Contributions to the design of policy instruments: an application to irrigation-induced salinity in Australia

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This thesis contains no material which has been accepted for the award of any other degree or diploma than specified in the cotutelle agreement between the Australian National University and the Université Montpellier I. To the best of my knowledge and belief, it contains no material previously published or written by another person, except where due reference is made in the text.

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Abstract

The focus of this thesis is on the design of policy instruments to manage environmental issues, with a particular interest in dynamic taxation schemes and cap and trade systems in the context of irrigation-induced salinity. This issue is of crucial importance in most irrigated areas throughout the world.

A first goal of the thesis is to investigate the use of dynamic taxation schemes based on a measure of group performance, as a way of implementing the notion of collective responsibility. The analysis focuses on the main characteristics of group performance based instruments, the interdependence they introduce among the agents. Indeed, when subject to group performance based instruments, agents' payoffs result from the effort provided by the group. The analysis is set at the catchment scale, and the collective result is the groundwater stock. Various taxes are investigated, including a time-independent standard input tax, a state-dependent ambient tax and a stock-dependent input tax. These taxes are illustrative of various ratios of individual performance and collective performance in the design of the policy instrument. One of the results of this analysis is that including a group performance component in the policy instrument is necessary to induce the agents to behave optimally along the whole time horizon. Hence the interest for mixing individual and collective incentives in the design of policy instruments.

A second aim of the thesis is to address the design of cap and trade systems to manage multiple coupled externalities. The catchments are replaced within the broader context of the river system, and the analyses relate to the interactions developing between water management initiatives at the catchment and the river scales. To manage both water scarcity along the river and irrigation-induced salinity in each catchment, two types of water rights are considered : standard diversion rights and recharge rights that allow right-holders to produce a certain amount of percolation. This analysis poses the question of the number of policy instruments needed to manage correlated externalities. It also raises issues associated with the implementation of cap and trade systems at different scales. The main result of this analysis is that the correlation existing between the externalities doesn't rule out the need for a policy instrument to manage each externality.

While the aim of this thesis isn't to compare these two policy strategies, by the analyses it provides it participates in the debate about the use of price-based or quantity-based instruments. Furthermore, the analyses apply to the general case of correlated environmental externalities.

Key-words : dynamic taxes, group performance, cap and trade, coupled externalities, non cooperative game theory, differential games, water markets, irrigation-induced salinity, Australia.

Résumé

Cette thèse est consacrée à l'élaboration d'instruments de politique environnementale. Elle vise plus particulièrement à étudier les schémas de taxation dynamique et les systèmes de type cap and trade pour gérer la salinité d'irrigation. Cette question est d'une importance cruciale en Australie, mais aussi dans de nombreux périmètres irrigués dans le monde.

Le premier objectif de ce travail est d'analyser l'utilisation de taxes dynamiques basées sur une mesure de la performance de groupe. L'analyse porte sur la caractéristique principale de ces instruments, l'interdépendance qu'ils introduisent entre les agents. En effet, leurs fonctions de gain dépendent alors de l'effort total procuré par le groupe. L'échelle d'analyse est le district d'irrigation et la performance du groupe se réfère au stock d'eau souterraine. Différents types de taxes sont analysées : une taxe sur les intrants, une taxe ambiante et une taxe sur les intrants dépendant du stock. Ces taxes illustrent différentes combinaisons de prise en compte de la performance individuelle et de la performance collective dans l'élaboration d'instruments de politique publique. Les résultats montrent qu'introduire une composante basée sur la performance de groupe est nécessaire pour inciter les agents à adopter un comportement optimal sur l'ensemble du sentier d'accumulation. D'où l'intérêt de combiner des incitations individuelles et collectives.

Le deuxième objectif de ce travail est d'analyser les conditions d'élaboration de marchés de droits pour gérer des externalités couplées. Les districts d'irrigation sont alors placés dans le contexte plus général du système rivière, de sorte que l'analyse porte sur les interactions entre les initiatives de gestion de l'eau à différentes échelles (district et rivière). Afin de gérer à la fois la pénurie d'eau et la salinité d'irrigation dans chaque district, deux types de droits sont considérés : des droits d'extraction et des droits de recharge. Ces derniers autorisent la production d'une certainte quantité de percolation qui recharge la nappe. Cette étude pose la question du nombre d'instruments à mettre en place pour gérer des externalités couplées. Elle met aussi en avant les problèmes liés aux échelles géographiques différentes des instruments. Le principal résultat de cette analyse est que l'existence de corrélationss entre les externalités ne réduit pas le nombre d'instruments nécessaire pour les gérer.

Si l'objectif de la thèse n'est pas de comparer les stratégies étudiées, elle participe de fait

au débat sur les instruments basés sur les prix ou les quantités. En outre, l'intérêt des résultats acquis s'étend au cas général de la régulation d'externalités environnementales couplées.

Mots-clés : taxes dynamiques, performance de groupe, cap and trade, externalités couplées, théorie des jeux non-coopératifs, jeux différentiels, marchés d'eau, salinité d'irrigation, Australie.

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List of Abbrevations

ACT : Australian Capital Territory

BSMS : Basin Salinity Management Strategy

COAG : Council Of Australian Governments

EC : Eletric Conductivity

EAA : Everglade Agricultural Area

EWR : Environmental Water Reserve

GL : Giga Litres

GPBI : Group Performance Based Instrument

HIZ : High Impact Zone

IA : Irrigation Area

ID : Irrigation District

IFD : Instream Flow District

LIZ : Low Impact Zone

MDB : Murray Darling Basin

MDBA : Murray Darling Basin Agreement

MDBC : Murray Darling Basin Commission

MHDS : Modified Hamiltonian Dynamic System

MIS : Murrumbidgee Irrigation Scheme

NAPSWQ : National Action Plan for Salinity and Water Quality

NCC : National Competition Commission

NCP : National Competition Policy

NHT : Natural Heritage Trust

NSW : New South Wales

NWI : National Water Initiative

QLD : Queensland

SA : South Australia

SDE : Salt Disposal Entitlement

SDS : Salinity and Drainageb Strategy

VIC : Victoria

WQT : Water Quality Trading

Chapitre 1

Introduction

The focus of this thesis is on the design of policy instruments to manage environmental issues, with a particular interest in dynamic taxation schemes and cap and trade systems to manage irrigation-induced salinity. These policy strategies follow the standard distinction between price-based and quantity-based instruments, illustrated by Weitzman's (1974) seminal paper. While the aim of this thesis isn't to compare these strategies, by the analyses it provides for each type of instruments, it participates in the above-cited debate.

Irrigation-induced salinity.

In all semiarid areas of the world, irrigation is responsible for extensive damage to the environment through what is known as the twin menace of irrigation. Salinization and waterlogging may occur naturally in some particular areas. However, these phenomena are broadly worsened by inappropriate irrigation practices. Salinization refers to the increasing concentration of dissolved salts in soils and waters. It has important impacts, both on-site and off-site. By on-site, understand the loss of productivity of irrigated soils and the damage to irrigation infrastructures at the paddock scale. Off-site impacts follow the hydrologic cycle and include damage to different types of infrastructures (including irrigation and drinking water networks, roads) as well as declining quality of surface water and groundwater. Saline water has consequences in terms of agricultural productivity, biodiversity, and public health. Waterlogging refers to the rising of the watertable, which leads to the soaking of soils above the underlying aquifer. It has consequences in itself, for instance plant asphyxia, but also worsens the processes of salinization. The extent of the problem is worldwide and increasing. The World Bank reports that 24 per cent of all irrigated areas are severely affected by irrigation-induced salinity (Umali 1993). It is responsible for putting out of production around 10 million hectares each year. The issue is particularly severe in Australia, where large-scale irrigation schemes have been promoted as the key process of the development project. The extent of this environmental issue is such that it questions the sustainability of irrigation itself (Umali 1993).

Irrigation-induced salinity is peculiar in many respects. First, it involves interactions between surface and underground water resources. Consequently, the geographical scale of analysis is important : while the catchment appears as a relevant management scale, it has to be replaced within the broader context of the river system to fully account for these interactions. Second, both quantitative and qualitative issues characterise irrigation-induced salinity. Indeed, while the underlying mechanisms are quantitative (diversion of water for irrigation, rising of the watertable), the resulting effects are both qualitative (root zone salinization, discharge of salty water) and quantitative (amount of return flows). However, the special nature of the pollutant 'salt' means that focus can be given to the key driving process, which is quantitative by nature. Indeed, it is the rising of the watertable that is responsible for the uptake of salts and their transport along the water cycle, from instream flows to groundwater, through the root zone.

In this thesis, the analyses develop around two axes, depending on which components are focused on. In Part II, the catchments are considered in isolation, and the analysis relates to groundwater accumulation as the key driving process. Consequently, irrigation-induced salinity is approached as a stock externality (or as correlated stock externalities). In Part III, the catchments are replaced within the broader context of the river system, and the analyses relate to the interactions developing between water management initiatives at the catchment and at the river scales. Consequently focus is placed on the spatial dynamics of river flows rather than on groundwater accumulation, so that environmental concerns are accommodated through the setting of static constraints.

Research goals.

The Australian policy setting provides the rationale for the choice of studied instruments. In this respect, the focus of this thesis is twofold. A first aim of the thesis, developed in Part II, is to investigate the use of dynamic taxation schemes based on a measure of group performance, as a way of implementing the notion of collective responsibility. It addresses theoretical questions such as the strategic interactions arising from the implementation of a policy instrument based on the observation of a collective result, the aggregation of individual performance. A second aim of the thesis, developed in Part III, is to address the environmental impacts of various water markets designs. In this respect, theoretical questions such as the number of policy instruments needed to manage multiple correlated externalities are addressed.

Outline of the thesis.

Part I presents the institutional background prevailing in Australia and the theoretical frameworks within which each subsequent Part develop. Chapter 2 illustrates the interactions that developed between the Australian environment and the policies aimed at managing water and environmental resources. It highlights the necessary adaptation of the policies, first aimed at increasing irrigated activities, then aimed at integrating productive and environmental concerns. The policy trends that form the basis of the analyses in this thesis are defined : on the one hand, the recourse to price-based policy instruments together with the consideration of the notion of collective responsibility at the catchment scale, and on the other hand, the recourse to water markets to manage water scarcity and the need to refine them to account for other environmental externalities. Chapter 3 reviews the use of group performance as a basis for policy design. It specifically addresses the main characteristics of group performance based instruments, the interdependency they introduce among the agents. Indeed, when an instrument based on group performance is implemented, the payoffs to the agents under the scheme result from the effort provided by the group. The review shows that group performance has an interest beyond the standard case of ambient taxes to manage nonpoint source pollution, and poses the question of the optimal use of individual and collective incentives in the design of policy instruments. Chapter 4 reviews the literature on water markets when environmental concerns, focused on the quantitative or qualitative features of the resource water, are accounted for. Beyond the theoretical rationale for implementing water markets, allocative efficiency, it illustrates the need to refine water markets to accommodate environmental impacts others than water scarcity. Indeed, the special nature of the traded good 'water' induces various problems following its reallocation, associated to either its qualitative or quantitative features. With regards to irrigation-induced salinity, the review puts in perspective the interest for recharge rights markets, and poses the question of their integration with the existing system of diversion rights market. Chapters 2, 3 and 4 are drawn together in Chapter 5 in order to develop the hypotheses that are tested in this thesis.

In Part II the use of dynamic taxation schemes to manage irrigation-induced salinity is analysed, focusing on the catchment as the relevant management scale. Chapter 6 details the modeling choices and assumptions, including the recourse to differential games. In Chapter 7 irrigation-induced salinity is analysed as a stock externality. First, individual individual agents, following either open-loop or feedback strategies, are shown to accumulate more and more rapidly than what is socially optimal. Then, various taxes are investigated, including a time-independent standard input tax, a state-dependent ambient tax and a stock-dependent input tax. These taxes are illustrative of various ratios of individual performance (input use) and collective performance (groundwater accumulation) in the design of the policy instrument. One of the results of this analysis is that including a measure of group performance in the policy instrument is necessary to induce the agents to follow the optimal path along the whole time horizon. Furthermore, it appears that the ratio between the individual performance component and the collective performance component needs to be greater when agents follow feedback strategies, than when they pre-commit to open-loop ones. In Chapter 8, the analysis is extended to a setting exhibiting correlated stock externalities.

Part III addresses water management over a river system, constituted of multiple catchment. It poses the question of the number of policy instruments needed to manage correlated externalities. The externalities considered in this Part are water scarcity along a river, and catchment-specific manifestations of irrigation-induced salinity. Chapter 9 presents the model, in particular the definition of environmental constraints by the regulator. Then two ways of enforcing these constraints are envisaged. Chapter 11 considers the enforcement of the constraints *per se*. Chapter 11 focuses on cap and trade systems. A result of this analysis is that the correlation existing between the externalities doesn't rule out the need for a policy instrument for each.

The thesis is concluded in Chapter 12 with a summary of the results. A discussion of research opportunities completes this final Chapter.

BACKGROUND AND CONCEPTUAL FRAMEWORK

This Part of the thesis is dedicated to a presentation of the conceptual frameworks within which the analyses carried out in Parts II and III are set. This work focuses on the management of the various manifestations of irrigation-induced salinity in the Murray Darling Basin, which comprises more than 70 per cent of irrigated areas of Australia. Irrigation was developed in this area as the main driving process of the settlement project of the young colonies of Australia, mainly New South Wales (NSW) and Victoria (VIC). Chapter 2 presents the interactions between the Australian environment and the policies aimed at managing water and environmental resources. It illustrates the necessary adaptation of the policies, aimed first at increasing irrigated activities and then at integrating productive and environmental concerns. It also puts in perspective the policy trends that form the basis of the analyses carried out in Parts II and III : the consideration of the notion of collective responsibility together with the recourse to price-based policy instruments, developed in Chapter 3, and the need to refine water markets to account for external impacts, analysed in Chapter 4. Chapters 2, 3 and 4 are drawn together in Chapter 5 in order to develop the hypotheses that are tested in this thesis.

Chapitre 2

Irrigation-induced salinity in Australia : management strategies

This Chapter presents the development and evolution of water and salinity management policies in Australia, with a particular focus on NSW and VIC where historically irrigation has developed to a greater extent. Section 2.1 provides an overview of the governance arrangements to manage water and salinity. Then in Section 2.2 a historical perspective is adopted in order to show the adaptation of the policy context to the evolving environment. Finally, Section 2.3 discusses the latest State policy developments, and illustrates the trends that will form the basis of the analyzes provided this thesis, among which the consideration of the notion of collective responsibility, the recourse to price-based policy instruments and the need to refine water markets to account for external impacts.

2.1 Levels of governance arrangements for water and irrigation-induced salinity

Governance arrangements about water and salinity have evolved, and are still evolving in Australia, with an increasing involvement of the Commonwealth, which has no direct constitutional power over the environment yet (Vourc'h and Price 2001). In particular, in all the States, the right to the use, flow and control of water is vested with the Crown. It is also of the States' responsibility to manage environmental issues. Indeed, the Constitution establishing Australia as a Federation was drafted when the existing Colonies, or States, already exerted power over water and land use management. A nation-wide environmental consciousness appeared only in the late 1970s and 1980s, marking the beginning of the involvement of the Commonwealth at a time when environmental issues relating to water and salinity were already stringent. In an attempt to reduce the duplication of activities at multiple levels, the Commonwealth has delineated National Initiatives, for which it acts as the initiator and coordinator, in cooperation with the States. Examples include the National Greenhouse Policy, Australia's Oceans Policy, the National Forest Policy, the National

Strategy for the Conservation of Australia's Biological Diversity (Vourc'h and Price 2001) and the various policies described in Section 2.2.

Transborder issues rapidly pushed the need for a Basin-wide management approach, next to the State and Commonwealth scales. The management of the waters of the Murray was one of the first issues addressed after Federation. The profound drought lasting from 1895 to 1902 increased pressure on the Murray River, which was already the object of dispute between VIC, NSW and South Australia (SA), the latter strongly in favor of the notion of minimum flows to support navigation activities. The River Murray Waters Agreement (RMWA) was signed in 1915 by the Commonwealth, and the Governments of NSW, VIC and SA. It set out the basics of the management of the waters of the Murray, which are still in force today, in particular the requirement that VIC and NSW supply SA with a guaranteed level of flows. The Murray Darling Basin Agreement (MDBA) was signed in 1987 as an amendment to the RMWA[1]. The purpose of the MDBA is 'to promote and coordinate effective planning and management for the equitable, efficient and sustainable use of the water, land and other resources of the Murray-Darling Basin' (Quiggin 2001, p.75). Figure 2.1 shows the location of the MDC. The MDBC is responsible for the design and implementation of policies that have had a profound effect on water and salinity management in Australia : the Cap on water diversions and the Salinity and Drainage Strategy. The former drew attention on the unsustainable level of water diversion, and introduced a then-temporary emergency measure to ensure the trend was at least stabilised. The latter introduced another type of cap on salt emissions, by issuing State level Salt Disposal Entitlements[2].

While the Commonwealth could override States' powers over the environment (due to its responsibility for protecting features of national importance, but also for external affairs and trades, which may have an indirect impact on natural resource management in the country) its strategy has been to avoid confrontation until very recently (Vourc'h and Price 2001). Indeed, recent initiatives, such as the appointment of a Federal Minister for Water and the Environment, mark a new era of involvement of the Commonwealth in the management of natural resources. This culminated late January 2007, when at the peak of a drought crisis in some parts of the country, the Prime Minister launched a new A$ 10 billion policy package (Prime Minister of Australia 2007). Central to this National Plan for Water Security is the requirement that the State members of the MDB Council (MDBC) cede their control over the Basin to the Commonwealth. Also, arguing for the lack of effectiveness of the MDBC[3] the Prime Minister proposed that it be turned into a Commonwealth Government Agency. As of March 2007, all signatories to the MDBA,

[1] Queensland (QLD) signed it in 1996, the Australian Capital Territory (ACT) in 1998.
[2] Both policies will be discussed in detail later in Section 2.2.
[3] Most criticisms concern the lack of effective sanctions for States in breach of the Cap, and the consensus-based principle central of the functionning of the MDBC, which is not considered conducive to efficient decision making.

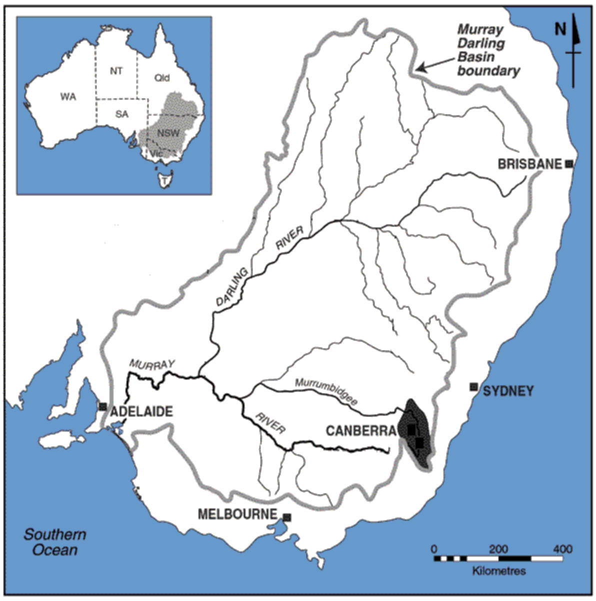

Fig. 2.1 – Location of the MBDC.
Source : CSIRO (2007).

except VIC, have agreed to hand over their powers over water, in exchange of the appointment of an independant body of experts.

Next to these decisional levels of governance, the operational levels of water and salinity management are the catchment management authorities and the irrigation areas and districts. Besides some individual riparian irrigators, most of irrigation development was undertaken within the framework of government schemes, irrigation areas (IA), or private schemes, irrigation districts (ID)[4]. In the last 20 years, Catchment Management Authorities (in VIC), Catchment Management Boards (in NSW) and Catchment Water Management Boards (in SA) have emerged as a relevant scale of management for environmental issues, especially those relating to the water cycle.

In figures 2.2 and 2.3 the current water and irrigation salinity governance settings are presented. Following Challen (2000), figure 2.2 illustrates the relations between the various levels of governance with respect to the allocation of water rights. The main property rights levels are shown : common property (Basin level and group level, areas or districts), State property and individual property. Figure 2.3 illustrates the relations between the various institutional levels with respect to the management of salinity. On the left hand side, the funding opportunities allowed by the Commonwealth involvement in environmental issues are presented. On the right hand side, is illustrated the main MDB policy development, the issuing of salt disposal entitlements (SDEs).

[4]In IAs, government irrigation agencies service individual irrigators, the infrastructure being owned by the government. In IDs, the distribution of water is done by private agencies, while the infrastructure is collectively owned and managed by the irrigators.

