k_i^j(t) = 0 \qquad \ldots(4.57),$$

$j = q, a, \beta$.

The time path of investment in each element of capital stock is derived as before in section 4.3.2:

$$\dot{I}_i^j = -C_i''(\cdot)^{-1} \cdot \left[(r + \delta_i^j)\mu_i^j - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} - \sum_s \left(t_{es} \cdot b_{is} \cdot \frac{\partial e_i^*}{\partial k_i^j} + t_{as} \cdot d_{is} \cdot \frac{\partial \beta_i^*}{\partial k_i^j} \right) \right] \qquad \ldots(4.68).$$

Comparing the FOCs of the cost minimising solution for a firm under the TSS and those for optimal pollution control under direct command in section 4.3.2, i.e. comparing Eq (4.68) with (4.61), the equilibrium achieved through the TSS would be the same as the optimal pollution control the environmental authority wishes to achieve through direct consents, if the following tax rates are set for effluent discharge and water abstraction:

$$t_{es} = -\lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial E_s} \text{ and } t_{as} = -\lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial H_s}, \quad j = q, a, \beta.$$

When these tax rates are used, not only will the equilibria under the two different policy instruments be the same point, but also this would ensure that the dynamic systems under the two policy instruments have the same properties of convergence and stability, i.e. the dynamic system under TSS has a saddle point equilibrium with only one trajectory that converges to the steady state equilibrium. Due to the difficulty in evaluating the appropriate value of the shadow price of water quality λ_s at each WQM site, tax rates in practice might not always have the correct value to induce the optimal equilibrium, thus the two equilibria are not coincident and the TSS is not cost efficient. However, recalling the Jacobian matrix A, it can be proved that the stability and convergence properties of the dynamics system under TSS will remain the same as long as the tax rates are of opposite sign to the values of $\frac{\partial f_s^*(\cdot)}{\partial E_s}$ and $\frac{\partial f_s^*(\cdot)}{\partial H_s}$.

4.3.4.b The TPP system

Under a TPP system each firm receives an initial quantity of "pollution" permits, comprising effluent discharge permits and water abstraction licenses. Initial distribution is either through auction or via "grandfathering" by the environment authority. The initial allocation are denoted as e_{is}^0 and β_{is}^0 for the pollution at site s from the firm at site i. The firm will demand more permits if its pollution emission and water abstraction effects exceed the amount of permits that it holds for the site it influences. If it is more costly to increase abatement, the firm will try to purchase the extra permit it requires from the market, or vice versa, supply its permits to the permit market if it has surplus amount of pollution permits.

Although the TPP system and the TSS are usually considered as having equivalent regulatory effects, there are still some differences between them. One obvious difference is that the optimal tax rate has to be chosen by the environment authority while value of permits under the TPP system would be set by market automatically without intervention from the environmental authority. There is another important difference relating to the dynamic nature of the problem. Pollution permits grant a right to pollute, therefore once permits are purchased, the pollution activity would be allowed since then until it expires. So the purchase of pollution permits is more to involve a lump-sum payment rather than the annual payment required under the TSS. Although pollution permits are not to be valid forever, the permit is renewable at a negligible price compared with the purchase payment. The expense or revenue generates for the firm through permit trading in the market is $P_{es} \cdot \dot{e}_{is}(t)$, where P_{es} is the price of effluent discharge permit at site s and $\dot{e}_{is}(t)$ are the additional permits purchased (or sold) in the market in a particular year. Since the initial permits could be obtained either through "grandfathering" or auction, the model used here only considers the pollution control costs incurred after the initial permit distribution.

The cost minimisation problem for a firm at site i (assuming its effluent discharge and water abstraction are carried out locally) can be indicated as below:

$$\text{Min} \int_0^\infty e^{-rt} \cdot \left(C_i(q_i, a_i, \beta_i, I_i^j) + \sum_s \dot{e}_{is} \cdot P_{es} + \sum_s \dot{\beta}_{is} \cdot P_{as} \right) \qquad \ldots(4.69)$$

s.t. Eq (4.49), $e(t) \geq 0, \beta(t) \geq 0$ and $k_i^j(0)$ is given. Different value of j represents the three elements of capital stock, production, effluent abatement and water abstraction.

Since

$$\dot{e}_{is} = \frac{\partial e_{is}}{\partial t} = b_{is} \cdot \frac{\partial e_i}{\partial t} = b_{is} \cdot \frac{\partial e_i}{\partial k_i^j} \cdot \frac{\partial k_i^j}{\partial t} = b_{is} \cdot \frac{\partial e_i}{\partial k_i^j} \cdot \dot{k}_i^j \qquad \ldots(4.70),$$

and similarly $\dot{\beta}_{is} = d_{is} \cdot \frac{\partial \beta_i}{\partial k_i^j} \cdot \dot{k}_i^j$, Eq (4.69) can be rewritten as:

$$\int_0^\infty e^{-rt} \cdot \left(C_i(q_i, a_i, \beta_i, I_i^j) + \sum_s b_{is} \cdot \frac{\partial e_i}{\partial k_i^j} \cdot \dot{k}_i^j \cdot P_{es} + \sum_s d_{is} \cdot \frac{\partial \beta_i}{\partial k_i^j} \cdot \dot{k}_i^j \cdot P_{as} \right) \ldots(4.71).$$

The current-value Hamiltonian for this cost minimisation problem is:

$$\begin{aligned} \hat{H} = {} & C_i(q_i, a_i, \beta_i, I_i^j) + \sum_s P_{es} \cdot b_{is} \cdot \frac{\partial e_i}{\partial k_i^j} \cdot (I_i^j - \delta_i^j \cdot k_i^j) \\ & + \sum_s P_{as} \cdot d_{is} \cdot \frac{\partial \beta_i}{\partial k_i^j} \cdot (I_i^j - \delta_i^j \cdot k_i^j) + \sum_j \mu_i^j \cdot (I_i^j - \delta_i^j \cdot k_i^j) \end{aligned} \qquad \ldots(4.72).$$

The FOCs of the cot minimising solution imply the following equations,

$$\frac{\partial \hat{H}}{\partial I_i^j} = 0 \Rightarrow \mu_i^j = -\left(\frac{\partial C_i^*(\cdot)}{\partial I_i^j} + \sum_s b_{is} \cdot \frac{\partial e_i^*}{\partial k_i^j} \cdot P_{es} + \sum_s d_{is} \cdot \frac{\partial \beta_i^*}{\partial k_i^j} \cdot P_{as} \right) \qquad \ldots(4.73);$$

$$\begin{aligned} \dot{\mu}_i^j = {} & (r + \delta_i^j)\mu_i^j - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} - \sum_s (P_{es} \cdot b_{is} \cdot \frac{\partial^2 e_i^*}{\partial k_i^{j2}} \cdot \dot{k}_i^j - P_{es} \cdot b_{is} \cdot \delta_i^j \cdot \frac{\partial e_i^*}{\partial k_i^j}) \\ & - \sum_s (P_{as} \cdot d_{is} \cdot \frac{\partial^2 \beta_i^*}{\partial k_i^{j2}} \cdot \dot{k}_i^j - P_{as} \cdot d_{is} \cdot \delta_i^j \cdot \frac{\partial \beta_i^*}{\partial k_i^j}) \end{aligned} \qquad \ldots(4.74);$$

$$\dot{k}_i^j = I_i^{j*} - \delta_i^j k_i^{j*} \qquad \ldots(4.56);$$

and the relevant transversality condition is

$$\lim_{t \to \infty} e^{-rt} \cdot \mu_i^j(t) \cdot k_i^j(t) = 0 \qquad \ldots(4.57),$$

$j = q, a, \beta$.

In order to examine the steady state solution and compare that achieved under the TSS, we differentiate Eq (4.58) with respect to time to obtain:

$$\dot{\mu}_i^j = \frac{\partial \mu_i^j}{\partial t} = \frac{\partial \mu_i^j}{\partial t} \cdot \frac{\partial I_i^j}{\partial t} = -\frac{\partial C_i^{'}(\cdot)}{\partial I_i^j} \cdot \frac{\partial I_i^j}{\partial t} = -\partial C_i^{"}(\cdot) \cdot \dot{I}_i^j \qquad \ldots(4.75).$$

By substituting the $\dot{\mu}_i^j$ and μ_i^j in Eqs (4.75) and (4.73) into Eq (4.74), we obtain:

$$\begin{aligned} \dot{I}_i^j = -C^{"}(I_i^{j*})^{-1} \cdot [-(r+\delta_i^j) \cdot \frac{\partial C_i^*(\cdot)}{\partial I_i^j} - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} - \sum_s P_{es} \cdot b_{is} \cdot \frac{\partial^2 e_i^*}{\partial k_i^{j2}} \cdot \dot{k}_i^j \\ - \sum_s P_{as} \cdot d_{is} \cdot \frac{\partial^2 \beta_i^*}{\partial k_i^{j2}} \cdot \dot{k}_i^j - \sum_s r \cdot P_{es} \cdot b_{is} \cdot \frac{\partial e_i^*}{\partial k_i^j} - \sum_s r \cdot P_{as} \cdot d_{is} \cdot \frac{\partial \beta_i^*}{\partial k_i^j} \end{aligned} \qquad \ldots(4.76)$$

At the steady state equilibrium where $\dot{k}_i^j = \dot{I}_i^j = \dot{\mu}_i^j = 0$, Eq (4.76) could be reduced to Eq (4.68) if $P_{es} = t_{es} / r$ and $P_{as} = t_{as} / r$. Therefore the TSS and TPP system lead to the same steady state equilibrium for investment and each element of the capital stock. This result is understandable because that through the purchase of one unit pollution permit the firm saves an infinite stream of tax payments which would otherwise be incurred for this unit of pollution. Therefore the firm needs pay an amount that is equal to the present-value of the aggregate tax payment (i.e. the present-value of the stream of tax payments into the infinite future, which equals t_{es} / r and t_{as} / r for effluent discharge and water abstraction respectively). It can be proved that the convergence and stability properties of the steady state equilibrium under the TPP system would be the same as those in the TSS (Xepapadeas 1997), hence also same as those discussed under the direct control and command option.

4.4 Comparative Statics

Comparative statics analysis studies the displacement of the equilibrium solution, evaluating how the equilibrium values of the variables respond to a change in one or more parameters. The response is examined by considering in which direction the steady state configuration moves to establish a new equilibrium to match the new configuration of parameters (Gandolfo 1997).

Changes in the values of exogenous variables will in general affect the optimal solution, and also vice versa, i.e. the optimal equilibrium can be modified by changing the exogenous changes. In our case of pollution control along the river, adjustment of tax rate is inevitable because of the difficulty in setting the optimal tax rate at the outset. As a policy instrument, changing the emission tax rate would influence the optimal equilibrium for investment and capital stock for each element and thus generate new equilibria for capital stock, investment, costs and water quality. Changing total amount of pollution permits in the market would function in a similar way to changing the tax rates. Therefore it is important to analyse the comparative statics of equilibrium under the tax and TPP schemes so that the environmental authority can ensure that the equilibrium would move to the desired direction when changes are made to tax rates or to the total amount of pollution permits in the market.

Taking the TSS as an example, two types of comparative statics analyses are carried out in the following section to indicate the effect of change of tax rates on the firm's pollution relevant activities and its investment decision. Short-run comparative statics indicate the effects of changing tax rate on the effluent discharge and water abstraction in a static system without considering the dynamic change in capital and investment; the steady state comparative statics, on the other hand, indicate the effects of tax policies on path of the accumulation of capital stocks and investment in each element in the long-run equilibrium.

4.4.1 Short-run Comparative Statics in the Static System

As specified in the static analysis, effluent discharge is a function of production output and abatement level, $e_i = Z_i(q_i, \alpha_i)$. Thus the FOCs for the optimal solution under a TSS, Eqs (4.27) and (4.28), can be rewritten as

$$\frac{\partial C_i^*(\cdot)}{\partial e_i} + \sum_s t_{es} \cdot b_{is} = 0 \qquad \text{...(4.27),}$$

$$\frac{\partial C_i^*(\cdot)}{\partial \beta_i} + \sum_s t_{as} \cdot d_{is} = 0 \qquad \text{...(4.28).}$$

Using the implicit function theorem (Gandolfo 1997; Xepapadeas 1997; Hoy *et al.* 2001), the short-run comparative statics analysis based on the Eq (4.27) and (4.28) can indicate the effects of changing effluent discharge tax rate on the effluent discharge and water abstraction, which are shown as below:

$$\begin{bmatrix} C_{ee}, & C_{e\beta} \\ C_{e\beta}, & C_{\beta\beta} \end{bmatrix} \cdot \begin{bmatrix} \partial e_i / \partial t_{es} \\ \partial \beta_i / \partial t_{es} \end{bmatrix} = \begin{bmatrix} -b_{is} \\ 0 \end{bmatrix} \qquad \ldots(4.77),$$

where C_{ee} represents the second order partial derivative of cost function with respect to effluent discharge. The abatement cost functions are assumed to have increasing marginal cost, i.e. $C_{ee}, C_{e\beta}, C_{\beta\beta} \geq 0$.

When $|D| = C_{ee} \cdot C_{\beta\beta} - C_{e\beta}^2 > 0$, by applying the Cramer's rule, it can be shown that

$$\frac{\partial e_i}{\partial t_{es}} = \frac{\begin{bmatrix} -b_{is}, & C_{e\beta} \\ 0 & C_{\beta\beta} \end{bmatrix}}{|D|} \leq 0,$$

$$\frac{\partial \beta_i}{\partial t_{es}} = \frac{\begin{bmatrix} C_{ee} & -b_{is}, \\ C_{e\beta} & 0 \end{bmatrix}}{|D|} \geq 0,$$ if $|D| > 0$, i.e. increasing the effluent tax rate reduces the effluent discharge, and on the other hand increases the water abstraction, because water abstraction now becomes a cheaper option of pollution comparing with effluent discharge, so the firm will increase the effort in effluent abatement but pay less effort in reducing water abstraction.

Similarly, the effects of water abstraction tax on the pollution activities can be obtained from

$$\begin{bmatrix} C_{ee}, & C_{e\beta} \\ C_{e\beta}, & C_{\beta\beta} \end{bmatrix} \cdot \begin{bmatrix} \partial e_i / \partial t_{as} \\ \partial \beta_i / \partial t_{as} \end{bmatrix} = \begin{bmatrix} 0 \\ -d_{is} \end{bmatrix} \qquad \ldots(4.78).$$

$$\frac{\partial e_i}{\partial t_{as}} = \frac{\begin{bmatrix} 0, & C_{e\beta} \\ -d_{is} & C_{\beta\beta} \end{bmatrix}}{|D|} \geq 0,$$

$$\frac{\partial \beta_i}{\partial t_{as}} = \frac{\begin{bmatrix} C_{ee} & 0, \\ C_{e\beta} & -d_{is} \end{bmatrix}}{|D|} \le 0 .$$

Therefore, it can be shown analytically that when the tax rate on one type of pollution (either effluent discharge or water abstraction) increases, the corresponding activity will be restrained by the more potent policy whereas the other activity will become a relatively "cheaper" option and hence will increase, assuming the tax on the "other" activity remains unchanged.

4.4.2 Steady State Comparative Statics in the Dynamic System

In the steady state equilibrium of dynamic optimisation, $\dot{I}_i^j = \dot{k}_i^j = 0$ as described in Eqs (4.56) and (4.68) so we have

$$-C_i''(\cdot)^{-1} \cdot \left[(r+\delta_i^j)\mu_i^j - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} - \sum_s \left(t_{es} \cdot b_{is} \cdot \frac{\partial e_i^*}{\partial k_i^j} + t_{as} \cdot d_{is} \cdot \frac{\partial \beta_i^*}{\partial k_i^j} \right) \right] = 0$$

...(4.79),

$$I_i^{j*} = \delta_i^j k_i^{j*} \qquad \text{...(4.80),}$$

$j = q, a, \beta$.

As $C_i''(\cdot)^{-1} \neq 0$, (4.79) can be reduced to

$$(r+\delta_i^j)\mu_i^j - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} - \sum_s \left(t_{es} \cdot b_{is} \cdot \frac{\partial e_i^*}{\partial k_i^j} + t_{as} \cdot d_{is} \cdot \frac{\partial \beta_i^*}{\partial k_i^j} \right) = 0 \qquad \text{...(4.81).}$$

Since $\delta_i^j > 0$ (all capital stock will depreciate), it will be concluded from Eq (4.80) that at the steady state equilibrium, capital stocks and investment in each element always move to the same direction when there is perturbation. Applying the implicit function theorem to Eq (4.79), gives

$$\frac{\partial k_i^j}{\partial t_{es}} = -\frac{\partial f/\partial t_{es}}{\partial f/\partial k_i^j} = -\frac{b_{is}\dfrac{\partial e_i^*}{\partial k_i^j}}{(\dfrac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j2}} + \sum_s t_{es} b_{is} \dfrac{\partial^2 e_i^*}{\partial k_i^{j2}} + \sum_s t_{as} d_{is} \dfrac{\partial^2 \beta_i^*}{\partial k_i^{j2}})} \quad \text{...(4.82),}$$

where f denotes the LHS of the Eq (4.81).

Having assumed the relationship between water quality and each element capital stocks in the stability analysis in 4.3.3, it can be concluded consequently that $\frac{\partial^2 e_i^*}{\partial k_i^{j2}} \geq 0, \forall j$, so the denominator of Eq (4.82) must be positive. Therefore, the overall sign of Eq (4.82) is determined by the numerator, therefore $\frac{\partial k_i^q}{\partial t_{es}} < 0$ and $\frac{\partial k_i^a}{\partial t_{es}} > 0$. For the same reason discussed in the short-run comparative analysis, there is $\frac{\partial k_i^\beta}{\partial t_{es}} > 0$. The change in effluent discharge tax rate has the same effect on the investment as it has on the capital stocks in each element. We can derive the effects of change in the tax rate of water abstraction on these variables through the same process. The results of comparative statics on the TPP system are very similar to those of TSS (Xepapadeas 1997), which are summarized in Table 4.1 and 4.2.

Table 4.1: Short-run comparative statics

	e_i	β_i
t_{es}	-	+
t_{as}	+	-
P_{es}[7]	-	+
P_{as}	+	-

[7] The equilibrium prices of pollution permits are not determined by the environment authority but the permit market, however the environment authority can raise (decrease) them by increase (decrease) the amount of pollution permits it distributes.

Table 4.2: Steady State Comparative Statics

	k_i^q	k_i^a	k_i^β	I_i^q	I_i^a	I_i^β
t_{es}	-	+	+	-	+	+
t_{as}	uncertain	uncertain	-	uncertain	uncertain	-
P_{es}	-	+	+	-	+	+
P_{as}	uncertain	uncertain	-	uncertain	uncertain	-

4.5 Conclusion

4.5.1 Theoretical Analyses

The choice of policy instruments in pollution control has been discussed in the literature for considerable time and most economists have agreed that, although there are still some limitations in implementing policies successfully into practical environmental management, the MBIs have superior properties to the direct command approach. The superiority of MBIs arises for several reasons, including cost savings realised in achieving the environment target and the continuous motivation provided to undertake the pollution control. In this thesis, taking into account both emission discharges and water abstraction as forms of pollution, I will explain why and how MBIs could be implemented to manage water quality in the tidal Ouse and also explain the potential benefits MBIs offer for river policies in water quality management, in situations when the actual location of emissions and abstraction matters.

The static and dynamic analyses of preceding sections have indicated the necessary conditions for the least cost solution of pollution control problem. Several conclusions can be drawn from the necessary conditions which arise from the static and dynamic analyses:

1. Because of the different location effects of pollution, a matrix of transfer coefficients needs to be produced by the environmental authority, in order to assign responsibility for pollution at each WQM site back to the pollution

source depending on its location. Therefore the pollution equilibrium for the least cost solution will vary from place to place, and time to time as these transfer coefficients vary following the variation of assimilative capacity in the river.

2. Static analysis has shown that when the least cost equilibrium is achieved, the ratio between the marginal cost of abatement and the marginal effect of abatement on water quality should be the same across all the options at all the binding WQM sites. This ratio will reflect the shadow price of river water quality at each WQM site at the cost effective equilibrium.
3. When the dynamic optimisation is considered with a positive discount rate, in addition to the first order conditions required by the static analysis, the least cost solution of pollution abatement in a dynamic system requires that at steady state equilibrium, investment in capital stock should be such that: the internal rate of return on that investment will be the same as the external rate of return which could be achieved by investment elsewhere in the economy.
4. The steady state equilibrium for the capital stock and investment is a saddle point equilibrium. The combination decisions on capital stock and investment must therefore follow a particular trajectory in order that the least cost solutions in each period of time will eventually lead to a stable equilibrium of the dynamic system. Since the capital stock is determined by investment, the industry must find this temporal investment path in order to achieve the stable equilibrium of the dynamic system.

4.5.2 Policy Implications

The primary objective of this research is to analyse the potential inefficiency that exists in current river policy as applied to the Humber system, and to illustrate that pollution control at reduced cost could be realised more easily through the use of integrated river policy and MBIs rather than direct CAC approach alone. From the results of both static and dynamic analyses, it is apparent that implementing an MBI would bring substantial cost savings in pollution control, either TSS or TPP system. This is mainly due to the flexibility they offer to pollution sources in achieving their pollution reduction targets. The flexibility property is particularly crucial when there is significant spatial variation in the impacts of pollution location on the river water quality. When water abstraction is also taken into account as a form of "pollution", the flexibility in these two MBIs allows integrated management of both effluent abatement and water abstraction. In this integrated management, effluent discharge and water abstraction are treated as substitutes for each other, therefore each plant is encouraged to find the least cost combination of the two options.

The environmental policy imposed on the pollution source seeks to induce them to approach the equilibrium which produces the desired environmental target at a least cost. Choices between the direct command approach and MBIs are analysed in static and dynamic situation. However, it is apparent that the choice of option depends, as we can see in reality, on the pertinent circumstances: the nature of the pollutant and its geographical conditions, and on various political and administrative considerations (Baumol and Oates 1988). Therefore, there is no panacea for all situations; each environmental issue needs to be considered on an individual basis. Nonetheless, our static and dynamic analyses remain valid in general, with the following implications for environmental policies:

1. It is almost impossible to achieve the least-cost solution in practice by a direct command approach. Market-based instruments are cost-effective themselves for any pollution control they achieve, but the environmental authority has to design them to carefully achieve the prescribed target.
2. A tax-subsidy scheme may incur less criticism than a simple emission charging scheme because it does not impose extra financial burden on plants and thus weaken their competitiveness. Neither would such a scheme increase the overall pollution level as what an abatement subsidy usually does. The choice between an effluent permit system and ambient permit system depends on many factors including how important location effects are in practice. A hybrid pollution-offset system may well be a means of differentiating the location effects among pollution sources without imposing too many impediments to the trading process.

3. In a dynamic system, a firm can decide its own time path of investment to accumulate its capital stocks which underpins production, abatement and abstraction. Hence these investment decisions can cause the firm to diverge from the optimal outcome for river water quality to different destinies. The direct command approach has limited influence on the firm's investment choices whereas comparative statics analysis shows that the firm's investment paths could be altered by a tax-subsidy scheme or a tradable pollution permit system.
4. Although a tax-subsidy scheme and a tradable pollution permit system have equivalent effects on the pollution control, the tax rate and permit price at the steady state equilibrium in a dynamic system are no longer identical as was the case in the static analysis. Through the purchase of one unit pollution permit the firm saves an infinite stream of tax payment in the future and therefore must pay a price for the permit equal to the present-value of the aggregate tax payment over this infinite time horizon, which equals to the tax rate divided by the interest rate.
5. A novel feature of my research is the integrated spatially explicit management of effluent discharge and water abstraction implemented through the use of different environmental policy instruments. Thinking of the interdependence between these two activities, environmental policies are proposed which will reduce their combined effects on river water quality rather than managing effluent discharge and water abstraction separately. Therefore, any environmental policy directed towards one activity would have a consequent impact on the other activity at the same time, as indicated by comparative statics analysis. This integration would enable environmental policy to reflect the interdependent effects on the water quality from both effluent discharge and water abstraction It would also provide flexibility in pollution control options which could allow substantial cost saving to be achieved in achieving the prescribed environmental targets.

Chapter 5 Data Sources and Methodology

5.1 Introduction

This chapter describes the data sources that were used to evaluate the cost effectiveness of alternative management actions for pollution abatement and water quality control in the tidal Ouse. Two, essentially separate, data sets were used.

Hydrological data for the tidal Ouse comprise the first data set. These data are supplied to the QUESTS1D river model to evaluate the improvement in river water quality which would be produced by alternative pollution abatement strategies. These hydrological data comprise, for the tidal Ouse and its tributaries:

- river flows;
- river water quality;
- ambient concentrations of water-borne substances;
- effluent discharges from STWs, local industries and other sources of emission.

The structure of the river system is embedded within the structure of the QUESTS1D model.

The second data set details economic aspects of the different options for pollution abatement. A comparison of cost effectiveness requires that the costs incurred by the different pollution abatement options are evaluated and then compared with the pollution reductions predicted by the QUESTS1D model for each abatement option. Abatement options are not regarded as static in this analysis. Data describing the capital cost of improving abatement capability are therefore required, as well as data describing the operating and depreciation costs of abatement facilities.

Modifications to effluent discharges and to water abstractions are evaluated in this research to investigate the potential which an integrated discharge and abstraction strategy offers for the management of water quality in the tidal Ouse. The interdependencies between discharges and abstractions must be accounted for appropriately within the hydrological model, and within the economic calculations. The QUEST model can accommodate water abstraction at specified locations. Economic data detailing the operating and depreciation costs of abstraction facilities are also required.

The capital cost of abatement facilities, and the operating and depreciation cost of abatement and abstraction equipment comprise the direct costs of water quality management.

This chapter comprises three further sections. The first describes the sources of the hydrological data used by the QUEST model, and explains how these data are pre-processed for utilisation by the model. The next section describes how questionnaires and interviews were used to obtain economic data. The processing of these data to provide the marginal costs of abatement for each management option is also described. The final section describes changes in the statutory consents for effluent discharges from industrial sources and STWs over the past 10 years. Water abstraction licences over the same period are also detailed.

5.2 Dataset for QUESTS modelling

5.2.1 Data requirement for QUESTS modelling

The QUESTS1D model is a one-dimensional representation of the tidal river system stretching from tidal limits of the Ouse, Wharfe, Aire, Don and Trent to the sea spurn. To utilize the model to simulate water quality under alternative pollution abatement options, the following data for the tidal Ouse and its tributaries are necessary:

- The river flows of the tidal Ouse and Trent, and their tributaries including the Wharfe, Derwent, Aire, Don and Hull.