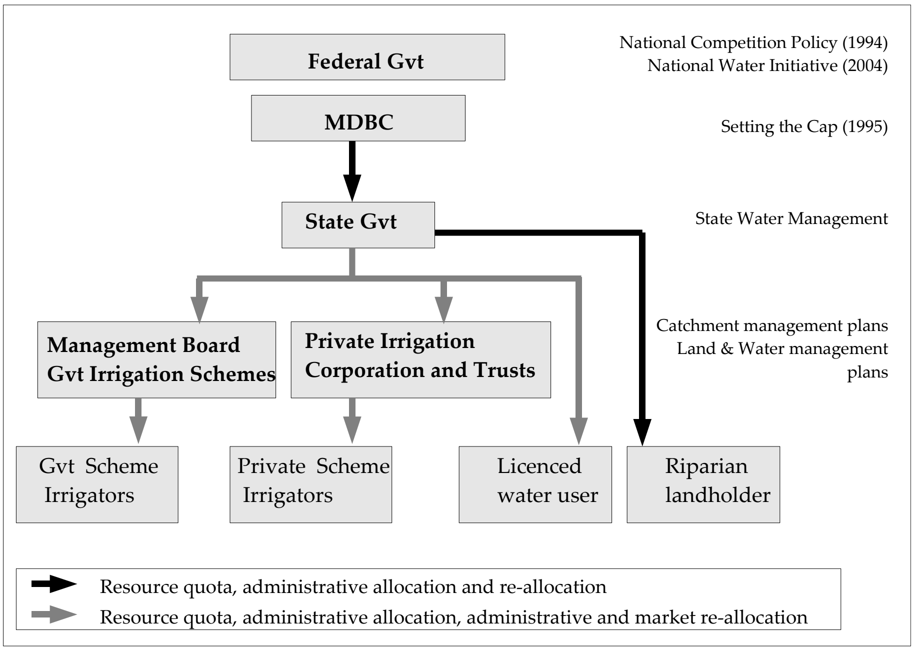

FIG. 2.2 – Institutional setting : water.
Source : Challen (2000).

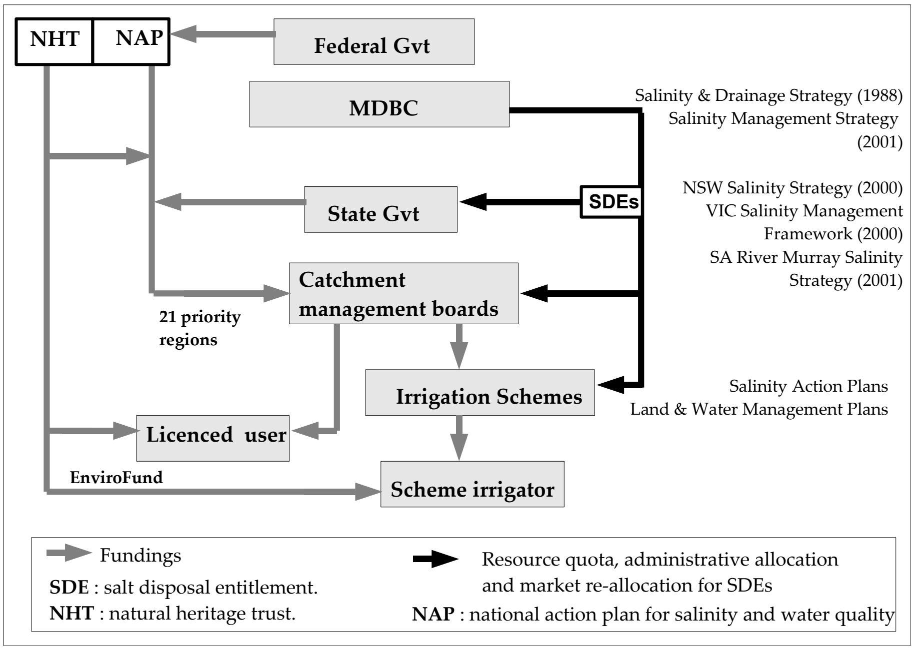

FIG. 2.3 – Institutional setting - salinity

In each case, early policies have been the initiative of the MDBC and were backed later on by National initiatives, that provide targets and guidance for subsequent State reforms. The group level (catchment or irrigation areas or districts) is important in the implementation phase.

2.2 Biogeography and policy development : links and clashes

This Section provides a historical perspective on water and salinity management to illustrate the rationales that prevailed for past management and that explain the evolution of current initiatives. The development of water and salinity strategies are illustrative of a continuous interaction between the policies and the environment, each impacting on the other. The first decades of European settlement in the colonies of Australia confronted the settlers with a totally new environment, that prompted the need to adapt the British common law they had inherited (see Section 2.2.1). The State governments endorsed the role of water developers in order to secure settlements on their lands, with impacts on the environment. The extent of environmental degradation, and a National impetus given to microeconomic reforms, then pushed the need for reforms, leading to the current policy context (see Section 2.2.2).

2.2.1 Early water policies : the common law faced to the Australian context

Water usage in the former Australian colonies reflected the English common law riparian doctrine, itself inherited from Roman law. Any person who held a land title to the bank of a river had the right to use the water from that source. The right was limited to the use of water, and did not give the right-holder ownership of the resource. This applied to ordinary uses, such as domestic uses. Other uses were permissible if they did not involve a sensible diminution of the volume or the quality or did not adversely affect downstream users [5]. The main industry at the time, gold mining, was highly dependant on water and needed secure supply (Taylor, McGlynn and Martin 2001), which could not be provided in the context of riparian rights[6]. Also, turning Australia into 'a bucolic clone of English cottage farms' (Tisdell, Ward and Grudzinski 2002, p.15) through the development of government-sponsored irrigation settlements meant that water supply had to be secure, not only along the river banks. As population developed, and competition between users arose, administrative and legal arrangements for controlling water rights became necessary. The devastating droughts of the years 1877-81 put water at the core

[5] Haisman (2005) notes that the notions of 'domestic use' and 'sensible diminution' lacked a clear definition.

[6] The same types of conditions led to the adoption of the prior appropriation doctrine in the Western States of the US.

of social and political stakes, and triggered the search for adapted legislation (Smith 2001).

At the end of the 19th century, each of the colonies conducted Royal Commissions of Inquiry into the management of water resources. They all recommended the replacement of the common law riparian system of water rights by a statutory law that would set up a system of administratively granted usufructuary water use rights. The most influential of these Commissions was the Victorian one, led by Alfred Deakin (Smith 2001), that led to the Irrigation Act 1886[7]. Deakin formulated his recommendations after a visit to the Western States of the US. While his principles are closer to the English common law than to the prior appropriation, the extent to which it subordinates the rights of the individuals to those of the State is greater (Smith 2001). Indeed, the Irrigation Act 1886 over-rode the common law rights of riparian landholders to take water flowing past their properties. However, these landholders retained the rights to divert water for domestic and stock purposes.

Table 2.1 presents the distinction made by Randall (1981) between the two phases of irrigation development, referred to as the 'expansionary' and 'mature' phases. The good condition of the infrastructure in the expansionary phase ensures an efficient supply, for instance by avoiding leakage externalities. The demand for water is such that no real competition exists between different sectors : no real constraint binds on water use. Furthermore, the perceived marginal benefit from irrigation is such that the social cost of subsidizing water use is assessed as very low. As population grows, the water economy enters a mature phase. The social cost of developing new irrigation schemes (and at some point of maintaining some) exceeds the benefits. Environmental issues and pressures on the resource become such that new policies to resolve disputes among users or to manage environmental issues need to be designed.

	Expansionary phase	Mature phase
Long-run supply of water	Elastic	Inelastic
Demand for delivered water	Low and growing	High and growing
Conditions of delivery systems	Good and fairly new	Poor
Competition for water among sectors	Minimal	Intense
Externalities	Minimal	Pressing
Social cost of subsidizing water use	Fairly low	High and rising

TAB. 2.1 – Characteristics of expanding and mature water economies.
Source : adapted from Quiggin (2001).

The next Section presents the policy context prevailing during the expansionary phase of irrigation development in Australia, during which the State governments took the role of water developers.

[7]For a review of the evolution of water rights in Victoria, refer to Harris (2005).

2.2.2 The expansionary phase of irrigation development and its impact on the environment

Increasing the storage capacity. Between 1900 and 1940, the storage capacity in NSW was multiplied by 45, as illustrated in Table 2.2. Then, from the 1950's, the Commonwealth started to get involved in the management of natural resources, in particular through the financing of irrigation development projects. The economic justification for the involvement of Federal funds into the construction of irrigation infrastructures was that 'in the long run, it is reasonable to expect that this cost will be offset by the extra revenue resulting from increased productivity and increased population' (Powell 1989).

Year	VIC	NSW	SA	QLD	Total
1900	0.13	0.08	0.02	0.01	0.25
1940	4.50	3.63	0.12	0.07	8.73
1990	12.22	25.41	0.26	9.8	87.26

TAB. 2.2 – Storage capacity in gigalitres (GL) of the major dams in the States members of the MDBC, 1900, 1940 and 1990.
Source : adapted from Smith (2001).

The Snowy Mountains Scheme is the most famous example of the Federal Government funding major irrigation works. Aiming at supplying the Murray and Murrumbidgee Rivers, this project cost A$800 millions for a storage capacity of 9,000 GL, a doubling of the country's capacity (Smith 2001). The dam became a national emblem, the example of Australia's success in managing water. The scheme was constructed for two reasons, to produce electricity and to increase the security of supply of irrigation water in the MDB. The economic reasoning underlying the project was that the returns on investments of the project would come only from the production of electricity, no contribution to the capital costs being expected from the irrigation sector. This amounted to a subsidy system for irrigated agriculture (Smith 2001). No more large dams were constructed after the 1980's. Indeed, the need for an economic rationale as well as environmental issues were emerging, a sign that the water economy was entering its mature phase (Smith 2001).

Ignoring the signals : first signs of salinity. Proust (2003) documents a series of accounts of manifestations of salinity by early settlers in South Eastern Australia. She also shows that examples of salinity-induced failures of irrigation schemes were available and documented in other parts of the British Empire, including India. The example of the Murrumbidgee Irrigation Scheme (MIS) is striking. The MIS was the first intensive irrigation project established in Australia ; consequently its development was highly influenced by the engineering profession, with concern primarily focused on the issue of water delivery. Together with the construction of the delivery infrastructure, soil surveys were undertaken as soon as 1903 on more than 1 million hectares. The Department of Agriculture provided a classification of soils according to their likeliness of being waterlogged. 'First class lands'

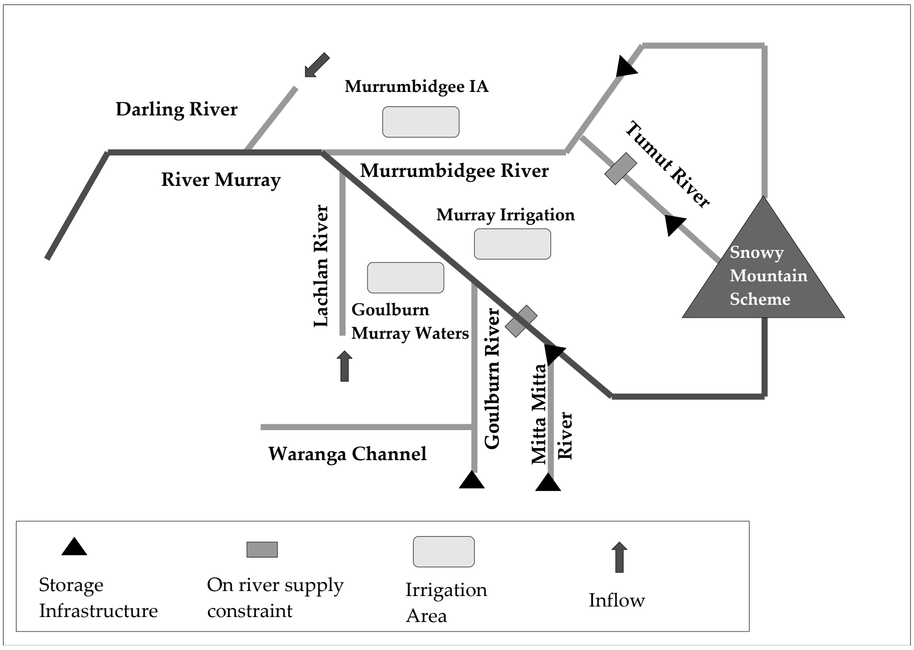

FIG. 2.4 – Schematic diagram of the MDB.
Source : Heaney et al. (2006).

were those with good natural drainage. 'Second class lands', poorer in natural drainage, were prone to waterlogging in the hands of inexperienced irrigators who tend to over-water the crops. The first irrigation blocks were leased in 1912 when water was made available to second class land. Proust (2003) identifies here an ignorance of signs of irrigation-induced salinity by the policymakers in NSW.

A highly regulated water system, near closure. The degree of regulation of a river is an indicator of the mature state of the irrigation sector. Figure 2.4 presents a schematic diagram of the MDB. It illustrates the role of storage infrastructures, in all parts of the Basin. The construction of dams and the release of water according to irrigation needs altered the seasonality of river flows. An illustration is provided by Figure 2.5. First, the quantity of water flowing is lower now that before irrigation development, especially outside of the irrigation season, during the winter months. Second, the seasonality of flows has been changed, with a fairly constant level of flows except during the peak of irrigation, which leads to an inversion of the seasonal flow. This has consequences on riparian and river species, which depend on the seasonality of the river flows to survive (Kingsford 2000).

The expansionary phase of irrigation in Australia led to the over-allocation of water, partly because of its low perceived value. One of the reason is the 'hidden' subsidy system to irrigated agriculture illustrated by Federal funds injected into the Snowy Mountain Project. Also, the water right system tied water ownership to land ownership. This had the consequence of locking water in unused or inefficient uses. Finally, the ageing infrastructures

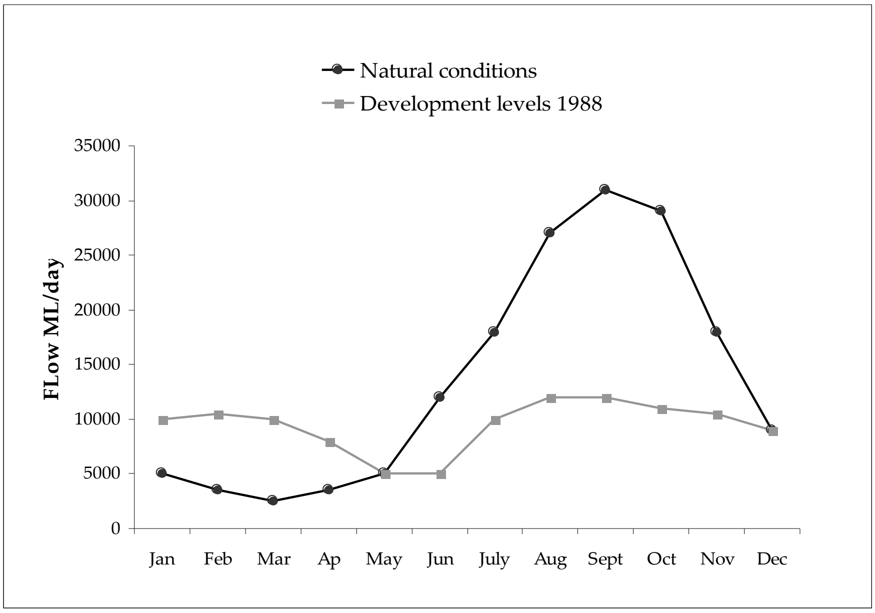

FIG. 2.5 – Median Monthly Flows : River Murray.
Source : http ://www.mdbc.gov.au.

needed investment, which was more and more difficult to find (Haisman 2005), because of a shift in the priorities of the Governments, and of society at large. Economic and environmental issues arising in the 1980's prompted the need for reforms. During the next twenty years, the Governments re-targeted their funds, from the construction of infrastructures to the financing of research projects to assess the impacts of irrigation (Smith 2001). The five founding myths of the the management of irrigation system were rejected (Smith 2001) illustrating the realization that : water is not a free commodity, it can't be managed irrespective of other considerations, the desert can't be turned into a garden, social values will change and the management of water isn't a technical problem only. Next the main national and regional water and salinity policy initiatives for water and salinity management are presented ; they form the framework within which State policies develop, as shown in Section 2.3.

2.2.3 A water reform agenda

COAG 94 : a national reform of the water sector. In 1994, the Council of Australian Governments[8] agreed on the need for a national water reform agenda, and issued a series of principles. These principles consolidated the reforms that had been emerging in the different States and promoted a nation-wide consistent effort. The main aim of the COAG reforms was to align the water sector with the National Competition Policy[9],

[8]COAG : comprises the Premiers of the States together with the Prime minister of the Commonwealth of Australia.

[9]NCP : a package of micro-economic reforms based on the principle of efficiency for all the sectors of the Australian economy, including the water sector.

through the separation of water rights ownership from land ownership, and the facilitation of water trade. The COAG also agreed on the need for environmental flows to restore rivers' health[10]. These recommendations were accompanied by a set of financial incentives[11]. Consequently, the strongest impetus towards water reforms was not for environmental reasons, rather than based on economic grounds (Hussey and Dovers 2006). This double discourse of rising environmental concerns and economic purposes is a characteristics of most Australian water initiatives.

The Cap 96. An audit of water use in the MDB conducted in 1995 showed a continued steady growth of water use as unused rights became activated, putting the Basin's river system under stress. The MDBC then agreed that a balance needed to be struck between consumptive and instream uses of water and introduced a (then interim) Cap on further increases in diversions or extractions. Each year the Cap on extraction is calculated basing on the level of development of the infrastructures that prevailed in the irrigation season 1993/94 and on the current climatic conditions. It is thus a dynamic measure. Such a regulatory instrument does not constitute an attempt at reducing extraction rather than a prevention against further increases in extraction. The main implication of the Cap is that all future growth in water based economic productivity must come from gains in water use efficiency or from water trade (Haisman 2005).

The combination of these two measures (capping water extractions and enabling or enhancing the potential for water trading) had various consequences. First, the level of diversion has been stabilised, and since the implementation of the Cap, most valleys have complied with it (National Competition Council 2004*a*). Second, the market has been successful in inducing structural changes in the irrigation sector, for example by moving water out of pasture areas to more productive and water efficient horticulture and wine areas, as argued by Turral et al. (2005). However, some concerns have been raised. Indeed, the reform package has put a pressure on irrigation communities, who felt that their water rights were not secure enough. This led to some reluctance to engage in water trading, and may explain why some institutional barriers to trade remain at the local level. Also, it has had the unexpected consequence of increasing the level of diversion, 'sleepers' and 'dozers'[12] becoming valuable and effectively used. These concerns, among others, prompted the design of a new national strategy, resulting in the National Water Initiative being launched

[10] Other key principles of the reforms include : consumption-based pricing, full cost recovery and transparency of cross subsidies and the approval of investment based on economic viability and ecological sustainability (Council of Australian Governments 1994).

[11] Satisfactory progress against the NCP reforms entitles a State or Territory to a *per capita* share of A$16 billion of transfers from the Commonwealth. The compliance and progress of the States with the NCP is assessed by the National Competition Commission (NCC). For instance, the NCC reported in 2004 about NSW not providing 'evidence to show that the ecological requirements of the downstream wetlands and the native flora and fauna of the system would be met' (National Competition Council 2004*b*, p.40) and consequently suspended the payment of A$26 millions as part of the NCP Payments 2004-05.

[12] Sleepers and dozers are the water licences that are either owned but not used (sleepers), or only partially used (dozers).

in 2004.

National Water Initiative 2004. In June, 2004, the COAG launched the National Water Initiative (NWI) to address two issues of national importance that are increasing the productivity and efficiency of Australia's water use while at the same time ensuring the health of the river and groundwater systems. To achieve this double target, the NWI outlines various objectives : to expand permanent trade, to provide more secure water access entitlements, to design more sophisticated water planning, to address over allocated systems and to better manage urban environments. In short, the objective is to increase the certainty of the environment and of investors in the water sector. By the national scale it endorses, the NWI was considered of unprecedent importance in the history of water management in Australia. However, Hussey and Dovers (2006) highlight potential problems with the implementation of the NWI, such as the lack of coordination of the various levels of governance within the Federal system. The recent National Plan for Water Security is a strong illustration of the Federal government's will to take responsibility for water management in the MDB.

2.2.4 A salinity reform agenda

Salinity has not been recognised as an environmental issue of national importance until recently. Earlier reforms were undertaken within the framework provided by the MDB.

MDBC Salinity and Drainage Strategy (SDS). The SDS[13] tried to meet two conflicting objectives, increasing the Murray river's quality while at the same time protecting irrigation zones from waterlogging, using financial transactions. It constituted a once-off agreement between the MDB States and the Commonwealth to finance a program of investments to improve River Murray salinity, through the construction of salt interception schemes. These schemes have proved successful in reversing the salinity trend between 1990-2000. However the long-term sustainability of such schemes is being questioned (Kefford 2000). Indeed, as an engineer solution to salinity, their main consequence is to move the salts from the river to evaporation basins, from where they eventually go back to the river system, by percolation or run-off. Another outcome of the SDS was the creation of salinity credits, the salt disposal entitlements (SDEs), granted to individual States in recognition of their financial participation to salt disposal schemes[14]. The SDS clarified the notions of past and present responsibility towards stream salinity. Each State was rendered responsible for its actions affecting stream salinity, the salinity levels over the period 1975-85 being adopted as the baseline for attributing impacts of all future actions.

[13] Issued in 1989 by the MDB Ministerial Council.

[14] States are then free to decide how to manage their SDEs within their boundaries. For instance, the State of VIC has devolved credit trading to the individual level within the State ; the right to dispose of saline water is granted to IDs and IAs provided that they contribute financially to the building and maintenance of interception schemes downstream (Newman and Goss 2000). Thus, no action that increases salinity is allowed unless offset by interception works.

Such a system clearly distinguished between future and past actions, States not being held responsible for past actions.

The SDS had the positive effect of accelerating the identification and implementation of solutions to tackle salinity by creating a value and accountability for salinity pollution[15]. Nevertheless, it soon became obvious that further actions were needed. The Salinity Audit conducted in 1999 predicted that salinity levels at Morgan[16] would increase by 50 per cent over the coming 50 years and that in some areas, salinity levels could preclude agricultural and rural needs (Newman and Goss 2000). This audit led to the issuing of the Basin Salinity Management Strategy.

Basin Salinity Management Strategy (BSMS). In 2000, the States members of the MDBC agreed that each would do its part to attain a global salinity target[17] and to respect catchment baselines. This constituted a policy shift from the SDS, from a budget-constrained to an environmental target-constrained policy. Also, it introduced the notion of living with salt, and the option of holding salt in place rather than mobilizing it, and insisted on dryland salinity as a potential major contributor to stream salinity. The BSMS also addressed the issue of responsibility towards future salinity increase caused by past actions that have not yet impacted on salinity concentration. Under the BSMS, the partners have agreed to offset salinity arising from past and future developments, by investing in new interceptions schemes. Each State is then free to choose its strategy, under the condition that it follows the MDBC protocols to account for salinity impacts. Thus different States have designed different initiatives, reflecting their different histories and locations. NSW and VIC constitute the historical irrigation areas, and have both issued detailed salinity strategies, focused on the catchment scale and the involvement of the communities. SA is special as it doesn't comprise any significant tributaries to the River Murray, so it has little impact on the global target ; however its capital city is highly dependant on the salinity target for its drinking water provision. QLD, not as affected as the other States, developed a program to review salinity hazard areas, and pushed towards the creation of Catchment Management Groups to put greater emphasis on salinity prevention.

COAG 2000 : taking salinity into account nationally. The recognition of salinity as a major threat to the country's sustainability was first embedded in broad environmental policies, with the creation of the Landcare Program, in 1992, and of the Natural Heritage

[15]As a result of the SDS, 14 major IDs have developed Salinity Action Plans or Land & Water Management Plans (Newman and Goss 2000).

[16]Morgan is a town in SA located just upstream of the water off-takes to Adelaide. Instream salinity at Morgan is thus an important indicator as this water constitutes the main source of potable water for the capital city of SA, Adelaide.

[17]to maintain salinity below 800 EC at the reference measurement point (Morgan) for 95 per cent of the time.

Trust, in 1997, both aiming at promoting the conservation and sustainable use of the environment through voluntary programs. It was only in 2000 that the COAG officially recognised salinity as a major issue for the country (Council of Australian Governments 2000). It initiated the National Action Plan for Salinity and Water Quality (NAPSWQ). Funded with A\$700 million over 7 years from the Commonwealth, and another A\$700 million from the States and Territories, it built on existing government initiatives to tackle salinity and water quality problems. Twenty-one highly affected areas were identified, and benefit from funding for their salinity management plans. These regional plans and investment strategies include practical remedies such as the protection and rehabilitation of waterways, improvements to native vegetation, engineering works changes to land and water use. Another point of the NAPSWQ was the focus on new approaches such as Market Based Instruments in order to tackle salinity[18].

Current water managers, and more generally environmental managers, are subject to constraints left by the history of water management in Australia (Freebairn 2003). It is now accepted that water has been over-allocated and that the current allocation is inefficient. The manager has a rationale for enhancing water trade to produce efficiency gains. However, Freebairn (2003) refers to an 'ownership assumption' as commercial users perceive they have a legitimate right to the property rights to their current uses of water. Also they have historically attached low prices to water. Furthermore, as illustrated in this Section, concerns for the environment are quite recent, the NWI 2004 putting the restoration of river health at the core of its strategies. The recognition of the environment as a legitimate user of water, that will be discussed in the next Section, is an essential part of the water market reform. Hence the difficulty to implement the reforms. With respect to salinity, the misunderstanding of the Australian landscape led to mismanagement strategies, which in turn led to increasing environmental issues. Engineering issues are now accompanied by catchment-based strategies.

2.3 Trends in water and salinity management : introducing the research goals

In this Section the latest State policy initiatives are illustrated, putting in perspective five main trends. First, there is an increasing recognition of the notion of collective responsibility at the catchment scale. Second, price-based instruments are increasingly used. This will form the basis of the analysis of group performance based instruments provided in Chapter 3. Then, to facilitate the development of water rights markets, the notion of unbundling of water rights is advocated. Also, at the same time, the environment is increasingly recognised as a legitimate user of water, while institutional barriers to trade

[18] In 2003 ten projects were chosen to benefit from A\$5 million. They cover the range of issues associated with dryland and irrigation-induced salinity.

are progressively removed. This forms the basis of the analysis of diversion rights markets when irrigation-induced salinity is an issue, provided in Chapter 4.

Recognition of a collective responsibility at the local scale. Increasingly, the catchment appears as the revelant operative scale to implement water-related initiatives, especially when they are targeted at integrating water with land and vegetation management (Hussey and Dovers 2006)[19]. Recently, the 'catchment care principle' was proposed by the Wentworth Group[20] as an alternative way of attributing the bearing of costs for natural resource management. The catchment care principle specifies that individual resource users have an obligation to avoid natural resource management practices that harm the long-term interests of resource users as a whole. Such an initiative is illustrative of the recognition of a collective responsibility, targeted at the totality of the users impacting on a resource. More precisely, this translates into the setting of end-of-valley, and within-valley, targets in terms of water quantity and quality.

Annual diversion targets per valleys. To assist in the implementation of the Cap on diversions, each identified valley in the Basin is constrained with respect to its annual diversions[21].

End of valley and within valley salinity targets required by the BSMS. Recently, to align to the draft Murray-Darling Basin Salinity Management Strategy 2001-2015, NSW has released a Salinity Strategy that provides the basis for the establishment of end of valley (and within valley) salinity targets. Hence, the scale of action has evolved from the State level to the catchment level. An end of valley salinity target is a 'river-based target for salinity at a point in the lower reach of a catchments main river' (NSW Department of Land and Water Conservation 2000, p.7), while a within valley target express 'the salinity levels desired to maintain important social, environmental and economic assets and values for locations or areas within the catchment' (NSW Department of Land and Water Conservation 2000, p.8). While the former are a requirement of the BSMS, the latter are left at the discretion of the States.

Pricing for sustainability : price-based instruments. 'Pricing for sustainability' is one of the key principle endorsed in VIC, as described in the latest water policy initiative, the White Paper (Victorian Government 2004). It follows recommendations of the NWI to have recourse to pricing mechanisms for water-related environmental externalities, for which the COAG has never provided any guidelines. Two initiatives, illustrating the use

[19]The Landcare movement is illustrative of this tendency, with more than 4,000 community groups that have volunteered to tackle land degradation, including salinity-related ones.

[20]The Wentworth Group of Concerned Scientists is an independent group of Australian scientists concerned with advancing solutions to secure the long term health of Australias land, water and biodiversity (Wentworth Group of Concerned Scientists 2007).

[21]See Schedule F of the Cap ; among the 22 valleys defined in Schedule F, seven don't have any cap yet (five in QLD, 2 in NSW).

of price-based instruments to account for irrigation-induced salinity, follow.

Environmental contributions raised by VIC's water authorities. A first illustration of the use of price-based instruments for the management of water-related issues is the environmental contribution raised by Victorian water authorities (rural and urban) 'to contribute to the costs of initiatives promoting the sustainable management of water and addressing the adverse environmental impacts of water use' (Productivity Commission 2006, p.14). Contributions vary between 5 per cent of the revenues, for urban authorities, and 2 per cent of the revenues for rural water authorities. In 2004-05 they generated a total of A$ 18.5 millions that were invested back into a range of programs to restore rivers and aquifers (Productivity Commission 2006). The underlying idea behind this mechanism is that the environmental contributions will be repercuted by the water authorities to the users through an increase of the price of water.

Sunraysia salinity levy, SA. The Sunraysia salinity levy, introduced in 2002, concerns the irrigation districts and the individual irrigators located along the Murray River from Nyah to the SA border. Water trade is very active in the area ; 46,000 ML of permanent water licences have been traded into the region between 1990 and 2000 (Duke and Gangadharan 2005). The scheme introduces constraints on water trades between irrigators located in various salinity impact zones. The zones have been defined over the area by landscape modelling, they differ according to their potential contribution to instream salinity : one High Impact Zone (HIZ) and four Low Impact Zones (LIZ). The funds raised by the scheme are used to finance salt interception schemes and drainage diversion schemes that offset salinity increases due to irrigation. The principles behind the scheme are as follows. First, only buyers pay the levy ; sellers incur no levy on sales. Second, potential buyers from the HIZ can only purchase from a seller located in the HIZ ; hence sales from a LIZ to a HIZ are forbidden. Third, buyers from a LIZ can purchase from any zone, incuring differentiated levies according to the zones. Consequently, this scheme has been criticised as introducing an asymmetry between buyers and sellers, but also between LIZ and HIZ. Indeed, simply forbidding sales from LIZ to HIZ potentially prevents highly efficient activities located in a HIZ from producing. Hence authors have proposed to introduce a levy for sales into HIZ, set at the appropriate level (Duke and Gangadharan 2005).

There has been some attempts at designing price-based instruments to account for irrigation salinity. They imply a recognition of the link between water use and salinity generation ; consequently the instruments apply to water price. They are also a way of recognizing the collective aspect of salinity generation, and water management, at a zonal scale[22]. Chapter 3 reviews the literature on policy instruments based on collective performance (and more precisely taxa-

[22] The zonal scale can be defined as the catchment, the irrigation district, a HIZ, a LIZ.

tion schemes) to address the design of efficient price-based instruments.

Recognition of the environment as a legitimate user of water In Victoria, the 2005 Water (Resource Management) Act provided a legal framework for the 'Environmental Water Reserve' (EWR), a legally protected share of water to be set aside to maintain the environmental values of both surface and underground water systems. Consequently, the water available for consumption is legally capped. VIC also engaged in recognizing existing entitlements ; hence it is accepted that in some instances the reserve may be 'inadequate in maintaining river health' (Victorian Government 2004, p.20). Nevertheless, the legal definition of the EWR is an important step. NSW has also defined the Environment as a legitimate user of water ; however the NCP has recognised that the accompanying measures necessary to provide appropriate environmental allocations of the most stressed rivers were not satisfactory. As a consequence, it recommended a suspension of the competition payments to NSW, up to 10 per cent (approximately A$13 million in 2004-05) ; the latest assessment lead to a continuation of the suspension for the period 2005-06 (National Competition Council 2004*a*) (National Competition Council 2005). Hence, if efforts are done to recognise environmental needs with respect to water, implementation is rendered difficult by numerous factors. Due to the degree of stress of the water systems, and on the existing entitlement system, implementation appears more difficult in NSW than in VIC, which has always undertaken more conservative water management practices, illustrated for instance in the higher security of entitlements.

Unbundling of water rights The unbundling of water rights from land titles has been recognised early as one of the key measures to facilitate water trading ; it is now the case in most Australian jurisdictions (Productivity Commission 2006). Young and McColl (2002) advocate a 'robust separation' of the three main entities constituting water rights : the entitlement (the share in a varying stream of periodic allocations), the allocations (a unit of opportunity -usually a volume- as distributed periodically) and the use licence (a permission to use allocations with pre-specified use conditions and obligations to third parties). While SA has not experienced unbundling to a greater extent than the separation of water rights from land titles, VIC, NSW and QLD have already separated water entitlements from water use licences. This facilitates trades as the water authority in charge of approving trade only has to approve the exchanged entitlements ; then only users holding water use licences will have the right to make use of the entitlement. VIC adds another layer of unbundling, as the new licence system includes : a water share (a legally recognised, secure share of water available for consumption), a delivery share (an entitlement to have water delivered to a property) and a water use licence (an entitlement to use water for irrigation on a property) (Victorian Government 2004).

Removal of barriers to trade. Finally, another key initiative to enhance water trading, as outlined in the NWI 2004, is the removal of barriers to trade 'that are not applied to protect the environment or ensure the practical management of trading [...] or provide a public benefit case for their continuance' (National Competition Council 2005, p.iv). SA, VIC and NSW all committed to remove all barriers to temporary trade, and to remove impediments to permanent trade up to an interim limit of 4 per cent of total entitlements (National Competition Council 2005) on the designated area[23]. The NCC has also identified new forms of barriers to trade implemented in some areas. For instance, in QLD, some irrigation districts are imposing exit fees to farmers that get out of irrigation to compensate the loss of funds to manage districts' infrastructure (National Competition Council 2005). Also, in VIC, a limit has been set on the number of non water-users allowed to hold entitlements : 10 per cent of total entitlements. This is a way of re-creating a link between land and water ownership, and thus constitutes a barrier to trade (National Competition Council 2005) .

The Federal Government, through the NWI 2004 and the recent NPWS, has created strong incentives to enhance water trading. At the same time, it recommended a series of 'safeguard' measures to account for a particular understanding of the environment, namely water scarcity. Voices arise to point out the need to take account of other consequences of trade, 'third party impacts' as usually understood, upstream-downstream interactions, but also other types of third party impacts, associated with water quality features Heaney et al. (2006). Chapter 4 reviews the literature on the impacts of water trade on the generation of environmental externalities, and analyze the special case of irrigation-induced salinity.

[23]The objective is to ensure full open trade by 2014 at the latest.

Chapitre 3

Group performance based instruments and environmental issues : a review

The analysis of the Australian policy context provided in Chapter 2 highlighted the increasing use of the notion of collective responsibility, at various scales, and of price-based policies to manage water and salinity. The notion of collective performance is attractive, as it resounds with notions of group interactions that have positive reputation (Itoh 1991) especially in the context of catchments where people are used to interacting and cooperating on certain matters (Millock and Salanié 2005). The objective of this Chapter is to analyze the various rationales that have been put forward to enhance the use of collective incentives in the design of price-based environmental policies. In the remainder of this Chapter, such instruments will be referred to as group performance based instruments (GPBI) as it focuses attention on the notion of group ; in the context of irrigation-induced salinity the group under study consists of the irrigators that impact on the underlying aquifer by their irrigation practices.

Recently, group performance based instruments have been addressed as a solution to environmental issues, especially in the context of catchment based management. GPBIs specify the desired policy outcome as the aggregation of individual performance but give the agents discretion in how they need to meet the outcome. Examples include the US Federal Government's threat to list salmon as an endangered species in Oregon, unless action is taken to restore its habitat. In Florida, to restore water quality in the Everglades wetlands, a tax 'for the privilege of conducing agricultural trade' was implemented. The tax base is the total phosphorus loading from an agricultural area ; each farmer is thus subject to a tax that depends on the aggregate pollution generated by the farmers from the area.

Performance based policies are often contrasted with design-based ones, which specify how the agents are to meet regulatory requirements. More precisely, interest in performance based policies arises from the relative failure of design-based strategies that have been traditionally used to manage environmental issues, such as command and control or voluntary schemes (Romstad 2003). While performance-based policies give the agents some flexibility on the choice of practices they make to attain the target, design-based policies rely on the application of Best Management Practices, available at the time of implementation of the policy, without offering rewards for exceeding this benchmark situation (Isik 2004). Also, performance-based policies provide incentives directly targeted at the environmental outcome. This constitutes a shift of strategy from the regulator, as the environmental awareness and responsibility of the agents are central to performance-based schemes. Another key difference between design-based and performance-based policies is that the latter provide more predictability to the regulator about the state of the environment, as the desired environmental outcome is identified as the policy target. Interest for group performance, as opposed to individual performance, quite intuitively stems from the fact that most environmental issues result from the actions of multiple agents. The context of catchment-based water management is particularly illustrative of this notion. Indeed, the interactions that exist between surface and underground water resources, and between the qualitative and quantitative features of these resources, make it necessary to account for all the contributors to the various environmental issues that arise from the exploitation of water.

Theoretical developments of GPBIs are numerous. They are typically associated with the ambient policies that have been proposed to manage nonpoint source pollution as a group moral hazard issue (Segerson 1988) (Shortle and Horan 2001). However, other strands of literature have analysed the inclusion of group performance in policy design, in contexts such as the contribution to a public good (Groves and Ledyard 1977), the management of stock pollution (Benchekroun and van Long 1998) and common pool resources (Schott, Buckley, Mestelman and Muller 2003), or biodiversity preservation (Krawczyk, Lifran and Tidball 2005). Consequently, the first objective of this review is to clarify the definition of GPBIs as solutions to environmental problems.

In a second step, this review focuses on the main characteristics of GPBIs, that is the interdependence they introduce among the group of agents. Indeed, when a GPBI is implemented, each agent under the scheme has a payoff function that results from the effort provided by the group. To take the above-cited example of ambient taxes, each agent is subject to a tax proportional to the amount of pollution in excess of an exogenous environmental target. This obviously introduces strategic interactions between the agents, which have consequences on the efficiency of the instruments. Hence, a second objective of this review is to derive the different types of strategic interactions that arise from the implementation of GPBI.

In this thesis, only a particular type of GPBIs is studied, namely dynamic taxes. The first reason is that environmental taxes are increasingly used and advocated as effective and efficient instruments to manage the environment (Vourc'h and Price 2001). Chapter 2 illustrated the increasing use of price-based instruments in Australia to manage water-related issues. A second reason is that the most stringent current environmental issues, such as acid rains, global warming or groundwater depletion, result from the accumulation of pollutants, or exploitation of resources, over a long period of time, and as such necessitate dynamic approaches. This is particularly true for irrigation-induced salinity.

This review is organised as follows. Section 3.1 defines the notions of group performance and GPBI in the environmental context. Then Section 3.2 provides a typology of strategic interactions arising from the implementation of GPBIs. Finally Section 3.3 develops some concluding remarks.

3.1 Performance and group performance based instruments : some definitions

In this Section, some definitions are provided to set the analysis in the context of environmental policy design. GPBIs are illustrated in both theory and practice.

3.1.1 Performance

The Oxford English Dictionary defines performance as 'the carrying out of a command, duty, purpose, promise'. In an economic context, this roughly translates into how well a task works. For instance, key macroeconomics measures of performance are the gross domestic product, inflation and unemployment rates. Applied to the environmental context, performance is usually associated with the consequences that carrying out a productive task has on the environment, for instance in terms of water scarcity level or area of remnant vegetation. Figure 3.1 illustrates different understandings of performance for the case of an agricultural pollutant, at both individual and collective levels.

Polluting inputs, such as fertilizers or herbicides, are applied for agricultural production. They are the traditional basis for design-based instruments, such as input norms or taxation schemes (Griffin and Bromley 1982). As a result of precipitations, fractions of polluting inputs are flushed out and end up in the surface water system. These individual emissions are either measured or estimated by way of modeling. Individual field losses aggregate to form stream loads, which constitute the most obvious group performance measure. They are affected by the state of the environment, through its natural absorption capacity for instance. Various water quality indicators exist, measuring various expressions of group performance. For instance, the occurrence of algal bloom illustrates a high nitrogen pollu-

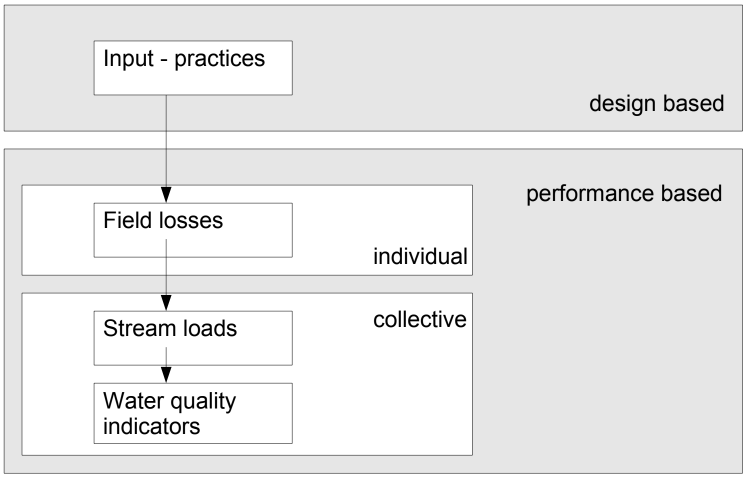

Fig. 3.1 – Individual and collective performance indicators - river pollution case.

tion rate in a water body. Hence group performance is measured for a specific ecosystem, at specific spatial and time scales.