- The concentration of various water-borne substances at the headwaters of each of the river reaches mentioned above.
- The effluent discharges from the major STWs, industries and other emission sources along the tidal Ouse, Trent, Humber and their tributaries, in terms of effluent flows and pollutant concentrations, such as BOD_5, NH_3 and suspended solids.

These data enable the QUESTS1D model to simulate hydrodynamics and water quality in the tidal Ouse, Trent and Humber system. Data regarding river flows and concentrations of the water-borne substances, particularly DO, BOD_5, NH_3 and suspended solids were recorded at six WQM sites along tidal Ouse (Naburn, Cawood, Selby, Long Drax, Boothferry Bridge and Blacktoft).

The data are compared with the simulated results to validate the predictions from the model. To evaluate the effects of water abstraction on the water quality, the following data are also required:

- The water abstraction levels in the tidal Ouse catchment
- The pattern and effect of water returns to the Ouse system.

5.2.2 Data sources for QUESTS

The EA provided most of the hydrological data from their routine surveys and monitoring, as well as the QUESTS1D model. The data from the EA was divided into three categories.

The first category comprised data embedded within the QUESTS1D model, or supplied to fulfil the data requirements for simulation. These data were obtained from routine sampling at the effluent sources and gauge stations along the tidal Ouse, Trent, Humber and their tributaries. Self-monitoring data from each effluent source are provided as checking data.

The data input at the river boundaries comprise the river flow of the tidal Ouse, Trent, Wharfe, Derwent, Aire and Don from 1993 to 2004, and concentrations of eight major water-borne substances at the headwater of these rivers from 1995 to 2004. The effluent data cover fifty point sources of effluent discharges, including the industries, STWs and other sources along the Ouse, Trent, Humber and their tributaries. It also covers various periods for the different point sources, but each contains data for the years 1995 to 2002, with exception of 1998. The QUESTS1D model simulation therefore covered the period of 1995-2002 to evaluate water quality under various pollution abatement options.

The second data category comprises the observed water quality data from the WQM sites along the river system. These data include all the WQM sites and a list of water-borne substances from the EA's routine sampling at different frequencies. This dataset stretches from 1994 to 2004 and is therefore used to validate the simulated results from the QUESTS1D model. The data from the LOIS Ouse dataset served the same purpose in validating the simulation results.

The third category of data details water abstraction from the tidal Ouse. The major water abstraction from the Ouse occurs at Moor Monkton, and from the River Derwent at Barmby and Elvington. Data detailing water abstraction licenses and actual water abstractions were only available for the years 1996, 1997, 2003 and 2004. Significant water is also lost from the tidal Ouse at Drax, where the Drax Power Station takes the river water for cooling and approximately half of it evaporates during the process. Only limited data are available to describe the amount of water lost and the temperature change in the returned water.

5.2.3 Hydrological data processing for QUESTS

The EA dataset for effluent discharge was incomplete. Some effluent discharges from the Selby industries after 2000 are missing. In order to evaluate the DO saturation with the possibility of improved discharge consents in Selby after 2000, two separate datasets were constructed for 2001 and 2002. One assumes the same level of effluent discharges as in previous years; the other is updated to allow for "future" effluent discharge consents, which are likely to be implemented by the EA (Cashman *et al.* 1999).

The data for validation of the simulation covered 1995 to 2004. However, data for 1998, 2003 and 2004 are insufficient or incomplete. Therefore other data elements are missing from the dataset. These issues were solved as follows:

1. Salinity data are estimated for the whole period. Salinity is defined as the mass of dissolved inorganic compounds in 1 kg of seawater, after all the bromide and iodide has been converted to chloride, and all carbonates converted to oxides. This can be calculated from the chlorinity following the Knudsen equation: $S‰ = 0.030 + 1.8050 \times \mathrm{Cl}\,(\mathrm{g\,Cl/l}) \times 1/\mathrm{P}$, where P is the density of seawater at that chlorinity. Since the river and tributaries upstream from the confluence of the Ouse and Trent are regarded as fresh water, P is same as the fresh water density 1000g/l.
2. Data detailing total phosphorus content are not available for these years. It is suggested that orthophosphorus, for which data are available, comprises approximately 80% of total phosphorus content for all the inputs, including riverhead water along the River Ouse (pers. comm. Trevor Hardy; Environment Agency). Therefore, in this research total phosphorus data are calculated accordingly from orthophosphorus.
3. Data detailing effluent discharges from BOCM and TLCA are missing. For both of the industry plants, effluent discharges before 2000 were assumed to be of the same level as reported in 1996 and 1997 (Cashman *et al.* 1999). Effluent discharges since 2000 were assumed to match the Environmental Agency's anticipated "future" consents described above. This is reasonable because since 2000 TLCA has managed to reduce its effluent discharge below the "future" consent, although the consent was not in force at that time. Effluent discharge from BOCM was assumed to be the same as that of 1996 and 1997 that reported by Cashman *et al.*(1999).

As the dynamic model of river water quality in order to generate simulation results at much higher frequency than the routine sampling, the data from the river boundaries and effluent sources have to be processed to produce a daily dataset. A statistical program called SHARE within the QUESTS1D under the Test Data Facility (Ellis *et al.* 1992; Clark and Ellis 1993; Slade and Morgan 1993a) is used to generate a rough description of input data. More than 20 samples within three consecutive years period are required for this process. After the SHARE program, the routine sampling data of both river boundaries and effluent inputs are extracted as a line of rough descriptors that describe the main statistical characteristics of the samplings. The rough descriptors from all the river boundaries and the fifty point sources of effluent then constitute aggregate descriptors, which are reconstructed into separate daily estimations of inputs for that year by the SYNTH program. A program called COLLATE then generates an auxiliary file from the results of SYNTH, which contains a time series of daily input data. This file is then fed into the QUESTS1D model for simulation.

The simulations of the QUESTS1D model provided good predictions of the water quality subject to changes in the level, location and timing of effluent discharges and water abstraction. A simplified function describing the influence of pollution abatement and water abstraction on the water quality is required in order to optimise the level and location of these activities. This simplified water quality function is derived from repetitive simulations of the QUESTS1D model, and therefore retains the accuracy of prediction of the model simulations. All the variables not subjected to change were regarded as constant, and adjacent effluent sources were aggregated to reduce the number of independent variables. This is valid provided that they are of similar composition and the transfer coefficient between the aggregated sources is sufficiently high. The DO saturations at three WQM sites, which usually suffers from DO sag, are regressed against one set of effluents and water flow, using the "System of Regression Equations" provided by Limdep (Econometric Software Inc. 1995). The dataset of effluents and water flow could predict the EWPCS composite score along the Ouse/Humber reaches or over the whole estuary catchment as well, to provide a more comprehensive constraint instead of the DO saturations at several discrete points.

5.3 Economic dataset

5.3.1 Economic data requirement

For evaluating the cost of pollution abatement at the effluent sources (industries and STWs), the capital cost and operational cost incurred in pollution abatement are required. Pollution abatement would be implemented by investing in new plant (increasing capital cost) or operating existing plant more intensively (increase in operational cost). It is also necessary to know the specific effluent discharge consents for each of the major effluent sources. Alternative pollution abatement options such as moving discharge locations or shifting discharge timing may require investment in additional capital stock such as tanks or pipes. Tradable water abstraction licences and the associated cost incurred by increasing or decreasing water abstraction level will also be considered in this research. The required economic data are therefore as follows:

a. The capital and operational costs of pollution abatement undertaken at the effluent sources to comply with the EA's effluent discharge consents.
b. The capital and operational costs of alternative pollution abatement strategies considered in the research.
c. The capital and operational costs associated with increasing or decreasing water abstraction in the catchment.
d. Any trading transactions of water abstraction licenses, including the price, quantity traded, and the cost incurred in the bargaining process.
e. Alternative technology or management introduced to reduce the effects of effluent discharges on the tidal Ouse or to improve the DO saturation in the river water.
f. Any commitments required by regulation and policy. Any fines and penalties for non-compliance.
g. The indirect costs of pollution abatement from each pollution abatement option, particularly on the local economy of Selby.

5.3.2 Economic data sources

Cost data for pollution abatement were provided by the three industrial plants at Selby for varying levels of effluent discharge and abatement. BOCM has now modified its production process and no longer discharges effluent to the Ouse. Yorkshire Water (YW) supplied cost data for the STWs considered in this research. STWs included were Barlby and Selby on the River Ouse, Snaith on the

River Aire, and Sandall and Thorne on the River Don. Water was abstracted at Moor Monkton on the River Ouse, and at Elvington and Barmby on the River Derwent, all beyond the tidal limits of the other rivers. Abstraction costs and details of abstraction licenses were obtained from YW and the EA.

Questionnaires and interviews with the managers of the industries in Selby, YW, and the EA were undertaken during summer 2005 to collect the economic data required for the research. Data detailing the cost of piped transfer of effluent and waste storage facilities were obtained from a report published by Ofwat (1999).

The following data were obtained from the questionnaire to the Selby industries:

a. Output levels of main product and changes in the last few years following modification of effluent discharge consents.
b. Current effluent discharge levels and changes required to comply with the new consents.
c. Changes in the effluent discharge consents granted by the EA.
d. Effluent treatment facilities implemented in the plants.
e. Capital and operational costs of the ETP in each industry, capital investment in the ETP, life expectancy and depreciation rate of capital stock, the changes of capital and operational costs in the plant to meet the EA's revised effluent discharge consents.
f. The physical effectiveness of ETP in terms of removal BOD_5 and other pollutants, past and current.
g. Rough estimate of the total production costs.

Similar questions were posed in the questionnaire to YW regarding its STWs and the water abstraction activities. Data required were:

a. Changes in the effluent discharge consents in the STWs as a consequence of the Urban Waste Water Treatment Directive (UWWTD) Regulation.
b. Current effluent discharge levels and changes required to comply with the UWWTD Regulation.

c. Capital and operational costs of sewage treatment.
d. Capital investment in the sewerage facilities, life expectancy and depreciation rate of capital stock, the changes of capital and operational costs as a consequence of the implementation of UWWTD.
e. The physical effectiveness of sewage treatment in terms of removing BOD_5 and other pollutants, past and current.
f. The levels of water abstraction from the River Derwent and the River Ouse, along with the water abstraction licenses held at these abstraction locations;
g. Capital and operational costs of water abstraction estimated by YW.

The cost to Selby industries and the major STWs of reducing the level of BOD_5 and other pollutants was estimated from the data described above. The costs which YW incurs in water abstraction were also estimated. The cost of capital investment in abatement facilities was also estimated for use in the dynamic economic model of water quality management. Views expressed by YW and Selby industries were taken into account when developing alternative options for water quality management in the research to ensure that the options considered were feasible and practical.

5.3.3 Economic data processing

The availability and details of economic data are restricted by confidentialities. It was therefore necessary to aggregate the cost data from all the industrial plants concerned into a single dataset for the industries. This inevitably introduces some deviation from reality, which however is considered acceptable because the plants all use anaerobic abatement facilities to reduce the load of BOD_5 in their effluent. The dataset also utilised cost data collected by Cashman *et al.*(1999). Cost data were also aggregated from the five STWs to estimate a cost function for their pollution abatement. The five STWs operated by YW provided more comprehensive and detailed data than the industries, although operational costs in each STW only covered three years since 2000.

The situation regarding water abstraction is more difficult. Because of its legal duty to provide potable water to the consumers in the catchment, YW has to maintain sufficient water abstraction to satisfy demand. YW would therefore be required to transfer water from other river systems if the total water abstraction within the Ouse catchment were to be reduced. YW has several contingency plans, so called "resource solutions", to meet any shortfall in water supply if water abstraction from the tidal Ouse catchment becomes insufficient. The cost of "resource solutions" allowed for estimating the cost of reduced water abstraction.

All the cost data are defined as the summation of operational cost and the depreciation of the capital stock throughout the year, inflated to 2004/05 prices using GDP deflator (HM Treasury). Capital depreciation differed between the industries in Selby and the assets of YW including STW plants and water abstraction facilities. Industries in Selby all assume an average life expectancy of 10 years for their ETPs. The capital value of ETPs in these industries thus depreciates at 10% of the remaining value each year. YW however assumes an average life expectancy of 40 years for its assets, and a depreciation rate of 2.5% net down on the original capital value is therefore used. Most of the STWs and water abstraction facilities were built during the 1960s and 1970s. Sandall STW, however, was built in 1947, while the Selby STW and water abstraction facilities at Moor Monkton were built much more recently in 1999 and 1996 respectively. This wide age range leads to the use of a constant depreciation rate for capital equipment in STWs and abstraction facilities which is much lower than that applied to ETPs in the industries. Applying 10% depreciation rate to the recently built plants would produce very high depreciation values, out of line with the old plants which have gradually depreciated for over 30 years. As a result, the much higher capital depreciation would overshadow the operational cost in these new plants, leading to overvalued abatement costs.

Once the costs of BOD_5 removal in the industries and STWs are revealed, a cost function of pollution abatement for BOD_5 removal in both industries and STWs can be estimated using regression techniques. The estimated cost functions for abatement and abstraction produced by this approach are smooth curves. The cost and marginal cost curves of pollution abatement and water abstraction in reality are more likely to be stepwise. The use of smoothed cost functions is not intended to suggest that actual abatement cost at a particular site would follow such a curve, but rather that the curve provides an approximation across the relevant industry as data allows. Similarly, a cost function for abstraction can be estimated by regression once the costs, which would be incurred by changing water abstraction levels, are known.

5.3.4 Opinion of new management options

The questionnaire sent to the industries in Selby enquired their opinion of alternative options for river management, including moving the location and timing of effluent discharge. The responses received reflected their priority and capability for pollution abatement, and proved helpful in assessing the acceptability of proposed options to improve water quality.

Three completely different responses were received from the Selby industries regarding their willingness to be involved in a permit market for effluent discharges to the tidal Ouse. Rigid Paper said "no" to this option as a middle-sized source of effluent. This suggests that it does not want to be bothered by the market instrument or to expend more effort to reduce the pollution load further, particularly considering the fact that its consents, and actual discharge, of BOD_5 has been increased significantly since 2002. TLCA, as the biggest source of BOD_5 in the Selby area replied "don't know" to this question, because it has the ability to reduce the load of BOD_5 at lower price than a plant of a smaller scale, but it might find it difficult to locate a buyer for substantial discharge permits in the catchment. Greencore, however, would welcome the introduction of a TPP system. Greencore has the smallest BOD_5 discharge consent among the three industries, making it more likely to buy discharge permit rather than sell them. Greencore supports a TPP system for river quality management partly because it already participates in an emissions trading scheme relating to greenhouse gas emissions.

Shifting effluent discharges from summer to winter was clearly rejected by TLCA and Rigid Paper, but again welcomed by Greencore. Greencore's support for season-dependent effluent discharge probably arises because its production is seasonal, subject to provision of raw material, and so is its discharge. Greencore had previously different levels of effluent discharge for summer and winter, but these have now been replaced by a uniform consent over the year. The other two companies rejected this option for various reasons, but the fact that both maintain constant production output over the year was clearly influential. Effluent storage required for this option is also an obstacle to acceptance.

None of the three companies showed any interest in moving the location of effluent discharges. Their major concern is the cost of laying pipes necessary to relocate discharges downstream. A rough estimate of £2 million pounds for 10 miles of pipes was given by Greencore, which believes this cost to be prohibitive. However, shared costs for a common pipe transporting effluent from the three plants and two nearby STWs would potentially reduce the individual cost for each plant substantially.

5.4 Effluent consents and water abstraction license

As mentioned previously, there have been some changes to the effluent consents of industries in Selby and the STWs in the catchment, subject to either PPC regulation or UWWTD. Effluent consents take different forms. Consents for BOD_5 are specified as either concentration (mg/L) or as flows (tonnes/day). Consents are also applied to total effluent flow (m^3/day). TLCA and Rigid Paper had their effluent consents reviewed under the PPC regulation in 2004 and 2002 respectively. Greencore's consent was not changed under the PPC regulation, but was amended separately by the EA. BOCM has now ceased effluent discharge to the tidal Ouse permanently. Among the five STWs considered in this research, Barlby is not yet regulated by the UWWTD due to its small size, and it is the only one with just primary treatment to the inlet effluent before discharge. The other four STWs all apply secondary treatment to the effluent using slightly different technologies. An upgrade of the Barlby STW to secondary treatment is being carried out currently. The four STWs that are subject to the UWWTD are of different scales. Selby and Thorne, as major STWs serving an agglomeration with a population equivalent of more than 15,000, were required to comply with the UWWTD by 31st December 2000. Snaith and Sandall are smaller, serving an agglomeration with a population equivalent of between 10,000 and 15,000. They were required to implement the UWWTD by 31st December 2005, but in fact have been compliant since early 2000s.

Effluent consents for TLCA have been continuously reduced from 13.89 tonnes/day in 1989 (Cashman *et al.* 1999) to 8 tonnes/day in 1994 for BOD_5, and to currently less than 3 tonnes/day. Total flow has been reduced from 15000 m^3/day to 9999 m^3/day since 1994, while its production increased. Rigid Paper has however seen an increase in its effluent consents since 1995 after the PPC procedure, when its flow consent increased from 1250 m^3/day to 1400 m^3/day in average, with a maximum of 2500 m^3/day. Its BOD_5 consent increased from 3.3 tonnes/day to 4950 mg/L at maximum, which equivalent to 6.9 tonnes/day for mean flow at 1400 m^3/day, and a possible maximum of 12.4 tonnes/day with maximum flow. Greencore expanded its flow consent recently from 1000 m^3/day since 1990s to 5500 m^3/day after June 2004. At the same time, the consent for BOD_5 discharge has been reduced significantly. Greencore used to have two separate consents seasonal for BOD_5, being 1 tonne/day between May to September and 2.5 tonnes/day between October to April. A uniform BOD5 consent of 0.75 tonne/day has now replaced them.

Changes to the effluent consents for the STWs are less substantial. Effluent flows from STWs have gradually reduced over the last ten years, but Snaith has increased its flow four-fold from 570 m^3/day to 2140 m^3/day. Improvement in the STWs on the rivers Aire and Don have been beneficial to the water quality of these tributaries, but STWs at Barlby and Selby are still partly contributing to the DO sag around Selby in the tidal Ouse.

The data detailing tradable water abstraction licenses are limited. No trade has taken place so far. YW is the major water company in the Humber catchment and holds the majority of water abstraction licenses. Licensed water abstraction by

YW is 82,500 thousand cubic meter per year (tcma) at Elvington and 33,440 tcma at Barmby on the River Derwent, 35,000 tcma at Acomb and 73,000 tcma at Moor Monkton on the River Ouse. Water abstraction varies over time, but is usually well below the amount granted by the licenses. There is also significant water uptake by Drax Power Station for cooling water. Approximately half of the volume abstracted is returned to the river after use. Water abstraction at Drax has been included in the water quality simulations produced by the QUESTS1D model, although no actual data are available to verify reported abstraction and return levels. Water abstraction from the Ouse for irrigation and agriculture is negligible in comparison with abstraction for potable use. Therefore a Business As Usual (BAU) approach is adopted for agriculture abstractions, and assuming no change for the purpose of this research.

Chapter 6 Static Optimisation

6.1 Introduction

This chapter describes the methods and results of the static cost minimization model. In this model the system of water quality functions is combined with economic cost functions of various options to identify the strategy that complies with water quality target at least cost.

For all the analyses below, the simulation results of QUESTS1D model are used to generate the system of water quality functions that can predict the water qualities at the EA's WQM sites. The system of water quality functions consists of five functions for five different cells in the QUESTS1D model around three WQM sites, which are likely to experience severe DO sag issue during low flow summer. In this research, the water qualities of cell 180 at Selby, cell 192 and 193 at Long Drax, and cell 197 and 199 at Boothferry Bridge were predicted through the system of water quality functions based on iterative simulations of QUESTS1D model, in term of 5%ile DO%. The functions predict the water qualities at these cells using the most significant determinants of water quality at each point. These determinants are the effluent discharge levels from various sources, water abstraction from river Ouse and tributary, and the effluent discharge locations. Since the predictions through the system of water quality functions were quite consistent with the simulation results of the QUEST1D model, the system is used to represent the water quality constraint in the optimisation model. The associated costs of these options are estimated using the data provided, and their sum is minimized subject to the achievement of given water quality targets.

The static optimisation is solved using the General Algebraic Modelling System (GAMS), and the sensitivity of the outcome to the assumptions made was tested over a range of scenarios. The least cost solution indicates the combination of effluent abatement levels in individual sources, water abstraction from the rivers, and where to discharge the effluent along the river Ouse. Despite differences among the optimal solutions for different scenarios, relocating the effluent discharges proved to be the most cost effective measure. With effluents from Selby area being discharged downstream of the river Ouse, water quality along the river Ouse could be significantly improved even in low flow conditions (as in 1996), but at much less cost than would otherwise occur. The feasibility of the least cost solution is also discussed, particularly in the light of Europe's Urban Waste Water Treatment Directive (UWWTD).

This chapter tests the sensitivity of the results to the assumptions of the cost function of effluent abatement. The least cost solution proved to be insensitive to the change. The sensitivity test also considered different water quality targets in the tidal Ouse. We find that under the low flow conditions, the options considered in the optimisation analysis are not able to achieve 5%ile DO% target higher than 40% in the tidal Ouse. One implication of this is that the establishment of water quality targets appropriate to flow conditions needs to be considered to avoid imposing excessive costs.

There are five sections in the chapter. The second section describes the constraint and objective functions of the static optimisation problem. It shows how the system of water quality functions is derived from simulations results of QUESTS1D model and why this is necessary. It also details estimation of cost functions for various options of improving water quality of river Ouse. The third section displays the results of static optimisation under different scenarios. The combinations of actions that satisfy the water quality target at the least cost are calculated through GAMS. This section then discusses the feasibility of the least cost solution in reality. A fourth section discusses the results of the sensitivity tests. Different cost functions of effluent abatement within the industries and STWs are applied to investigate the possible change in the optimal solution. It also discusses the reason for the consistent optimal solution. Different water quality targets for the tidal Ouse together are also considered. The last section summarises the outcome of the optimisation analysis and points to the policy implications of the outcome (to be addressed in the following chapters).

6.2 Constraint and Objective Functions of Static Optimisation

6.2.1 Constraints: the System of Water Quality Functions

In this research, water quality along the river Ouse is treated as a constraint that needs to be satisfied through river management. The DO% of the river water was chosen as indicator of water quality. The EA monitors water quality at several WQM sites along the river using a series of indicators including DO%. The DO% is directly linked with the DO sag issue in the river Ouse. Since the EA only monitors at certain sites, the optimisation only includes constraints at these sites. Nevertheless, the water quality of the whole river system was checked afterwards against the same constraints to ensure compliance at each point along the river.

The QUESTS1D model is a comprehensive dynamic water quality model, which takes into account many influencing factors. This is very useful to assess the impacts on river water quality of some particular management options when the change is known and manageable in the model. By simply changing the parameter values or structure of the model, new management options can be easily assessed and modified according to the outcome of simulations. However, this kind of "black-box" feature becomes less convenient if one wishes to find out the best solution for a river that is unknown beforehand. It is used here to identify the most cost effective river management option for particular environmental targets, which need to combine the cost function of pollution abatement with the effects of pollution on river water quality as a function of various abatement levels. A simplified system of functions for water quality under different scenarios is used to identify the combined option without knowing it beforehand.

In order to reduce the modelling work, the simplified system of water quality functions predict water quality at specific points instead of along the whole river length, but the predicted results from the simplified functions are checked against the simulation of QUESTS1D model to ensure compliance at all points. Five points around the EA WQM sites where DO sag issue is likely to occur during the summer are chosen. The functions are derived from repetitive simulations of the QUESTS1D model, and therefore reflect the predictive accuracy of the model. The most significant factors at each of the points were carefully chosen and assessed in different forms. All variables not subjected to change were treated as parameters and adjacent effluent sources were aggregated to reduce the number of independent variables. In the research, the effluent discharge around Selby area was aggregated as a single effluent source because (a) they are all discharging organic effluent of similar composition and with similar impact on the DO% of river water, and (b) they are located close to each other and the transfer coefficients between them are sufficiently high that their effluents can be treated as perfectly mixed. This is also helpful for the estimation of abatement costs discussed in the next section, as the abatement cost data do not allow abatement cost functions to be estimated individually for each source. The water quality predicted at each point is the 5%ile DO% of the cell in the QUESTS1D model, using the "System of Regression Equations[8]" provided by Limdep (Econometric Software Inc. 1995). The same dataset of effluents and water flow can also be used to predict the EWPCS composite score along the Ouse/Humber reaches or over the whole estuary catchment, providing a more comprehensive prediction instead of the DO saturations at several discrete points.

The simplified system of water quality function for the following points in 1996 is shown in table 6.1. The first column is the cell number of points predicted through the simplified system. Cell 180 is at the WQM site at Selby, cells 192 and 193 are located at Long Drax and cells 197 and 199 are at Boothferry Bridge. The WQM sites of Naburn Weir and Cawood were not regarded to be at risk as their DO% are more than 60% even in the worst conditions in 1996, therefore the water quality functions did not take into account these two sites. The same applies to Blacktoft, where water quality is dominated by the flow of Trent and is insensitive to various management options in the river Ouse. The water qualities at the five points are predicted simultaneously through this system of equations giving the 5%ile DO% of the cell.

[8] The regression results from System of Regression Equations will only be the same as equation by equation ordinary least squares if the estimators of each equation are the same and there is no linear restriction imposed. Otherwise, the results differ. In our case, the result will be different as three STWs on the tributaries are omitted from the first three equations of the regression system.

Table 6.1: Coefficients table of the system of water quality functions

Cell	Constant	X	X^2	ln(SBOD)	ln(Ouse)	ln(Derw)	ln(Sna)	ln(Sand)	ln(Tho)
180	-442.09	1.474	-0.042	-3.604	128.210	9.220	None	None	None
192	-113.406	-0.028	-0.020	-9.238	37.174	23.418	None	None	None
193	-79.943	-0.424	-0.011	-9.432	28.993	23.206	None	None	None
197	37.749	-1.552	0.019	-9.032	1.060	17.697	0.141	-0.228	0.085
199	42.566	-1.518	0.020	-8.922	-0.763	16.800	0.160	-0.261	0.098

The following nine variables are the estimators of water quality. The first is the constant. X in the second and third column is the distance from discharge location to the Trent Falls in kilometres. SBOD is the total tons of BOD_5 discharged from the sources around Selby per day. Ouse and Derwent are river flows (m^3s^{-1}) of rivers Ouse and Derwent while the flows of other tributaries remains unchanged. Sna, Sand and Tho are three different STWs in the tributaries Aire and Don, and have no effects on the first three points. The location of effluent discharges is best fitted to water quality as a quadratic function; improvement being quite slow when X is large (upstream) or small (downstream), but much faster in the mid-range of tidal Ouse. The effect of BOD_5 discharge on water quality is best described as logarithmic function, as is the effect of river flow. This is understandable as both factors have diminishing marginal impacts on water quality. See Appendix 3 for the details of the regression analysis.

The results obtained from the reduced system of water quality functions is in close agreement with simulations from the QUEST1D model, which has been carefully calibrated and validated against water quality observations throughout the years. It is therefore, reasonable to conclude that the simplified system of water quality function is reliable for purposes of determining the most cost effective river management option. However, it is important to point out that the results reflect the data on which the reduced system was calibrated. The system of water quality functions was estimated based on flow conditions in 1996, which is a dry year with high risk of DO sag. The functions, therefore, are best able to predict water quality under similar low flow conditions. A year with much higher flow such as 2002 would doubtless have higher assimilative capacity and therefore produce a different picture of the dynamics of water quality. Even under the similar flow conditions, the water quality function needs to be applied with caution. The effect of BOD_5 discharge on water quality has been estimated over a wide range of variation, but the same is not true for water abstraction. The effect of water abstraction was estimated around the actual levels, which were between 29 to 35 m^3s^{-1} for river Ouse and 7 to 12 m^3s^{-1} for river Derwent. For water abstraction much different to this, i.e. much higher or lower water flows from the two rivers, the water quality functions may not be as reliable. But for the purposes of this research, the variation of water abstraction is reasonable. The coefficient of Ouse flow was diminishing along the river, having less impact on downstream water quality. It becomes negative at cell 199, which is probably due to the fact that the small dilution impact at Boothferry Bridge was overwhelmed by the DO consumption from resuspended sediments caused by the flow. As seen, the coefficients of the STWs on the tributaries Aire and Don (Snaith, Sandall and Thorne) have no effects on water quality upstream of Drax, and have much less impact on water quality at Boothferry Bridge compared with BOD_5 sources in Selby. This is because they all locate in tributaries far from the river Ouse. While they do have an impact on water quality, they have fewer impacts on the water quality of river Ouse than on their own tributaries.

6.2.2 Objectives: Pollution Abatement Cost Functions

The objective is to achieve the least cost solution for the given water quality target in the river Ouse. Three different options for improving water quality were taken into account, and the aggregate cost of all three was minimised. These comprise the cost of effluent abatement within individual industry and STW, the cost of reducing water abstraction from rivers Ouse and Derwent, and the cost of moving effluent discharges along the river Ouse. The least cost solution involves a combination of the three options. All costs in the functions were annual costs of these options in million British Pounds (£m).

6.2.2a Cost function of effluent treatment in industries and STWs

One firm has ceased its effluent discharge to the river Ouse and it was reluctant to provide previous cost data. A cost function for effluent treatment for the Selby industries was estimated based on the data from the remaining major industrial sources of pollution in the town. Due to the confidentiality of much cost data, there are only seven observations available from three discrete years. In this research, two abatement cost functions were derived from the aggregated abatement cost data corresponding to the industries and STWs respectively. The Selby industries were treated as a single source in the water quality functions, as discussed in last section. Although the abatement technologies implemented in each industry are not exactly the same, all are based on similar methods of anaerobic treatment. The paucity of observations inevitably casts some doubt on the reliability of the estimation; however, the consistency of the results obtained from regression gives some confidence. Details of the regression result are shown in Appendices 4 and 5. A two-stage process was adopted in the treatment of the industrial and STW effluent abatement. The abatement cost functions were calculated from available data and used to optimise abatement levels from each cluster. Following this an analysis aws carried out to consider the efficient allocation of abatement between the individual industries and STWs. The first stage is discussed in terms of the static (Chapter 6) and dynamic (Chapter 7) optimisation models. Chapter 8 then discusses the allocation of abatement responsibilities among the sources.

The estimated cost function of effluent treatment for the industries in Selby took the following form, where a is the abatement level of the effluent treatment in each industrial plant in terms of tons of BOD_5 removal per day.

$$Cost_{ind} = 0.256e^{0.109a} \qquad \ldots(6.1),$$

The estimated cost was a function of abatement levels only, without taking product output or input into account. The reason for not using variables of output or input is as follows. First, although the different output or input level will result in different load of BOD_5, the working efficiency of the anaerobic treatment remains relatively constant. This is because the Effluent Treatment Plant (ETP) manager will always maintain the inload BOD_5 concentration at the most favourable level for bacterial growth in the plant, through such manoeuvres as varying the residence time of effluent. Therefore the efficiency of abatement is to some extent independent of inload, as is the cost of abatement. Second, the complexity of multi-outputs from the same industry and the wide range of inputs make it difficult to convert them into commensurable units.

The cost data for effluent treatment in the STWs provided by YW are better than those of industries. However, they were not sufficient to evaluate the cost function of STWs separately. The five STWs in the river Ouse and the tributaries that are considered in this research were therefore also combined as for the industries.

As in the industries, the cost data for STWs came from three discrete years, with twelve observations altogether. The cost function was estimated against the abatement level in each STW. The resulting cost function of effluent abatement in the STWs is as follows, where a is tons of BOD_5 removal per day by each STW. The details of the regression for STWs are given in Appendix 5.

$$Cost_{STW} = 0.249e^{0.245a} \qquad \ldots(6.2)$$

Exponential abatement cost functions have been widely found in empirical work, particularly in water pollution control (Baumol and Oates 1988; Hanley *et al.* 1997; Perman *et al.* 1999; Tietenberg 2001), with increasing marginal cost of abatement (MCA). Needless to say, in reality, there is almost no smooth abatement cost function. The exponential function provides a reasonable approximation of the stepped cost function seen in reality. The results are discussed in section 6.4.

The application of the cost function for the industries needs to be carried out with caution. The cost function was only able to predict the change of abatement cost incurred by marginal changes in the abatement level. This is to say, the cost function only applies to effluent treatment under the current abatement facilities, and the predicted cost only takes into account operating cost, interest, capital depreciation, and maintenance. It does not include the cost of replacement of new facilities or applications of new techniques. When there is major change of abatement facilities, either in facilities or in techniques, the cost function would not be able to reflect the change of costs.

6.2.2b Water abstraction cost functions in the river Ouse catchment

As part of its legal duty, YW is responsible to provide sufficient potable water to the residents and to satisfy the various water demands. Therefore the cost of water abstraction per se, is not the cost incurred for pollution control in the river basin, but the cost of production as a water supplier. However, since the water abstraction has adverse impacts on the water quality, the cost incurred from reducing water abstraction could be regarded as the cost of ameliorating water quality reductions caused by water abstraction. In the case of the Ouse, YW is unable or very unlikely to reduce the water supply for the whole catchment, which is currently at 360 Mega Litre (Ml) per day or 4.167 m^3s^{-1} (Mega Litre = 1 million Litre). If there has to be reduction in the water abstraction in the Ouse catchment, YW has to find enough water sources from somewhere else. Because of this, the cost of water supply in the Ouse catchment accounts for two aspects of cost. The first is the abstraction cost of water from the Ouse and the Derwent. The second is the cost of water supply from alternative water resource options.

The cost of water abstraction from the Ouse and Derwent comprises the capital depreciation and operating costs. According to YW, water abstraction could be switched between Ouse and Derwent at negligible additional cost within the licenses since the cost mainly comes from electricity usage and basic treatment. Therefore, apart from different capital depreciation, there is almost no difference in abstraction cost for water abstracted from different sites. On the other hand, YW also has estimated the possible costs for the alternative water supply sources or options. The possible sources include leakage control, pipeline option from Elvington, Ouse bank side storage and desalination at Hull. Each option has different water yields and associated costs, and the choice between the options was based on their yields and cost of water supply.

The cost function for water supply was estimated against different levels of water abstraction from the catchment. The cost is the aggregate cost from both water abstraction and alternative water resource options. The operational cost of water uptaking was provided from YW for both rivers. The alternative water supply sources were generally more expensive than water abstraction within the catchment. The cost function of water supply was estimated by various water abstraction levels from rivers Ouse and Derwent in aggregate. The cost of waster supply followed an exponential function of the aggregated water abstraction when it ranged from 0-50% reduction of current levels (see details in Appendix 6). The function is shown below, where β_1 and β_2 are the respective levels of water abstraction from rivers Ouse and Derwent. Currently, the maximum water abstraction rate of YW allowed by the water abstraction license in an annual average is 3.530 m^3s^{-1} from the river Derwent and 0.833 m^3s^{-1} from the river Ouse.

$$Cost_{abs} = 39.607e^{-0.472(\beta_1+\beta_2)} \qquad \ldots(6.3)$$

It should be stressed that, this function is derived from the data obtained from YW, in which only up to 50% reduction of water abstraction were evaluated due to data limitations. Because of this, some alternative options have not been taken into account because they are relatively more costly. Had all the alternative options for reducing water abstraction from river Ouse been considered, the cost would have been higher than estimated. Therefore, the estimated cost function of reducing water abstraction is reliable only when abstraction ranges between 50% and 100% of current level, but less reliable for levels of abstraction below 50% of current level.

6.2.2c Effluent relocation costs

The cost of moving effluents from Selby to other points along the river was estimated using the method adopted in Cashman et al. (1999). Piping and storage costs were based on the benchmark price of UK water industries (Ofwat 1999).

The effluents from Selby are from five main sources: Industry A, Industry B, Industry C, STW A and STW B. They are scattered in a small area around Selby, discharging effluents within a 3 km section of the river Ouse. Before moving them downstream, the aggregate effluents would need to be piped into a central collection point within Selby, which a storage facility is able to adapt to variations in the effluents. Preliminary treatment in the storage facility before pumping the effluents downstream is possible, but not always necessary. A pipe along the river would then transfer the effluents downstream of the river Ouse, where environmental targets may be met at minimal cost. The construction of pipes collecting the effluents within Selby to the central storage tank might be costly as it entails pipeline construction within an urban area (Cashman *et al.* 1999). The storage facility is assumed to have a four-hour capacity in order to balance the variations in flows and act as buffer in case of emergency. The aggregate flow from the five sources in 2004 was about 1 ML per hour over a 24-hour working time. Therefore the storage tank needs a capacity of 4 ML. The diameter of pipes within Selby is 150 mm and the main pipe to transfer the effluents downstream is 300mm. Details of other capital investments are available in Appendix 7. The majority of the operational costs of transferring the effluent are the pumping costs to transfer the effluent downstream. Details of operating costs can also be found in Appendix 7. The annual cost of moving effluents is the depreciation of all the capital investments required plus the estimated operating cost.

As may be expected, the cost of moving effluents is a linear function of the distance from the new discharge location to Selby. In order to be consistent with the water quality function, the cost function is evaluated against X, the distance from the new discharge location to the Trent Falls in kilometres, and the distance from Selby to the new discharge location is $(41-X)$ km. The resulted cost function is therefore as follows:

$$Cost_{mov} = 0.820 - 0.005X \qquad \ldots(6.4)$$

Unfortunately, this underestimates the full opportunity cost of transferring effluent. Establishing the collecting point is not technically a problem in Selby and most of the construction of pipeline occurs in rural areas where the impacts could be minimized through good planning and practice (Cashman *et al.* 1999). But possible obstacles of this option could be anticipated from those living downstream as well as landowners and interest groups. Therefore, a proper consultation processes would be needed to address this issue and this is costly in terms of time and money.

6.2.3 Costs of Changing the Timing of Effluent Discharge

Another option that could result in significant water quality improvement in the river Ouse without changing the location of effluent discharge is to change the timing of effluent discharges seasonally from summer to winter. Since the DO sag issue happens mostly in summer months, this option would store effluents from Selby during the summer months, and discharge in the winter at double rate. This 25% storage scenario described in section 3.4.3.b is able to increase the 5%ile DO% at Selby in 1996 from less than 10% to around 20%.

However, this option does not come at a low price. Cashman et al. (1999) estimated a similar option of storing 75% of the effluent discharges from the four industries during the summer to involve a capital investment of more than £27m required by this option and £0.2m operating costs, making it less cost effective than the option of effluent abatement in each individual industry. The total effluents from Selby sources in this research were double as much as that considered by Cashman et al. (1999), so this option was not considered in this research.

6.3 Static Optimisation Analysis

Having estimated the cost functions of effluent treatment, abstraction reduction, discharge relocation, and the system of water quality functions, the cost of meeting an arbitrary water quality target at the water WQM sites is minimised. The static optimisation takes the following form, where Q_s and $\overline{Q}_s$ are water quality prediction and targets at cell s in terms of DO%:

Minimize

$$C_{total} = \sum_{ind} Cost_{ind}(a_{ind}) + \sum_{stw} Cost_{STW}(a_{STW}) + Cost_{abs}(\beta_1, \beta_2) + Cost_{mov}(X)$$

s.t.

$$Q_s = f_s(X, SBOD(\sum_{Sel}(B_{Sel} - a_{Sel}), Ouse(\beta_1), Derw(\beta_2), Sna(a_{sna}), Sand(a_{san}), Tho(a_{tho})) \geq \overline{Q}_s \quad \ldots(6.5).$$

In the function, a_{ind} is the abatement level in the industry and a_{STW} is for the STWs. β_1 and β_2 are the water abstraction levels in the Ouse and Derwent. B is the original BOD_5 inload in the source before any abatement therefore $SBOD(\sum_{Sel}(B_{Sel} - a_{Sel})$ represents the aggregated BOD_5 discharge from the sources around Selby, including the remaining industries in Selby and two STWs. The cells s predicted in this research are cells 180, 192, 193, 197 and 199, reflecting the water qualities at Selby, Long Drax and Boothferry Bridge that are at risk of DO sag during the summer. The arbitrary water quality target for these cells are assumed to be 30% DO% at 5%ile (in order to protect the return of salmon). All the three options have effects on water quality improvements at different prices. Analysing the effects on water quality and economic cost of the trade-off among these options, the static optimisation is able to find the best combination that meets the quality target at least cost. When no other constraints are applied, the least cost solution is at the point where each option has the same marginal cost of water quality improvement.

6.3.1 Business as usual (BAU) scenario

Under the BAU scenario, it is assumed the EA would achieve the target of at least 30% DO% at 5% at the WQM sites in a dry year, through tightening the effluent discharge consents for the three industries and STWs (STW A and STW B) near Selby. Effluents from the STWs C, D and E are discharged from the tributaries Aire and Don, with only little impact on the water qualities at the predicted cells. They are therefore not affected. The water abstraction reduction and effluent relocation options were not considered in the BAU scenario. The only variable allowed for variation in Eq (6.5) was SBOD, the aggregated BOD_5 discharge from Selby into the river Ouse. Therefore the BAU scenario implies the costs of water quality improvement incurred within the Selby industries and STWs in order to comply with the water quality target during a dry year. It turns out to be infeasible to achieve the water quality target at the five cells through the reduction of SBOD alone, however. That means no matter how much BOD_5 to be abated through the industries and STWs in Selby, the DO% of at least one of the cells would still fail to comply with the 30% requirement at 5%ile, even if the industries and STWs could somehow manage to afford the financial costs of abatement. The simulation of QUESTS1D model also confirmed this. When there were no BOD_5 effluent discharges from the five sources around Selby, the river water quality in terms of DO% at 5% along the river Ouse was as shown in Figure 6.1.

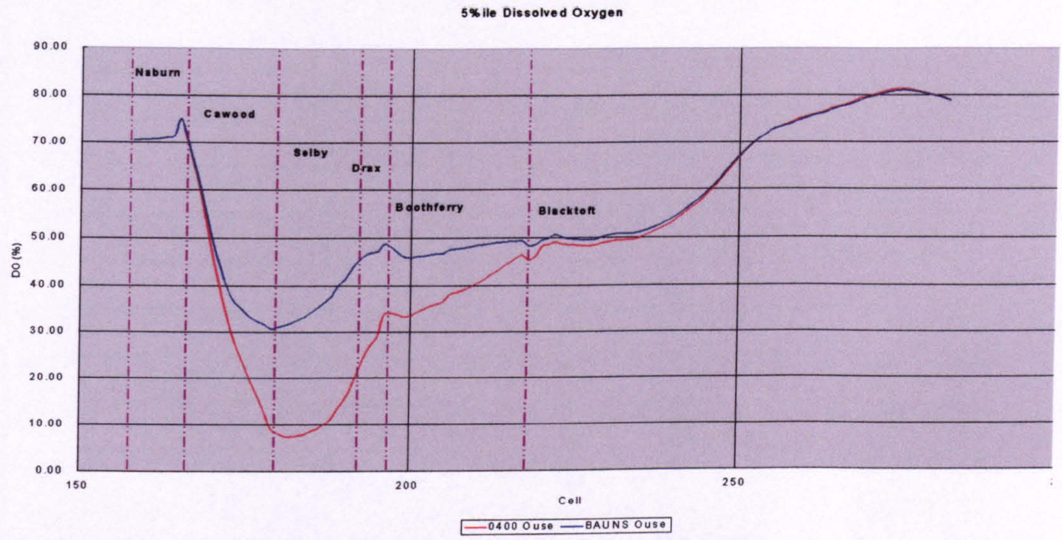

Figure 6.1 DO% of BAU & No SBOD (BAUNS)

The red line in Figure 6.1 is the simulated 5%ile DO% over the river Ouse during a dry year under the current effluent discharge consents, without changes in abstraction and discharge location. The blue line that has better water quality is the result of zero-emissions to the river Ouse from the Selby area. Even so, the DO% around the WQM site of Selby was just at 30%, although water quality at Long Drax and Boothferry Bridge was significantly improved. The predictions from simplified system of water quality functions are even more pessimistic than the QUEST1D simulation indicating that even with zero emissions in Selby, the 30% DO% requirement could not be met. In both of the predicted results, the DO sag around Selby is attributed to the upstream transportation of resuspended solids mentioned in earlier research (Cashman *et al.* 1999; Freestone 2003). Therefore the arbitrary water quality target of 30% DO% at 5%ile is infeasible (either technically or economically) just through variation of effluent abatement.

6.3.2 No Constraints (NC) Scenario

Under this scenario, effluent treatment, water abstraction reduction and discharge relocation were all allowed as options to meet the water quality targets at the WQM sites. The water abstractions considered in the Ouse catchment are from the Ouse and Derwent, whereas their impacts on the water quality are different due to the nature of the rivers. Therefore the option of reducing water abstraction also included the possibility of switching water abstraction between the river Ouse and the river Derwent. The range of water abstraction reduction was up to 50% of current abstraction levels, and the solution did not consider the situation with higher than current water abstraction levels. The relocation of effluent discharges was evaluated within the section of river Ouse, between Selby and the Trent Falls. Moving effluent discharges further downstream is possible, but the impacts need to be evaluated after taking into account the dilution effects from river Trent, which is beyond the range of this research. Since the industries in Selby are implicitly assumed to operate under the same cost function and the same location of discharge, it might be expected that aggregate abatement levels for the effluents of industries would be equally divided among them in the optimal solution. In reality, the industries have different abatement capacities and do face different abatement costs, and such solution might be infeasible. The aggregate reduction in emissions would thus have to be allocated between industries using a mechanism such as tradable emission permits. This will be discussed in Chapter 8.

The required average abatement levels in the industries and STWs are given as tons of BOD_5 per day (t/d); the reduction in water abstraction from the river Ouse and river Derwent is given in m^3s^{-1} while X is the distance from new discharge location to the Trent Falls in kilometres. Table 6.2 indicates the optimal abatement levels from each effluent source, the water abstraction levels in the Ouse and Derwent and the new effluent discharge location for the least cost solution under the NC scenario. For example, the first three cells indicate that an average of 2.036 t/d of BOD_5 should be abated by each of the three Selby industries. The next two cells indicate optimal abatement by the two STWs around Selby, and the last three cells indicate optimal abatement levels for the three STWs on the tributaries Aire and Don. Under the least cost solution, none of the STWs are required to abate their effluent at all, which means the inload BOD_5 could be discharged without any abatement. The two cells for Ouse and Derwent suggest that 0.637 m^3s^{-1} water should be abstracted from the site on the river Ouse, and 3.530 m^3s^{-1} of water is to be abstracted from the river Derwent. The value of X is the optimal location of Selby effluent discharges under the least cost solution, 14.673 km upstream from the Trent Falls.

Table 6.2: Static Optimal Solutions (NC Scenario)

Industry A	Industry B	Industry C	STW A	STW B	Ouse	Derwent	X	STW C	STW D	STW E
2.036	2.036	2.036	0.000	0.000	0.637	3.530	14.673	0.000	0.000	0.000

Table 6.3: Water qualities at WQM sites (NC Scenario)

Site	Selby	Long Drax		Boothferry Bridge	
Cell	Q_{180}	Q_{192}	Q_{193}	Q_{197}	Q_{199}
DO%	30.000	34.494	34.262	32.539	30.000

Table 6.4: Cost of river management (NC Scenario)

	Abatement	Abstraction	Relocation	Total
Cost (m£)	2.205	5.541	0.747	8.493

In the NC scenario, only the three industries were required to abate their effluent discharges, while there was no requirement at all for the STWs to abate their effluents. This is understandable since the marginal cost of abatement by the industries is less than that of the STWs. Moreover, the relocated BOD_5 discharges from the STWs could be absorbed by much diluted river water without failing water quality target. Instead of purer and more manageable effluent inload to their treatment plants in the industries, the STWs have to deal with mixed effluents out of their control from all kinds of sources, such as small industries, households and other sectors. This inevitably involves higher abatement costs.

Under this scenario, the static least cost solution implies water abstraction occurs in the river Derwent at the maximum of license permission, and the rest of water abstraction is from river Ouse to satisfy the water supply demand. No reduction in water abstraction is required due to the higher costs that would incur from alternative resources in terms of improving water quality. This concluded that (a) under the NC scenario the marginal effect of water abstraction on the water quality was in general higher in the river Ouse than in the river Derwent, therefore it was better to abstract from river Derwent first; (b) the marginal cost of improving water quality through reducing water abstraction was all time higher than the other two options so there was no reduction needed in the optimal solution.

Relocation of effluent discharges proved to be very effective in improving water quality along the tidal Ouse, considering the slack conditions required for the two management options above. The optimal location to discharge effluent from Selby was 14.673 km upstream of the Trent Falls, which is about 1 km upstream of the confluence of river Don. The dilution effects from tributaries Aire and Don seemed quite promising according to this choice of discharge location.

The annual costs of river management under the least cost solution for the NC scenario was £8.493m and yielded water quality of at least 30% DO% at Q180 and Q199 (Table 6.3 and 6.4). It needs to be pointed out though, that £5.541m out of the total annual costs was the cost of water abstraction, as production cost of YW to provide water supply, accounting for over 65% of the total cost of river management. The costs of improving water quality through treatment and discharge relocation was only £2.952m in total, less than 60% of the current effluent treatment costs (£4.936m) that incurred in the industries and STWs.

However, despite the obvious cost advantages of the least cost solution under the NC scenario, it is not an easy solution. One constraint is the European Directive of UWWTD. One of the elements of UWWTD is the secondary treatment of discharges from the STWs. It is inappropriate to have the inloads of STWs discharging without any treatment. UWWTD also requires the STWs to reduce nutrient inputs to sensitive areas, in order to prevent eutrophication problem in the water bodies. The abatement of effluents is therefore necessary even without of the issue of DO% sag. The STWs have all been continuously investing during the last decades to comply with UWWTD. The cost of closing down these facilities (as abatement was not required in the STW according to the solution) is not considered in this research, but will certainly be unacceptable to the water company. The second problem is due to nature of research that mainly considers the water quality along the river Ouse. If the STWs in the Aire and Don were closed, there would be considerable deterioration in water quality those rivers, as well as in the section of river Ouse below their confluence. As a result, some constraints have to be placed on the static optimisation.

6.3.3 UWWTD Constraints (UC) Scenario

For the reasons given above, changing abatement levels in industries is likely to be more possible than changing abatement levels in the STWs. Two of the industries have been using their effluent treatment plants for quite a long period and would have to install new plant if the effluent discharge consents become more stringent. Given competitive pressures, both industries and the local economy of Selby would benefit following the least cost solution from less restrictive emission requirements. In the UC scenario, all five STWs were assumed to be working at no less than the current levels in compliance with the UWWTD requirements, while abatement in industries, water abstraction and effluent discharge location were assumed to be choice variables.

Table 6.5: Static Optimal Solutions (UC Scenario)

Industry A	Industry B	Industry C	STW A	STW B	Ouse	Derwent	X	STW C	STW D	STW E
1.081	1.081	1.081	0.599	1.955	0.637	3.530	14.890	0.498	7.902	2.954

Table 6.6: Water qualities at WQM sites (UC Scenario)

Site	Selby	Long Drax		Boothferry Bridge	
Cell	Q_{180}	Q_{192}	Q_{193}	Q_{197}	Q_{199}
DO%	30.000	34.231	33.968	32.481	30.000

Table 6.7: Cost of river management (UC Scenario)

	Abatement	Abstraction	Relocation	Total
Cost (m£)	4.074	5.541	0.746	10.361

The optimal abatement levels of the three industries in Selby, water abstraction levels in Derwent and Ouse, and effluent discharge location are given in Table 6.5. The least cost abatement level for the three Selby industries under UC scenario is 1.081 t/d of BOD_5, in aggregate only 16% of their current abatement level. Given the water supply cost function, reducing water abstraction levels was quite costly as a means of improving water quality than the other options. Therefore no reduction in the water abstraction was required. The pattern of water abstraction under the UC scenario is the same as NC scenario. In addition, the optimal discharge location under UC scenario was 14.890 km upstream from the Trent Falls, several hundred metres upstream than that in the NC scenario. Unlike the NC scenario, the abatement levels of STWs are same as the current levels because of the UWWTD constraint. Although higher abatement levels in the STWs are possible, it is not cost effective to do so hence the STWs are suggested to remain at their current abatement levels. Water quality in the Don and the Aire were maintained as the STWs are discharging at current levels along these two tributaries. Furthermore, the good quality of water in the tributaries improves water quality in the tidal Ouse through dilution effects. This is reflected in the optimal discharge location, which takes advantage of the dilution effects of the tributaries Aire and Don.

Table 6.6 shows water quality at the five points concerned under the UC scenario. The two points at which the water quality constraints are binding are same for both NC and UC scenarios, Q180 and Q199. Under the UC scenario, water quality at the other three sampling points is similar to that under the NC scenario. Water quality along the river Ouse is improved between Selby and Boothferry compared to the BAU scenario, and the DO sag disappears from the river Ouse even in a low flow year as 1996. The QUESTS1D simulation using the pattern of emissions and abstraction identified in the least cost solution confirms the prediction from the water quality functions. Figure 6.2 indicates the 5%ile DO% along the river Ouse simulated by the QUESTS model following the least cost solution generated by GAMS.