This analysis is transposable to any productive task which impacts on the environment. In the case of fisheries, group performance can be understood as the size of the collective catch (Schott et al. 2003) or the extent of the remaining stock of fish (Jensen 2001). The same applies to the management of a forest (aggregate amount of logged wood vs remaining stock), or to the contribution made to a public good provision (aggregate amount of contributions in terms of practice change vs impact on the environmental amenity) (Krawczyk et al. 2005).

A GPBI specifies the desired policy outcome, defined as the aggregation of individual performance, but gives the agents discretion in how they need to meet the outcome. Theoretical interest in them is not new, as noted by Randall (2003), however it is only recently that they have become a real focus of interest from an implementation perspective. The following points illustrate this gap, as there are still few applications of numerous theoretical developments.

3.1.2 Group performance based instruments

Tax/subsidy schemes

In an environmental context, GPBIs are usually understood as the ambient tax/subsidy instruments firstly proposed by Segerson (1988). Building on the team production literature, Segerson (1988) addresses nonpoint source pollution, characterized by the lack of observability of individual emissions, as a group moral hazard problem. Group moral hazard is defined by Holmstrom (1982, p.324) as 'the problem of inducing agents to supply proper amounts of productive inputs when their actions cannot be observed and contracted for'. Inducing workers to produce an effort when they cannot be monitored is equivalent to preventing polluters from over-emitting when their emissions cannot be measured. In both cases, group performance is the only observable policy basis. Agent-specific taxes (subsidies) are charged (paid) when the ambient pollution exceeds (falls below) the exogenous environmental target :

$$\begin{aligned} T(a) &= t_i(a - a_0) + \tau_i \quad \text{if } a > a_0, \\ &= s_i(a - a_0) - \beta_i \quad \text{if } a \leq a_0, \end{aligned}$$

where t_i is a tax rate, s_i a subsidy rate, τ_i and β_i lump sum penalty and bonus, a_0 the exogenous target and a the ambient pollution. Authors have provided a number of extensions of the Segerson scheme[1], exploring issues such as the multiplicity of equilibria (Spraggon 2002), the non budget-balancing nature of the instrument (Xepapadeas 1992), the assumption of non-cooperative behavior (Millock and Salanié 2005) or introducing an individually observable component as another base for the instrument (Xepapadeas 1995, Kritikos 2000).

GPBIs have also been analysed in contexts other than nonpoint source pollution, including stock pollution (Ko, Lapan and Sandler 1992, Karp and Livernois 1992, Benchekroun and van Long 1998) and landscape change as a public good (Krawczyk 2005), for reasons other than circumventing the lack of observability of individual emissions, as will be shown in Section 3.2.

The Everglades Forever Act provides one of the few examples of implementation of a tax/subsidy scheme based on group performance. It was signed in 1994 to establish Florida's commitment to restore water quality. Major projects for draining the area, together with the development of agricultural production, severely affected the wetlands. Among the initiatives designed to reduce agricultural pollution, the Act imposes an annual tax for conducting agricultural trade in the Everglade Agricultural Area (EAA) and the C-139 Basin. Known as the 'privilege tax', it concerns over 650,000 acres in total. Landowners are given the opportunity to decrease their tax liability, down to a minimum of US $ 24.99 per acre, by earning incentive credits for reducing phosphorus load discharges from the area. Individual credits reward individual performance when it is measurable[2]. Area-wide credits are available to the totality of the area's landowners, when measured phosphorus loads fall below the baseline discharge[3]. Hence both individual and group performance are

[1] See Shortle and Horan (2001) for a review of this topic.

[2] EAA landowners must meet phosphorus reduction standards established by the Act in order to be eligible for these credits.

[3] The baseline discharge is defined as the total estimated load that would have occurred during the base period 1979-1988. Credits are available when the reduction exceeds 25 per cent.

rewarded through this scheme. Another example is provided by the Tar Pamlico Nutrient Trading program, where point source polluters are considered as a group ; when their aggregate emissions exceed a pre-determined target, they have to pay an incentive fee used to finance agricultural cost-sharing programs targeted at nonpoint source polluters of the Basin (Hoag and Hughes-Popp 1997).

'Voluntary instruments'.

A strand of literature has focused on voluntary approaches for environmental management. Segerson and Miceli (1998) analyse the level of abatement negotiated between a firm and the regulator. Dawson and Segerson (2004) consider that the regulator has defined an industry-wide pollution target, and uses an emission tax as a background threat if the required abatement is not attained voluntarily. Segerson and Wu (2006) consider an ambient tax as the background threat[4] if the so-called voluntary approach[5] is unsuccessful in meeting the environmental target. They show that first-best conditions can be obtained under such a scheme even if the ambient tax is not applied. The threat of imposition of the tax is then sufficient to induce voluntary compliance to the collective target.

Karp (2005) reports the extreme example of the US Federal Government threatening to list salmon as an endangered species in the State of Oregon if nothing is done to restore its habitat. This prompted the State to develop a plan to enhance voluntary citizen cooperation for the restoration of the salmon's natural habitat (Green Plans 2007). Millock and Zilberman (2004) report the example of the Bureau of Reclamation threatening to stop providing water to a group of irrigators until they had taken action to reduce selenium loading into the Kesterson Reservoir of the San Joaquin Valley[6].

Under the Kyoto protocol, the European Union 15 committed to an aggregate 8 per cent reduction in greenhouse gaz emissions. The reduction is redistributed amoung the member States according to a burden sharing agreement which specifies individual targets. If the aggregate target is met, all the States will be considered in compliance with their requirements. However, if the 8 per cent reduction is not reached, then only the States not in compliance with their individual target are held responsible (Rübbelke and Dijkstra 2006).

Contracting approaches

Some studies have included group performance in the design of contracting mechanisms specified to deal with adverse selection. Romstad (2003) designs a contract where a principal allows nonpoint sources of pollution to choose between two alternatives : some stan-

[4]This ambient mechanism can take the form of reduced governmental subsidies.
[5]Indeed their voluntary nature is facilitated by the threat of ambient tax.
[6]This drainage-related issues was found responsible for extensive wildlife death in the region.

dard, design-based, regulatory setting and a contract. This contract is favorable to the agents as a team relative to the first option, provided that the team of agents meet the targeted emission level, but unfavorable to the team if the target is not met. Taylor (2003) analyses a team contract that can be used as the basis of exchange within a group performance based water quality trading market between point sources of pollution. With this system, a point source polluter can purchase abatement from various nonpoint sources of pollution, whose compliance can be verified by the measurement of actual ambient pollution.

This Section provided some definitions and illustrations of the concepts of group performance and GPBIs in the environmental context. The remainder of this Chapter focuses on the main characteristics of GPBI, the dependency they introduce between the individual agents' payoff functions, with a particular interest in dynamic taxes.

3.2 Group performance based tax/subsidy schemes and strategic interactions

Instruments based on a measure of collective performance are relevant in cases where the agents are aware that they can influence their policy burden by the decisions they make. Recognizing the effect of their actions on the level of the policy instrument means that the agents exhibit a strategic behavior with respect to the decision maker (Karp and Livernois 1992). In addition to this, when a group of agents is subject to a common policy instrument, the agents interact through strategic responses. Hence implementing a GPBI supposedly induces two types of strategic interactions, one specifically between the regulator and the agents individually, and one among the group of agents.

Most analyses of dynamic tax/subsidy schemes are set in the framework of differential games. Central to the resolution of such games is the informational structure of the agents (Dockner, Jorgensen, Van Long and Sorger 2000, Karp 2005). Two main informational structures are usually used in dynamic games : open-loop and feedback strategies. In the former case, each agent takes the rival strategies as simple functions of time when he determines the optimal trade-off between the current and future effects of his actions (Dockner et al. 2000). In the latter case, an agent considers that rival actions are functions of the state of the game at the same period ; consequently he takes into account the modification of rival actions after the change he introduces to the state variable, through feedback effects (Dockner et al. 2000)[7].

Consequently, the resulting open-loop Nash equilibrium captures the strategic interactions vis à vis the regulator (Karp 2005), and overlooks inter-agent strategic responses, while

[7]The informational structure used in differential games will be discussed in more details in Chapter 7.

the feedback Nash equilibrium also captures the strategic interactions vis à vis the other agents (Karp 2005, Dockner and Fruchter 2004). As a result, the results shown in Section 3.2.2 that focuses on inter-agent strategic interactions were obtained under the assumption that the agents follow feedback strategies.

3.2.1 Group performance : more than an answer to informational asymmetries

A review and synthesis of various studies of dynamic collective tax/subsidy schemes in an environmental context shows that reliance on group performance as a policy basis is motivated by the strategic interactions that arise between the regulator and the individual agents, typically in two types of contexts. The first, and most studied, is when there exist informational asymmetries between the decision maker and the agents, which renders the use of standard instruments based on individual performance difficult. There exist a second case where, irrespective of informational issues, including group performance allows improving the efficiency of individual instruments.

Collective performance as an answer to informational asymmetries

The first and most obvious reason to have recourse to group performance is that no other instrument basis is satisfactory. Informational asymmetries between the policymaker and the agents prevent standard individual instruments from inducing optimal decisions by the agents. Two types of information asymmetries are developed here : unknown individual abatement costs and unobservable individual emissions. Group performance based instruments are a way of overcoming the high transaction costs associated with generating private information. Basing on the observation of collective results, the policymaker designs explicit incentives to induce the agents to make optimal decisions from a social viewpoint.

Abatement cost. In the context of a flow pollutant, Karp and Livernois (1994) consider that individual abatement costs are unknown to the policymaker. Consequently, he cannot compute the optimal individual rates for an emission tax. The authors design a tax which adjust automatically according the difference measured between the observed and desired levels of aggregate pollution. Noting X_t aggregate pollution at time t, X^* a predetermined pollution target, s_t the tax at time t and α an adjustement parameter, the tax adjustment rule is as follows :

$$\dot{s_t} = \alpha(X_t - X^*) \quad \text{if} \quad s > 0, \text{ or } s = 0 \text{ and } X_t \geq X^*,$$
$$\dot{s_t} = 0 \quad \text{if} \quad s = 0 \text{ and } X_t < X^*.$$

Reliance on a tax adjusting automatically according to observed aggregate pollution allows the regulator to induce the polluters to conform to the social optimum.

Individual emissions. Following Segerson's (1988) seminal paper, various studies analysed the ambient tax scheme in dynamic nonpoint source contexts. Xepapadeas (1992) studies the case of a nonpoint source pollution accumulating over time. He shows that there exist inter-temporal schemes that induce the agents to attain a desired level of accumulated pollution in the long-run. The efficient scheme takes the form of an effluent charge, based on the deviation between the desired and observed pollution accumulations. Karp (2005) adapts Karp and Livernois' (1994) adjusting tax to a dynamic nonpoint source problem without pollution accumulation. Consequently, the state variable of the game is the tax adjustment rule, according to the difference measured between the observed and desired levels of ambient pollution. Karp (2005) shows that when only the strategic interactions between the decision maker and the agents individually are taken into account the adjustment mechanism induces the agents to overreact to the current level of the tax by decreasing their emissions in order to forestall the future level of the tax.

In both cases, recourse to GPBIs is a way to circumvent informational asymmetries. The characterization of a GPBI as an emission tax that adjusts according to the group performance takes advantage of the strategic interaction between the decision maker and the agents so that the latter are induced to restrain from polluting in order to impact on the future level of the tax.

Collective performance as a way to improve the efficiency of individual instruments

GPBIs have not only been studied to circumvent informational issues but also as ways of improving the efficiency of instruments based on individual incentives only, in contexts of perfect information where other constraints impede the policymaker from inducing the agents to perform optimally.

Constrained budget. Krawczyk et al. (2005) propose a coupled incentive scheme to allow a budget-constrained policymaker to induce agents to produce a given quantity of a positive externality[8]. They consider a group of farmers who by changing their field mix composed of intensely cultivated areas and less cultivated areas can impact on the level of biodiversity in a given region. Indeed, by grouping less cultivated areas together, they can promote biodiversity, as wildlife reproduces faster in aggregate wilderness than in scattered backwoods. Krawczyk et al. (2005) consider the case of a budget-constrained policymaker. They first show that an individual incentive scheme reimbursing the agents the cost of technology shift will induce a null reaction if the policymaker is not able to finance the scheme fully. That is why they analyze a coupled incentive scheme, a subsidy based on both

[8]While set in a static framework, this analysis can be extended easily to a dynamic setting (Krawczyk et al. 2005).

individual and collective performances. They show that a budget-constrained policymaker can induce the agents to perform a target level of technology shift at a lower cost by using the coupled-incentive scheme.

Time consistency. A stream of literature has focused on dynamic taxation schemes. Contrary to the nonpoint source pollution studies, in this case information is not an issue, so that the authors concentrate on the accumulative process of pollution and its impact on the decision making process of the agents. Ko et al. (1992) adapt the standard Pigouvian tax to a dynamic setting and compare time-dependent taxes with time-independent taxes, inflexible but easier to implement. The latter can be adjusted once according to the observation of accumulated pollution. Bergstrom, Cross and Porter (1981) also derive time-dependent tax/subsidy schemes that induce a monopolist to comply to the efficient exploitation of a non-renewable resource. Karp and Livernois (1992) show that this scheme, being time-dependent, is subject to strategic manipulation by the monopolist if the regulator cannot commit to the entire sequence of tax. In order to ensure time-consistency of the policy scheme, they derive a Markov perfect tax rule that induces the monopolistic to exploit optimally the resource. The tax is Markovian because it depends on the current level of the stock, not its entire accumulation history. It is also subgame perfect. Benchekroun and van Long (1998) derive a Markov perfect emission tax to induce oligopolistic polluters to follow an optimal pollutant emission path. The tax rate is conditional upon the level of the stock of pollution, so that the agents receive the message that their emission decisions affect the level of the tax they are, and will be, subject to.

To sum up, reliance on group performance in the design of dynamic taxes is motivated by two main arguments : to circumvent informational issues, when collective performance is the only observable item available to the policymaker ; and to improve the efficiency of policy instruments based on individual performance only, when information is not an issue. The following point is devoted to inter-agent strategic responses to the implementation of GPBIs.

3.2.2 Strategic interactions among agents : when the informational structure matters

Inter-agent strategic responses arise when the agents are assumed to be aware that their actions have an impact on the decision making process of the other agents. Hence each will account for the others' decisions, and for the impact of his decision on the decision making of the others. The nature of the strategic incentive that will arise from the implementation of a GPBI typically depends on two items. First, the sign of the externality at stake (positive or negative). Second, the nature of the strategic relations between the agents' actions (complementarity or substitutability)[9]. This review focuses on negative externalities

[9]See Jun and Vives (2004) for a detailed taxonomy of strategic incentives arising in duopolistic interactions when there are adjustment costs.

(such as the accumulation of pollution or the exploitation of a non-renewable resource). However, the complement/substitute nature of the agents' actions is not *a priori* given.

Strategic complementarity/substitutability. The distinction between strategic complementarity and substitutability was first introduced in industrial organization theory (Bulow, Geanakoplos and Klemperer 1985) and applies to other settings. Actions are defined as strategic substitutes (complements) when the marginal benefit of increasing one's own action decreases (increases) if his opponent increases his action. Consider two agents, i and j, with control variable u_k and benefit function π_k, $k = i, j$,

$$\frac{\partial^2 \pi_i}{\partial u_i \partial u_j} < 0 \quad \Rightarrow \text{static substitutability,}$$
$$\frac{\partial^2 \pi_i}{\partial u_i \partial u_j} > 0 \quad \Rightarrow \text{static complementarity.}$$

In dynamic contexts, one refers to inter-temporal substitutability (resp., complementarity) as the concept relies on the stock rather than directly on the reaction functions (Jun and Vives 2004). Denote x_k the pollution stock ($x_i{=}x_j$ is the usual stock pollution case where all the agents contribute to the same stock),

$$\frac{\partial u_i}{\partial x_j} < 0 \qquad \Rightarrow \text{inter-temporal strategic substitutability,}$$
$$\frac{\partial u_i}{\partial x_j} > 0 \qquad \Rightarrow \text{inter-temporal strategic complementarity.}$$

Note that absent from this formulation is one's impact on the change in the state variable.

Even in the absence of any policy instruments, agents may develop strategic interactions. Indeed, their payoff functions can be interdependent even when no policy instrument based on group performance is applied. This is the case of certain cases of pollution which aggregation has a direct impact on agents' payoffs. For instance, in the context of irrigation-induced salinity, each irrigator suffers from the rising of the watertable that results from too much percolation water being produced by a group of irrigators (Wichelns 1999). Another example is the phenomenon of acid rains, the deposition of sulphur and nitrogen oxides above a critical load, which directly affects polluting countries (Mäler and de Zeeuw 1998). In this case, one may talk of 'endogenous' strategic interactions arising between the agents as it depends on the context only. The remainder of this Section focuses on 'exogenous' strategic interactions that develop because of the implementation of a policy instrument.

Changing nature of the interaction. Karp's (2005) adjusting tax has the property of inducing the agents' actions to be either strategic substitutes or complements. When the tax rate is low, agents pursuing feedback strategies will be induced to lower their emissions in order to keep the tax low, reflecting the strategic complementarity concept. However, when the tax rate gets higher, agents have a greater incentive to increase their emissions, because the resulting increase in the tax will discourage the others from polluting, as emissions become strategic substitutes. Karp (2005) departs from the other analyses of dynamic taxation schemes as the state variable of the game is the tax adjustment rule rather than the aggregate level of pollution.

Negative incentives. Xepapadeas (1992) and Benchekroun and van Long (1998) analyse stock pollution, either in a perfect information (Benchekroun and van Long 1998) or imperfect information (Xepapadeas 1992) setting. While Xepapadeas (1992) assumes strategic substitutability for agents' strategies, Benchekroun and van Long (1998) show that agents effectively develop strategic substitute strategies. In both cases, the agents reason as follow : 'if I don't pollute, the others will benefit from it by over-polluting, following the signal sent by the tax. Consequently, I'd better pollute more myself now'.

3.3 Concluding remarks

While relying on collective performance bears interesting features for the regulator, according to the informational structure of the agents under study, there may be some negative consequences.

In both Xepapadeas' (1992) and Benchekroun and van Long's (1998) studies, agents' awareness of their decisions' impact on the others' payoff function generates negative incentives. Agents tend to free-ride on the others' efforts to restrain from polluting. The consequence in terms of policy design differs according to the possibility to rely on another type of incentive, individual incentive. On the one hand, in Xepapadeas' (1992) model, the regulator relies on collective performance only. Hence, to induce the agents to behave optimally the tax rate has to be higher in the feedback case than in the open-loop case so that it becomes too costly to cheat. On the other hand, Benchekroun and van Long (1998) analyze a policy instrument based on both individual and collective incentives. They show that the optimal collective part of the tax has to be lower in the feedback case. The regulator's strategy is here to reduce the incentive to cheat by increasing the individual part of the policy instrument. In Part II of the thesis, this type of analysis will be extended to provide a comparison of instruments based on individual performance, collective performance, and a mix of both, to assess whether there is an optimal mix of each type of incentive. Also, the impact of the agents' informational structure will be investigated.

The underlying model in use will supposedly be of importance to assess the precise impact of GPBIs in the case of irrigation-induced salinity. The consequence of having endogenous, or coupled externalities, will also be tested.

Chapitre 4

Water markets and environmental externalities : a review

The analysis of the Australian policy context in Chapter 2 showed the increasing interest in markets for water rights in order to manage the growing water scarcity concerns. It also stressed the increasing concerns for environmental externalities others than water scarcity (in particular related to irrigation-induced salinity) and for the impact that the development of water markets may have on those externalities.

This Chapter provides a literature review of water markets when environmental concerns, focused on the quantitative or qualitative features of the resource water, are accounted for. Starting from the theoretical rationale for implementing water markets, allocative efficiency, the need to refine water markets when environmental impacts are present is highlighted. Indeed, the special nature of the traded good water induces various problems following its reallocation, associated to either its qualitative or quantitative features.