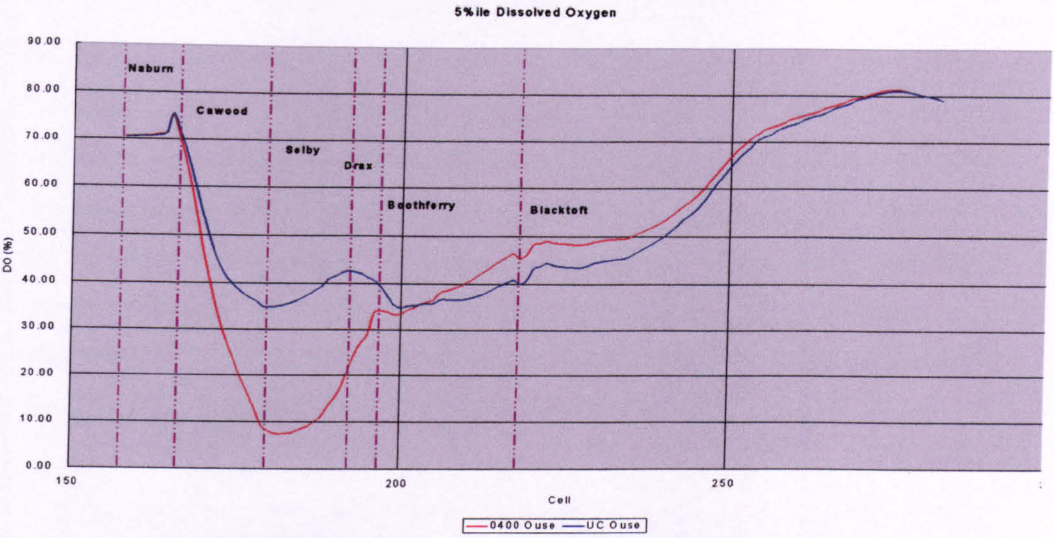

Figure 6.2 DO% under the least cost solution in the UC scenario

The least cost solution of optimisation results in slightly better water quality in the QUESTS1D model simulations than predicted by the system of water quality functions, with reduced risk of non-compliance. The DO% of the river Ouse under the least cost solution decreases near Naburn due to the tidal inflow and the resuspended sediments, increases after Selby and reaches a local maximum around Drax, then decreases again, but finally becomes stable around 35% and eventually recovers towards saturation after the confluence with river Trent.

The least cost solution involves an aggregate cost of £10.361m to comply with the 30% DO% requirement. As in the NC scenario, almost half of this is accounted for by abstraction costs. Costs of effluent abatement and relocation of the effluent discharge account for £4.820m, achieving much better water quality along the river Ouse at slightly less cost than that is currently endured by the industries and STWs. Abatement in the STWs account for more than 60% of the costs of effluent abatement and discharge relocation. The relocation of effluent discharge only accounted for 15% of the costs but had obviously much significant impact on the river water quality.

There are some uncertainties associated with the least cost solution under the UC scenario. The first one involves the cost of the infrastructure needed to transfer effluents to the new discharge location. Although the annual cost of transfer is small, the capital investment required to build the storage facility and lay down the pipes is close to £10m. This cost could be a possible obstacle to the acceptance of the solution. The second is because the cooperation between industries and STWs, which is critical to the solution, is not guaranteed under this scenario. Since the STWs have to maintain their current abatement levels because of UWWTD, they would be reluctant to pay for the effluent relocation infrastructure that does not affect their abatement levels. If the industries have to bear the cost of infrastructure alone, the required investment of capital would be too high to be acceptable in current trading conditions.

Reallocation of the benefits among the industries and STWs through negotiation might be sufficient to ensure the STWs' participation, since industries could save costs from the abatement done by the STWs. Alternatives would be either an emission tax-subsidy scheme or a TPP system. On the other hand, although the STWs are unable to reduce their abatement levels under the UC scenario, they still benefit from future expansion allowed by the fact that the DO sag issue is removed from the Selby area. That is, they can respond to increasing demand for sewage services due to the growth of the population and the economy in North Yorkshire and Lincolnshire (Jarvie *et al.* 1997b). The STWs would also benefit from a reduction in the risks of failure during extreme conditions such as flood or storm.

6.4 Sensitivity Tests

The best-fit individual abatement cost functions for the industries and STWs is in fact a power function and a linear function implying a constant MAC. Since the data are limited it was decided to test the sensitivity of the outcome to the functional form of the abatement cost function. Using a power function for the industries and a linear function for the STWs (for the estimation of these see Appendix 8 and Appendix 9),

$$Cost_{ind} = 0.181a^{0.660} \qquad \ldots(6.6)$$

$$Cost_{STW} = 0.172 + 0.194a \qquad \ldots(6.7)$$

the system was re-optimised under the UC scenario, and yielded the following solutions for abatement levels in the industries and STWs (Table 6.8).

Table 6.8: Static Optimal Solutions (Sensitivity Tests)

Industry A	Industry B	Industry C	STW A	STW B	Ouse	Derwent	X	STW C	STW D	STW E
0.000	0.000	0.000	0.599	1.955	0.416	3.530	10.461	0.498	7.902	2.954

Table 6.9: Water qualities at WQM sites (Sensitivity Tests)

Site	Selby	Long Drax		Boothferry Bridge	
Cell	Q_{180}	Q_{192}	Q_{193}	Q_{197}	Q_{199}
DO%	30.000	33.990	33.826	32.558	30.000

Table 6.10: Cost of river management (Sensitivity Tests)

	Abatement	Abstraction	Relocation	Total
Cost (m£)	3.558	6.151	0.752	10.461

Comparing with Tables 6.5-6.7, the STWs end up at the same levels of abatement, as under the UC scenario before but now the industries do not have to abate their effluent at all, which means they could discharge their inload effluent directly into the tidal Ouse without any treatment. Accompanied with this, the water abstraction in the river Ouse is reduced compared with 0.637 m^3s^{-1} in the UC scenario. More significant change comes from the location of Selby effluent discharges. The new location is 10.461 km upstream of the Treat Falls, 4 km further downstream of the optimal location of UC scenario. The change in the function of STWs' abatement cost seemed irrelevant to the solution of STWs' abatement levels, but ruled out the need of abatement in the industries because of the change in function of industrial's abatement function. As a result, water abstraction in Ouse was reduced and the location of effluent discharge from the Selby sources has to be moved further downstream.

A second set of sensitivity tests relate to the choice of water quality targets. The actual water quality targets are RE4 classification (pers. comm. Peter Stevenson; Environment Agency) at these three WQM sites along the tidal Ouse, which requires 50% DO% at 10%ile value. As the water quality functions were derived from 5%ile value of DO%, new functions based on 10%ile would be needed to apply the optimisation analysis. Here we consider different water quality targets at 5%ile at these three sites using the water quality functions. Tables 6.11 and 6.12 indicate different optimal solutions for various water quality targets, and the resulted cost of river management. Changes of abatement levels within the STWs were disallowed, because of the effect of the UWWTD.

Table 6.11: Static Optimal Solutions for various water targets

Target	Industry A	Industry B	Industry C	Ouse	Derwent	X
20	0.000	0.000	0.000	0.833	3.334	29.235
25	0.000	0.000	0.000	0.833	3.334	19.978
30	1.081	1.081	1.081	0.637	3.530	14.890
35	6.540	6.540	6.540	0.637	3.530	18.170
40	N/A	N/A	N/A	N/A	N/A	N/A

Table 6.12: Cost of river management for various water targets

Target	Abatement (m£)	Abstraction (m£)	Relocation (m£)	Total (m£)
20	3.978	5.541	0.674	10.193
25	3.978	5.541	0.720	10.239
30	4.074	5.541	0.746	10.361
35	4.776	5.541	0.729	11.046
40	N/A	N/A	N/A	N/A

Once again, there is no need for the industries to abate their effluents until the target reaches 30% DO% at 5%ile value. This is because, with the abatement undertaken in the STWs, the effluents from the industries are within the assimilative capacity of the river when discharged at the optimal location, thus their effluent could be naturally degraded without abatement needed before discharge. When the DO% target was raised to 30%, each industry in Selby was required to abate an average of 1.081 t/d BOD_5, and maximum water abstraction was required from the river Derwent to reduce the impacts of water abstraction from the river Ouse. The allocation of abatement responsibility among the three industries' different abatement capacity is a separate issue, and will be discussed later in this thesis. Both tradable pollution permits system and pollution tax-subsidy scheme could be appropriate options.

Water quality target of 40% DO% at 5%ile along the tidal Ouse is not achievable in the optimal solution. This does not rule out the possibility of achieving better water quality, but implies that it is not attainable along the river given the current water quality management options, for the given low flow year. Better water quality may still be possible if more effective instruments could be identified and in the years with better flow conditions. Figure 6.3 indicated the change of aggregate costs of river management in the tidal Ouse for different water quality targets. The cost displayed in the chart is the aggregated cost of effluent abatement and discharge relocation alone. The water abstraction cost was not included since the total abstraction remains unchanged during the optimisation and the cost is purely for water supply rather than improving water quality.

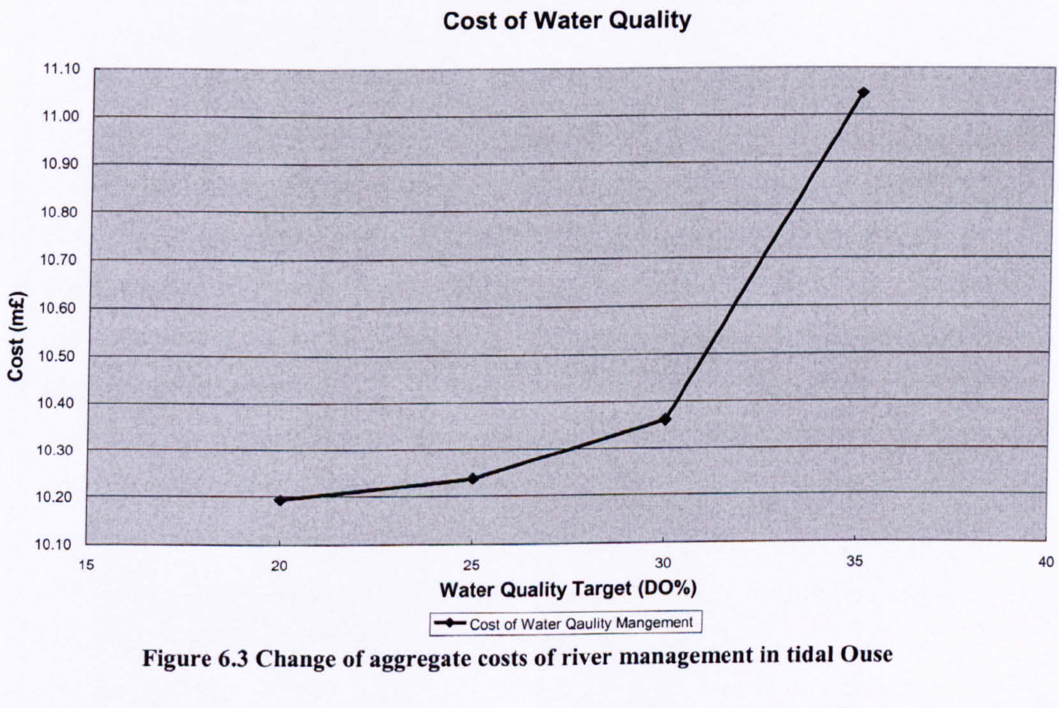

Figure 6.3 Change of aggregate costs of river management in tidal Ouse

The aggregate costs of river management were only slightly increased when the water quality target increased. But a sheer increase was observed after the target became 35% DO% as the industries in Selby were require to increase their abatement levels significantly. If the target increases further and the option of reducing water abstraction from either Ouse or Derwent had to be implemented, the cost of river management would increase more rapidly. Therefore it is important to choose water quality target carefully for the compliance of regulations.

6.5 Conclusions

This chapter has considered the least cost solution for water quality management in the tidal Ouse for a particular water quality target. The problem was posed as the minimization of the cost of water quality management through a mix of management options, subject to minimum water quality targets being achieved at a number of sampling points along the river. The least cost solution complied to UWWTD requirement involves the relocation of the point of effluent discharges from the three industries and two STWs around Selby to a point 14.890 km upstream from the confluence the river Trent and river Ouse at the Trent Falls. The discharge location is close to the main tributaries of the river Ouse, and not far from its confluences of the river Trent. It makes it possible for the effluents to be sufficiently diluted to maintain water quality at all receptor points. Shifting water abstraction from the Ouse to the Derwent was also suggested in order to reduce the impacts of water abstraction on DO% in the Ouse. The solution takes account of the variation in assimilative capacity of river water along the tidal Ouse, in order to avoid excessive cost in water quality management. With abatement levels unchanged in the STWs, the industries are allowed to abate much less of their effluents than they currently do. Compared with the current water quality regulatory regime, which is mainly through the reduction in the on-site effluent discharges from various sources, this least cost solution yields significant water quality improvements and removes the DO sag around Selby and Drax, whilst yielding cost savings of £116,000 a year.

The least cost solution is calculated for a low flow year (1996) in which it has been shown that the 30% DO% target is infeasible through the on-site effluent abatement within the industries and STWs alone. The least cost solution involves an integrated water quality management regime which is sensitive to the variation in assimilative capacity and abatement costs. Of course the arbitrary water quality target of 30% DO% for all WQM sites (to allow salmon passage throughout the tidal Ouse and the tributaries) is unlikely to be the only criteria of water quality in the tidal Ouse. For example, the current WQOs for the tidal Ouse are RE2 for Naburn, RE3 for Cawood, RE4 for Selby, Drax and Boothferry Bridge. These are 70%, 60% and 50% DO% at 10%ile respectively. Using the framework of cost minimisation discussed above, the management regime could be used to identify the least cost solution for any water quality targets. As the water quality target becomes more stringent, the costs of water quality management rise. In general, determining ecologically sound and economically reasonable water quality targets for the tidal Ouse that reflect public attitudes and ethical choice is a very challenging task for the policy maker. How to determine the appropriate water quality target is beyond this research, however, the framework is able to determine the most effective way to comply with any target once the environmental authority decides the target.

A separate issue that has not been addressed here is the optimal allocation of abatement between the Selby industries in particular, but also between the STWs and water abstraction plants. Fixed effluent discharge consents or water abstraction licenses, as given in the optimal solution, could jeopardise the cost savings and flexibility offered by the integration of water quality management. They are especially inappropriate when the required abatement level is implemented to various sources with different abatement costs or when there is demand for a change in the scale of production or discharge from an effluent source. This problem may be solved by implementing economic instruments of river policy, either a TPP system or an emission tax-subsidy scheme. This will be discussed in details in Chapter 8 for the results of both static and dynamic optimisation after the discussion of dynamic optimisation in the next chapter.

Chapter 7 Dynamic Optimisation

7.1 Introduction

Chapter 6 discussed the optimal solution to water quality management in a static situation, in which all parameters are assumed constant over time. The values of variables are fixed once the optimisation for one point of time is done, and no changes are expected from both the internal and external systems. More particularly, the optimal solution indicated the most cost effective water quality management given the capabilities of effluent abatement in the industries and STWs, and given water abstraction levels. This assumption excludes change over time and the uncertainty it might bring.

In fact, the industries and STWs have both been investing in ETPs over the last few decades in response to various pressures, including output growth, population increase in the catchment, and more stringent effluent consents imposed by the EA. Therefore, the effluent abatement capability of ETPs has changed over time, implying the need for a dynamic analysis. In this research, we assume that the variables that drive the change in capabilities of ETPs are their capital stocks and the investment, which builds up the capital stock through time. It is of particular interest as well, for the research to not only investigate the impact of river policy on the industries and STWs' abatement levels, as we did in Chapter 6, but also to identify how the policy would affect investment decisions that indirectly affect the water quality in the long term.

This chapter accordingly aims to investigate the dynamic changes in the water quality management options discussed in the last chapter. Due to limited data, this research only investigated the dynamic changes in the effluent abatement levels of ETPs in the STWs and Selby industries. As the effluent abatement capabilities are driven by endogenous investment decisions in the STWs and industries, we expect the dynamic analysis to reveal the impact of river policy on the investment decisions of the STWs and industries. On the other hand, it is assumed that the river policy or water quality targets in the dynamic analysis are exogenous to the industries and STWs, while environmental regulation and policy instruments are considered time independent in this analysis.

As has been shown before (Gandolfo 1997; Barro and Sala-i-martin 1999; Shone 2002), the dynamic problem is not easy to assess, particularly when nonlinearity leads to more than one equilibrium. When multiple equilibria are present in the dynamic system, only local stability properties can be investigated through linear approximation. A dynamic equilibrium would be much less meaningful in economic management if it were unstable; therefore, the stability of equilibrium and the path approaching to the equilibrium are as important as the dynamic equilibrium itself.

In this research, the minimization of the overall cost of water quality management over time was difficult due to both data limitations and the nonlinearity of the cost and water quality functions. To carry out the dynamic analysis, some critical simplifying assumptions had to be made. These assumptions will be explained in more detail in the following sections.

This chapter consists of five sections. The second introduces the dynamic system, listing the dynamic elements in the system and the way they are involved in change over time. The third discusses optimal water quality management. The impacts of various conditions (constraints) on the dynamic optimum are explored and compared. In the fourth section, the local stability of the dynamic optimum is investigated for a simple case. This section also identifies the investment path that leads to the dynamic optimum for the industries' ETPs. The last section points out the policy implications of the outcomes in these two chapters. These policy implications are discussed further in Chapter 8.

7.2 The elements of dynamic optimisation

The dynamic analysis was carried out based on the model discussed in Chapter 4. Since the effect of labour cost was negligible in this research and the operational cost was assumed positively correlated to the capital stock, effluent treatment capacity was therefore assumed to be a function of the capital stock of the ETP. In contrast to the static analysis, the abatement cost of the ETPs in the dynamic analysis was not determined by the BOD removed in each day, but by the capital stock and investment in ETPs. Decisions on investment in the ETPs are what determine water quality in the tidal Ouse in the long-term.

GAMS was utilised to find the long-term equilibrium level of capital stock and investment for each ETP to comply with particular water quality targets cost effectively. The constraints in the dynamic optimisation were the water quality at the monitoring sites, and the dynamics of capital stocks, i.e. the dynamics of investment and depreciation. To simplify the problem, water abstraction and discharge relocation were treated as in the static optimisation problem. Only effluent abatement by the industries and STWs was considered in the dynamic analysis. Due to the limited data, the results produced are illustrative only. They are insufficient to support decision making, and are less convincing than the static results discussed in Chapter 6. This chapter aims to explore the methods and procedures which would be used in further research based on a more adequate dataset.

7.2.1 Dynamics of effluent treatment capacity

The capacities of effluent treatment in the ETPs of industries and STWs were treated as a function of capital stock only. The capital stocks in the ETPs are driven by the investment decisions of each pollution source. The investments in the ETPs of the STWs and industries in the tidal Ouse are mainly ETP upgrades. As a consequence of the UWWTD, the STWs were obliged to implement secondary treatment before effluent disposal. This has been done in four of the five STWs with similar technology. The exception is Barlby STW, whose upgrading to secondary treatment is currently in progress. The industries have long been using various treatments before discharging effluents to the river, except for BOCM, which shut down its effluent discharge by the time of research. These technologies include aerobic and anaerobic treatment, and waste minimisation technologies. The capital depreciation in each ETP was also taken into account, the depreciation rate being 2.5% in the STWs and Selby industries. The relationship between effluent treatment capacity and capital stock was estimated based on pooled data across all the industries and STWs.

The approach taken involves the estimation of abatement functions. Data for the ETPs of STWs and industries were aggregated respectively. Separate estimation for each of the industries would be more reasonable had data on their capital stocks been available. The results of the estimation suggest a consistent relation between capital stocks and ETP capacities. The estimated effluent treatment capacity functions were as follows, reflecting the diminishing marginal effect of increasing capital stock on abatement capability.

Industries: $a_{ind}(k_{ind}^{a}) = 3.180 + 3.977\ln(k_{ind}^{a})$, $(R^2 = 0.855)$ ...(7.1)

STWs: $a_{STW}(k_{STW}^{a}) = -4.108 + 3.224\ln(k_{STW}^{a})$, $(R^2 = 0.768)$ ...(7.2)

Details of the estimation procedure can be found in Appendices 10 and 11. The logarithmic functions of capital stocks in both STWs and industries imply diminishing marginal effectiveness of investment. As in Chapter 6, capital stocks k^a are measured in million Sterling Pounds (m£) and ETP capacity a is measured in tons BOD_5 removed per day (t/d). The estimated functions imply that the capital stock requirement for ETPs in the industries was less than that required in the STWs for same abatement. This is consistent with the finding in Chapter 6, and reflects the large amount of effluents to be treated and the complexity of incoming wastewater to STWs.

7.2.2 The effluent abatement cost function

The costs of pollution abatement were assumed to be function of current capital stock and investment in ETP during the same period. Due to the confidentiality of this information, only limited data were available from the industries in Selby. On the other hand, the STWs were able to provide data on investment in ETPs during the last ten years. However, these estimations are rather approximate at best. As a result the estimation of pollution abatement costs is based on pooled data from the STWs and industries. The estimated pollution abatement function is shown in Eq (7.3), details of which are given in Appendix 12. As shown by the t-value, investment is not a significant explanatory of abatement costs in the regression. This is expected to be improved by future research with access to more detailed and accurate investment data. Because of this, the dynamic optimisation discussed later can provide only a rough approximation to the dynamic optimum of capital stocks in the industries and STWs. Nevertheless, the analysis below identifies a methodology for understanding the dynamic problem of pollution control in the Ouse.

$$C_i^a(k_i^a, I_i^a) = 0.068k^{0.952}I^{0.017}, \quad (R^2 = 0.980) \qquad \ldots(7.3)$$

In equation 7.3, capital stock, investment and abatement costs are all in units of m£. Abatement costs are annual costs for each source; capital stocks are the current-value of ETPs at the beginning of the period and investment is the value of ETP upgrades during each period.

7.2.3 The dynamic optimisation problem

The water quality targets used in this dynamic analysis are same as that in Chapter 6, and have to be complied with at all points in time during the period. Water quality at the five receptor points is predicted by the system of water quality functions. It is based on the low flow conditions in 1996. The original water quality target is 30% DO% at 5%ile. Due to the insufficiency of data on investment decisions in the past, and lack of evidence for the cost of changing water abstraction to achieve water quality improvement; changes in water abstraction were not considered in this analysis. The total amount of water abstraction was fixed at current levels, although the allocation of abstraction between the rivers Ouse and Derwent was still treated as variables. The location of effluent discharge was taken as given in the optimisation. It is treated as an exogenous variable that only takes two values, corresponding to the two locations that might be chosen for effluent discharges. The first is the optimal location of effluent discharge in the UC scenario of Section 6.5: 14.890 km upstream of the Trent Falls, just before the confluence of the River Don. The second is the current location of effluent discharges in Selby, around 40.7 km upstream from the Trent Falls. The dynamic problem considers the impacts of different choices as to effluent discharge location, on capital stocks of the ETPs and on the compliance with water quality targets. The difference between the optimal and current capital stock in the ETPs provides a target for investment, given depreciation and interest rates.

7.3 The dynamic optimum

The general form of the dynamic optimisation problem is as follows:

$$Minimize_{I_i^a} \int_0^{\infty} e^{-rt}[C_i^a(k_i^a(t), I_i^a(t)) + Cost_{abs}(\beta_1, \beta_2) + Cost_{mov}(X)]dt \quad \ldots(7.4),$$

$s.t.$

$$Q_s(t) =$$
$$f_s(X, SBOD(a_i(k_i^a(t))), Ouse(\beta_1), Derw(\beta_2), Sna(a_{sna}(t)), Sand(a_{san}(t)), Tho(a_{Tho}(t))) \geq \overline{Q}_s \quad \ldots(7.5);$$

and $$\dot{k}_i^a = I_i^a - \delta_i^a k_i^a \quad \ldots(7.6),$$

$k_i^a(0)$ is given.

This minimizes the costs of water quality management in the tidal Ouse by choice of the level of investment in ETPs through $I_i^a(t)$. SBOD is BOD_5 discharged from the three industries and two STWs near Selby, which is determined by the initial total BOD_5 inloads (constant) and the sum of their aggregated effluent abatement levels, $\sum_i (BOD_5 - a_i(K_i^a))$. The BOD_5 discharged from the other three STWs, Snaith (Sna), Sandall (San) and Thorne (Tho), is represented by their initial BOD_5 inload (constant) and their existing abatement levels. The water quality targets $\overline{Q}_s$ are set at 30% for 5%ile DO% at the five monitoring points. Under some conditions however, this water target has to be reduced, as it becomes infeasible. Eq. (7.6) expresses the dynamics of capital and investment. Capital stocks change over time, $\dot{k}_i^a$, because of investment

$I_i^a(t)$ and depreciation $\delta_i^a k_i^a$ at any instant of time. At steady state equilibrium, Eq. (7.6) implies that $\dot{k}_i^a = I_i^{a*} - k_i^{a*}\delta_i^a = 0$, in which k_i^{a*} and I_i^{a*} are the optimal levels of capital stock and investment in each ETP. That says at the equilibrium, the capital depreciation is just offset by the new investment to maintain the equilibrium level of capital stock.

The discrete time equivalent of this problem is as follows.

$$Minimize_{I_{i,t}^a} \sum_{t=0}^{\infty}\sum_{i} \rho^t \cdot [C_{i,t}^a(k_{i,t}^a, I_{i,t}^a) + Cost_{abs}(\beta_1, \beta_2) + Cost_{mov}(X)] \qquad \ldots(7.7),$$

s.t.

$$Q_{s,t} =$$
$$f_{s,t}(X, SBOD(a_{i,t}(k_{i,t}^a)), Ouse(\beta_1), Derw(\beta_2), Sna(a_{sna,t}), Sand(a_{san,t}), Tho(a_{Tho,t})) \geq \overline{Q} \qquad \ldots(7.8);$$

and $k_{i,t+1}^a - k_{i,t}^a = I_{i,t}^a - \delta_i^a k_{i,t}^a$ ...(7.9),

$k_{i,0}^a$ given.

In both the continuous and discrete time problems, water abstraction levels and effluent discharge locations were treated as constant over time to simplify the situation. A time horizon of 10 years and a discount rate of 4.5% annually (UK average in the same period) was adopted for the dynamic optimisation, and the capital depreciation rate was assumed to be 2.5% annually across the industries and STWs (taken from the industries and STWs). At the end of the 10 years, all the costs from the date onwards were summed up through integration.

To render the problem tractable, the objective was reduced to the minimization of the annual cost of water quality management. The capital stock and investment level corresponding to the minimized annual cost were then assumed to give the steady state equilibrium for capital stock and investment. This offers a rough approximation of the real dynamic equilibrium. The problem solved using GAMS was of the form:

$$Minimize_{I_i^a} \sum_{i} C_i^a(k_i^a, I_i^a) + Cost_{abs}(\beta_1, \beta_2) + Cost_{mov}(X) \qquad \ldots(7.10),$$

s.t.

$$Q_s = f_s(X, SBOD(a_i(k_i^a)), Ouse(\beta_1), Derw(\beta_2), Sna(a_{sna}), Sand(a_{san}), Tho(a_{Tho})) \geq \overline{Q}_s \quad ...(7.11);$$

and $I_i^a = \delta_i^a k_i^a$ ...(7.12).

Objective Eq (7.10) is similar to Eq (7.4) but without integration over time. That is, it solves a current rather than a present-value problem. Capital stocks, investment and abatement are all solved for the steady state, as is the constraint function for water quality. Eq (7.12) simply requires investment to be equal to depreciation in order to hold the capital stock at the equilibrium level.

The main purpose of the exercise is to make it possible for policy scenarios analysis. Scenario 1 analyses the equilibrium levels of capital stock in the ETPs when the UWWTD requirement was applied to the STWs. Scenario 2 assumes a step up from scenario 1 in which the inload of BOD_5 to all the ETPs is increased by 50%. In the first two scenarios, the effluents from the Selby sources are discharged at 12.472 km upstream from the Treat Falls, as suggested under the UC scenario in Chapter 6. In scenario 3, the effluents of Selby sources are discharged from their current locations.

7.3.1 Scenario 1: Dynamic optimisation with the UWWTD constraint

The UWWTD requires STWs to apply secondary treatment to effluent before discharge unless is exempted. This is not just to address DO consumption in the river water but also to protect the river system from eutrophication caused by nitride and phosphorous compounds. In the scenario 1, the UWWTD requirements for the STWs were adopted. Abatement of BOD_5 in the ETP of each STW was required to be no less than their current levels, so the water qualities in the rivers Aire and Don will be maintained. The Selby effluents were discharged 14.890 km upstream of the Trent Falls and the total water abstraction level in the catchment was fixed.

Under this scenario, the dynamic equilibrium for the capital stock and investment in each ETP, derived from the solution to Eq (7.10) subject to Eqs (7.11) and (7.12) was similar to that in static analysis 6.3.3, shown in Tables 7.1-7.3. The BOD_5 in STWs at the dynamic equilibrium a_i was abated at the current levels. The abatement level required for the ETPs of the industries is 1.082 t/d BOD_5 removals, the same as that under the UC scenario in Chapter 6. Water quality in this scenario was much better than current in the tidal Ouse and its tributaries in a low flow year such as 1996, and saw a saving of £270k over the £4.936m current cost of effluent abatement.

No change in the capital stocks of the ETPs in the STWs is required at this dynamic equilibrium, since abatements are maintained at current levels. On the other hand, the steady state capital stocks of ETPs in the industries are less than current. In other words, the conclusion is similar to that reached under the UC scenario in section 6.3.2, which highlighted the possible saving in costs and significant water quality improvement if the effluent discharges from Selby were relocated to14.890 km upstream of the Trent Falls.

Table 7.1: Dynamic equilibrium with UWWTD

	Industry A	Industry B	Industry C	STW A	STW B	STW C	STW D	STW E
K_i^a	0.590	0.590	0.590	4.306	6.557	4.173	41.479	8.939
I_i^a	0.015	0.015	0.015	0.108	0.164	0.104	1.037	0.223
a_i	1.082	1.082	1.082	0.599	1.955	0.498	7.902	2.954
C_i	0.038	0.038	0.038	0.263	0.395	0.255	2.360	0.533
Ouse (m^3s^{-1})			Derwent (m^3s^{-1})			X		
0.637			3.530			14.890		

Table 7.2: Water qualities at monitoring sites with UWWTD (Dynamic)

Site	Selby	Long Drax		Boothferry Bridge	
Cell	Q_{180}	Q_{192}	Q_{193}	Q_{197}	Q_{199}
DO%	30.000	34.231	33.969	32.481	30.000

Table 7.3: Cost of river management with UWWTD (Dynamic)

	Abatement	Abstraction	Relocation	Total
Cost (m£)	3.921	5.541	0.746	10.208

7.3.2 Scenario 2: Dynamic optimisation with increased BOD_5 load

One of the advantages of an integrated river policy is that the STWs and industries would be able to increase output using current abatement facilities. This is because the policy makes better use of the assimilative capacity of the river. This section aims to predict the consequent variation in abatement levels and capital stock of the ETPs in the industries and STWs at Selby if the total inload of BOD_5 to the ETPs increased by 50%. This is an approximation for increasing pollution due to industrial growth, population growth and economic development in the river basin.

In this scenario, the effluent discharge location is the same as in scenario 1, so is the total water abstraction. UWWTD requirements were also applied to the ETPs in the STWs. The optimal level of capital stock K_i^a and investment I_i^a are listed in Tables 7.4-7.6, along with the resulting water qualities at the monitoring sites, and river management costs.

Table 7.4: Dynamic equilibrium with 50% BOD increase

	Industry A	Industry B	Industry C	STW A	STW B	STW C	STW D	STW E
K_i^a	2.022	2.022	2.022	4.306	6.557	4.173	41.479	8.939
I_i^a	0.051	0.051	0.051	0.108	0.164	0.104	1.037	0.223
a_i	5.981	5.981	5.981	0.599	1.955	0.498	7.902	2.954
C_i	0.126	0.126	0.126	0.263	0.395	0.255	2.360	0.533

Ouse (m^3s^{-1})	Derwent (m^3s^{-1})	X
0.671	3.496	14.890

Table 7.5: Water qualities at monitoring sites with 50% BOD increase (Dynamic)

Site	Selby	Long Drax		Boothferry Bridge	
Cell	Q_{180}	Q_{192}	Q_{193}	Q_{197}	Q_{199}
DO%	30.000	34.529	34.279	32.525	30.000

Table 7.6: Cost of river management with 50% BOD increase (Dynamic)

	Abatement	Abstraction	Relocation	Total
Cost (m£)	4.186	5.541	0.746	10.472

The results indicate that if the inload of BOD_5 were increased by 50%, then at the equilibrium the STWs would still retain their current abatement levels at the dynamic equilibrium. However, the ETPs in the industries would be required to abate at a level of 5.981 t/d BOD_5. This is still less than their current abatement levels. In 2004, the BOD_5 abatement from the three industries in Selby was 19 t/d in aggregate, 1 t/d more than that suggested in Table 7.4. For all ETPs in the STWs and industries, the equilibrium abatement level under scenario 2 would be 18 t/d BOD_5 more than that required in the scenario 1, while a 50% increase implies an extra 20 t/d BOD_5 in the inload to the ETPs. Therefore, by integrated management the river system absorbs an extra input of nearly 2 t/d BOD_5, a third of the total BOD_5 discharge from the Selby industries. Water qualities at the monitoring sites comply with the stringent water quality target in a low flow year, whilst the demands of growth in industry, population and economic output are satisfied. With 50% more BOD_5 inload, the minimal annual abatement cost in scenario 2 was about £265k more than that under scenario 1. The minimised cost under scenario 2 from an integrated river management solution is equivalent to the current abatement cost in industries and STWs, a scheme that fails to comply with the water quality target with *current* BOD_5 inload.

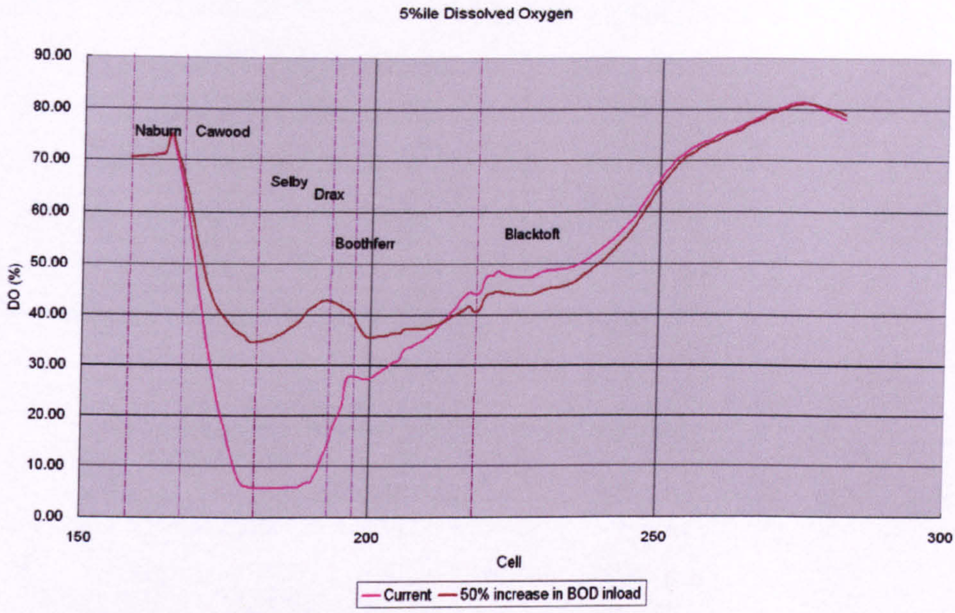

Figure 7.1 The simulated DO% of tidal Ouse under scenario 2

Figure 7.1 shows the simulation from the QUEST1D model, for the DO% in the tidal Ouse under scenario 2. The purple line is DO% under the current abatement conditions while the brown line shows the improvement in DO% achieved under this scenario 2 with integrated management. The DO% simulated by the QUEST1D model is slightly better than the prediction from the water quality functions, which was slightly above 30% DO% along the whole length of the river. Relatively low DO% occurred around Selby and downstream of Boothferry Bridge. Therefore it is reasonable to believe that the system could cope with a 50% increase in the BOD_5 inloads in all the major sources to the tidal Ouse and still maintain the water quality in the tidal Ouse.

7.3.3 Scenario 3: Dynamic optimisation with current discharge locations and current abstraction levels

The last scenario to be discussed assumes that effluent from the industries and STWs in Selby are discharged at their current locations around Selby. Water abstraction from the rivers Ouse and Derwent are also fixed at current levels, so increasing abatement from the ETPs is the only option to improve water quality under this scenario. The UWWTD requirements are applied to all STWs as before. The solution under this scenario identifies capital stocks of the ETPs in the industries and STWs that meet water quality targets through effluent abatement alone. A comparison between scenarios then reveals the different impacts on water quality and management costs of an integrated river policy versus current regulation. As they are now the only option to improve water quality, the BOD_5 abatements required in the ETPs are higher than in other scenarios, and more investment is required. The results are reported in Tables 7.7-7.9.

Table 7.7: Dynamic equilibrium with discharge in Selby

	Industry A	Industry B	Industry C	STW A	STW B	STW C	STW D	STW E
K_i^a	3.000	3.000	3.000	4.306	6.557	4.173	41.479	8.939
I_i^a	0.075	0.075	0.075	0.108	0.164	0.104	1.037	0.223
a_i	7.549	7.549	7.549	0.599	1.955	0.498	7.902	2.954
C_i	0.185	0.185	0.185	0.263	0.395	0.255	2.360	0.533
Ouse (m^3s^{-1})			Derwent (m^3s^{-1})			X		
0.637			3.530			40.700		

Table 7.8: Water qualities at monitoring sites with discharge in Selby (Dynamic)

Site	Selby	Long Drax		Boothferry Bridge	
Cell	Q_{180}	Q_{192}	Q_{193}	Q_{197}	Q_{199}
DO%	15.000	23.311	26.129	37.770	37.381

Table 7.9: Cost of river management with discharge in Selby (Dynamic)

	Abatement	Abstraction	Relocation	Total
Cost (m£)	4.362	5.541	N/A	9.903

It should be stressed that the water quality target under this scenario is less stringent than in other scenarios. The original water quality target for 5%ile DO% over 30% was not applied because the BOD_5 abatement required was so high that almost no BOD_5 could be discharged. Part of the problem is that the predictions of the simplified system of water quality functions were inaccurate when the BOD_5 discharge from the industries was very low, as it was not calibrated on data with low BOD_5 discharge. To avoid these difficulties, a water quality target of 15% DO% was chosen for this scenario. The water quality around Long Drax was still struggling to reach the 30% due to the persistent DO consumption due to effluent discharge, but water quality became much better at Boothferry Bridge after the dilution effects from the Aire and Don tributaries. Heavy DO sag that is sufficient to prevent the return of salmon, could be seen around Selby and Long Drax even after a significant improvement in BOD_5 abatement.

The required BOD_5 abatement in the ETPs of STWs were, as before, due to the constraint of the UWWTD, while the industries were required to abate much more BOD_5 under this scenario than before. Specifically the industries in Selby were required to abate 22.647 t/d BOD_5 from an aggregate BOD_5 inload of 25.509 t/d. This is equivalent to an aggregate effluent discharge of only 2.8 t/d BOD_5 for the three industries, about a half of their current discharges.

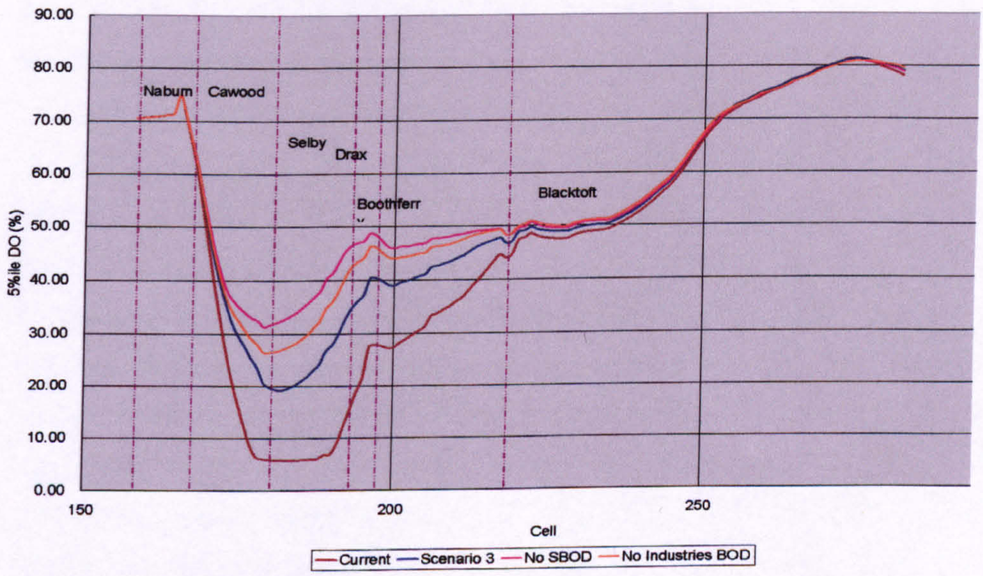

Figure 7.2 The simulations of DO% in the tidal Ouse (a) under current effluents conditions; (b) under the dynamic equilibrium of scenario 3; (b) without BOD_5 discharge from all the Selby sources; and (d) without BOD_5 discharge from the Selby industries.

The increased abatement requirements under this scenario imply higher investment in ETPs. Since the abatement cost function was derived from the data on STWs (due to data shortage discussed in 7.2.2), the abatement costs for the industries are not precise. However comparison between abatement costs under the different scenarios nevertheless sheds light on the problems faced by the Selby industries. Under this scenario, the annual abatement cost of ETP in each industry was 0.185 m£, 50% more than the cost under an integrated river management scenario which handled a 50% increase in BOD_5 inload and still complied with the 30% DO% targets. Furthermore, under the scenario 3, the water quality target of 30% DO% was not attainable, as shown in the Figure 7.2 of the QUESTS1D simulations.

Figure 7.2 simulates the 5%ile DO% in the tidal Ouse under different conditions when the effluents are discharged at Selby without relocation. The brown line predicts DO% under current discharge conditions, which would result in very severe DO sag around Selby and Long Drax in low flow years. The DO% in the worst area would fall to just 5% due to the DO consumption of pollutants. It could be improved to 20% DO% under optimum abatement as shown by the blue line (the simulation from QUESTS1D model is higher than the prediction from the derived system of water quality functions). This is slightly higher than the 15% DO% target but still remains poor quality and prevents the return of salmon. The orange and purple lines indicate the best possible DO% that could result if there was no BOD_5 discharge from the industries in Selby or no BOD_5 discharge from all the Selby sources, including the industries and Barlby and Selby STWs. As indicated by the orange line, the consequent 5%ile DO% was still significantly less than 30% even when no BOD_5 was discharged from the industries. This implies a 100% abatement of the BOD_5 inload, which is far beyond the capabilities of current ETPs in Selby. The purple line, which resulted in just about 30% DO%, assumed the BOD_5 discharges from the Barlby and Selby STWs could also be completely removed. Therefore, the QUESTS1D simulations produce a very clear indication that with the current discharge locations in Selby, it is not economically practical to achieve the target of 30% DO% in the tidal Ouse in low flow years such as 1996.

Although the total costs under scenario 3 are somewhat lower than those in other scenarios, this scenario may not be favourable to either the EA or the industries. Even for an unacceptable water quality target to the EA, 15% DO% at

5%ile, more than 94% of BOD_5 inload has to be abated in the ETPs of industries. This requires a higher efficiency of BOD_5 treatment than is currently in operation in any industrial ETP. Current technologies applied in the industries are unlikely to achieve this target. If new abatement were to be adopted in the future, the costs could be much different to the cost function derived from current data. The practical and management difficulties in installing new ETPs should also be taken into account when comparing the different scenarios. Even more than that, the reduced DO% target will probably not be acceptable to other interested parties.

7.3.4 Shadow Costs of Water Quality

As shown by table 7.8, under scenario 3 the only binding water quality target constraint was at cell 180 in Selby. As a result, for the dynamic equilibrium of scenario 3, we have $\lambda_s = 0$ (the shadow cost of water quality equals zero) for all the four points except for λ_s at cell 180. This simplifies the calculation of the shadow cost of water quality. This is mainly due to the geographic location of effluent discharges from the Selby sources. For similar reasons, the calculation of λ_s at the dynamic equilibria under scenario 1 and scenario 2 can also be simplified[9]. Since the dynamic equilibria of the three scenarios were actually approximated by the static system defined by Eqs. (7.10) – (7.12), the only non-zero λ_s under each scenario can be estimated approximately from the first order necessary conditions of static optimisation. This is so because the marginal cost of abatement $\partial C_i^*(\cdot)/\partial a_i$ and the marginal effect on water quality of abatement $\partial f_s^*(\cdot)/\partial a_i$ can be calculated. The estimated shadow prices of water qualities at the binding points under different scenarios is 0.023 m£/DO% at cell 199 under scenario 1, 0.074 m£/DO% at cell 199 under scenario 2, and 0.037 m£/DO% at cell 180 under scenario 3. The values of shadow prices indicate the changes which would arise in the minimized cost of river management under

[9] Under the scenario 1 and 2, both cell 180 and cell 199 are binding. But under these two scenarios, the effluents of Selby were discharged from 12.472km upstream of the Trent Falls. According to the transfer coefficients matrix shown in section 3.4.3, the effluents would have no effects on the water quality at Selby, which means $\partial f_{180}^*(\cdot)/\partial a_i = 0$ for Selby. This produces the situation in which $\sum_s \lambda_s \cdot \dfrac{\partial f_s^*(\cdot)}{\partial a_i} = \lambda_{199} \cdot \dfrac{\partial f_{199}^*(\cdot)}{\partial a_i} = \dfrac{\partial C_i^*(\cdot)}{\partial a_i}$ for scenario 1 and 2.

these scenarios, if a small change were to be made in the water quality target. It should be pointed out that the water quality target was 30% at 5%ile for scenarios 1 and 2, but only 15% at 5%ile for scenario 3, (recall that there is no change in the location of effluent discharges from the Selby sources under scenario 3.

7.4 The investment path

The equilibria discussed above are, as explained, not the exact dynamic equilibria but rough approximations obtained by solving a current-value cost minimisation problem instead of a present-value problem. The equilibria found by this method would only be accurate if discounting could be ignored, which is often not the case. Although the current-value estimations of the above three scenarios do provide estimates of the equilibria for capital stocks and investment levels, they do not give information about the saddle patterns; neither do they provide information regarding the interdependent variation in capital and investment. In order to investigate these dynamic characteristics of capital stocks and investment levels, and to identify the saddle arms of the equilibria, the analysis below tries to solve the dynamic system described by Eqs (7.4) – (7.6), i.e. the system of equations which describes the discounted net present-value of cost rather than current-value of total cost. The dynamic system can only be solved analytically when simplified cost and abatement functions are assumed.

As discussed in Chapter 4, the steady state equilibrium of the optimised dynamic systems of $\dot{I}_i^j, \dot{k}_i^j$ is saddle-point equilibrium. This implies that the dynamic optimum can only be reached by following the stable saddle arm. For the dynamic system of $\dot{I}_i^j, \dot{k}_i^j$, the saddle arm leading to the dynamic optimum indicates the optimal investment for each level of capital stock, through which the system can approach to the dynamic equilibrium of cost minimisation. This stable saddle arm is therefore regarded as the optimal investment path, i.e. the investment level to be allocated at any time to reach the steady state equilibrium of the dynamic system.

In this section, I will try to identify the optimal investment path which the Selby industries should follow to manipulate the capital stocks of their ETPs following the dynamic equations that describe the variation of I_i^j, k_i^j. Changes in the STWs were not considered because of the constraints of the UWWTD. The three industries in Selby were treated identically in the proceeding analysis, therefore the dynamic system for their investment and capital status are identical, and so are their investment paths. As explained in Chapter 4, the dynamic systems described by $\dot{\mu}_i^j, \dot{k}_i^j$ and $\dot{I}_i^j, \dot{k}_i^j$ describe the same dynamic change through different variables and thus portray the same change on different phase planes. To give more concrete results, we choose the dynamic system of $\dot{I}_i^j, \dot{k}_i^j$ for the analysis. Since abstraction was treated as *static* variable, the capital and investment in the dynamic analysis are for abatement only, $j \equiv a$. Other factors such as water abstraction, STWs abatement and Selby effluent discharge location were assumed unchanged from their current-values as in scenario 3. This is for two reasons: (a) the capital stock for Selby industries would be higher than current, as discussed in 7.3.3, hence there is a demand to upgrade abatement from current levels, hence the optimal investment path is of interest; and (b) due to the water quality recovery after effluent discharge, the water quality target was binding only at cell 180 in Selby, making the calculation simpler.

7.4.1 Dynamic equilibrium of I_i^a and k_i^a

The first step in analysing the dynamic system of $\dot{I}_i^a, \dot{k}_i^a$ is to locate the dynamic optimum of I_i^a and k_i^a. This is not available for the dynamic optimisation problem described by Eqs (7.4) – (7.6) with the original cost and abatement functions. Simplifications to the original cost and abatement functions were made in order to make it possible to identify the steady state equilibrium of dynamic optimisation. Although this doubtlessly reduces the empirical relevance of the results, it does help us to explore the characteristic behaviour of the dynamic system, and to investigate the issues confronting the industries. It needs to be noted, however, that the results are illustrative only and cannot be used for the purpose of policy making in reality. The equation system in $\dot{I}_i^a, \dot{k}_i^a$ describing the dynamics of I_i^a and k_i^a has been discussed in section 4.3.3.b of Chapter 4 as following.

$$\dot{k}_i^a = I_i^{a*} - \delta_i^a k_i^{a*} = F(k_i^a, I_i^a) \quad \ldots(4.56)$$

$$\dot{I}_i^a = -\frac{1}{C_i''(I_i^{a*})} \cdot [-(r+\delta_i^a) \cdot C_i'(I_i^{a*}) - \frac{\partial C_i^*(\cdot)}{\partial k_i^a} + \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^a}] = G(k_i^a, I_i^a) \ldots(4.65)$$

The dynamic steady state optimum of I_i^a and k_i^a is the point at which $\dot{I}_i^a = 0$ and $\dot{k}_i^a = 0$. This cannot be solved without knowing the value of λ_s. This problem must therefore be solved numerically with the values of the shadow costs of water quality at the five points to find the dynamic optimum. However, the shadow price of water quality at the dynamic optimum described by Eqs. (7.4) – (7.6) cannot be calculated directly as was described in 7.3.4 for situations in which is actually a static equilibrium. For cell 180 in Selby under scenario 3, it gives $f_{180}^*(\cdot) = 15$. Putting this together with the equations (4.56) and (4.65), λ_{180}, I_i^{a*} and k_i^{a*} could be found, if the following simplifications made to the cost and abatement functions of industries:

(a) the abatement cost function specified by Eq (7.3) is simplified as below:

$$C_i^a(K_i^a, I_i^a) = \underset{stedv:0.520}{-0.320} + \underset{0.078}{0.6}\ln(K^a) + \underset{0.492}{0.07}\ln(I^a), \qquad (R^2 = 0.874) \ldots(7.13),$$

and

(b) the effluent treatment capacity function for the ETPs in the industries, Eq (7.1) is replaced by a simplified function, in which the effluent treatment capacity is a linear function of the capital value of ETPs,

$$a_{ind}(K_{ind}^a) = \underset{stedv:0.400}{0.69} + \underset{0.000}{1.641} * K_{ind}^a \qquad (R^2 = 0.915) \quad \ldots(7.14).$$

The simplifications are assumed to allow simpler forms of derivatives so the system of equations could be solved. The cost function accordingly takes a simpler form than in Eq (7.3). Substituting Eqs (7.13) and (7.14) into (4.56) and (4.65), and setting $f_{180}^*(\cdot) = 15$, the optimal values of I_i^{a*}, k_i^{a*}, λ_{180} and μ_i^{a*} for the Selby industries were calculated as in Tables 7.10 and 7.11.

Table 7.10: Dynamic Optimum of River Management Options

	Industry A	Industry B	Industry C	STW A	STW B	STW C	STW D	STW E
k_i^{a*}	4.180	4.180	4.180	4.306	6.557	4.173	41.479	8.939
I_i^{a*}	0.105	0.105	0.105	0.108	0.164	0.104	1.037	0.223
a_i^*	7.549	7.549	7.549	0.599	1.955	0.498	7.902	2.954
μ_i^{a*}	-0.670	-0.670	-0.670	N/A	N/A	N/A	N/A	N/A

Ouse (m^3s^{-1})	Derwent (m^3s^{-1})	X
0.637	3.530	40.700

Table 7.11: Water qualities at monitoring sites (Dynamic Optimum)

Site	Selby	Long Drax		Boothferry Bridge	
Cell	Q_{180}	Q_{192}	Q_{193}	Q_{197}	Q_{199}
DO%	15.000	27.847	30.859	42.451	41.994
λ_s	0.033[10]	0	0	0	0

The results suggest that at the dynamic optimum, capital stocks and investment levels for the three industries in Selby are identical, with identical abatement levels at 7.549 t/d BOD_5, as in scenario 3. The STWs abatement was fixed at their current level, as was water abstraction from the Ouse and Derwent. Effluent from Selby sources was discharged at their current locations which is about 40.7 km upstream of the Trent Falls. The industries' abatement levels were same as those in scenario 3 because they were both determined by the binding water quality condition $f_{180}^*(\cdot) = 15$. The two analyses produced different values of λ_{180} however, because of the changes in the cost functions and the differences between the static and dynamic optimisation. At the dynamic optimum $\lambda_{180} = 0.037$ m£/DO%, implying a cost increase of 0.037 m£ for the water quality management if the water quality target at Selby were set higher at the margin.