This Chapter is structured as follows. Section 4.1 introduces the main merits of a water trading system and presents some cautionary theoretical and empirical analyses. They illustrate the need for refined market designs in order to fully account for the environmental externalities potentially generated by the introduction of a water market. Section 4.2 presents how these externalities have been accounted for in the literature. Finally, Section 4.3 addresses the context of irrigation-induced salinity and raises some questions that will form the basis of Hypotheses 2a and 2b of this thesis.

4.1 Allocative efficiency through water trade ?

4.1.1 Theoretical foundation

The merits of alternative allocative systems have been much studied in a context of ever increasing pressure on water resources around the world (Weber 2001). Numerous water sources in the world are fully allocated so that pressure arises to reallocate water to suit new conditions and uses. The main theoretical rationale for implementing water markets is that, under four assumptions which economists agree seldom exist in practice (a set of well defined property rights, atomicity of the market, reliable and available information and goods which are mobile and easily shifted to different uses and users (Tan 2005)) water will be reallocated to its highest value use. Promotion of water markets is a direct corollary of the First Theorem of Welfare Economics (Griffin 2006). Figure 4.1.1 illustrates the basics of water reallocation through the use of markets.

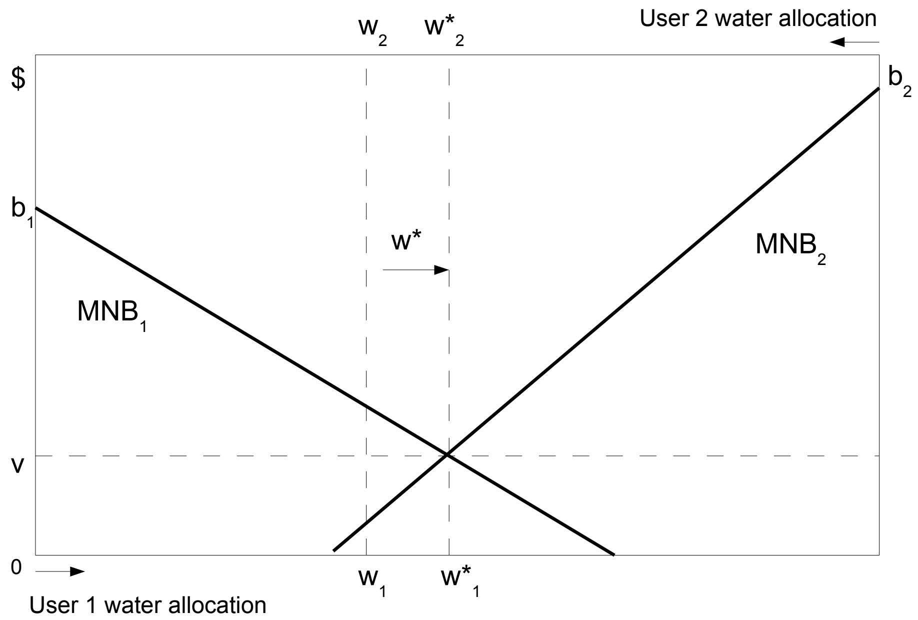

FIG. 4.1 – Basics of water trade.
Source : Griffin (2006).

Suppose 2 water users, denoted by 1 and 2, sharing a quantity Q of water. They hold initial allocations of water rights w_1 and w_2 such that $w_1 + w_2 = Q$. Their respective marginal benefits functions are linear, $MNB_i = b_i - c_i w_i$. After the initial issuing of rights, $MNB_1 > MNB_2$. User 1 has an incentive to buy rights, for a price p, as long as $MNB_1 \geq p$. User 2 has an incentive to sell rights as long as $MNB_2 \leq p$. Then, letting transaction costs aside[1], it is privately optimal for the agents to trade up to the point where $MNB_1 = MNB_2$, with a market price set at $\mathbf{v}$[2].

Water markets have the same property than other markets : to ensure equality of marginal benefits among traders. They have the potential to '... [unlock] the resource from

[1]See Stavins (1995).
[2]Refer to Dinar and Letey (1991) for an illustration of the impact of quotas and water markets on individual agents' revenues.

low-value applications' (Griffin 2006, p.207) by motivating the rights holders to appraise their water use strategies. In the short term, the less efficient users are induced to sell their access rights to those who use it in a more efficient way. In the longer term, users will improve their productivity by investing in water-saving facilities (Lahmandi-Ayed and Matoussy 2002). Markets are also preferred to other management options because they constitute a politically soft process : the government is not perceived as arbitrarily choosing the winners and losers of the reallocation process (Weber 2001)(Lahmandi-Ayed and Matoussy 2002).

However the equality of marginal benefits corresponds to economic efficiency if the private marginal benefits equal the social marginal benefits, that account for marginal external costs and marginal social benefits (Freebairn 2003). Section 4.2 illustrates that the setting of water management, particularly in dentritic systems, is conducive to the rising of environmental externalities that induce an inequality between social and individual marginal benefits.

Before that, in Section 4.1.2 some empirical studies of water markets are presented. Indeed, relatively few papers develop formal models of water markets. Burness and Quirk (1979) provide one of the first analyses of the efficiency of a system of transferable water rights in the context of the prior appropriation doctrine[3]. Other developments, in more general contexts, include Griffin and Hsu (1993) and Weber (2001). In contrast, there exists an abundant literature on site specific aspects of water markets, including simulations based on catchment specific data, description of the institutional context and description of the functioning of local markets and associated problems (Weinberg, Kling and Wilen 1993) (Vaux and Howitt 1984)(Hearne and Easter 1997). If simulations provide optimistic results (Hearne and Easter 1997), empirical analyses show more contrasted results, as addressed in the next Section.

4.1.2 Empirical studies : 'siren songs' ?

Chile and the Western States of the US concentrate the most studied examples of water markets. A couple of decades of implementation allow empirical studies to give relevant insights about the factors of success or failure of the markets. Water markets have been experienced in Australia more recently, but their fast and growing development makes it

[3]In this system, the establishment and maintainance of water rights depend on the actual use of water, and individual profit is a function of the amount of water used and of the diversion capacity developed by each user. Following the principle of 'first in, first served', rights holders have different seniorities, such that the water available to a junior user depends on the aggregate amount of claims to water by senior users. This system is in place in most western States of the US, where the English common law riparian doctrine cannot apply and the development of irrigation required ways to secure investments. Burness and Quirk (1979) show that unequal risk sharing and diversion capacity among firms explain the inefficiencies of the prior appropriation allocation. They argue that establishing a market for water rights allows the establishment of an efficient allocation with a system of prior appropriation (Burness and Quirk 1979).

an interesting case study.

Since the introduction of water markets in 1981, the Chilean model is presented as the leading example of water management through the use of markets. Hearne and Easter (1997) show how the introduction of the market potentially generates substantial gains in this context. However, they document the examples of the Elqui and Maipo Valleys where the lack of adapted infrastructure prevents trade between farmers. Hearne and Estear's optimism regarding the actual efficiency of water markets is not shared by Bauer (1997)(2004) who argues that even if water trade allowed some reallocation of water from low to high value uses, the overall efficiency gains are limited, and well below the theoretical predictions. Bauer (1997) identifies three main causes of the failure of water markets in Chile : the lack of infrastructure to allow water to move between traders, cultural attitudes that led farmers to keep hold on water and the lack of an appropriate administrative framework (for instance, there is no registration of water rights). Furthermore, environmental impacts of trade are not accounted for. Indeed, instream flows don't have any legal status in the Chilean Water Code (Mentor 2001). Also, the definition of water rights does not mention any constraints on the use of water [4]. Finally, if trades have to be approved by water users associations, government agencies are given no role in assessing third party impacts associated with trade (Bauer 1997).

In the Western States of the US, trade also remained mostly limited to the same type of users in close proximity to each others (Dellapenna 2001). Trade is rendered difficult in California by the coexistence of the riparian and appropriation systems (ACIL Tasman 2003). Authors explain the relative dearth of water markets[5] compared to theoretical predictions by the rise of local resistance to water trades. Mentor (2001) documents the example of California. Irrigation districts have developed strategies to prevent, to a certain extent, water to move out of their areas, for environmental and social reasons. The concept of 'no injury' is afforded a status in state Water Code, but only for surface water. Indeed, interactions between surface and underground systems are not recognised, mainly because groundwater is not regulated by the State. The introduction of water markets has enhanced groundwater mining, which has in turn had consequences on water availability both underground and at the surface. Also, districts are recognizing a pecuniary externality : selling water rights out of a district, and thus renouncing to agricultural activities, constitutes a loss to the local economy, in terms of jobs, sales, tax revenues and of stranded irrigation assets which become less profitable to manage (Mentor 2001). In the absence of State-level protection, local restrictions on water markets have developed.[6]. The US example illustrates that, apart from socioeconomic impacts illustrated by cultural atti-

[4] In contrast with the notion of 'beneficial use' in the riparian doctrine for example.

[5] Exceptions include the California Water Bank and the Colorado/Big Thompson project, that show the ability of water markets to move over long distances and among different users (Easter and Archibald 2002).

[6] In a theoretical approach, Vaux and Howitt (1984) have even suggested restricting water trade in California within counties to avoid negative consequences on local businesses.

tudes to water trades, third-party impacts of water markets are not restricted to the rising scarcity of instream flows. In particular, interactions with groundwater have become an issue.

The formal introduction of water market in Australia is more recent that in the previous examples. However, a number of legislations have ensured its rapid uptake, especially in the South-Eastern States of the country. A recent study by Turral, Etchells, Malano, Wijedasa, Taylor, MacMahon and Austin (2005) provides an analysis of the development of water markets in this particular region. As in the previous examples, inter-sectoral trade, let it be permanent or temporary, is very limited. However, the authors point out that this may due to the fact that most industrial activities are based on the coastline of the country, in watersheds independent from the ones supporting agriculture, so there is no real competition with the agricultural sector. Also, permanent trade is still extremely limited, accounting for 1 per cent of transaction in volume only, while temporary trade is more frequent and growing. The authors argue that permanent trades, even if very restricted, have proved an efficient tool to accompany structural adjustment, illustrating this with the example of Northern Victoria where inefficient grazing was abandoned at the benefits of highly efficient dairy farms. To explain the low number of transactions, Tisdell and Ward (2003) have documented Australian farmers' attitudes towards water markets, and argue that cultural attitudes to trade need to be accounted for to ensure the success of a water trading scheme. In the Australian context, a certain reserve to take part into the water market can be partly explained by the fact that farmers have lost confidence in their water rights because reforms are undertaken at a particularly rapid pace. Furthermore, as in the US, irrigation districts have developed ways to prevent trading water out of their jurisdiction. Murray Irrigation Corporation has stated that it will not allow individual irrigators to trade water out of their irrigation system (Turral et al. 2005). Barriers to trade are in some instance explained by the need to accommodate environmental impacts ; for instance in the Lachlan system, to prevent transmission losses, trades are restricted within small geographical areas (Turral et al. 2005).

This brief analysis of empirical experiences with water trade highlighted a number of reasons why markets for raw water may be confronted to difficulties reducing their efficiency. First, cultural attitudes towards water markets are a crucial feature of success or failure of water markets. Second, the legal and administrative frameworks underpinning the implementation of water markets condition their efficiency. Shi (2005) has inventoried more than 400 water products in the Southern Connected River Murray System differing on the level of reliability, the degree of transferability, use conditions. This obviously renders the process of exchanging rights difficult. Finally, various authors have pointed out the rising of local resistance to water trade for environmental reasons ; to protect groundwater and its indirect impact on surface water flows, in California ; or to avoid excessive transmission losses of surface water in the Lachlan valley of NSW. Often, this is due to the inability of water markets, in their current design, to account properly for the environmental externalities they generate. Analysing the theoretical approaches to this issue is the purpose of the next Section.

4.2 Taking environmental issues into account : refining water markets

Three main issues arise in assessing the efficiency of markets for water rights in a river system. First, flow constraints can become binding, when the reallocation of water from downstream to upstream users reduces the capacity of intermediate users to divert enough water to fulfill their water rights (Weber 2001). Second, consideration of the qualitative features of the resource water can impact on the efficiency of a market for diversion rights, as the reallocation of water can have effects on the assimilative capacity of a river, or parts of rivers (Weber 2001). Finally, there is increasing interest in rendering explicit the interactions between surface and underground water, hence identifying the river-aquifer system as a relevant management scale.

4.2.1 The return flow externality

Reallocation of water between users located along a river has an effect on various features of a river flow : quantity, timing, location[7]. It has been recognised early that when some users rely on the return flows produced upstream[8], any change in the patterns of diversion and use upstream can have an impact on users located downstream. This 'return flow externality' has the potential to induce water misallocation in a river system (Griffin and Hsu 1993). With the increasing recognition of environmental needs, and the requirement of minimum flows to support these environmental needs, the question of the return flow externality becomes even more stringent. In the recent literature, the problem has been addressed by considering rights defined in terms of consumptive use, instead of diversion. This approach contrasts with the previous literature on water rights, which insisted on the necessity to keep the variables 'diversion' and 'return flows' separated, preventing the use of consumption alone as a decision variable.

Recent vs earlier literature : are consumptive rights sufficient ?

In order to deal with the return flow externality, it has been shown that recourse to consumptive rights instead of diversion rights is efficient (Anderson and Johnson 1986) (Johnson, Gisser and Werner 1981). The definition of water rights as a net amount of water used, instead of in terms of gross water diverted, introduces a certain security of outcome in terms of water available in the river for diversion. By adding in their analysis the requirement that at each diversion point, the amount of instream flows be greater

[7]Leaving quality issues aside for now.
[8]Either directly or indirectly through the underground system.

than the quantity of diversions, Johnson et al. (1981) find that this constraint is violated in rare occasions. However, when return flows are used to expand the amount of surface water, which is the case in fully allocated systems, then the constraint automatically binds (Weber 2001), and the market is inefficient. Weber (2001) analyzes a consumptive rights market encompassing the reallocation of return flows, and shows that efficiency requires that the shadow price of a consumptive right be location specific, in order to reflect the dependance between upstream and downstream users.

Griffin and Hsu (1993) question the relevance of considering consumptive rights as the solution to the return flow externality in the light of an earlier literature illustrated by (Hartman and Seastone 1970) and (Hirshleifer, De Haven and Milliman 1960). Indeed, the depiction of the river system provided in (Hartman and Seastone 1970) (Hirshleifer et al. 1960) and (Griffin and Hsu 1993) is more complex than in the recent literature, in the sense that they precisely describe the interactions between instream flows, diversion and return flows. For instance, Griffin and Hsu (1993) account for the fact that water use on a given location can generate return flows that discharge into the river system at various locations downstream. The main contribution of these studies is to highlight the need for a two-tiered system of water pricing, involving charges for diverted water and credits for return flow water. Griffin and Hsu (1993) argue that the simplification of the framework operated in the recent literature may explain the theoretical support of marketing for water rights.

Environmental participation in the market ?

The above-cited studies differ with respect to their treatment of environmental flows' management. Johnson et al. (1981) or Weber (2001) treat them as constraints on instream flows enforced at each diversion point. This approach is consistent with the most widespread instream flow management strategy currently, which is the definition of instream flows requirement associated with the regulatory review of the proposed transfers[9]. This management strategy guarantees that a baseline environmental quality is achieved, but has the drawbacks of traditional command-and-control instruments, such as the lack of flexibility or the absence of incentive to exceed the target (Murphy, Dinar, Howitt, Rassenti, Smith and Weinberg 2004).

In recent years, there has been increasing interest in the various modalities of participation of 'the environment' in the market for water rights - diversion or consumption- in order to ensure that environmental flows needs are met. The environment, recognised as a legitimate water user, needs to be properly defined. Indeed, participation in the water market can translate into the government taking part in it, by purchasing water rights in order to ensure that his environmental flows constraints are met. Murphy et al.

[9]In consistency with the no injury principle of the riparian doctrine.

(2004) provide an experimental analysis of the participation of an 'environmental agent' on a market for consumption rights assuming that the environmental agent has perfect information about the social benefits of these flows. Hence the system they test amounts to transforming the instream flow requirement regulation into a cap and trade program (Murphy et al. 2004). They conclude about the potential gains from the participation of an environmental agent in the market ; however they also warn against the potential strategic behavior of this agent, who could understate its willingness to participate in the market. This leads them to point out that should the environmental agent decide not to participate in the market, the baseline instream flow constraint will still be met, amounting to the command-and-control regulation. Griffin and Hsu (1993) consider two types of users : diverters and instream users, differing by the consumptive or non consumptive nature of the use they make of water. The authors show that optimality of a system of markets for diversion and consumption rights requires that instream users organise as Instream Flows Districts (IFD) for each portion of the river between two diversion points. In both studies is made the assumption that the environmental agent, or IFD, is equipped with the relevant information about the social damage from the lack of instream flows. This assumption could be difficult to sustain in real settings, an IFD could either over or under estimate the need for instream flows.

The return flow externality has been addressed broadly in the literature. The first lesson is that if consumption rights appear attractive as a solution to this problem, this comes from a simplification of the framework. Indeed, more detailed analyses, considering that the constraint on instream flows between users is binding (Weber 2001), or considering different types of users and more detailed flows (Griffin and Hsu 1993), show the need for spatially differentiated prices, special institutional structures or a combination of various water rights markets[10].

4.2.2 Surface water quality

Water is the medium of transport and dilution of numerous assimilative pollutants[11]. Few studies have addressed the impact of implementing water markets on the damage from pollution such as agricultural nonpoint source pollution. However, reallocating water may have consequences on the absorbing capacity of a river system, by changing both the location and timing of this capacity. Unger (1978) analyzes a system of input charges able to manage efficiently input discharge into a river, with the characteristics that damage

[10]Weber (2001) agrees that the transaction costs of introducing a spatially differentiated market could impede its implementation. Griffin and Hsu (1993) also address the great information costs of detailing return flow parameters, but introduce an instream water district that partly simplifies the process. This higher ability of a 'water agency', a 'trading house' (Bell 2002) or a 'water district' to process information relating to the environmental impacts of water trade, compared to individual users, forms the basis of Bell (2002)'s analysis of a central trading house to accommodate salinity-related externalities of water use.

[11]According to Tietenberg (1985)'s classification of pollutants as assimilative/cumulative and uniformly/non uniformly mixed.

arise from the accumulation of the pollutants and their interactions. This study shows that a differentiated pricing system is needed in order to optimally manage pollution in a river system, to account for upstream-downstream interactions, but it doesn't mention water markets. Hung and Shaw (2005) provide an analysis of a trading ratio system for a pollution rights market with participants located along a river. They show the efficiency of such a scheme, as the trading ratios ensure that location effects are accounted for. Weber (2001) adds to their study the water market dimension. She analyses a market for pollution damage rights associated with a market for consumptive rights and shows that both markets must be capable of supporting localized prices, that account for the interactions between both features of the river, quantity and quality.

In almost every studies of the management of river through the use of markets, groundwater is implicitly accounted for, at least by way of reference to return flows. However, fewer studies explicitly address the impact of surface water transfers on the management of groundwater, both in terms of quantity and quality. Specific integration of groundwater in the study of surface water markets include the impact on groundwater exploitation, as an alternative to surface supplies, and the impact on the generation of poor-quality discharge from the aquifer back to the river system.

4.2.3 Surface-groundwater interactions

Analyses of the interaction between surface and underground systems that focus on quantity issues address the conjunctive use of both water resources, as an alternative to each others. Knapp and Olson (1995) show the impact of stochastic surface supplies on groundwater depletion, that appears as a buffering source of water. Amigues, Gaudet and Moreaux (1997) analyze the reservoir value of an aquifer when the population is growing, as the aquifer participate in intertemporal transfers of surface water. Knapp, Weinberg, Howitt and Posnikoff (2003) show how transfers of surface water out of an agricultural production area impacts on the management of groundwater. While leaving the question of the sign of the overall impact open, they highlight the various impacts that introducing a market may have, according to the existing arrangements over groundwater management.

Both studies by Weinberg et al. (1993) and Dinar and Letey (1991) address the externalities associated with drainage generated from irrigation. Indeed, percolation of water left after uptake by the plants may contain polluting effluents, that degrade the quality of groundwater. Also, excessive percolation is at the root of irrigation-induced salinity. Both studies show how the introduction of a market for water rights has a positive impact on the drainage issue, as a result of a general increase of water use efficiency.

In the next Section, the specific setting of irrigation-induced salinity is addressed. The three approaches to the analysis of the environmental impacts of water trade detailed above are relevant in this context. First, water is a scarce resource, managed by way of markets, and impetus is given towards removing barriers to trade, opening the way to exchanges that have the potential to reallocate water between users located upstream and downstream of a river. Second, irrigation-induced salinity is typically linked to the management of aquifers, the main underlying mechanism being an excessive recharge compared to the natural assimilative capacity of the aquifer. Third, the discharge of saline water into the river system is a uniformly mixed assimilative pollutant problem. However, the setting of irrigation-induced salinity is specific in the sense that aquifers are to be approached as 'receiving tanks' of water rather than[12] sources of water.