The shadow price of capital accumulation in abatement, μ_i^{a*}, represents the price of one extra unit of capital stock at the dynamic optimum in units of

[10] The value of λ_{180} is different to that under scenario 3, which is 0.037 m£/DO%. This is due to the simplified cost abatement functions (7.13) and (7.14).

current-value cost at the time of equilibrium. It is the imputed marginal value of state variable, in this research the capital stock of abatement k_i^{a*}, which is equal to the marginal increase to the current-value of minimum cost that incurred to the industry when the capital stock of abatement k_i^{a*} increases by a small amount. At the dynamic optimum of this research, $\mu_i^{a*} = -0.627$, implying that the current-value cost of pollution abatement would be roughly 0.627 m£ less for each 1 m£ more capital stock of abatement made in the ETP. However, this is the cost saving to the industry only, not taking into account the environmental cost of water quality at the same time caused by pollution. Therefore the value of μ_i^{a*} at the dynamic optimum should be so to equate the internal and external rate of return of capital as indicated by the equation below, taking into account the environmental cost of water quality, which was explained in details in the section 4.3.2 of Chapter 4:

$$\frac{1}{\mu_i^{a*}} \cdot \left(\frac{\partial C_i^*(\cdot)}{\partial k_i^a} - \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^a}\right) - \delta_i^a = r \qquad \ldots(4.63).$$

Doubtless that the simplifications made have weakened the results. For the same reason, the cost of integrated river management under the dynamic optimum was not evaluated. Nevertheless, the results do enable us to explore the stability of the real dynamic equilibrium of integrated water quality management.

7.4.2 Stability of dynamic optimum

Equations (4.56) and (4.65) form a 2×2 simultaneous system of differential equations for the investment and capital stock. Since this system is non-linear, we can only investigate the local stability of the steady state equilibrium using the linearization method (Gandolfo 1997). The method of linearization has been discussed in Section 4.3.3. After linearization, the Jacobian matrix of the dynamic system defined by Eqs (4.56) and (4.65) is represented by

$$A \equiv \begin{bmatrix} \dfrac{\partial F(\cdot)}{\partial k_i^a}, & \dfrac{\partial F(\cdot)}{\partial I_i^a} \\ \dfrac{\partial G(\cdot)}{\partial k_i^a}, & \dfrac{\partial G(\cdot)}{\partial I_i^a} \end{bmatrix}_{\left(\begin{smallmatrix} k_i^a = k_i^{a*} \\ I_i^a = I_i^{a*} \end{smallmatrix}\right)} = \begin{bmatrix} a_{11}, & a_{12} \\ a_{21}, & a_{22} \end{bmatrix}.$$

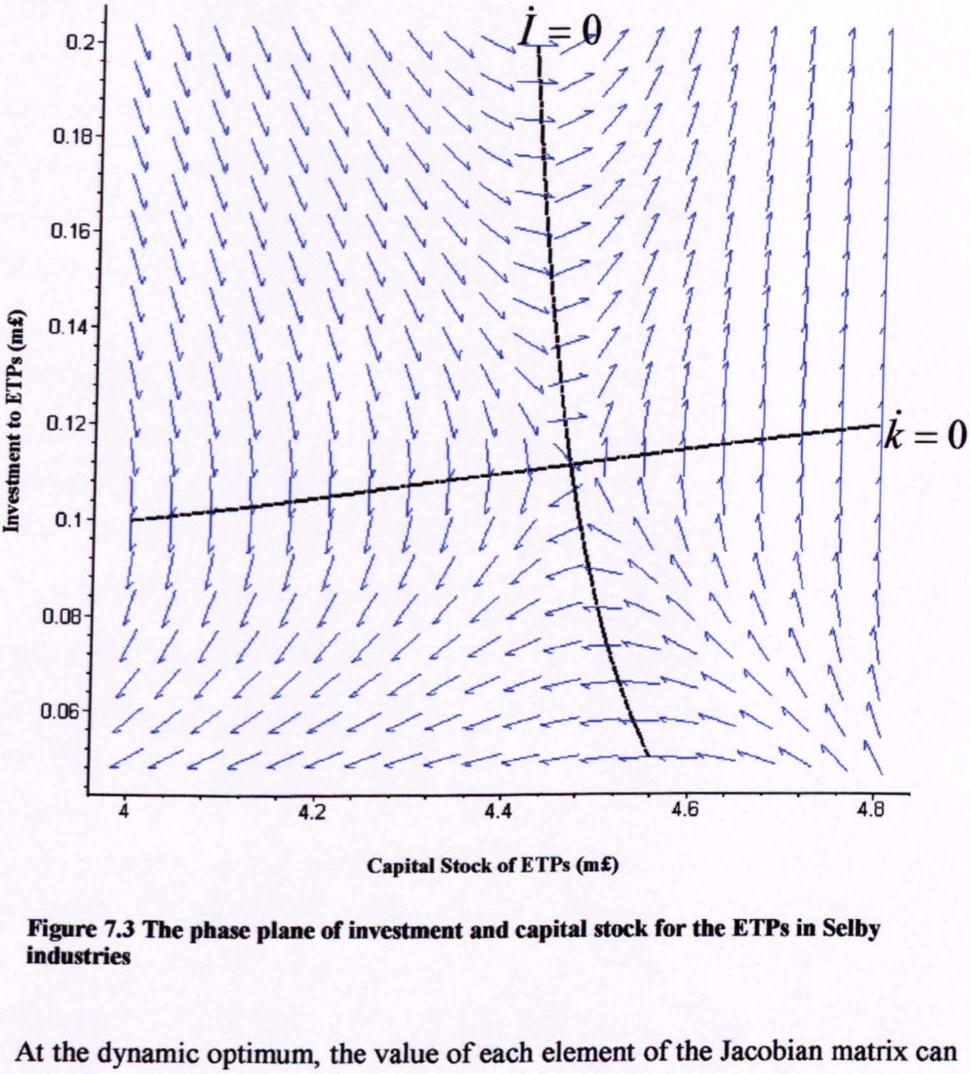

Figure 7.3 The phase plane of investment and capital stock for the ETPs in Selby industries

At the dynamic optimum, the value of each element of the Jacobian matrix can be calculated as below,

$a_{11} = -\delta_i^j = -0.025$,

$a_{12} = 1$,

$$a_{21} = -C_i^{"}(I_i^{j*})^{-1} \cdot (\sum_s \lambda_s \cdot \frac{\partial^2 f_s^*(\cdot)}{\partial k_i^{j2}} - \frac{\partial^2 C_i^*(\cdot)}{\partial k_i^{j2}}) = 0.083,$$

$a_{22} = r + \delta_i^j = 0.070$.

The determinant of the Jacobian matrix is $\det A = a_{11}a_{22} - a_{12}a_{21} = -0.085 < 0$, therefore the non-linear differential system has a saddle-point equilibrium in the neighbourhood of steady state point, which is regarded as the optimum of water quality management scheme in 7.4.1. The phase plane representing the dynamics of investment and capital stock is displayed by Figure 7.3. This phase plane was produced by Maple 8, and describes the motion of the system for different combinations of existing capital stock and on-going investment. Trajectories in the phase plane portrayed the changes in the investment and capital stock from each initial combination (I_i, k_i) following their motion equations (4.56) and (4.65).

Trajectories in the phase plane are indicated by the blue arrows in Figure 7.3, while the two black lines show the isoclines of $\dot{I} = 0$ and $\dot{k} = 0$. Their intersection at (4.180, 0.105) is the dynamic optimum of investment and capital stock of the ETPs. The maximum level of capital stock in ETPs is $k^a_{i,\max} = 4.795$ since total BOD_5 abatement cannot exceed total BOD_5 inload. Each point in the phase plane represents a combination of capital stock and investment level for the ETPs. The two isoclines $\dot{I} = 0$ and $\dot{k} = 0$ divide the phase plane into four sections each of which produces different directions of change. According to the properties of saddle-point equilibria (Gandolfo 1997; Xepapadeas 1997; Barro and Sala-i-martin 1999), only one investment path will eventually converge to the dynamic optimum and only the points initiated from the path will reach the dynamic optimum by following the motion of trajectory. This path is called the stable saddle arm to differentiate it from the unstable saddle arm, which moves away from the optimum. The stable saddle arm for the phase plane of investment and capital stock in Figure 7.3 reflects the optimal investment decisions at each point of time for the ETPs, defining an investment path to the dynamic optimum.

7.4.3 Investment path

Because the non-linear system of simultaneous differential equations had a saddle-point equilibrium around the neighbourhood of the steady state equilibrium, the two saddle arms of the phase plane in Figure 7.3 may be determined from the eigenvectors of the linearised simultaneous system (Shone 2002). The linearised simultaneous system of differential equations for (4.56) and (4.65) was defined by $\dot{Y} = A(Y - Y^*)$ where $Y = \begin{bmatrix} k_i^a \\ I_i^a \end{bmatrix}$ and $A \equiv \begin{bmatrix} a_{11}, & a_{12} \\ a_{21}, & a_{22} \end{bmatrix}$ as defined in 7.4.2. The stable path of the saddle-point corresponds to a negative eigenvalue (Gandolfo 1997), one of the solutions to $|A - \eta I| = 0$. The solutions could be found by solving $\eta^2 - (a_{11} + a_{22})\eta + (a_{11}a_{22} - a_{12}a_{21}) = 0$ ...(7.16).

The two values of η for the simultaneous system defined above are $\eta_1 = 0.314$ and $\eta_2 = -0.269$. Here η_1 and η_2 are the eigenvalues (or characteristic roots) of the Jacobian matrix A. Each of the two eigenvalues is associated with an eigenvector (or characteristic vector) for the linearised system of differential equations. The two eigenvectors are determined from the two values of λ as below (Gandolfo 1997; Shone 2002).

The unstable arm r:

$$(a_{11} - \eta_1)(k_i^a - k_i^{a*}) + a_{12}(I_i^a - I_i^{a*}) = 0$$

or

$$a_{21}(k_i^a - k_i^{a*}) + (a_{22} - \eta_1)(I_i^a - I_i^{a*}) = 0,$$

and the eigenvector is $\begin{bmatrix} 1 \\ \dfrac{\eta_1 - a_{11}}{a_{12}} \end{bmatrix} or \begin{bmatrix} 1 \\ \dfrac{a_{21}}{\eta_1 - a_{22}} \end{bmatrix} = \begin{bmatrix} 1 \\ 0.339 \end{bmatrix}$.

The stable arm s:

$$(a_{11} - \eta_2)(k_i^a - k_i^{a*}) + a_{12}(I_i^a - I_i^{a*}) = 0$$

or

$$a_{21}(k_i^a - k_i^{a*}) + (a_{22} - \eta_2)(I_i^a - I_i^{a*}) = 0,$$

and the eigenvector is $\begin{bmatrix} 1 \\ \dfrac{\eta_2 - a_{11}}{a_{12}} \end{bmatrix} or \begin{bmatrix} 1 \\ \dfrac{a_{21}}{\eta_2 - a_{22}} \end{bmatrix} = \begin{bmatrix} 1 \\ -0.244 \end{bmatrix}$.

After simplification, the two saddle arms in the phase plane of Figure 7.3 are illustrated in Figure 7.4 together with the isoclines $\dot{I} = 0$ and $\dot{k} = 0$ and the steady state equilibrium - produced using Maple 8 (Shone 2002). The two saddle arms across the steady state equilibrium are marked by the blue line r and red line s. As discussed before, the line r represents the unstable arm with direction away from the steady state equilibrium while line s represents the locus of all points moving to the steady state equilibrium following the arrows in Figure 7.3. All (I_i, k_i) pairs along line s will eventually converge to the steady state equilibrium following the direction indicated by the arrows. When approaching the equilibrium, the motion represented by $\dot{I}$ and $\dot{k}$ also tends to zero so it will, in theory, take infinite time to reach the equilibrium.

Because of the saddle-point stability of the dynamic steady state equilibrium, all other points on the phase diagram will ultimately diverge away from the steady state equilibrium, as indicated by the arrows. The probability that the initial state of the system happens to be located on the stable arm approaches zero. In a controlled system, however, choice of the control variable makes it possible to locate on the stable arm. For example, in the phase plane of Figure 7.4, the ETPs of the industries could choose their initial investment level freely, so as to position the starting point of their investment trajectory on the stable arm s in accordance with their initial capital stock. More specifically, if the stable arm for the simplified dynamic system was known sufficiently accurately, the linear function of the stable arm (by substituting $\eta_2 = -0.269$ into the function of stable arm given above), $I_i^a = -0.244 \cdot k_i^a + 1.202$ could be utilized to find the appropriate investment levels corresponding to the various capital stocks ranging from zero to $k_{i,\max}^a$ (4.878 m£).

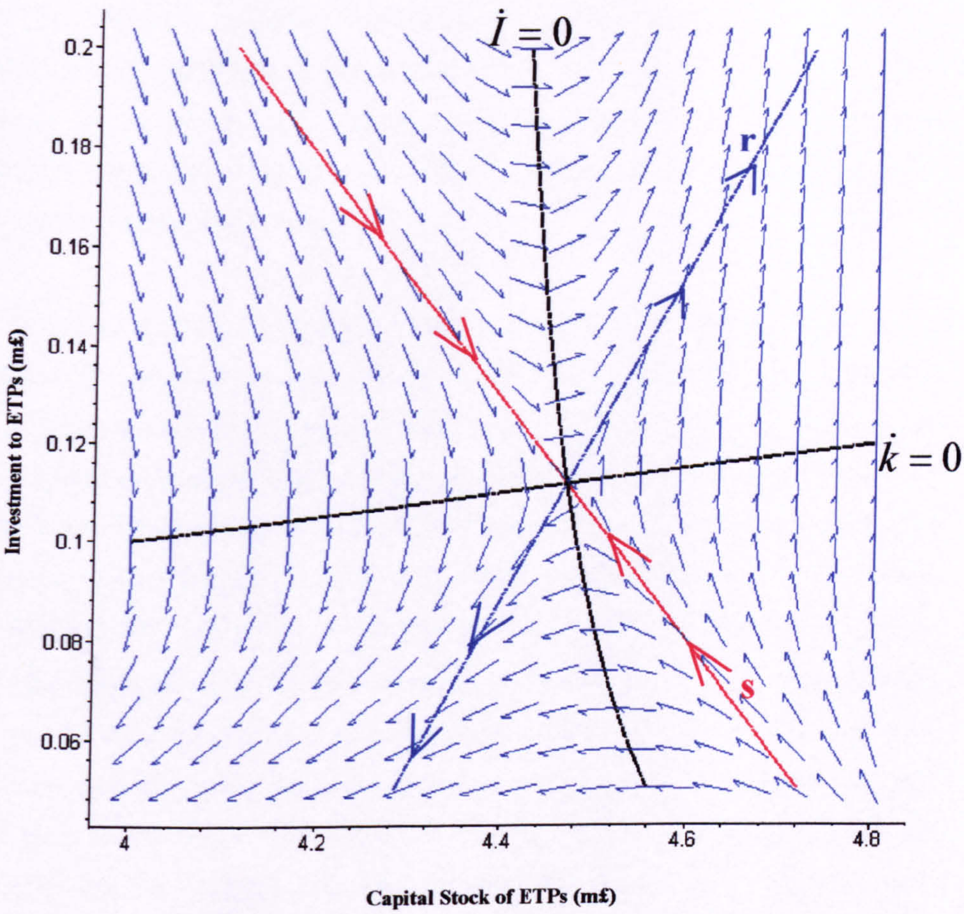

Figure 7.4 The saddle arms of steady state equilibrium

7.5 Conclusion

In this chapter, I investigated the dynamic equilibrium of water quality management, minimising the overall cost of water quality management over time in the tidal Ouse subject to particular water quality targets. As already mentioned in the introduction, the limited dataset and nonlinearity of the system has made the analysis in this chapter less rigorous than the conclusions produced from the static analysis in Chapter 6. Nonetheless, the analysis in this chapter is still indicative of the control problem for improving water quality. It provides a useful discussion of the way to develop policy, since there are as yet relatively few studies in this area (Xepapadeas 1997; Dellink 2005). The chapter investigated only the most common option for improving water quality, abatement in the ETPs of pollution sources. Different scenarios were designed to test the impacts of various conditions on the dynamic equilibrium. The chapter also discussed the local stability of the steady state equilibrium for the simplified dynamic system and identified the stable saddle arm of the saddle-point equilibrium, analysed using a simplified "current-value" approach.

The three different scenarios in section 7.3 discussed three different policy options. Not surprisingly, scenario 1 in section 7.3.1, whose conditions are similar to the UC scenario in Chapter 6, pointed to similar solutions. In scenario 2, I investigated the possible change in the effluent inload to the STWs and industries. The chosen 50% increase in the BOD_5 inload to all the STWs and Selby industries reflected the demands of growth in Yorkshire and Humber area. In the equilibrium analysis, the only changes in response to the increased BOD_5 inload were found in the abatement levels of ETPs in the Selby industries. However, adopting the solution of an integrated river policy with relocation of Selby effluent discharge, the total abatement of BOD_5 required by the Selby industries was still 1 t/d less than their current level, and this produced much better water quality than that currently observed despite the 50% increase in BOD_5 inload. The river also absorbed 2 t/d more BOD_5 than at present, by following the integrated policy solution. In the final scenario, I evaluated the consequence of current policy without changing effluent discharge location or water abstraction. This result indicated that more abatement would be required from the Selby sources and that water quality in Selby would be lower than under the other scenarios. This demands higher investment in ETPs and requires the introduction of new technology or wastewater management.

The dynamic optimum for achieving the least total cost of water quality management over time was represented by optimal levels of capital stock and investment level for the ETPs in the Selby industries, while abatement levels of the STWs were constrained by UWWTD. The global stability of the dynamic equilibrium could not be investigated, however the local stability around the neighbourhood of dynamic optimum was investigated through linearization. The Jacobian matrix of the linear approximation suggests that the optimal equilibrium is a saddle-point equilibrium, which is only conditionally stable. A method to achieve the stable equilibrium was also discussed following identification of the stable saddle arm. Since the industries could (relatively free) select their investment level, identification of the stable saddle arm indicates how the initial point of investment could be selected, knowing the start-up level of abatement capital, to ensure that on-going investment could drive the system following the stable saddle arm to the dynamic equilibrium.

One assumption has to be emphasized before closing the discussion of these two chapters. The cost data for effluent abatement were aggregated to yield functions for STWs and industries respectively. The situation for the STWs is constrained by the UWWTD and the tributaries on which the STWs were located, so abatement levels in the integrated policy solution were differentiated for each STW according to their own inload. The Selby industries, on the other hand, share the same cost function and discharge at almost the same location under all the scenarios discussed in these two chapters. This was mentioned in Chapter 6.2.2a when deriving the cost function for the Selby industries. They were treated as a single source with the same water quality and abatement cost function, although in reality the industries have quite different abatement levels and capacities. This assumption was applied throughout the analyses in Chapter 6 and 7. In the next chapter, mechanisms that allow efficient allocation of abatement responsibilities among the sources will be discussed, referring to the different policy instruments discussed in Chapter 4. This will allow the integrated river policy to take into account the difference between industries.

Chapter 8 Policy Implications

8.1 Introduction

The optimisations in last two chapters are based on the important assumption that the industries in Selby have identical effluent abatement capacities and costs. Therefore the impacts of their effluents on the river are identical too. This is not the case in reality. The assumption simplified analysis of the problem and made it tractable given the limitations of the data. From the perspective of the physical pollution problem, the assumption is reasonable in that the river at that point is close to being completely mixed (Freestone 2001) and the effluents from various sources are of similar composition and effect. However, it does leave the question of the allocation of pollution burden among the Selby industries, which is to be determined for the practical implementation of an integrated river policy in the tidal Ouse.

The results from the static and dynamic optimisations indicate the optimal levels of total abatement under various scenarios. The allocation of that abatement is not considered in the optimisations but can be achieved through specific policy instruments, which will be discussed in this chapter. The policy instruments that are discussed most in the literatures are command and control (CAC) and market-based instruments (MBIs). The CAC approach is more favoured in the current administration and management, but has been accused of lacking flexibility and being cost ineffective by many economists (Hanley *et al.* 1997; Tietenberg 2001). The MBIs generally recommended as alternatives to command and control involve two policy instruments, emission charges and tradable pollution permits (TPP). The MBIs are more flexible in achieving compliance, more cost effective and provide a continuous incentive to reduce effluents, which make them more appealing to many economists. However, each of these MBIs is likely to be problematic in managing water quality in the tidal Ouse.

The discussions in this chapter are based on two scenarios. According to the discussion in this chapter, the tax and subsidy scheme (TSS) derived from emission charge and discussed in the section 4.2.3.b, was considered suitable to act as a complementary policy instrument in the tidal Ouse together with the CAC approach. The TPP system was believed not to be very suitable to control the non-uniformly mixed pollutant along the whole length of a river system such as the tidal Ouse, particularly because of the spatial differences along the river length and the multiple constraint points on water quality in the river system. The small number of industries also makes it difficult for the TPP system to be effective (Tietenberg 2006).

This chapter reviews the general selection criteria for policy instruments, and the effectiveness of each instrument within the study setting against those criteria. We then discuss the possible implementation of TSS and TPP systems under two different scenarios. Finally, the choice of policy instruments is made based on the practical feasibility and the potential policy implications investigated.

8.2 Applications of the policy instruments

In the following section, I investigate specific instruments for water quality management in the tidal Ouse, based on the optimal level of total BOD_5 abatement among the industries obtained from the static and dynamic optimisations. The TSS scheme and TPP system are investigated, and their advantages and disadvantages are discussed given the criteria for policy instrument selection. Two scenarios are considered as representative of the static and dynamic optimisation problems to be solved, the UC scenario in the section 6.3.3 and scenario 3 from the section 7.3.3 (I hereafter call it 'the Selby scenario') in the dynamic analysis.

8.2.1 General criteria for selection of policy instruments

There has been a controversial debate for decades over the choice of policy instrument in environmental management. The instrument that is most commonly implemented takes the form of fixed standards and consents. This kind of CAC regulates the quantity of outputs or inputs, or the technology used in production processes, and is called quantity-based instrument. The other kind of policy instrument is economic or price-based incentive instrument, and uses taxes or prices to regulate emissions. As they act through market forces to regulate the emission and abatement without obligatory commands they are called MBIs (Turner *et al.* 1994; Hanley *et al.* 1997; Perman *et al.* 1999). There are also other forms of policy instruments such as voluntary regulations and liability rules, and moral suasion (Common 1995; Perman *et al.* 1999), but they are likely to act as complementary instruments only. In the case of the tidal Ouse, research has been done to explore the possibility of implementation of MBIs, in particular the TPP system, for water quality management (Cashman *et al.* 1999), but no MBI instrument has yet been implemented. The current water abstraction license in the UK has some of the characteristics of MBIs, as it is tradable among the agents taking water from the river, but in reality there have hardly been any trades since the license system was introduced.

The literature suggests that MBIs have considerable advantages over CAC in some criteria, such as cost-effectiveness, information requirements, flexibility and dynamic incentive. The criteria are as follows:

- Cost-effectiveness: whether the instrument can reach the target at the least cost?
- Information requirement: what information the instrument requires the control authority to posses to effectively use the instrument?
- Flexibility: capability of the instrument being adapted quickly and cheaply to accommodate the changes in economic circumstances?
- Dynamic incentive: does the instrument encourage the adoption of new technologies or new production process to continuously reduce pollution?

On the other hand, CAC approaches are not always inferior to MBIs. Here are some examples of criteria by which the CAC approaches perform better.

- Dependability: how reliable does the instrument deliver the desired target?
- Monitoring and enforcement: feasibility and cost to monitor pollution abatement and to enforce compliance.
- Political acceptability: the distributional effects of instruments.
- Costs under uncertainty: the cost when the instrument is used with incorrect information or when dealing with environmental incidents and hazards?

In this chapter, the TSS scheme and TPP system are investigated for their potential application to the tidal Ouse in delivering the optimal solutions obtained from the analysis. It is worth stressing here that when the total amount of water abstraction in the catchment is more flexible, the two instruments could also be evaluated for inclusion in an integrated river policy to regulate both effluent discharges and water abstractions based on their impacts on the water quality, as shown in Chapter 4. For simplicity, this chapter only considered the applications to effluent discharges.

8.2.2. Tax and Subsidy Scheme (TSS)

One important advantage of the effluent charge or tax scheme is its relative simplicity in administration and management. The EA only needs to set an appropriate tax/subsidy rate for the industries in Selby. The industries will choose their own level of abatement according to their abatement cost. In this thesis, abatement cost was assumed to be the same across the three industries in the absence of industry-specific data, and as a result this leads to the same abatement levels among them. This is not the case in reality and the industries will end up with different abatement levels.

There are two disadvantages to the TSS scheme. The first is the financial burden it imposes on industry. This is the main concern of the industries, particularly in current trading conditions. The second is the uncertainty surrounding the water quality obtained from any given effluent tax rate. The effluent tax cannot guarantee water quality without incurring monitoring and enforcement costs to the EA that may initially be greater than those incurred in the implementation of pollution consents, and the effects are often lagged. Therefore it ends up as a 'trial and error' process to find the appropriate effluent tax rate. The unique feature of TSS is that it potentially combines tax and subsidy in one system, and takes into account the necessary abatement level needed for target compliance. The industries either pay a tax or receive a subsidy according to their abatement effort in relation to a target effluent discharge level, determined by the EA. Under this scheme, industries are encouraged to implement the optimal solution. The industries pay taxes on the amount of effluent over the target but receive subsidies if they are willing to abate more. Therefore the financial burden may be minimized. The industries still have flexibility in choosing the abatement level they want, but also have an incentive to adopt more efficient abatement technology due to tax saving or subsidy seeking. At the same time, cost effectiveness may be achieved, as the TSS is a cost effective instrument. The difficulties for the EA arise from the quantification of the appropriate tax/subsidy rate and the determination of target effluent discharge level for each industry.