4.3 Water markets and irrigation-induced salinity

The context of irrigation-induced salinity, the focus of this Section, is characterised by the need to manage both surface water scarcity and excessive[13] recharge to the aquifer. First some options to manage recharge to an aquifer are addressed. They differ from the standard analyzes of groundwater as a source of water for irrigation. Then the integration of these strategies within the management framework for surface water is analysed. This raises some questions that will form the basis for the development of Hypothesis 2.

4.3.1 Managing the recharge...

Recent groundwater legislations worldwide illustrate that groundwater is losing its private property connotation (Burchi 1999). Indeed, groundwater users' rights are not necessarily linked to the ownership of the overlying land, but to the granting of a right from the Government (Burchi 1999). This analysis applies to groundwater as a source of irrigation or drinking water. However, when it comes to considering an aquifer as a receiving tank of water, no legislation applies. Hence, *de facto*, farmers have a poorly defined right to manage the recharge produced from their farm as they want to.

Strategies to induce a more efficient recharge management have relied on the prescription of best management practices and planning, as it the case for the management of nonpoint source pollution to aquifers (Burchi 1999). Currently being tested in the Colleambally Irrigation Area, a cap and trade mechanism for recharge management has been proposed by Whitten, Collins and Khan (2003). The basic idea behind this scheme is that a limit on aggregate recharge can be calculated for a given recharge area, in consistency with the adsorptive capacity of the aquifer. This target is then divided into recharge credits, issued to participants in the scheme - mainly irrigators, as they are the main users of water in the areas, but any agent taking remediation works can participate. A recharge right is then defined as a share of the adsorptive capacity of the aquifer that the group of participants

[12]According to its salinity, groundwater can remain a provider of water ; in this case 'rather than' has to be replaced by 'as well as'.

[13]Excessive means above the natural adsorptive capacity ; it is a state-dependant and evolving notion.

have in common. Participants are then allowed to trade their recharge credits ; for instance by increasing their irrigation efficiency they reduce their recharge production, hence they can sell their rights in excess. The recharge cap and trade is defined at the recharge zone level - this implies that any trade of these rights is geographically restricted. It is also important to note that the definition of the target is dynamic. Indeed, the adsorptive capacity of the aquifer might evolve each year, or each relevant time step.

The theory behind cap and trade systems is well-known, and predicts substantial gains. However, the recharge to an aquifer is particular with respect to the difficulty of observation and monitoring of individual emissions, so that successful implementation is not warranted (Whitten et al. 2003) (Whitten, Khan and Collins 2004) .

Designing a recharge cap and trade amounts to assigning rights to return flows, understood as the 'indirect' return flows that pass through the underground system before coming back to the surface system. In this respect, it is consistent with Griffin and Hsu (1993)'s prescriptions to distinguish between the three entities that are diversion, consumption and return flows. The next point addresses the integration of this geographically restricted cap and trade for recharge with the strategies designed to cope with water scarcity in a fully allocated system.

4.3.2 ... in the context of scarce water resources

The analyses carried out by Dinar and Letey (1991) and Weinberg et al. (1993) have highlighted that the implementation of a surface water market reduces drainage issues, as a consequence of induced water restrictions and irrigation technology upgrade. A first question is then : are there conditions under which a market for water rights is efficient in managing both water scarcity and recharge ?

Whitten et al. (2003) (2004) have analysed of a recharge cap and trade system, as a way to constrain the recharge to aquifers at risk of inducing salinity damage. Constraining the recharge amounts to either reducing water use or increasing irrigation or abatement efficiencies. Could a recharge cap and trade be efficient in managing both recharge and water scarcity ?

The two previous questions considered the implementation of one type of strategy, in order to manage two types of externalities. With reference to the Tinbergen principle (Tinbergen 1950), that states that to each issue should correspond a policy instrument, the remaining questions focus on the combination of the two types of strategies. More precisely, each market may have a different scale of implementation - the recharge cap and trade is by definition constrained within recharge areas, while the market for surface water rights is not necessarily. Indeed, theoretical predictions are in favor of an opening of the market in order to increase to opportunities for efficiency gains. Few theoretical analyses exist on this topic. Garrido (2000) analyses intra-districts and inter-district water trades, and compares efficiency gains in both settings. However, each setting is considered in isolation. Caplan and Silva (2005) analyse the conjoint implementation of a market for a local pollutant and a global market for green house gas emissions. They highlight the necessity to account for the interactions existing between correlated externalities in the design of markets to manage each. Hence the third question raised is : what is the impact of implementing a set of recharge cap and trade on the functioning of markets for surface rights, at various scales ?

Chapitre 5

Synthesis of Part I : hypotheses

In this Chapter the insights gained from Chapters 2, 3 and 4 are summed up to develop research hypotheses, each focused at a direction taken in the field of water and salinity management in Australia : on the one hand, recourse to price based policies and to the notion of collective responsibility to manage environmental issues, on the other hand, increasing reliance on water trade to manage water scarcity.

Collective responsibility and price-based policies to account for irrigation salinity. Individual catchments are responsible for achieving water policy targets, such as to comply with end-of-catchment flow limits under the State Caps, or to achieve end of valley - and even within valley - targets in terms of instream salinity. Policy strategies have increasingly been based on a collective performance standard, such as the extent of irrigation salinity - measured in terms of instream salinity, groundwater salinity or sensitivity to salinity. Some catchments are subject to price-based policies to account for irrigation salinity. In order to deepen the analysis of 'group responsibility' for the management of environmental issues, Chapter 3 provided a literature review of the inclusion of 'group performance' in the design of dynamic tax/subsidy schemes.

A conclusion of the literature review was that there had been no real comparison of policy instruments based on different combinations of individual and collective performances. Both policy bases have advantages and drawbacks, mainly in terms of the signals they send to the agents subject to the policy and in terms of the information burden they cause. Hence the interest to investigate combinations of individual and collective performances instruments.

A second conclusion was the importance of the nature of the externalities on the optimal combination of individual and collective performance instruments designed to manage them. Various models of irrigation salinity can be designed, with an increasing degree of complexity. The impact of the underlying hydrological model on the efficient combination of individual and collective instruments will be assessed.

These two theoretical issues give rise to the first hypothesis :

Hypothesis 1. In the context of dynamic externalities, a policy instrument based on a combination of individual and collective performances is more efficient than a policy instrument based on one type of performance alone.

This hypothesis will be tested in two hydrological settings, with recourse to the methodology of differential games, a specific type of game theory. This methodology is useful to understand the interactions arising between agents. It allows the assessment of the impact of implementing a policy instrument on agents' behaviors, and thus on the environment.

Water markets and environmental externalities. Water scarcity, which has become a crucial issues in recent years especially with the recurrent drought experienced by most States, is increasingly managed through the use of water rights trading. Chapter 2 highlighted that the most recent policy initiatives are mainly targeted at facilitating the trading of rights - by refining the definition of the rights and removing institutional barriers to trade - while recognizing the environment as a legitimate user of water, in order to secure minimum flows.

The theoretical analysis of water markets presented in Chapter 4 stressed the need to carefully identify and account for the range of environmental externalities that may be affected by the implementation of a water market, in order to derive appropriate rules for governing the functioning of a market. In particular, when water use is associated with externalities - positive or negative - then the impacts of water trade have to be accounted for. The question of the design of a policy instrument to address the induced externalities may then be posed. In the context of irrigation salinity, water use and recharge to the aquifer are positively correlated, so that using more water worsens the rising of the watertable. Consequently, the question arises of whether a policy instrument is needed to manage irrigation salinity when a market is in place to manage water scarcity - say by imposing a cap on water used. This question gives rise to the following hypotheses.

Hypothesis 2a : spatial extent. Where coupled externalities exist, a series of zonal cap and trades is more efficient than a regional cap and trade.

Hypothesis 2b : number of instruments. Where coupled externalities exist, combining two types of instruments is more efficient than implementing only one type of instruments.

Hypothesis 2c : spatial extent x number of instruments. Where coupled externalities exist, a regional diversion market associated to a

series of zonal recharge markets is more efficient than a series of zonal markets (either on diversion or on recharge).

PART II : GROUP PERFORMANCE AND IRRIGATION-INDUCED SALINITY

This part is devoted to the testing of Hypothesis 1. The catchments are considered as the relevant scale of decision-making and they are treated as independant entities. In order to address the design of optimal taxation schemes, an economic model of benefit maximisation is associated with an hydrological model describing the mechanisms at the root of irrigation-induced salinity.

As already mentionned, the main mechanism underlying the various manifestations of irrigation salinity (instream, root zone and groundwater salinity, waterlogging) is the rising of aquifers due to an hydrological imbalance : a recharge in excess of the discharge capacity. Then it is the interaction between this stock of groundwater with individual root zone salt and water stocks that is responsible for the uptake, transport and accumulation of salts in the various parts of the catchment. Building on previous studies of groundwater management, and salinity management, dynamic models of accumulation of groundwater and root zone salts, expressed in continuous time as differential expressions, are used. Consequently, the relevant methodoly to analyze such dynamic models is differential games (Dockner et al. 2000).

Chapter 6 presents the modeling choices, including the model structure and the agents' strategy spaces. Chapter 7 presents an analysis of taxation schemes when irrigation salinity is approached as a single stock problem, when groundwater accumulation is retained as the key mechanism. Then Chapter 8 addresses irrigation salinity as a set of interacting stocks, focusing on a collective stock of groundwater and individual root zone salt stocks.

Chapitre 6

Dynamic taxation schemes : modeling choices and assumptions

This Chapter presents the modeling choices made to address the optimal design of dynamic taxation schemes to manage irrigation-induced salinity. Section 6.1 presents the basic structure of the model, and introduces the control and state variables under study. Then Section 6.2 discusses the type of information upon which the agents are assumed to condition their strategies. The definition of the strategy space of the agents appears as an important step in the modeling process. Contrasted situations exist in this respect in Australia. Finally Section 6.3 analyses the informational requirements of the models.

6.1 Basic structure of the models

The two models used to test Hypothesis 1 are based on the same assumptions ; however introducing individual root zone salt stocks modifies the baseline model used in Chapter 7 to describe the interactions arising between the stocks properly.

The key common features are the following. Consider n agents, indexed by i, producing an irrigated agricultural output. By their irrigation practices, they contribute to accumulating groundwater in a confined aquifer ; the resulting rising of the aquifer has a detrimental impact on the production of the totality of the irrigators with land located above the aquifer, either directly or also indirectly through the root zone.

Production function Consistent with the irrigation economics literature (Dinar and Xepapadeas 2002), and backed by empirical studies (Dhatta, Sharma and Sharma 1998), a quadratic expression of the production function is used :

$$F(u_i, \mathbf{S}) = a + bu_i - \frac{c}{2}{u_i}^2 - d(\mathbf{S}),$$

where u_i is agent i's irrigation water use, $\mathbf{S}$ is a vector of detrimental stocks, and $d(.)$ is a quadratic damage function[1].

Groundwater stock dynamics. The evolution of groundwater over time is modelled as a stock accumulation. This contrasts with standard analyses of groundwater as an input to production (Gisser and Sanchez 1980) (Provencher and Burt 1993) (Rubio and Casino 2003) (Roseta-Palma 2003). Indeed in these studies of the optimal management of groundwater as a renewable resource, agents' impact on the evolution of the resource over time is modelled as follows :

$$\dot{H} = \frac{1}{AS}\left[R + (a-1)\sum_i w_i\right],$$

where H is the watertable elevation above some arbitrary reference level, w_i is the agent i's extraction, AS the area under irrigated agriculture, a the return flow rate and R a natural recharge rate. In such models, focus is given to the impact of irrigators' pumping rates on the decreasing level of the watertable head and the resource being exploited is groundwater. The assumption of a constant recharge R is assumed to be acceptable 'within the economically relevant range of groundwater reserves' (Rubio and Casino 2001, p.1122), when no leakage from a confined aquifer to the surface system occurs because irrigators' mining activities keep the watertable low. The justification for such a description of groundwater's evolution over time was introduced by Gisser and Sanchez (1980) and extensively used in subsequent studies.

When irrigation-induced salinity is an issue, groundwater becomes a stock pollution (Greiner and Cacho 2001)(Wichelns 1999) and remains an input to production under conditions only[2]. The resource under exploitation is then the adsorptive capacity of the aquifer (Whitten et al. 2004) : irrigators compete for the possibility to produce percolation water. Consequently, focus is shifted toward the impacts that agents have by producing percolation water, and groundwater's evolution over time is described as follows :

$$\dot{X} = \sum_i u_i - \delta X, \tag{6.1}$$

where u_i is agent i's percolation and δ is a discharge rate ($0 \leq \delta \leq 1$).

The stock of groundwater is bounded, between the initial stock $X_0 \geq 0$ and a limit value $\overline{X}$ that corresponds to the situation where the aquifer's discharge capacities are exceeded. However this study focuses on the design of policy instruments to induce individual agents

[1]The formulation of the damage function will depend on the type of stocks under study, see Chapters 7 and 8.

[2]Hence in salinity affected areas, groundwater becomes too salty to be used for irrigation, unless it is mixed with fresh water or goes through desalinization processes, which prove expensive.

to follow the optimal path, within the economically relevant range of the groundwater storage capacity. Hence it is assumed that the limit value $\overline{X}$ isn't reached under the socially optimal management, so that this study abstracts from the issues relating to groundwater management in the neighborhood of the limit value. Also, a more simplifying assumption is made that, even under non-cooperative management, this limit value is not reached. For alternative analyses of how a limit value on the stock accumulation may alter resource extraction or pollution emissions, refer to Farzin and Tahvonen (1996), Amigues et al. (1997) or Chakravorty, Magné and Moreaux (2006)[3].

Absent from this formulation is the consideration of stochastic events, such as rainfall's contribution to groundwater accumulation. Indeed, the objective of this analysis is to design instruments that affect individual agents' decision making ; in this respect the purpose of the modeling is to put in perspective agents' impact on groundwater accumulation. However, rainfall contribution could be added in the model through the definition of a random precipitation variable, in order to obtain an expected groundwater accumulation function. More importantly, unexpected episodic climatic events, which have the potential to affect durably the environment under study (Whitten et al. 2003), could be approached as random events which density functions are unknown to the regulator. Roseta-Palma and Xepapadeas (2004) provide a robust control analysis of water management, in a context where uncertainty is assumed regarding the probability distribution for the stochastic variable, namely precipitations. They show that this framework is conducing to the emergence of a precautionary principle. The MDB is currently subject to a particularly dry period (Australian Government 2007), which means that watertables are not put under as much pressure regarding irrigation-induced salinity as in wet years. Consequently, the potential contribution of rainfall to groundwater accumulation is not as critical as it might be in events of strong precipitations.

Note also that the agents are considered homogeneous with respect to their production function and irrigation efficiency, implicit in equation (6.1), that links irrigation to percolation generation. This builds on the assumption that irrigators are more homogeneous within than between irrigation districts : consequently, within the recharge management area of a watertable, homogeneity is assumed. The impact of this assumption on tax design will be addressed in the subsequent chapters.

[3]Farzin and Tahvonen (1996) address the optimal timing of an emission tax when a stock pollution causes irreversible damage above a certain threshold. To do so, they consider a state-dependent damage function. Amigues et al. (1997) extend the standard analysis of groundwater mining when the population may evolve over time. The assumption of increasing population assigns a reservoir value to the aquifer, besides the values usually attached to the water it stores. Consequently, the aquifer has to be addressed as a reservoir with a limited capacity ; its dynamics is state-dependent, as when the reservoir is full, recharge can't exceed aggregate extractions. Finally, Chakravorty et al. (2006) address how a (regulatory) ceiling on the stock of pollution affects the Hotelling rule. The ceiling is treated as a constraint on the stock of pollution, which can be binding or not. As Chakravorty et al. (2006) argue, such a formulation may be considered as a special case of a threshold-based damage function.

The main difference between the models lies in the definition of individual root zone salt stocks interacting with the collective groundwater stock in the model developed in Chapter 8.

Root zone salts dynamics. In the process of designing a more detailed model of irrigation-induced salinity, a reservoir approach is adopted, alike the one governing the development of SALTMOD, a software used for the prediction of soil, groundwater and discharge water salinities (Saltmod 2007). In SALTMOD, four reservoirs are considered, each characterized by two equations, salt and water balance equations. In order to ensure the analytical tractability of the results in this study, two economically relevant reservoirs are considered, the groundwater stock and the root zone salt stocks. Such a simplification is rendered possible by the particular nature of the pollutant 'salt'; indeed its mobilization, transport and accumulation are directly linked to the flows and stocks of water. This is a way of highlighting the importance of the evolution of the watertable head as the main mechanism to account for in contexts where waterlogging and salinity are an issue. Root zone salt stocks appear as intermediary stocks impacted on by both individual and collective decisions.

The first source of salinity in the context of irrigation is poor quality irrigation water. Denoting S^W the salt concentration of irrigation water, the first inflow of salt is $S^W u_i$. The second source of salts to the root zone is the capillary rise from the watertable. Groundwater is an important tank of salts, bringing salts to the root zone by capillary rise which constitutes the second source of salts for the root zone. Denoting S^G the salt concentration in groundwater, γ_i an individually-specific capillary rise factor $(0 \leq \gamma_i \leq 1)$, this inflow is : $\gamma_i S^G X$.

Percolation represents an outflow of salts from the root zone, to the watertable. Denoting S^R the salt concentration in the root zone, then the amount of percolated salts is $S^R P_i$, where P_i is the quantity of water percolating. Water remaining the the root zone, available for uptake by plants, and water percolating are function of the quantity of applied water and the efficiency of the technology in use, β[4] : $R_i = \beta u_i$ and $P_i = (1-\beta)u_i$.

Taking these mechanisms into account, an equation describing the dynamics of salt accumulation in the root zone at location i can be formulated as follows, with Q_i, the quantity of salts for farm i, and $S^R = Q_i/R_i$:

$$\dot{Q}_i = S^W u_i + \gamma_i S^G X - \frac{1-\beta}{\beta} Q_i. \tag{6.2}$$

[4]With the assumption that $0 \leq \beta \leq 1$.

The quantity of groundwater X, is a variable common to all the farmers, so is the concentration of salt S^G. Furthermore, S^W and S^G are treated as exogenous variables. However, water remaining in the root zone and percolating water are individual variables, as they depend on individual choices, regarding the quantity of water applied and the irrigation technology. The introduction of these individual stocks modifies the groundwater stock dynamics as follows, with $\gamma = \sum_i \gamma_i$:

$$\dot{X} = \sum_{i=1}^{n} P_i - (\delta + \gamma)X. \tag{6.3}$$

These are the main features of the games studied in Chapters 7 and 8. Irrigators make their decisions by implementing their control (irrigation water use) along a time horizon extended to infinity. Their payoffs are based on the production function $F(u_i, \mathbf{S})$. The state variables (groundwater stock and root zone salt stocks) are payoff relevant as they impact negatively on the production function. Next, a feature central to the definition, and resolution, of differential games is addressed : the agents' strategy space.

6.2 Differential games and strategy space description

In differential games, one needs to specify the information players condition their strategies on. The various types of strategies that have been analysed can be classified according to their informational requirements. At one extreme, agents only know the initial state of the state variable and current time t, at the other extreme agents know the whole history of the game (Dockner et al. 2000). The choice of strategy space is a matter of the 'institutional environment in which the game is played' (Dockner et al. 2000, p.29). Indeed, it is the type of information available, as well as the nature of the commitment the agents can engage in, that condition the agents' strategy space.

Two main informational structures are usually used in dynamic games : open-loop and feedback strategies[5]. In the former case, each agent takes the rival strategies as simple functions of time when he determines the optimal trade-off between the current and future effects of his actions(Dockner et al. 2000). In the latter case, the choice of action is conditionned upon time and the state of the environment at the current time [6].

Informational requirements. A major difference between open-loop and feedback strategies is the informational burden they suppose. Open-loop strategies don't require any other information than the current time and the initial state of the state variable; all other information about the state of the environment is left aside, either because the

[5] Feedback strategies are also referred to by some authors as 'closed-loop' strategies or Markovian strategies (Dockner et al. 2000, p.30).

[6] An implicit assumption is that the game history doesn't matter, as the current state of the environment summarises adequately past actions (Dockner et al. 2000).

players chose to, or because they don't have access to that information. Indeed, in the open-loop case, agents commit to a predefined set of decisions for the whole time-path, and don't reassess it trough time as the stock evolves. Agents making use of feedback strategies are in need of information about the state of the environment at each point of time in order to formulate their decisions.