Following the discussion of policy instruments in sections 4.2.3.b and 4.3.4.a in Chapter 4, under both static and dynamic scenarios, the TSS needs set the tax/subsidy rate at each WQM site as shown below: $t_{es} = -\lambda_s \cdot \frac{\partial f_s(\cdot)}{\partial E_s}$ and $t_{as} = -\lambda_s \cdot \frac{\partial f_s(\cdot)}{\partial H_s}$, for effluent discharge or water abstraction respectively. In another word, it equals the product of the shadow price of water quality at the WQM site and the impacts on the WQM site of increasing effluents or water abstraction. For each pollution 'source', either effluent discharge or water abstraction, the total tax or subsidy to be paid or received is, as defined in these two sections above, $T_{ie} = \sum_s \left(e_i - e_i^0\right) \cdot b_{is} \cdot t_{es}$ and $T_{ia} = \sum_s \left(\beta_i - \beta_i^0\right) \cdot d_{is} \cdot t_{as}$, where e_i^0 and β_i^0 are the target levels of effluent discharge and water abstraction for each source, b_{is} and d_{is} are the transfer coefficients indicating the effects of effluents and abstraction from site i to the WQM site , defined in section 4.2.2.a.

In following analyses, $\frac{\partial f_s(\cdot)}{\partial E_s}$ and $\frac{\partial f_s(\cdot)}{\partial H_s}$ were not evaluated, but we could quantify $\frac{\partial f_s(\cdot)}{\partial e_i}$ and $\frac{\partial f_s(\cdot)}{\partial \beta_i}$ through the system of water quality functions. As $\frac{\partial E_s}{\partial e_i} = b_{is}$ and $\frac{\partial H_s}{\partial \beta_i} = d_{is}$, $\frac{\partial f_s(\cdot)}{\partial E_s} \cdot b_{is} = \frac{\partial f_s(\cdot)}{\partial e_i}$ and $\frac{\partial f_s(\cdot)}{\partial H_s} \cdot d_{is} = \frac{\partial f_s(\cdot)}{\partial \beta_i}$. There are

$$T_{ie} = \sum_s \left(e_i - e_i^0\right) \cdot b_{is} \cdot t_{es} = \sum_s -\lambda_s \cdot \left(e_i - e_i^0\right) \cdot \frac{\partial f_s(\cdot)}{\partial e_i} \qquad \ldots(8.1),$$

$$T_{ia} = \sum_s \left(\beta_i - \beta_i^0\right) \cdot d_{is} \cdot t_{as} = \sum_s -\lambda_s \cdot \left(\beta_i - \beta_i^0\right) \cdot \frac{\partial f_s(\cdot)}{\partial \beta_i} \qquad \ldots(8.2).$$

Therefore, the optimal tax/subsidy rate to achieve the water target can now be quantified if the shadow prices of water quality at each WQM site are found. The shadow prices vary at the static and dynamic equilibria, as we indicate below.

- **TSS under the UC scenario**

The optimal result for the UC scenario was derived from static optimisation in section 6.3.3, in which the integrated river policy achieves the water quality target at least cost through a combination of management options. As discussed before, the three industries were treated identically in the static optimisation under the UC scenario, regardless of their specific effluent treatment capacity and effluent inloads. This section discusses the introduction of a TSS into the regulating system under this scenario, to solve the issue of abatement allocation based on the actual situations encountered with the industries.

Under the least cost solution for the UC scenario, the three industries in Selby were required to abate an average of 1.081 t/d of BOD_5 compared with their current discharges which, in 2004, averaged above 5 t/d. This means less abatement effort than their current levels. Once the tax/subsidy rate is determined, each industry will choose the effluent discharge level at which its marginal cost of abatement is equal to the tax rate.

Since none of the industries abstracts a substantial amount of water from the river, the total tax or subsidy to each industry was defined by Eq (8.1). The target effluent discharge level for each industry e_i^0 can be determined by the BOD_5 load to the ETP and the optimal abatement levels required under the UC scenario, which is 1.081 t/d. For the same reason discussed in section 7.4.1, under the UC scenario cell 199 is the only point at which water quality is a binding constraint on the industrial effluent discharges, therefore the shadow cost of water quality at cell 199 can be determined. According to section 7.4.1, $\lambda_{199} = 0.079$ m£/DO%. So the complete form of effluent tax or subsidy to each industry is $T_{ie} = -0.079 \cdot \frac{\partial f_s(\cdot)}{\partial e_i}(e_i - e_i^0)$. In this function, $\frac{\partial f_s(\cdot)}{\partial e_i}$ is the marginal effect of effluent discharged from site i on the water quality at site s, which can be evaluated through the water quality function for cell 199. Under the river flow and effluent condition of the UC scenario, $\frac{\partial f_s(\cdot)}{\partial e_i} = -0.398$ DO%/(t/d BOD_5), and assuming the common abatement costs, the target effluent discharges are 5.499, 4.438 and 12.329 t/d BOD_5 for the industries A, B and C.

To sum up, under the UC scenario, an effective TSS to meet the water quality target at least cost, taking into account the differences among industries, is shown below for the three industries in Selby.

$T_{ie} = 0.031 \cdot (e_A - 5.499)$ for Industry A; and

$T_{ie} = 0.031 \cdot (e_B - 4.438)$ for Industry B; and

$T_{ie} = 0.031 \cdot (e_C - 12.329)$ for Industry C.

Facing this TSS scheme, the industries will choose their abatement levels and effluent discharge accordingly. When the cost function of abatement is accurate enough, the marginal cost of abatement of each industry at its target effluent discharge level should be the same as the tax/subsidy rate it faces. Because of the increasing marginal cost of abatement, a rational industry aiming at minimizing its abatement cost will choose to abate its effluent at the target level. If the industry could mange to abate BOD_5 at less cost than predicted, it would abate more to receive the subsidy. On the other hand, if its marginal cost were higher than the function predicted, the industry would choose to pay the tax rather than abate. The TSS scheme cannot completely remove the "trial and error" process because of the uncertain responses from the industries to the tax and subsidy rate. However, the target effluent discharges clearly indicate the desired effluent discharges from the perspective of the EA. The industries also have an incentive to reduce their abatement cost and to abate more than the required minimum in order to obtain the subsidy. Hence the water quality compliance risk could, to some extent, be reduced.

- **TSS under the Selby scenario**

For the dynamic equilibrium in the Selby scenario, the story is quite different. Although the value of tax/subsidy rate is still determined by the same function and the shadow price of water quality has already been found in 7.4.1, the problem arises from the increased aggregated optimal abatement level that is required to comply the water target, 22.647 t/d BOD_5. In average, that is 7.549 t/d BOD_5 for each industry, which exceeds the BOD_5 inloads to their ETPs for two of the three industries. Therefore it is impossible for the industries with less BOD_5 inloads to reach the average abatement level if they were so required.

In this case, equal abatement levels across the industries are inefficient and impossible. Since the total abatement of BOD_5 effluent under the Selby scenario is 22.647 out of a total 25.509 t/d BOD_5 inload, a 90% removal rate is assumed. Each industry was required to abate 90% of their BOD_5 inload to comply with the abatement target of Selby scenario. Under the Selby scenario, the binding point on water quality is around Selby at cell 180. The shadow price of water quality under this scenario has been evaluated in section 7.4.1, at 0.037 m£/DO%. The marginal effect of effluent discharge from site i on the water quality at site s is $\frac{\partial f_s(\cdot)}{\partial e_i} = -1.201$. Therefore the TSS based on the uniform 90% removal from the industries is shown as below.

$T_{ie} = 0.044 \cdot (e_A - 0.658)$ for Industry A; and

$T_{ie} = 0.044 \cdot (e_B - 0.552)$ for Industry B; and

$T_{ie} = 0.044 \cdot (e_C - 1.341)$ for Industry C.

Comparing the TSS tax/subsidy rates under these two scenarios, the industries are subject to higher tax for their excessive effluent discharges (and higher subsidies for extra effort to reduce effluent discharges) under the Selby scenario than the UC scenario.

The discussions of TSS under the static UC scenario are also applicable to the dynamic equilibrium under the Selby scenario. The industries were treated more equally in the Selby scenario as they remove the same percentage rather than same amount of BOD_5, but the aggregate cost is higher because they are at different levels of marginal cost of abatement. When effluent is discharged at Selby under this scenario, the industries have to abate most of their BOD_5 inloads to comply with their target effluent discharge levels. Instead, if an industry is unable to improve its ETP performance, either because of the large capital investment required or because its abatement level is already high, they may choose to pay the tax for excessive effluent, compromise the water quality as a result.

8.2.3. Tradable Pollution Permit (TPP) System

Along with TSS scheme, the TPP system is another MBI for environmental policy. A detailed comparison of the two MBIs in theory has already been offered in Chapters 2 and 4. One of the advantages of the TPP system prior to the TSS scheme is its certain outcome with regard to the environmental target. Unlike a regulation on the rate of tax or subsidy, the TPP system directly controls total pollution level by the amount of permits issued. Therefore introducing the TPP system into the river policy for the tidal Ouse will ensure the total effluent discharges from the sources match the target level, without "trial and error" readjustment unless the water target itself changes.

But the TPP system also requires particular conditions in order to work effectively, such as low transaction costs, and a reasonably large number of potential traders in the permit market. In the case of the tidal Ouse, if only industrial effluent discharges are involved, the transaction costs of permit trade may be small, because the industries are closely located and know each other well. However the permit market may be too thin to operate effectively (Tietenberg 2006). It is possible to expand the TPP system to the two STWs around Selby or even to those in tributaries, but the UWWTD constraint and their diverse impacts on the water quality will add complexity to permit trading. This will be discussed with the specific examples of UC and Selby scenarios.

It should be stressed that the TPP systems in the static and dynamic analyses are different. Although the method of permit allocation would be the same, the price of permits reflects the different values of permits in the static and dynamic cases. This has been discussed in section 4.3.4.b of Chapter 4. The price of pollution permits in the static case is the same as the tax rate needed to achieve the same target, but is different in the dynamic case, where $P_i = t_i / r$.

If permits are issued for each WQM site, the equilibrium price of permits for each WQM site is $P_{es}^{*} = -\lambda_s \dfrac{\partial f_s(\cdot)}{\partial E_s}$ and $P_{as}^{*} = -\lambda_s \dfrac{\partial f_s(\cdot)}{\partial H_s}$ for effluent and abstraction permits respectively, identical to the tax/subsidy rate for each WQM site in 8.2.2. They become $P_{es}^{*} = -\dfrac{\lambda_s}{r}\dfrac{\partial f_s(\cdot)}{\partial E_s}$ and $P_{as}^{*} = -\dfrac{\lambda_s}{r}\dfrac{\partial f_s(\cdot)}{\partial H_s}$ in the dynamic case. Consequently, the equilibrium prices of pollution permits for each source under the static analysis are equal to the sum of prices at each WQM site weighted by their transfer coefficient. Therefore, we have

$$P_{ei}^{*} = \sum_s b_{is} \cdot P_{es}^{*} = \sum_s -\lambda_s \cdot \frac{\partial f_s(\cdot)}{\partial e_i} \qquad \ldots(8.3),$$

$$P_{ai}^{*} = \sum_s d_{is} \cdot P_{as}^{*} = \sum_s -\lambda_s \cdot \frac{\partial f_s(\cdot)}{\partial \beta_i} \qquad \ldots(8.4).$$

If the pollution source has impacts on more than one WQM site, this complicates trading because traders have to acquire pollution permits from each WQM site they affect. The dynamic equilibrium prices of the pollution permits are determined in the same way as their static counterparts, as discussed above. To simplify the process, a Pollution Offset (PO) system as discussed in section 4.2.3.c may be implemented, in which the permits regulate effluents from each source instead of WQM site, and can be traded between sources at a rate determined by the impacts of each source on all WQM sites, at which the water quality constraint is binding. The rate is defined as in section 4.2.3.c of Chapter 4. With the total amount of pollution permit determined by the static optimisation, the EA could either distribute them to the industries for free (grandfathering) or auction them.

- **TPP under the UC scenario**

Under the UC scenario, no change is suggested to the water abstraction from the rivers Ouse and Derwent. The pollution permit thus only refers to the effluent discharge permits for the three industries. To make the situation simple, effluent permits are assigned to industries instead of to each WQM site, and the trade of permits is based on the PO system. The total amount of effluent permits to be allocated among the three industries under the UC scenario is not difficult to quantify and equals the sum of target effluent discharges under the TSS scheme. The total BOD_5 discharge permits for the three industries are 22.266 t/d under the UC scenario. Either through grandfathering or auction to distribute the permits among the industries, the EA can always guarantee that the total effluent discharges from the Selby industries match the target levels.

There is an incentive for the industry to initiate trade of pollution permits if they can make cost savings by so doing. Trade will be initiated whenever there are differences in marginal abatement costs, and will continue up to the point at which marginal costs weighted by transfer coefficient are equalized. Comparing Eqs (8.3) and (8.1), the equilibrium price of permits is identical to the tax/subsidy rate of TSS scheme at 0.031 m£ for 1 t/d BOD_5 discharge from the any of industries. The final allocation of permits will match the aggregate effluent discharge target. At equilibrium, the three industries will all have the same abatement, if the aggregated abatement cost function represented the individual function of the industries. Otherwise the industries will choose to abate at different levels according to their actual individual abatement cost functions. But they will always reach the equilibrium with the least aggregated cost.

Since there is only aggregated effluent from the three industries in the TPP system, the situation is considerably simplified because all three discharge at the same location. The effluents from the three industries have similar composition and affect the same WQM site, cell 199, to the same extent. This facilitates trading among the industries, as they can trade effluent permits on the "one-to-one" basis. There is also no need to worry about pollution "hot-spots", which is a considerable risk associated with TPP systems. In this case trade in permits has no spatial impact on water quality. This would be true even if the TPP system were expanded to include the Barlby and Selby STWs, which also discharge at the same location. But difficulties would arise if the TPP were extended to include STWs in other tributaries, or to integrate the water abstraction issue. The details are discussed in 8.2.3.c.

- **TPP under the Selby scenario**

The total amount of effluent permit under the Selby scenario is small due to the high level of abatement required. Altogether, the effluent permits to be allocated among the industries are 2.862 t/d BOD_5, only half the BOD_5 effluent load from the three industries in 2004. Due to the demand for effluent permits and the long-lasting effects which are reflected in a dynamic analysis, the equilibrium price of effluent permits is much higher than under the UC scenario.

The price of permits reflects the discounted stream of abatement costs saved by the purchase of one effluent permit. The equilibrium price of effluent permit was shown in 8.2.3 to be:

$$P_{ei}^{*} = \sum_{s} b_{is} \cdot P_{es}^{*} = \sum_{s} -\frac{\lambda_s}{r} \cdot \frac{\partial f_s(\cdot)}{\partial e_i} \qquad \ldots(8.5)$$

Compared with Eq (8.1), $P_{ei}^{*} = t_{ei}^{*} / r$, where t_{ei}^{*} is the tax/subsidy rate of the TSS under the Selby scenario. The equilibrium price, therefore would be 0.987 m£ for 1 t/d BOD_5 discharge.

Given the high price of effluent permits under this scenario, and given the small number of potential traders, one would expect trading to be very thin. So the initial allocation from the EA needs to be as close to the optimal allocation as possible. Another way to increase the vigour of the thin permits market is to expand it to include other major pollution sources, the STWs in Selby and other tributaries. This then leads to the same problem associated with the UC scenario, which is discussed below.

8.2.4 Disadvantages of TPP system in the tidal Ouse management

The previous discussions pointed out that in this research the trade of effluent permits between the three industries could be on a "one-to-one" basis with no risk of creating pollution "hot spots". This is because effluents pre- and post- trade are discharged at the same location at the same total amount, and influence water quality at the same WQM sites. However, things would be much more complicated if this were not the case. Two potential disadvantages of permits trade are investigated below.

a. If the effluent sources of trade involve different locations but still affect water quality at the same WQM sites.

In this case, the situation would be slightly more complicated, but is still amenable to the PO rules. This is because the binding point of water quality will not change during the trade. The ratio of the trade for effluent permits was defined in section 4.2.3.c of Chapter 4, to be $\delta^{e}_{i,j} = \dfrac{\sum_{s} P^{*}_{es} \cdot b_{is}}{\sum_{s} P^{*}_{es} \cdot b_{js}}$. Since P^{*}_{es} is nonzero only when the water quality at the WQM site is binding, the rate at which effluent permits trade is $\delta^{e}_{i,j} = \dfrac{b_{is}}{b_{js}}$ under the UC and Selby scenarios, in which site s refers to cell 199 and cell 180 respectively. The Transfer

Coefficient Matrix (TCM) developed in Chapter 3 would be the appropriate reference for the determination of trade ratio under this situation.

b. If the effluent sources of trade involve different locations and affect water quality at different WQM sites.

WQM sites are not relevant to the permit trade unless water quality at that site is a binding constraint. Here we refer to the situation when new binding WQM sites emerges or the binding WQM site changes as a consequence of the trade. Since the water quality system is spatially heterogeneous, new pollution from industries or STWs in other tributaries are likely to result in the second situation. In this case, it is difficult to apply the PO rules to determine the trade ratio as the shadow price of water quality at the binding WQM site, P_{es}^{*}, varies. For example, in the UC scenario, if a new effluent source enters the river system at a point upstream of the industrial discharge location around the confluence with the river Don, it needs to enter the market by purchasing effluent permits from some of the industries. However, if the new effluent source only has small impact on the water quality at cell 199, which is binding (possibly because it is far from the Boothferry Bridge) but mainly affects other WQM sites, say Selby, then a single permit sold by a downstream industry must be converted to multiple permits for the various WQM sites affected by the new entrant following PO rules. This discharge from upstream would then violate water quality targets at upstream WQM sites such as Selby and Long Drax well before it could affect water quality at cell 199. This would be the same for the Selby scenario in which the industries discharge their effluents at Selby. A new entrant downstream will possibly result in water quality deterioration at downstream WQM sites. The only way to guarantee water quality compliance at all the WQM sites is to check the effects of effluent discharge on each WQM site, and to make sure that after the trade the water quality will not be deteriorating below the water quality target at each site. This would complicate the trading process and make it even less appealing to the effluent sources. At the same time, whenever there is new source of effluent discharged into the tidal Ouse system, the EA has to re-evaluate the impact on water quality at each WQM site. When pollution pressure is imposed on the new-binding WQM site after the trade, extra action has to be undertaken by the EA to remove the excessive permits from the market, which is very costly.

8.3 Conclusion: Policy options for the tidal Ouse

Neither the TSS nor the TPP system can be introduced into the river policy for the tidal Ouse without some difficulty. Although these MBIs have been proved theoretically superior to CAC methods in terms of cost saving, information requirement, providing incentives etc., (Hanley *et al.* 1997), they are not necessarily as convenient in practice, especially in the control of non-uniformly mixed pollutants in a spatially heterogeneous system such as a river or an estuary. Taking these factors into account, the appropriate policy instrument can only be found on a case-by-case basis.

The CAC approach is currently most favoured by the EA for the tidal Ouse, although its information requirements are large and it is, in most cases, not cost effective. Some obvious advantages of the CAC approach to the regulator are easy management, simple administration and the certainty of the outcome. It is not realistic to advise the EA to abandon the CAC approach for MBIs that offers only theoretical advantages in most cases, particularly when successful examples of MBI implementation in the water quality management are rare (Hanley *et al.* 1998; Cashman *et al.* 1999).

We have shown that the TSS and TPP system can not be applied to the problem of controlling effluent discharges in the tidal Ouse without difficulty. They may, however, be complimentary to the CAC approach, offering more flexibility than the current regulations alone. Some successful examples of MBIs have been found to work well in controlling air pollution in US and Europe (Tietenberg 2006), but there is still no good evidence of the value of such instruments in river pollution management. Of these two MBIs, the introduction of the TPP system would be more complicated because of the non-uniformly mixed pollutant in the river water and the spatial difference of effluent discharges along the river when effluent sources at location other than Selby are included. Section 8.2.3.c investigated the disadvantages of implementing the TPP system for water quality management in the tidal Ouse, and particularly discussed the difficulties associated with trade when water quality constraint were binding at more than a single point. Considering the complexity that would be added to the operations of industries and the EA, it may be inappropriate to choose the TPP system for water quality management in the tidal Ouse. The thin permit market, more or less dominated by one large pollution source, and the high price of permits could also be drawbacks for the effectiveness of the TPP system. We also conclude from the discussion that the PO system, which works well for uniformly mixed pollutants, is capable of dealing with non-uniformly mixed pollutant only if one constraint (WQM site in our case) is binding, but not practical when there are multiple constraint points.

The TSS scheme is more manageable for the EA and clearer to the industries. Although the uncertainty in the outcome would persist even with accurate estimation of abatement cost functions, its negative impact could be minimized by the subsidy-seeking behaviour of industries. If the TSS scheme acts as complementary instrument to the CAC approach, this uncertainty could simply be ruled out by effluent consents on the total discharges. For example a total effluent consent for the three industries could be imposed by the EA to ensure compliance with the water quality target, while the industries submit a plan estimating their average daily discharge over the next year to claim subsidy or pay tax. Therefore the EA will be able to have a rough estimation of the next year's effluent discharges for the three industries, based on which the EA can approve or reject their plans. This kind of combination could guarantee compliance as well as offering flexibility in the means of compliance. By setting up their target effluent discharges appropriately, the equity issue can also be avoided in the TSS scheme.

In summary, it appears that for water quality management in the tidal Ouse, the CAC approach will probably remain the first choice of the environmental authority for the near future, while a TSS may be developed as a complementary approach. Other forms of policy instruments, such as moral suasion, can also contribute to river policy for the tidal Ouse. There is no panacea for the complex issue of pollution management in the estuary.

Chapter 9 Conclusions and Future Research

9.1 Introduction

Facing the imminent requirements of the European Water Framework Directive (WFD) for water pollution control, one of the priorities of river policy is to achieve the required improvement in water quality without incurring disproportionate cost. The cost of the WFD was estimated at between £450 and £630 million, with an estimated benefit between £105 and £522 million per year (Defra and Welsh Assembly Government 2003). Although these ranges are wide and values are possibly over- or under- estimated due to uncertainty, the water quality improvement in a cost effective manner is a fundamental requirement of the WFD. This in turn requires a careful review of the river policy decision process to improve regulatory efficiency.

One of the novel contributions of this research to the literature is the method implemented to evaluate the effectiveness of river policy. In contrast to comparing the consequences of arbitrary changes in specific activities as different policy options, which is normal in scenario analyses, this research aims to provide a comprehensive optimisation-based analysis of policy options subject to a given target. By taking into account various activities influencing the water quality including water abstraction, the integrated optimisation combines their effects through inter-linked hydrological and economic models, to determine the optimal level required for each activity to achieve the desired target. The optimisation results can then be compared to current policy options to assess possible improvement in cost effectiveness. The framework combining hydrological and economic models to identify the potential for integrated and cost effective river management options provides a useful framework for the river policy maker confronted with the forthcoming WFD requirements. The construction of this framework and its application to the tidal Ouse catchment has been the major objective of this research. The main findings of the research are summarised in this chapter, along with potential policy implications that arise from the empirical application.

Despite the call for cost effective policies for water quality improvement, the current regulatory regime dominated by a command and control (CAC) approach is likely to persist for some time. The tradable permits or allowances schemes that are seen more often in the policy of air pollution control are not very suitabl for pollution control in river policy. However, as water quality targets becomes more and more stringent, the issue of the economic costs of water quality improvement has attracted increasing attention, not only from regulated pollutions sources, but also from regulatory bodies, as well as the general public. Therefore further efforts to develop a comprehensive policy for river catchment management can be expected in the near future, together with the integration of water management with other policy sectors that have impacts upon the water environment.

9.2 Main Findings and Conclusions

In England and Wales, the Environment Agency (EA) has developed various hydrological and water quality models to assist on the design of regulation for managing rivers, lakes and coastal environments. However, very few economic analyses of these regulations have been carried out, neither have economic costs been assumed to play an important role in the design of these regulations. Also, although various factors that affect the water quality, such as spatial and temporal differences in discharge location, changes in water volume and anthropogenic disturbances, are typically included in the hydrological models of water quality, current regulatory regimes either fail to take these differences into account or regulate them in a disjointed manner. This thesis has shown that an integrated river policy derived from a combined hydrological and economic modelling framework can improve understanding of the water quality management problem in a spatially heterogeneous river system. This approach potentially can also allow comprehensive analysis of resource distribution for water quality management in the river system. Through this research, the author aimed to address the following questions.

1. Does the current regulatory regime include all the options to tackle the water quality problem? If not, are there any other options the regulatory regime could, and should, take into consideration?
2. If the current regulatory regime is inefficient and incurs unnecessary costs in water quality management, how do we develop a more cost effective river policy?
3. What are the policy implications of the findings from the analyses? How could the current regulatory system of river policy be improved, and in which aspects?

To address the first question, Chapter 3 discusses the effectiveness of the current regulatory regime for improving water quality, using the dynamic hydrological model QUESTS1D for the tidal Ouse. Simulations from the modelling indicate that current river policy is unable to comply with the desired water quality target during the summer of a typical dry year. Several alternative water quality management options are evaluated through simulations. These alternative options include changes in the location and timing of effluent discharge from the Selby sources and changes in water abstraction levels. These options are investigated as potential components of an integrated river policy for further analysis. Comparison of their effectiveness indicates that a shift in either the location or the timing of effluent discharges from the Selby sources could produce significant improvement in the DO% sag experienced in the tidal Ouse, while changes in the location or amount of water abstraction are at best considered as complementary measures, unable to tackle the DO% sag issue alone. The findings from the QUESTS1D model simulations suggest that the conventional mechanisms for effluent load control could collaborate with some effective alternative options to achieve an integrated river policy, which makes good use of the assimilative capacity of river water and improves water quality significantly to comply with the EA's desired target of water quality.

The background theory for cost minimising in pollution control in an integrated management policy subject to a particular water quality target is developed in Chapter 4. Optimisations are developed for both static and dynamic systems. These analyses produce two conditions to be satisfied for a cost effective river policy when alternative management options are included. For the static system, the ratio between marginal cost of abatement and the marginal effect of abatement on water quality should be the same across all the options. This ratio is captured in the shadow price of river water quality at the cost-effective equilibrium. For the dynamic system, the internal rate of return on investment in the capital stock equilibrium for pollution abatement should be the same as the return on investment made elsewhere in the economy. The dynamic equilibrium of capital and investment is characterised as a saddle point equilibrium, which can only be approached by the stable arm. These analyses embed the integration of various options for water quality management into a comprehensive river policy, covering both effluent discharge and water abstraction, and also taking into account variation in the assimilative capacity of the river. These analyses also compare three different policy instruments for water quality management: command and control (CAC) approach, Tax and Subsidy Scheme (TSS) and Tradable Pollution Permit (TPP) system in terms of delivering the required regulation targets among the regulated industries in an efficient and practically convenient manner.