The question arises of whether such an informational burden is not too high in some contexts. Indeed, the state variable may refer to entities that are not necessarily observable by individual agents ; consequently they need the regulator, or some agency, to provide them with this information. In the case of irrigation-induced salinity, the state variable 'groundwater' has to be treated differently according to the catchment under study. Indeed, a lot of modeling and monitoring effort has been done in some catchments, identified as the most at risk, so that information is available with sufficient certainty, and is transmitted to individual irrigators (Whitten et al. 2004). Hence the institutional background in these 'pilot' catchment is conducive to the recourse to feedback strategies. In contrast, some catchments still lack this type of information, so that agents are *de facto* reduced to open-loop strategies[7].

Commitment. Another difference between open-loop and feedback strategies is the assumption that in the open-loop case agents are able to commit to a set of decisions at the beginning of the game, and stick to this set of decisions along the whole time horizon. Contrary to this, in the feedback case, agents can reevaluate their decisions with regards to the state of the environment, which reflects the rivals' actions.

The necessity of commitment is often highlighted as the main drawback of open-loop strategies, as it may appear unreasonable to assume that agents will not pay attention to the evolution of the environment. Amir and Nannerup (2006) stress this contradiction between a static-like behavior and an explicitly dynamic framework. Thinking of agents as making their decisions at each point of time, as described by feedback strategies, may appear more realistic. However, Dockner et al. (2000) point out that in the environmental setting, the level of commitment implied by open-loop strategies may be approached as a far sighted environmental awareness.

Besides these considerations of information and commitment, the most critiziced aspect of open-loop strategies is there inability to ensure sub-game perfectness. Indeed, open-loop strategies ensure that agents' decisions are optimal along the equilibrium state trajectory, but not off this trajectory, while feedback strategies have this property of subgame perfectness. Consequently, open-loop strategies comply with the minimal requirements of credibility only.

[7]Dockner et al. (2000) stress that instances of information-constrained settings are numerous, so that open-loop strategies should not be considered irrelevant.

Hence feedback and open-loop strategies differ with respect to various aspects, such as their informational burden and the level of commitment they require, their potential to support weak or strong time consistency[8]. However, as Dockner et al. (2000) point out, the choice of strategy is of the same nature as the definition of the model's assumptions, it is driven by the context under study and its institutional background.

6.3 Informational requirements

The informational requirements associated with the models studied in this Part are of two types : access to the model's parameters, necessary to derive the optimal tax parameters, and observability of the policy bases, in order to implement the taxes. Both types of information are conducive to the generation of transaction costs that may alter the practical implementation of the policies.

Has irrigation-induced salinity been turned into a point source pollution ? Some authors state that in reason of the extensive modeling effort that has occurred, and of the resulting confidence in the hydrological models available, irrigation-induced salinity has been turned from a nonpoint source pollution into a point source pollution (Duke and Gangadharan 2005)(Pakula 2004). The characteristic feature of nonpoint source pollution is that individual emissions are not observable at a reasonable cost, in reason of a high number of polluters, the lack of knowledge of hydrological or pedological features or the stochasticity of mechanisms that render the measurement of individual emissions prohibitively costly (Segerson 1988). As Duke and Gangadharan (2005, p.2) put it :

> '...recent advances in hydrology modelling have however improved knowledge about the cause and effects between production and water quality, making it possible to estimate the relationship between production practices and external environmental impacts.'

Indeed, as addressed in Chapter 2, during the last ten years, Federal funds have been targeted at research projects aimed at tacking salinity, essentially through a better understanding and monitoring of its mechanisms. Numerous models have been developped, at various levels, to answer various types of questions. The Practical Index of Salinity Models (PRISM 2007) lists more than 90 models that are useful for Australian environmental policymakers in the context of irrigation-induced salinity. Table 6.1 presents a selective illustration of the most influential. It appears that important recent policy initiatives, including the pilot Interstate Trade, or the salinity zoning policy in SA, were informed by hydrological models. Consequently, the notion that salinity can be treated as a point source pollution in Australia is emerging.

[8] Subgame perfectness is sometimes referred to as 'strong time consistency'.

Model	Use in policymaking	References
SWAGMAN	community education	(Whitten et al. 2003)
SALSA	MDB salinity strategy, NAPSWQ	(Beare, Heaney and Mues 2001)
SIMPACT	salinity zoning SA	(Miles, Kirk and Meldrum 2001)
SIMPACT	pilot Interstate Trade exchange rates	(Miles et al. 2001)

Tab. 6.1 – Models used for policymaking in relation with irrigation salinity in Australia.

This poses the question of the precise definition of a nonpoint source pollution. In one of the first depiction of the range of informational issues associated with nonpoint source pollution, Braden and Segerson (1991) argue that what constitutes the characteristics of a nonpoint source pollution, as opposed to a point source pollution, is a combination of three imperfect monitoring and measurement : impossibility to measure individual emissions, to infer emissions from observable inputs and to infer individual emissions from environmental quality. In this regards, the second and third items have been largely improved in some identified catchments in the MDB. The cost of getting this information, borne by all levels of Goverments, has reduced the nonpoint source nature of irrigation-induced salinity. However, it is still practicaly impossible to have access to individual emissions with certainty.

Hence in this thesis, while irrigation salinity is considered as a nonpoint source pollution, the informational setting is such that recourse to input-based instruments is supposedly capable of efficiently managing the issues at stake. Consequently, the policy basis illustrative of the individual incentive part of the taxation schemes under study is an input - more precisely, irrigation water. Furthermore, it can be assumed that in some pilot catchments, the transaction costs of providing the type of models needed to design the appropriate tax instruments are low, as they have already been borne by various levels of government in the past two decades. However, in the remaining catchments that have not been coined pilot, where degradation were less severe, the transaction costs of generating this information is supposedly high.

The next point discusses the monitoring requirements to have access to both components of the taxation schemes under study - the individual and the collective parts.

Monitoring capacities. Improvement of the availability of data about water in general, and groundwater in particular, is part of all the latest water legislations in Australia. Signatories to the NWI have already committed to improving the information collected on groundwater and its connectivity with surface water (Council of Australian Governments 2004). A country-wide initiative, aiming at reconciliating existing data and improving the transfers of future data accross States, is being developped within the Australian Government Groundwater Infrastructure (AGDI). One of the main task is to agree on Standards ; for instance groundwater depth is defined as the depth in meters from ground surface to watertable, measure at monitoring bores or piezometers, at specified points in time. Also, in order to assess the performance of initiatives such as the NAP or the NHT, the Federal NRM provides guidelines to develop an efficient monitoring bore coverage to provide consistent data. Furthermore, individual States have their own groundwater data, most of which is available freely on the internet, like the NSW water information website (NSW Groundwater Works 2007) or the South Australian Obswell (Government of South Australia 2007). While the current coverage of groundwater monitoring is still not satisfactory, the States have committed to improve the level of monitoring. However, there exist salinity 'hotspots' that have been extensively studied and are very well documented[9] and where policy instruments such as groundwater-contingent taxation schemes could be implemented as pilot schemes.

Remains the question of the observability of water inputs. In the Australian irrigation context, data availability will depend on the type of irrigator at stake. Indeed, there are mainly three irrigation contexts : regulated systems, unregulated surface systems and groundwater extractions. For example, in Victoria, regulated systems account for 80 per cent of consumptive use of water, while unregulated rivers and groundwater extraction account for around 10 per cent each. Unregulated and groundwater systems are supposedly the most difficult to monitor. Metering is explicitly put forward in the NCC as one of the objectives to be achieved by the States (National Competition Council 2004*a*). Metering is also an essential part of the new tariff policies to be implemented ; in Victoria, the tariff scheme should include several fixed parts (service, storage and infrastructure access fees) and variable parts (infrastructure use and additional service point fees). The Victorian Government is due to subsidy meters installation (up to A$400 a meter) for existing unmetered users. Hence, the transaction costs of accessing this type of information have been accepted by most levels of government.

6.4 Concluding remarks

This Chapter laid the framework within which the models used to test Hypothesis 1 are developed. The main methodological choice is to consider irrigation-induced salinity as a stock pollution problem, highlighting the watertable rising as the key mechanism. Hence the analysis departs from standard analyses of aquifers as an input to production.

Absent from the formulation of the models are the issues of stochasticity and heterogeneity of decision-makers. The former refers mainly to rainfall contributions, while the modeling process is voluntarily focused on agents' contribution to groundwater accumulation. The latter refers to agents' production and percolation functions. Indeed, in this

[9] Refer to the example of the Colleambally Irrigation Area (Whitten et al. 2003)(Whitten et al. 2004).

thesis is considered that agents are homogeneous with respect to these features. This bases on the assumption that in a same irrigation district, farms are usually of a similar size, and network effects tend to homogenize production and techniques. This assumption is of course simplifying, but doesn't prevent the analysis of agents' strategic interactions.

Reference to the Australian context led to formulate some assumptions. First, the availability of aquifer-related information means that the nonpoint source nature of irrigation-induced salinity is weakened ; hence the reliance on input-based instruments. Second, the contrasted situations existing in the MDB regarding aquifer-related information means that both open-loop and feedback strategies prove relevant in the analysis of the agents' decision-making process.

Chapitre 7

Managing irrigation-induced salinity as a stock pollution : dynamic state dependent water taxes

This Chapter[1] presents an analysis of dynamic taxation schemes to manage irrigation-induced salinity from a qualitative viewpoint, in the sense that the movements of the watertable head are considered as the main mechanism underlying the rest of the manifestations of irrigation-induced salinity.

Various taxation schemes have been proposed in the dynamic taxation literature. Ko et al. (1992) derive a dynamic Pigovian tax, in the context of a stock pollution emitted by a single activity. Their first-best solution, a time-dependent emission tax, is rather intuitively set as the opposite of the co-state variable. Arguing for the practical difficulties of implementing such a tax, in particular due to political or legislative adjustment costs, they analyse a second-best solution, a time-independent tax.

In the context of the monopolistic exploitation of a non-renewable resource, Bergstrom et al. (1981) derive a time-dependent tax. Karp and Livernois (1992) show that this taxation scheme is prone to strategic manipulation by the monopolist, if the authority cannot commit to the entire sequence of the tax. That is why they develop a linear Markov tax rate, that induces the monopolist to recognise that by choosing its current rate of extraction, it is able to influence the future values of the tax. Benchekroun and van Long (1998) apply this concept to the setting of a polluting oligopoly, setting which exhibits strategic interaction between the polluters. They develop an emission based tax, in which the rate

[1]Parts of these results were presented in : Legras, S. and Lifran, R. (2006) Dynamic taxation schemes to manage irrigation-induced salinity, *Environmental Modeling and Assessment*, 11 :157-67.

is conditioned upon the level of the stock.

Horan, Shortle and Abler (1998) analyze the Segerson (1988) ambient tax in a dynamic framework. They show that its efficiency depends on the dimensionality of agents' choice sets, and propose a state-dependant ambient tax, linear in ambient pollution.

The analysis of optimal taxes to manage irrigation-induced salinity will be based on adaptations of the above-cited taxation schemes. Hence the following general tax formulation is considered :

$$T(u_i, X) = (\tau_1 + \tau_2 X)u_i + \tau_3 X. \tag{7.1}$$

Such a formulation accounts for : a time-independent standard input tax ($\tau_2 = \tau_3 = 0$), a state-dependent ambient tax ($\tau_1 = \tau_2 = 0$) and a stock-dependent input tax ($\tau_3 = 0$). These taxes have been analyzed in different settings (stock pollution, nonpoint source pollution) for different reasons, as addressed in Chapter 3. The aim of this Chapter is then to analyse, and compare, the efficiency of such schemes[2] in the setting of irrigation-induced salinity. Hence is assessed the potential of instruments based on a measure of individual performance only (τ_1), a measure of collective performance only (τ_3) or a mix of both (τ_2) to induce individual agents to behave optimally. It will appear that according to the definition of optimality (along the whole path or at the steady state only) the efficient taxation scheme varies.

This Chapter is organised as follows. The main features of the model are recalled in Section 7.1. Section 7.2 presents a comparison of the optimal solution and the non-cooperative equilibria, and illustrates the need for a policy instrument. Section 7.3 highlights the differentiated consequences that various tax parameters, associated with a measure of individual performance (input use), a measure of collective performance (groundwater stock) or a mix of both, have on the level of groundwater at the steady state, and on its accumulation rate. The optimal tax parameters are derived in Section 7.4. Section 7.5 provides somes illustrations, and Section 7.6 concludes.

7.1 The model

Consider n agents, indexed by i, that produce a homogeneous good from a unique input u_i, irrigation water. The time horizon of the problem considered is extended to infinity, $t \in [0, \infty[$. We assume that the agents are identical and all their discount rates equal.

Plants do not use the totality of the water that is applied for irrigation purposes. Some water percolates to reach the watertable where it accumulates. This percolation water e_i

[2] These taxation schemes are analysed in the framework of correlated stocks in Chapter 8.

that each agent emits is a flow variable at each t. The relation between u_i and e_i is supposed to be known and described by $e_i = \theta_i u_i$, with $0 \leq \theta_i \leq 1$. By an appropriate choice of units, the percolation rate is set at $\theta_i = 1$.

The stock variable is the amount of groundwater. Its accumulation dynamics is described by the following expression :

$$\dot{X} = \sum_i u_i - \delta X, \tag{6.1}$$
$$X(0) = X_0 \geq 0,$$

where δ, the groundwater discharge fraction, is a positive parameter. For expositional clarity, the time indicator is omitted in the following wherever possible.

The benefit of any agent at each instant of time can be written as a function of its input use and of the stock of groundwater : $F(u_i, X)$, with $\frac{\partial F}{\partial u_i} > 0$, $\frac{\partial^2 F}{\partial u_i^2} < 0$, $\frac{\partial F}{\partial X} < 0$ and $\frac{\partial^2 F}{\partial X^2} < 0$, with :

$$F(u_i, X) = a + bu_i - \frac{c}{2}{u_i}^2 - \frac{d}{2}X^2,$$

where d is a marginal individual damage term due to the rising of the aquifer.

7.2 Individually vs socially optimal decisions

This Section is devoted to the derivation of the optimal outcome from a cooperative solution concept. Indeed, considering that the damage affects the irrigators located above the aquifer only, leaving the issue of stream salinity aside, then the optimal irrigation and stock paths are the outcome of a cooperative game involving the n agents. The optimal solution is then compared to the non-cooperative outcomes, that can be characterised by different informational structures. It is expected that non-cooperative optimisation problems lead to over-pollution as agents do not fully account for their involvement in the accumulation of groundwater. However, open-loop and feedback strategies are also expected to lead to different levels of over-pollution.

7.2.1 Optimal solution

The optimal outcome is obtained for agents who jointly maximise their benefits :

$$\max_{u_1,\ldots,u_n \geq 0} V = \sum_i \int_0^\infty F(u_i, X)e^{-rt}dt, \text{ subject to equation (6.1),}$$

where r denotes the discount rate.

Proposition 1. *Cooperative agents internalise the stock externality, by considering the impact of their irrigation decisions on the group. Assuming interior solutions, the resulting optimal control, state and costate variables are :*

$$
\begin{aligned}
&X^{SO}(t) = X^{SO}_\infty(1 - e^{\rho^{SO}t}) + X_0 e^{\rho^{SO}t},\\
&u^{SO}(t) = X^{SO}(t)\frac{\delta + \rho^{SO}}{n} - X^{SO}_\infty \frac{\rho^{SO}}{n},\\
&\lambda^{SO}(t) = X^{SO}(t)\frac{c(\delta + \rho^{SO})}{n} - X^{SO}_\infty \frac{c\rho^{SO}}{n} - b,\\
&\quad \textit{where } X^{SO}_\infty = \frac{nb(r+\delta)}{c\delta(r+\delta) + n^2 d},
\end{aligned}
$$

and ρ^{SO} is the negative root of the equation : $\rho^2 - r\rho - [\frac{n^2 d}{c} + \delta(r+\delta)] = 0$.

Démonstration. The current value Hamiltonian is :

$$H^C(u_i, X, \lambda^{SO}) = \sum_i (a + bu_i - \frac{c}{2}u_i^2 - \frac{d}{2}X^2) + \lambda^{SO}(\sum_i u_i - \delta X),$$

where λ^{SO} is interpreted as the dynamic shadow cost of groundwater accumulation. The necessary conditions for optimality are :

$$\lambda^{SO} = cu_i - b, \tag{7.2}$$

$$\dot{\lambda}^{SO} = (r+\delta)\lambda^{SO} + ndX, \tag{7.3}$$

along with (6.1) and the transversality condition, $\lim_{t\to\infty} e^{-rt}\lambda^{SO}(t)X(t) = 0$.

As $H^C_{u_i u_i} = -c < 0$, these conditions are also sufficient. The optimal steady state values are obtained by setting $\dot{\lambda}^{SO} = \dot{X} = 0$:

$$u^{SO}_\infty = \frac{\delta}{n}X^{SO}_\infty, \tag{7.4}$$

$$b - cu^{SO}_\infty = -\lambda^{SO}_\infty = \frac{nd}{r+\delta}X^{SO}_\infty. \tag{7.5}$$

The optimal stock path $X^{SO}(t)$ is obtained from the Modified Hamiltonian Dynamic System (MHDS) :

$$
\begin{aligned}
&\dot{\lambda_C} = (r+\delta)\lambda_C + ndX,\\
&\dot{X} = \frac{n}{c}\lambda_C - \delta X + \frac{nb}{c}.
\end{aligned}
$$

Its characteristic polynomial is : $\rho^2 - r\rho - [\frac{n^2 d}{c} + \delta(r+\delta)] = 0$. Its negative root is chosen to ensure stability. Resolution of the MHDS, keeping in mind the initial and terminal conditions, leads to the proposed expression of $X^{SO}(t)$. $\square$

At the steady state, the valuation of the individual marginal benefit equals the present value of the stream of marginal adverse effect on group production caused by the additional pollution generated by it, as shown in equation (7.5). This means that cooperative agents internalise the stock externality, by considering the impact of their irrigation decisions on the group.

From equation (7.2), the optimal irrigation decision at each point of time should be such that the marginal cost of groundwater accumulation equals the net marginal benefit of input use. Also, equation (7.3) describes how the shadow value of pollution accumulation should evolve over time : at the rate $r + \delta$, and increased by a term of collective damage. Consequently, the stock of groundwater rises monotically from the initial level to the steady state level, if $X_0 < X_\infty$[3]. At the same time, it appears that input use follows the opposite direction. Starting from $u_0 > 0$, the optimal level of inputs then falls to reach u_∞^{SO} at the steady state. In other words, the cooperative aggregate level of input use falls over time.

7.2.2 Open-loop equilibrium

The individual optimisation program is :

$$\max_{u_i \geq 0} V_i = \int_0^\infty F(u_i, X).e^{-rt}.dt, \text{ subject to (6.1).}$$

Proposition 2. *Agents following open-loop strategies do not account for their contribution to the damage experienced by the other members of the group. Consequently, they over-emit compared to the social optimum case ; assuming interior solutions, the open-loop stock and irrigation paths are given by :*

$$X^{OL}(t) = (X_0 - X_\infty^{OL})e^{\rho^{OL}t} + X_\infty^{OL},$$
$$u^{OL}(t) = X^{OL}(t)\frac{\delta + \rho^{OL}}{n} - X_\infty^{OL}\frac{\rho^{OL}}{n},$$
$$\textit{where } X_\infty^{OL} = \frac{nb(r + \delta)}{c\delta(r + \delta) + nd},$$

and ρ^{OL} is the negative root of the following equation : $\rho^2 - r\rho - [\delta(r + \delta) + \frac{dn}{c}]$.

Démonstration. The resolution method is similar to the cooperative case, as the setting is symmetric. Noting λ^{OL} the shadow cost of groundwater accumulation, the current value

[3] As a detrimental stock is considered, the analysis is restricted to this case.

Hamiltonian is :

$$H^{OL}(u_i, X, \lambda^{OL}) = a + bu_i - \frac{c}{2}u_i^2 - \frac{d}{2}X^2 + \lambda^{OL}(u_i + \sum_{j\neq i} \hat{u_j} - \delta X)$$

The first order conditions are then :

$$\lambda^{OL} = cu_i - b,$$
$$\dot{\lambda}^{OL} = (r+\delta)\lambda^{OL} + dX.$$

The steady state values of the variables are given by the following expression :

$$\lambda_\infty^{OL} = cu_\infty - b = -\frac{d}{r+\delta}X_\infty. \tag{7.6}$$

□

The open-loop steady state stock exceeds the optimal one. At the denominator of X_∞, the term of damage is multiplied by n in the open-loop case, and n^2 in the optimal case. Equation (7.6) states that, at the steady state, the valuation of the individual marginal benefit by an open-loop agent equals the present value of the stream of marginal adverse effect on his own production, caused by the additional stock generated by the group. This shows that in the open-loop case, agents do not account for their contribution to the damage experienced by the other members of the group. However, one should note that open-loop agents operate a partial internalization of the externality, in reason of its endogenous nature. This expresses directly into the utility function through the term in d. By setting $d = 0$, the obtained steady state stock gets even higher. The higher the marginal damage, the lower the steady state stock.