Chapters 6 and 7 apply the static and dynamic optimisations developed in Chapter 4 to the tidal Ouse catchment, using the hydrological and economic dataset constructed in Chapter 5. Some modifications have to be made because of insufficient data, particularly economic data regarding the cost of effluent treatment and water abstraction. Chapter 6 considers minimum cost pollution abatement in a static system to achieve a particular water quality target. The analysis recommends an integrated river policy towards the implementation of the WFD in the near future, and it also takes account the requirements imposed currently on STWs by the UWWTD, together with more modest and realistic suggestions for river policy regulations on the tidal Ouse. Details of the integrated river policy are based on the simulations of options laid out in Chapter 3 and specific costs evaluated from data in Chapter 5. Options considered are: moving effluent discharges downstream, shifting water abstraction between the rivers

Ouse and Derwent, and effluent abatement on-site. The benefits of implementing an integrated river policy for the tidal Ouse include a significant improvement in the water quality, compliance with the EA's water quality target even in a dry year, together with annual abatement cost savings of £116,000 to the industries and STWs. The integrated river policy achieves these outcomes mainly through better utilisation of the assimilative capacity of the river. The results also point out though, 40% DO% at 5%ile along the tidal Ouse is not achievable with any of the alternative options as considered in this research. Higher level of water quality however still remains possible given that more effective options could be identified in the future. Due to the modifications and assumption necessary to make the dynamic analysis of Chapter 7 tractable, the research does not undertake a detailed discussion of investment paths and capital stocks for the industries in the dynamic setting, but tries to assess the capability of this framework to address dynamic management if a future research can be less constrained by data availability. Nonetheless, the dynamic analysis still illustrates the mechanism required to identify the stable arm of the saddle point equilibrium, and the corresponding optimal investment path for the industries and STWs.

The final question is answered by the discussion in Chapter 8 to some extent, as river policy is unlikely to be determined by economic concerns alone. Two MBIs, a TSS scheme and a TPP system are discussed in the light of selection criteria for policy instruments when applied to the tidal Ouse to allocate effluent abatement responsibilities among the industries in Selby. After this comparison, the author concludes that the CAC approach will still remain the first choice instrument of EA for the near future, but a TSS has the potential to be developed into a complementary instrument, which could improve the overall cost effectiveness of the river policy. It appears inappropriate to introduce a TPP system for the tidal Ouse because of the spatial heterogeneity and multiple constraints within the river system.

9.3 Further Research

This research has produced a relatively accurate optimisation of river policy in a static system and has explored capital and investment interactions within a dynamic system. However, this research is more of a beginning rather than a conclusion for the comprehensive framework of river policy analysis, and there are several critical assumptions that should be addressed in the future studies.

The first assumption arises from the use of a common function for the abatement cost for the industries or for the STWs respectively. This was necessary because insufficient abatement cost data were available for the individual industry and STW, and therefore cost data had to be pooled for industries and STWs respectively. This was done on the assumption that they followed the same cost function because of the similar composition of their effluents and treatment technologies. Although the data do not reject this assumption, the subsequent optimisations produce the same abatement level and marginal cost for all industries as a consequence. This could be improved to reflect individual differences among the industries and to make the analysis more meaningful when different forms of economic instruments for river policy are indtroduced. This also applies to the STWs. If a cost function could be estimated for each STW, the optimised abatement levels for STWs could be more reflective of the effluent impact and abatement cost of the individual STWs. More data on abatement cost from each pollution source would also help us to understand the impacts of the technologies used on the abatement cost and effluents.

The second assumption was stressed in the dynamic analysis of Chapter 7. The investment data available for the industries and STWs are insufficient to permit accurate estimation for the impacts of investment on the industries and STWs' abatement cost functions. More investment data need to be collected from the interviews or questionnaires to the industries and STWs in order to produce a better representation of the interaction between capital investment and cost in effluent abatement. New technologies are also an important factor to consider when estimating abatement cost. These improved estimations would generate a more precise estimation for the stable saddle arm which leads towards the dynamic optimum for capital stock and investment for each industry and STW.

While more data on investment in abatement capacity remains one constraint of identifying the dynamic optimum, the functional forms assumed for the abatement capacity and cost functions are other concerns worth considering. One of the problems encountered in the dynamic analysis in this research was the inability of the GAMS to identify optimal solutions using more complicated nonlinear cost and abatement functions. These constraints could be improved through further research with programming assistance.

This research successfully integrated changes in water abstraction levels alongside abatement policies as a means of reducing pollution and improving water quality. But the level of aggregate water abstraction from the tidal Ouse was not changed in either static nor dynamic optimisations, because of the high marginal cost of reducing water abstraction and still meeting the requirement for water supply. The alternative water resource options considered in the research were provided by experts in the water company, ranked by required water yield and average cost of water supply. This situation could change when new alternative water resource options are identified, if their costs of water supply under these options become low enough to make reduction in water abstraction from the Ouse and Derwent an economically viable option to improve water quality in the integrated river management.

Due to the limitations of the water quality model, this research only considers effluent impacts from point sources along the tidal Ouse system. This is reasonable, given the relatively significant impacts imposed directly by point source pollution around Selby and Drax. However, agricultural activities are believed to be one of the biggest pollution sources in river systems, particularly so for the upland river systems. This is a difficult issue to address because pollution arises from the runoffs from diffuse sources. Managing diffuse pollution is also an objective for comprehensive integrated river management. The urgency and importance of diffuse pollution has been emphasised clearly in many researches and national legislations, as well as in the EU WFD (Lewis *et al.* 1997; European Commission 2000; Defra 2003b; Defra 2005c). In England and Wales, diffuse water pollution from agriculture accounts for 43% of phosphorus and 60% of nitrate in the water body (Amin-Hanjani 2006). The uncertainty and difficulties in quantifying diffuse pollution make it very difficult for current regulatory frameworks to tackle the problem effectively (D'Arcy and Frost 2001; O'Shea

2002). One of the very meaningful challenges to this research is to expand the framework from point sources to include diffuse sources from farming and other distributed land usage, to better assist decision making in river policy. This would require not only a more powerful hydrodynamic model to estimate impacts on water quality from diffuse sources, but also changes in the economic decision making and behaviour patterns of farmers facing a range of different incentives from the regulatory system. Some ground-breaking discussion has arisen following Defra's consultations in this area (Anthony 2006). It is to be expected that social-economic analysis will play a key role in developing an integrated, comprehensive river policy system to cope with the water quality requirements of WFD, as well as management of water resources. Rosegrant *et al.* (2000) used a similar approach to manage the water resource in the Maipo river basin in Chile, by using a rough model of the economic behaviour of farmers facing different management instruments.

9.4 Policy Implications and Broader Application

Following the results obtained from this research, the main recommendation for river policy in the tidal Ouse is to implement an integrated river policy, which includes the main factors affecting the water quality, i.e. both effluent discharges and water abstraction, to develop a comprehensive systematic set of regulations for the river system. The most cost effective single measure to improve water quality in the tidal Ouse would be to relocate the effluent discharges from Selby sources (collaborating with changes in levels of effluent discharge and water abstraction in the tidal Ouse). TSS could be a useful addition to the current regulatory toolkit for the tidal Ouse, helping to deliver integrated and cost effective river management options in a more efficient way.

The framework developed here has combined a hydrological water quality model with an economic model to provide a quantitative analysis for the activities, which have impacts on both water quality and economic outcomes. The optimisation approach used has been able to balance the outcomes of activities against specific criteria and targets. This is expected to offer advantages to the decision making of river policy to regulate water quality and related activities.

This framework can also be applied to decision making surrounding other environmental issues when both environmental targets and economic constraint are present in a spatial setting. For example, this framework could also be used to investigate the cost effectiveness of measures for air pollution control, to capture the impacts of different pollution mitigation options coordinated in the modeling to improve cost effectiveness. Depending on the specific modeling undertaken, the framework has considerable breadth of application, which could be explored in future research.

Appendices

Appendix 1 Proof of Expression (4.3.3.a)

Consider a system of generic non-linear, autonomous differential equations as $\dot{Y} = F(Y)$, where Y is an n x 1 vector and F is a 1x n vector. Let Y^* be the steady state point of the system.

Expanding the equation in the first-order Taylor approximation, produces (Xepapadeas 1997):

$$\dot{Y} = F(Y^*) + \sum_i \frac{\partial F(Y^*)}{\partial y_i}(y_i - y_i^*), y_i \in Y, i = 1,2,...n \qquad ...(A1)$$

Because at the steady state equilibrium, $F(Y^*) = 0$,

$$\dot{Y} = \sum_i \frac{\partial F(Y^*)}{\partial y_i} \cdot y_i - \sum_i \frac{\partial F(Y^*)}{\partial y_i} \cdot y_i^* = \sum_i \frac{\partial F(Y^*)}{\partial y_i} \cdot y_i + B, y_i \in Y \qquad ...(A2)$$

where B is a $1 \times n$ vector.

If we assume that $\frac{\partial F(Y^*)}{\partial y_i} = a_{ij}$, then $A \equiv \begin{bmatrix} a_{11} & a_{12} & \dots & a_{1n} \\ a_{21} & a_{22} & \dots & a_{2n} \\ \dots & \dots & \dots & \dots \\ a_{n1} & a_{2n} & \dots & a_{nn} \end{bmatrix}$ is the Jacobian matrix of the function F evaluated at the steady state equilibrium. If the equilibrium point in the linear approximation is globally stable, then it is also locally stable in the original non-linear system. The converse is not necessarily true (Xepapadeas 1997).

Appendix 2 Proof of Expression (4.3.3.b)

Substitute $\mu_i^{j*} = -C_i'(\cdot)$ into Eq (4.61), it becomes:

$$\dot{I}_i^j = -\frac{1}{C_i''(\cdot)} \cdot [-(r+\delta_i^j)\cdot C_i'(\cdot) - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} + \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^j}] = G(k_i^j, I_i^j) \qquad \ldots(A3).$$

The value of element a_{22} in the Jacobian matrix is then determined as below:

$$a_{22} = \frac{\partial G(\cdot)}{\partial I_i^j} = -C_i''(\cdot)^{-2} \cdot \{[-(r+\delta_i^j)\cdot C_i''(\cdot)\cdot C_i''(\cdot) - C_i^{(3)}(\cdot)\cdot [-(r+\delta_i^j)\cdot C_i'(\cdot) - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} + \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^j}]\} \qquad \ldots(A4),$$

where $C_i''(\cdot)$ and $C_i^{(3)}(\cdot)$ denote the second and third order derivatives of cost with respect to investments in each of the three elements respectively. Since the Jacobian matrix of the functions is evaluated at the steady state equilibrium, $\dot{I}_i^j = 0$ when $I_i^j = I_i^{j*}$. From Eq (A3), it follows that:

$$-(r+\delta_i^j)\cdot C_i'(\cdot) - \frac{\partial C_i^*(\cdot)}{\partial k_i^j} + \sum_s \lambda_s \cdot \frac{\partial f_s^*(\cdot)}{\partial k_i^j} = 0 \qquad \ldots(A5).$$

Therefore Eq (A4) can be reduced to the function below:

$$a_{22} = \frac{\partial G(\cdot)}{\partial I_i^j} = -C_i''(\cdot)^{-2} \cdot [-(r+\delta_i^j)\cdot C_i''(\cdot)\cdot C_i''(\cdot)] = (r+\delta_i^j)$$

Appendix 3 System of Water Quality Functions

Estimated through Limdep 7.0

```
+--------------------------------------------------------------------+
| Estimates for equation: DO180                                       |
| Generalized least squares regression     Weighting variable = none  |
| Dep. var. = DO180      Mean=     .1181930748     , S.D.=    10.48815805    |
| Model size: Observations =       190, Parameters =    6, Deg.Fr.=     184 |
| Residuals:   Sum of squares= 912.0611517      , Std.Dev.=         2.22640 |
| Fit:          R-squared=   .954700, Adjusted R-squared =           .95347 |
|               (Note:   Not using OLS.   R-squared is not bounded in [0,1] |
| Model test: F[   5,      184] =   775.56,      Prob value =         .00000 |
| Diagnostic: Log-L =    -418.6232, Restricted(b=0) Log-L =     -715.6439 |
|                 LogAmemiyaPrCrt.=      1.632, Akaike Info. Crt.=     4.470 |
| Log-determinant of W      -4.5483          Log-likelihood         -915.9013 |
| Durbin-Watson Stat.=       1.2860          Autocorrelation    =       .3570 |
+--------------------------------------------------------------------+
+---------+-------------+---------------+--------+---------+----------+
|Variable | Coefficient   | Standard Error |b/St.Er.|P[|Z|>z] | Mean of X|
+---------+-------------+---------------+--------+---------+----------+
 Constant -442.0904854        17.187262        -25.722    .0000
 LOCATION   1.473521240        .10889670        13.531    .0000   27.136842
 LOCA_2    -.4215780044E-01   .19716017E-02   -21.383    .0000   831.33684
 LOGSBOD   -3.603665789        .65229110        -5.525    .0000   .00000000
 LOGOUSE    128.2103381        4.3375640        29.558    .0000   .00000000
 LOGDERWE   9.220113769        1.4394565         6.405    .0000   .00000000


+--------------------------------------------------------------------+
| Estimates for equation: DO192                                       |
| Generalized least squares regression     Weighting variable = none  |
| Dep. var. = DO192      Mean=     .1682783934     , S.D.=    11.80294199    |
| Model size: Observations =       190, Parameters =    6, Deg.Fr.=     184 |
| Residuals:   Sum of squares= 1146.512735      , Std.Dev.=         2.49621 |
| Fit:          R-squared=   .955035, Adjusted R-squared =           .95381 |
|               (Note:   Not using OLS.   R-squared is not bounded in [0,1] |
| Model test: F[   5,      184] =   781.62,      Prob value =         .00000 |
| Diagnostic: Log-L =    -440.3567, Restricted(b=0) Log-L =     -738.0833 |
|                 LogAmemiyaPrCrt.=      1.861, Akaike Info. Crt.=     4.698 |
| Log-determinant of W      -4.5483          Log-likelihood         -915.9013 |
| Durbin-Watson Stat.=       .8038           Autocorrelation    =       .5981 |
+--------------------------------------------------------------------+
+---------+-------------+---------------+--------+---------+----------+
|Variable | Coefficient   | Standard Error |b/St.Er.|P[|Z|>z] | Mean of X|
+---------+-------------+---------------+--------+---------+----------+
 Constant -113.4061995        19.270108        -5.885    .0000
 LOCATION -.2811491401E-01    .12209340         -.230    .8179   27.136842
 LOCA_2    -.1988166572E-01   .22105312E-02    -8.994    .0000   831.33684
 LOGSBOD   -9.238499987        .73133931       -12.632    .0000   .00000000
 LOGOUSE    37.17412156        4.8632138         7.644    .0000   .00000000
 LOGDERWE   23.41763058        1.6138978        14.510    .0000   .00000000


+--------------------------------------------------------------------+
| Estimates for equation: DO193                                       |
| Generalized least squares regression     Weighting variable = none  |
| Dep. var. = DO193      Mean=     .1731819945     , S.D.=    10.78493647    |
| Model size: Observations =       190, Parameters =    6, Deg.Fr.=     184 |
| Residuals:   Sum of squares= 1060.842202      , Std.Dev.=         2.40113 |
| Fit:          R-squared=   .950170, Adjusted R-squared =           .94882 |
|               (Note:   Not using OLS.   R-squared is not bounded in [0,1] |
| Model test: F[   5,      184] =   701.71,      Prob value =         .00000 |
| Diagnostic: Log-L =    -432.9788, Restricted(b=0) Log-L =     -720.9456 |
|                 LogAmemiyaPrCrt.=      1.783, Akaike Info. Crt.=     4.621 |
| Log-determinant of W      -4.5483          Log-likelihood         -915.9013 |
| Durbin-Watson Stat.=       .8000           Autocorrelation    =       .6000 |
+--------------------------------------------------------------------+
+---------+-------------+---------------+--------+---------+----------+
```

Variable	Coefficient	Standard Error	b/St.Er.	P[|Z|>z]	Mean of X
Constant	-79.94286489	18.536174	-4.313	.0000	
LOCATION	-.4241536192	.11744327	-3.612	.0003	27.136842
LOCA_2	-.1072496116E-01	.21263395E-02	-5.044	.0000	831.33684
LOGSBOD	-9.432453421	.70348505	-13.408	.0000	.00000000
LOGOUSE	28.99258921	4.6779903	6.198	.0000	.00000000
LOGDERWE	23.20610946	1.5524298	14.948	.0000	.00000000

Estimates for equation: DO197
Generalized least squares regression Weighting variable = none
Dep. var. = DO197 Mean= .1886351801 , S.D.= 6.551599510
Model size: Observations = 190, Parameters = 9, Deg.Fr.= 181
Residuals: Sum of squares= 469.4357960 , Std.Dev.= 1.61046
Fit: R-squared= .939257, Adjusted R-squared = .93657
(Note: Not using OLS. R-squared is not bounded in [0,1]
Model test: F[8, 181] = 349.85, Prob value = .00000
Diagnostic: Log-L = -355.5265, Restricted(b=0) Log-L = -626.2418
LogAmemiyaPrCrt.= .999, Akaike Info. Crt.= 3.837
Log-determinant of W -4.5483 Log-likelihood -915.9013
Durbin-Watson Stat.= 1.0401 Autocorrelation = .4800

Variable	Coefficient	Standard Error	b/St.Er.	P[|Z|>z]	Mean of X
Constant	37.74924817	12.427447	3.038	.0024	
LOCATION	-1.552492007	.78737644E-01	-19.717	.0000	27.136842
LOCA_2	.1880900270E-01	.14255648E-02	13.194	.0000	831.33684
LOGSBOD	-9.032303071	.47231295	-19.124	.0000	.00000000
LOGOUSE	1.060207076	3.1364845	.338	.7353	.00000000
LOGDERWE	17.69696814	1.0415746	16.991	.0000	.00000000
LOGSNAIT	.1409821105	.12094459	1.166	.2437	.00000000
LOGSANDA	-.2284539360	.12589559	-1.815	.0696	.00000000
LOGTHORN	.8543278927E-01	.14409510	.593	.5533	.00000000

Estimates for equation: DO199
Generalized least squares regression Weighting variable = none
Dep. var. = DO199 Mean= .1796468144 , S.D.= 5.925286548
Model size: Observations = 190, Parameters = 9, Deg.Fr.= 181
Residuals: Sum of squares= 376.3486606 , Std.Dev.= 1.44197
Fit: R-squared= .940463, Adjusted R-squared = .93783
(Note: Not using OLS. R-squared is not bounded in [0,1]
Model test: F[8, 181] = 357.39, Prob value = .00000
Diagnostic: Log-L = -334.5301, Restricted(b=0) Log-L = -607.1505
LogAmemiyaPrCrt.= .778, Akaike Info. Crt.= 3.616
Log-determinant of W -4.5483 Log-likelihood -915.9013
Durbin-Watson Stat.= 1.1077 Autocorrelation = .4462

Variable	Coefficient	Standard Error	b/St.Er.	P[|Z|>z]	Mean of X
Constant	42.56558169	11.127295	3.825	.0001	
LOCATION	-1.518255987	.70499653E-01	-21.536	.0000	27.136842
LOCA_2	.1959711230E-01	.12764140E-02	15.353	.0000	831.33684
LOGSBOD	-8.921968974	.42314942	-21.085	.0000	.00000000
LOGOUSE	-.7628892410	2.8084073	-.272	.7859	.00000000
LOGDERWE	16.80019207	.93289009	18.009	.0000	.00000000
LOGSNAIT	.1603493699	.12898231	1.243	.2138	.00000000
LOGSANDA	-.2610663257	.13426234	-1.944	.0518	.00000000
LOGTHORN	.9815368896E-01	.15367136	.639	.5230	.00000000

Appendix 4 Exponential Cost function of effluent treatment in the Selby industries

Estimated through SPSS 11.0

Variables Entered/Removed[b]

Model	Variables Entered	Variables Removed	Method
1	BOD_REM[a]	.	Enter

a. All requested variables entered.

b. Dependent Variable: LNC

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.901[a]	.811	.773	.274245

a. Predictors: (Constant), BOD_REM

ANOVA[b]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	1.616	1	1.616	21.485	.006[a]
	Residual	.376	5	.075		
	Total	1.992	6			

a. Predictors: (Constant), BOD_REM

b. Dependent Variable: LNC

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	-1.364	.195		-7.013	.001
	BOD_REM	.109	.024	.901	4.635	.006

a. Dependent Variable: LNC

Appendix 5 Exponential Cost function of effluent treatment in the STWs

Estimated through SPSS 11.0

Variables Entered/Removed[b]

Model	Variables Entered	Variables Removed	Method
1	BOD_REM[a]	.	Enter

a. All requested variables entered.

b. Dependent Variable: LNC

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.888[a]	.788	.767	.458993

a. Predictors: (Constant), BOD_REM

ANOVA[b]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	7.827	1	7.827	37.151	.000[a]
	Residual	2.107	10	.211		
	Total	9.934	11			

a. Predictors: (Constant), BOD_REM

b. Dependent Variable: LNC

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	-1.389	.185		-7.498	.000
	BOD_REM	.245	.040	.888	6.095	.000

a. Dependent Variable: LNC

Appendix 6 Exponential cost function of reducing water abstraction in the rivers Ouse and Derwent

Estimated through SPSS 11.0

Variables Entered/Removed[b]

Model	Variables Entered	Variables Removed	Method
1	ABSTRACT[a]	.	Enter

a. All requested variables entered.

b. Dependent Variable: LNC

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.951[a]	.904	.880	.231794

a. Predictors: (Constant), ABSTRACT

ANOVA[b]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	2.019	1	2.019	37.586	.004[a]
	Residual	.215	4	.054		
	Total	2.234	5			

a. Predictors: (Constant), ABSTRACT

b. Dependent Variable: LNC

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	3.679	.225		16.366	.000
	ABSTRACT	-.472	.077	-.951	-6.131	.004

a. Dependent Variable: LNC

Appendix 7 Cost Details of moving effluent discharges from Selby

Estimated Capital Costs

Item	Cost (m£)
Site Establishment	0.842
Pumping Station	0.561
Pipeline within Selby	0.202
Storage Tank	0.685
Main Pipeline	7.362-0.16X
Subtotal	9.652-0.16X
Consultancy fees (10%)	0.965-0.016X
Access rights, legal fees	0.500
Contingencies (10%)	0.965-0.016X
Total	**12.082-0.192X**

X is the distance from new discharge location to the Trent Falls in kilometer.

Estimated Operational Costs

Item	Cost (m£)
Labour	0.056
Pumping Costs	0.410
Maintenance Cost	0.050
Total	**0.516**

Appendix 8 Power cost function of effluent treatment in the Selby industries

Estimated through SPSS 11.0

Variables Entered/Removed[b]

Model	Variables Entered	Variables Removed	Method
1	LNBR[a]	.	Enter

a. All requested variables entered.

b. Dependent Variable: LNC

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.971[a]	.942	.931	.151540

a. Predictors: (Constant), LNBR

ANOVA[b]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	1.877	1	1.877	81.741	.000[a]
	Residual	.115	5	.023		
	Total	1.992	6			

a. Predictors: (Constant), LNBR

b. Dependent Variable: LNC

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	-1.711	.135		-12.632	.000
	LNBR	.660	.073	.971	9.041	.000

a. Dependent Variable: LNC

Appendix 9 Linear cost function of effluent treatment in the STWs

Estimated through SPSS 11.0

Variables Entered/Removed[b]

Model	Variables Entered	Variables Removed	Method
1	BOD_REM[a]	.	Enter

a. All requested variables entered.

b. Dependent Variable: COST

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.982[a]	.964	.960	.135780

a. Predictors: (Constant), BOD_REM

ANOVA[b]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	4.880	1	4.880	264.701	.000[a]
	Residual	.184	10	.018		
	Total	5.064	11			

a. Predictors: (Constant), BOD_REM

b. Dependent Variable: COST

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	.172	.055		3.139	.011
	BOD_REM	.194	.012	.982	16.270	.000

a. Dependent Variable: COST

Appendix 10 Effluent treatment capability function (logarithmic) of capital stock in the ETPs of the industries of Selby

Estimated through SPSS 11.0

Variables Entered/Removed[b]

Model	Variables Entered	Variables Removed	Method
1	LNIKS[a]	.	Enter

a. All requested variables entered.

b. Dependent Variable: IBOD

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.924[a]	.855	.834	1.940590

a. Predictors: (Constant), LNIKS

ANOVA[b]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	154.924	1	154.924	41.139	.000[a]
	Residual	26.361	7	3.766		
	Total	181.285	8			

a. Predictors: (Constant), LNIKS

b. Dependent Variable: IBOD

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	3.180	.764		4.161	.004
	LNIKS	3.977	.620	.924	6.414	.000

a. Dependent Variable: IBOD

Appendix 11 Effluent treatment capability function (logarithmic) of capital stock in the ETPs of STWs

Estimated through SPSS 11.0

Variables Entered/Removed[b]

Model	Variables Entered	Variables Removed	Method
1	LNSKS[a]	.	Enter

a. All requested variables entered.

b. Dependent Variable: SBOD

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.876[a]	.768	.747	1.700932

a. Predictors: (Constant), LNSKS

ANOVA[b]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	105.431	1	105.431	36.441	.000[a]
	Residual	31.825	11	2.893		
	Total	137.256	12			

a. Predictors: (Constant), LNSKS

b. Dependent Variable: SBOD

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	-4.108	1.269		-3.237	.008
	LNSKS	3.224	.534	.876	6.037	.000

a. Dependent Variable: SBOD

Appendix 12 Pollution abatement costs function of capital stock and investment in the ETPs

Estimated through SPSS 11.0

Variables Entered/Removed[b]

Model	Variables Entered	Variables Removed	Method
1	LNI, LNK[a]	.	Enter

a. All requested variables entered.

b. Dependent Variable: LNC

Model Summary[b]

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.990[a]	.980	.961	.207909

a. Predictors: (Constant), LNI, LNK

b. Dependent Variable: LNC

ANOVA[b]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	4.326	2	2.163	50.043	.020[a]
	Residual	.086	2	.043		
	Total	4.413	4			

a. Predictors: (Constant), LNI, LNK

b. Dependent Variable: LNC

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	-2.684	.244		-10.994	.008
	LNK	.952	.098	.983	9.708	.010
	LNI	1.654E-02	.050	.033	.328	.774

a. Dependent Variable: LNC

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