7.2.3 Feedback equilibrium

Proposition 3. *There exists a unique symmetric MPNE in linear strategies, $f(X) = w - zX$, with $w > 0$, $z > 0$, that ensures the convergence of X(t) to a steady state.*

Démonstration. Deriving the Markov perfect equilibrium requires solving n simultaneous differential equations. Let $V_i(X)$ be the optimal value function for agent i. The Hamilton-Jacobi-Bellman equation for i's problem is :

$$rV_i(X) = \max_{u_i\geq 0}\{F(u_i, X) + V_i'(X)(u_i + \sum_{j\neq i} u_j(X) - \delta X)\}, i, j \in \{1, .., n\}. \tag{7.7}$$

The formulation of the problem is the same for the $(n-1)$ other players. The right hand side maximisation gives the formulation of the instantaneous reaction function of player i,

$\hat{u}_i(X)$:

$$\hat{u}_i(X) = \frac{b + V_i'(X)}{c}, i \in \{1, .., n\}.$$

The coefficients of $\hat{u}_i(X)$ are derived by the undetermined coefficients method. Suppose that the value function is a second order function of X :

$$V_i(X) = \frac{A}{2}X^2 + BX + C \Rightarrow V_i'(X) = AX + B \Rightarrow \hat{u}_i(X) = \frac{A}{c}X + \frac{b+B}{c}.$$

Replace u_i, $V_i(X)$ and $V_i'(X)$ by their quadratic values in equation (7.7) to obtain a second-order equation in X, which has to equal zero for every X. Then terms of equal degree in X are equated to zero :

$$X^2 : \frac{A^2(2n-1)}{2c} + A(-\frac{r}{2} - \delta) - \frac{d}{2} = 0, \tag{7.8}$$

$$X : \frac{bA}{c} - \frac{A(b+B)}{c} + B(-r - \delta + \frac{nA}{c}) + \frac{An(b+B)}{c} = 0, \tag{7.9}$$

$$Cte : -rc + a + \frac{b(b+B)}{c} - \frac{(b+B)^2}{2c} + \frac{Bn(b+B)}{c} = 0. \tag{7.10}$$

The negative root A^{FB} is selected from equation (7.8) to ensure convergence to a steady state. Then B^{FB} follows from equation (7.9) :

$$A^{FB} = \frac{c(r+2\delta) - \sqrt{\triangle^{FB}}}{2(2n-1)} < 0 \quad \text{and} \quad B^{FB} = \frac{nbA^{FB}}{c(r+\delta) - (2n-1)A^{FB}} < 0,$$

where $\triangle^{FB} = c^2(r+2\delta)^2 + 4cd(2n-1)$.

The groundwater dynamics is :

$$\dot{X} = [\frac{nA^{FB}}{c} - \delta]X + n\frac{B^{FB}+b}{c} \Rightarrow X_\infty^F = \frac{n(B^{FB}+b)}{c\delta - nA^{FB}},$$

and the equilibrium state trajectory is obtained by integration :

$$X^{FB}(t) = (X_0 - X_\infty^{FB})e^{(\frac{nA^{FB}}{c} - \delta)t} + X_\infty^{FB}.$$

□

Hence agents following feedback strategies in this context tend to react to the rising of the watertable by decreasing their input use, $f(X) = w - zX$, with $w > 0$, $z > 0$. There is strategic substitutability between the agents' decisions.

Proposition 4. *Agents accumulate faster in the feedback case than in the open-loop case.*

Démonstration. The accumulation rate in the feedback case is :

$$\rho^{FB} = \frac{nA^{FB}}{c} - \delta = \frac{rc - \sqrt{c^2(r+2\delta)^2 + 4(2n-1)dc} - 2\delta c(1-1/n)}{c(2-1/n)} < 0,$$

to which the accumulation rate obtained in the previous case is obtained :

$$\rho^{OL} = \frac{rc - \sqrt{c^2(r+2\delta)^2 + 4cdn}}{2c} < 0.$$

Note that $2c > c(2-1/n) > 0$; also, $n > 1 \Rightarrow 2n-1 > n$, then :

$$\left| rc - \sqrt{c^2(r+\delta)^2 + 4(2n-1)dc} - 2\delta c(1-1/n) \right| > \left| rc - \sqrt{c^2(r+2\delta)^2 + 4cdn} \right| > 0.$$

Then, agents accumulate faster in the feedback case than in the open-loop and the social optimum cases[4]. □

An agent is induced to over-emit expecting a lowering of the others' emissions when the watertable rises. Each agent reasonning this way, the aggregate result is an over-accumulation of groundwater.

Non-cooperative strategies, either open-loop or feedback, don't lead to the optimal solution and cause over-accumulation of groundwater. In the open-loop case, agents don't internalise the total impact of their irrigation decisions ; in the feedback case they also act strategically with respect to their opponents. As a consequence, the regulator has a rationale for intervention in order to restore an optimal management of the groundwater stock. In the next Sections, the use of taxation schemes to achieve this outcome is investigated.

7.3 Impact of a general taxation scheme on the accumulation rate and the steady state values

This Section provides some first insights into the impact of various combinations of tax parameters as described in equation (7.1) on the resulting accumulation of groundwater, at the steady state and along the path, through the accumulation rate.

Proposition 5. *The three types of tax parameters (a measure of individual performance τ_1, a measure of collective performance τ_3 or a mix of both τ_2) impact on the level of the stock at the steady state. However, only the mixed term τ_2 has an impact on the accumulation rate and thus has the potential to induce the agents to produce the optimal stock path along the whole time horizon. This is true when agents make use of open-loop and feedback strategies.*

[4]The second part of this result was established in the previous Section.

Démonstration. The open-loop case. The current value Hamiltonian of an agent i subject to $T(u_i, X)$ and the associated first-order equations are :

$$H^{\tau}(u_i, X, \lambda^{\tau}) = F(u_i, X) - T(u_i, X) + \lambda^{\tau}(u_i + \sum_{j \neq i} u_j - \delta X),$$

$$b - cu_i + \lambda^{\tau} - (\tau_2 X + \tau_1) = 0, \tag{7.11}$$

$$\dot{\lambda^{\tau}} = (r + \delta)\lambda^{\tau} + dX + \tau_2 u_i + \tau_3. \tag{7.12}$$

The MHDS is obtained by replacing the value of u_i obtained from equation (7.11) in equations (6.1)-(7.12). Solving it leads to the following expression of the groundwater stock :

$$X^{\tau}(t) = X^{\tau}_{\infty}(1 - e^{\rho^{\tau} t}) + X_0 e^{\rho^{\tau} t}, \tag{7.13}$$

where ρ^T is the negative root of the following equation :

$$(\rho^{\tau})^2 + \rho^{\tau}\left[\frac{(n-1)\tau_2}{c} - r\right] - \frac{1}{c}[(r + \delta)(c\delta + n\tau_2) + \delta\tau_2 + nd] = 0, \tag{7.14}$$

$$\text{and } X^{\tau}_{\infty} = \frac{n(r + \delta)(b - \tau_1) - n\tau_3}{c\delta(r + \delta) + nd + \delta\tau_2 + n\tau_2(r + \delta)}. \tag{7.15}$$

Note that τ_1 and τ_3 are absent from the expression of the accumulation rate obtained from equation (7.14). □

Démonstration. The feedback case. First the existence and unicity of the Markov perfect Nash Equilibrium, when the tax rule is of the form $T(u_i, X) = (\tau_1 + \tau_2 X)u_i + \tau_3 X$, has to be checked. It is shown that there exists a unique Markov perfect Nash equilibrium in linear strategies that ensures the convergence of the stock to a steady state. This strategy is of the form $K(X) = w - zX$ with $w \geq 0$ and $z \geq 0$. The same methodology as in the feedback case in the absence of tax is used. Assuming a value function of the form $A^{\tau}X^2/2 + B^{\tau}X + C^{\tau}$, the resulting players' strategy is :

$$u(X) = \frac{A^{\tau} - \tau_2}{c}X + \frac{B^{\tau} + b - \tau_1}{c}, \text{ with :}$$

$$A^{\tau} = \frac{rc + 2n\tau_2 + 2\delta c - \sqrt{(rc + 2n\tau_2 + 2\delta c)^2 - 4(\tau_2 + cd)(2n - 1)}}{2(2n - 1)},$$

$$B^{\tau} = \frac{n(A^{\tau} - \tau_2)(b - \tau_1) - \tau_3 c}{(\delta + r)c + A^{\tau}(1 - 2n) + n\tau_2}.$$

Then the groundwater accumulation rate and steady state value are :

$$\rho^{\tau} = n\frac{A^{\tau} - \tau_2}{c} - \delta \text{ , and } X^{\tau}_{\infty} = \frac{n(B^{\tau} - b - \tau_1)}{c\delta - n(A^{\tau} - \tau_2)}.$$

□

The three tax parameters τ_1, τ_2 and τ_3 impact on the steady state stock. Furthermore, $\frac{\partial X_\infty^\tau}{\partial \tau_1} < 0$, $\frac{\partial X_\infty^\tau}{\partial \tau_2} < 0$ and $\frac{\partial X_\infty^\tau}{\partial \tau_3} < 0$. However, only τ_2 impacts on the value of ρ^τ, and $\frac{\partial \rho^\tau}{\partial \tau_2} < 0$. This leads to the rather intuitive result that the higher the parameter of the tax, the lower the steady state value of the stock.

Consequently, a standard input tax, of the form $T(u_i) = \tau_1 u_i$, with $\tau_1 > 0$, impacts on the stock paths only indirectly, through the steady state level. However, the rate of accumulation is unchanged, compared to the open-loop case, as appears from equation (7.14). For instance, by setting $\tau_2 = 0$, $\rho^\tau = \rho^{OL}$ as defined in Section 7.2. If such a tax can be derived in order to induce the agents to attain the optimal stock at the steady state, it doesn't have the potential to induce agents to follow the optimal path along the whole time horizon.

In the same manner, a state-dependant ambient tax, of the form $T(X) = \tau_3 X$, only impacts on the steady state stock. The same conclusions are thus derived.

On the contrary, a stock-dependent input tax, inspired from the emission tax derived by Benchekroun and van Long (1998), has the potential to induce non-cooperative agents to follow the optimal stock and irrigation decisions paths. It impacts both on the steady state level and on the rate of accumulation of groundwater.

The next Section addresses the derivation of the tax parameters able to induce individual agents to behave in an efficient manner.

7.4 Deriving the optimal taxes

This Section is devoted to the derivation of the optimal taxation schemes, with respect to two definitions of efficiency. In a first step, the regulator requires the agents to comply with the optimal level of groundwater at the steady state. In this regard, it constitutes an application of Xepapadeas (1992)'s definition of an efficient scheme[5]. In a second step, the regulator requires a more standard definition of efficiency, which the is requirement that the agents comply with the optimal accumulation path along the whole time horizon.

7.4.1 Open-loop strategies

Proposition 6. *To induce agents following open-loop strategies to reach the optimal level of groundwater at the steady state, the regulator needs to have recourse to either a pure ambiant tax $\hat{\tau}_3$ or a pure input tax $\hat{\tau}_1$:*

$$\textit{with } \hat{\tau}_1 = \frac{ndb(n-1)}{c\delta(r+\delta) + n^2 d} = \frac{\hat{\tau}_3}{(r+\delta)}.$$

[5] A similar approach in the context of public good provision is provided by Fershtman and Nitzan (1991).

Démonstration. The derivation of the the tax parameters that induce the agents to reach the optimal steady state stock of groundwater consists in solving the following equation : $X_\infty^T = X_\infty^{SO}$ with respect to various combinations of the tax parameters, as shown in Table 7.1. □

Solve w.r.t.	Parameters	Solve w.r.t.	Parameters
τ_1, τ_2,τ_3	$\tau_1(\tau_2,\tau_3)$	τ_1 $[\tau_2=\tau_3=0]$	$\hat{\tau_1}$
τ_1, τ_3 $[\tau_2=0]$	$\tau_1(\tau_3)$	τ_2 $[\tau_1=\tau_3=0]$	$\hat{\tau_2}=\frac{nd(n-1)}{nr+\delta(n+1)}$
τ_1, τ_2 $[\tau_3=0]$	$\tau_1(\tau_2)$		
τ_2, τ_3 $[\tau_1=0]$	$\tau_3(\tau_2)$	τ_3 $[\tau_1=\tau_2=0]$	$\hat{\tau_3}$

TAB. 7.1 – Optimal steady state tax rates : open-loop case

Hence when the 'pure' tax parameters are available, they are sufficient to induce the agents to reach the optimal steady state groundwater stock.

Proposition 7. *There exists an incentive scheme that induces non cooperative agents to follow the optimal input use and stock paths :*

$$T(u_i,X)=(\tau_2^{OL}X+\tau_1^{OL})u_i,$$

and the optimal parameters are :

$$\tau_2^{OL}=\frac{n(n-1)d}{nr+\rho(1-n)+\delta(n+1)}\geq 0 \text{ and } \tau_1^{OL}=\tau_2^{OL}\frac{(1-n)X_\infty^{SO}\rho^{SO}}{n(r+\delta)}\geq 0.$$

Démonstration. The resolution process follows Benchekroun and van Long (1998). The objective is to have $\{u^{SO}(X),T(\tau_1,\tau_2,\tau_3)\}$ fulfilling the first order conditions as defined in equations (6.1), (7.11) and (7.12). Keeping in mind that there is an optimal reaction function,

$$u^{SO}(X)=\frac{(\delta+\rho^{SO})X^{SO}-\rho^{SO}X_\infty}{n},$$

derivation of equation (7.11) with respect to time and its equalization to equation (7.12) leads to :

$$cu'(X)\dot{X}+\tau_2\dot{X}=(r+\delta)(-b+cu(X)+\tau_2X+\tau_1)+dX+\tau_2u(X)+\tau_3, \tag{7.16}$$

which simplifies as a first order equation in X. Each sum of terms of equal degree in X is equated to zero, to stay in compliance with the condition that (7.16) is true for every X :

$$X:\tau_2(r+\delta-\rho^{SO}+\frac{\delta+\rho^{SO}}{n})+d-\frac{c}{n}(\delta+\rho^{SO})(\rho^{SO}+r+\delta)=0 \tag{7.17}$$

$$Cte:(r+\delta)\tau_1+\tau_3+\tau_2\rho^{SO}X_\infty^{SO}\frac{n-1}{n}-b(r+\delta)+\frac{c}{n}\rho^{SO}X_\infty^{SO}(\rho^{SO}-r)=0 \tag{7.18}$$

Then τ_2^{OL} is obtained from equation (7.17). From equation (7.18), it appears that any combination of τ_1, τ_3 such that $\tau_1 + \tau_3/(r+\delta) = \tau_1^{OL}$ is a candidate to induce optimal choice. Hence combining the three tax parameters constitutes a case of over-information. The policy mix then reduces to either $(\tau_2^{OL}, \tau_1^{OL})$ or $(\tau_2^{OL}, \tau_3^{OL})$ with $\tau_3^{OL} = (r+\delta)\tau_1^{OL}$. $\square$

It appears that a mixed tax scheme is necessary to induce the agents to follow the optimal groundwater accumulation path : $(\tau_1 + \tau_2 X)u_i$ or $(\tau_2 u_i + \tau_3)X$. In the remainder of this Section, only the first formulation, the stock dependent input tax, is analysed. Its rate is always positive. This contrasts with the study of a polluting oligopoly by Benchekroun and van Long (1998). A somewhat surprising result of their study is that it might be optimal to subsidise polluters, for some initial interval, even if their *laissez-faire* output is higher than the social one at each point of time ; indeed oligopolists tend to underemit due to the market power effect. In our study, market power is not an issue. Indeed, agents are linked by environmental, and not market, variables.

In the context of irrigation-induced salinity, such a tax tells the polluters that, as the water percolates from their field and affects the level of the watertable, the more they irrigate now, the higher their future tax liability. Even if their current emissions are optimal, the level of the tax is conditioned upon the current state of the aquifer, which is affected by their past actions. Next an analysis the impact of the various parameters of the model on the value of τ_1^{OL} and τ_2^{OL} is provided.

Impact of the discount rate : $\frac{\partial \tau_2^{OL}}{\partial r} < 0$, $\frac{\partial \tau_1^{OL}}{\partial r} < 0$. A higher discount rate induces a lower taxation burden. The optimal solution arises from the cooperative outcome, in which the discount rate taken into account is the same as the agents'. As a consequence, a higher discount rate means that the agents, individually but also collectively, do not value future damage as much as if the discount rate had been lower.

Impact of the discharge rate : $\frac{\partial \tau_2^{OL}}{\partial \delta} < 0$, $\frac{\partial \tau_1^{OL}}{\partial \delta} < 0$. The groundwater discharge rate plays a role similar to the discount rate. With a higher discharge rate, water accumulates underground less rapidly. Consequently the necessary tax needs not be as high.

A comparison of the partial derivatives shows that δ has more influence on the level of the tax parameters than r. Indeed, δ has a more direct impact on the level of the stock than r. δ refers to mechanical effects, when r affects economic variables. Also, the value of r (resp. δ) impacts on the relative importance of δ (resp. r) on the accumulation of the stock. Indeed, when the discharge fraction is very low, the level of the discount rate has even more importance on groundwater's accumulation rate.

Impact of the individual damage term : $\frac{\partial \tau_2^{OL}}{\partial d} > 0$ and $\frac{\partial \tau_1^{OL}}{\partial d} > 0$. This result might seem puzzling at first. Indeed, it states that the more the agents are affected by the rising of the watertable (and individual perception of the damage dX^2 increases), the more they get taxed. At first sight, d drives each agent to partly internalise the externality. However, given the definition of the social damage, which amounts to the sum of individual damages, d is also linked to the damage imposed to the other agents. Hence the signs of the partial derivatives.

Impact of the number of agents : $\frac{\partial \tau_2^{OL}}{\partial n} > 0$ et $\frac{\partial \tau_1^{OL}}{\partial n} > 0$. This simply states that when the number of agents increases, each will tend to ignore its contribution to the rising of the watertable. Hence the need for a more severe tax.

7.4.2 Feedback strategies

Proposition 8. *To induce agents following feedback strategies to reach the optimal level of groundwater at the steady state, the regulator needs to have recourse to either a pure ambiant tax $\hat{\tau}_3$ or a pure input tax $\hat{\tau}_1$.*

Démonstration. Refer to Proposition 6 : the same type of relations are obtained in the feedback case. □

Proposition 9. *There exists an incentive scheme, of the form $T(u_i, X) = (\tau_2^{FB} X + \tau_1^{FB})u_i$ that induces agents to follow the optimal input path as a symmetric MPNE.*

Démonstration. With the same method as in the open-loop case, the optimal incentive scheme parameters are obtained :

$$\begin{aligned}
\tau_2^{FB} &= \frac{nd + c(\delta + \rho^{SO})[r + \delta/n + \rho^{SO}(1-2n)/n]}{-n[rn + 2\delta + 2\rho^{SO}(1-n)]}, \\
\tau_1^{FB} &= \frac{\rho^{SO} X_\infty^{SO}(1-n)}{rn + \delta + \rho^{SO}(1-n)} \tau_2^{FB} \\
&+ \frac{\rho^{SO} c X_\infty^{SO}(\rho^{SO}(1-2n) + \delta + nr - n\delta) + nb(rn + \delta + \rho^{SO}(1-n)) - n^2\tau_3}{n(rn - n\rho^{SO} + \delta + \rho^{SO})}.
\end{aligned}$$

As in the open-loop case, focus is given to stock dependant input taxes and τ_3 is set equal to zero without affecting the optimality of τ_1^{FB} and τ_2^{FB}. □

Hence, whatever the informational structure of the agents, the type of tax necessary to induce the agents to reach the two types of efficiency are of similar structure. While standard instruments are sufficient to ensure the optimality of the level of accumulation at the steady state, compliance with the optimal accumulation path necessitates having recourse to mixed instruments, incorporating elements of individual and collective performance. In the next point, the impact of the informational structure of the agents on the relative importance of the individual and collective performance items is assessed.

7.4.3 Comparing the feedback and open-loop taxes

Proposition 10. *The ratio τ_1/τ_2 is greater in the feedback case than in the open-loop case.*

Démonstration. See Appendix A. □

This means that faced by agents making use of feedback strategies, the regulator needs to adjust the tax rate, compared to the open-loop case, so as to decrease the relative part of collective incentives, represented by τ_2 , compared to individual incentives, represented by τ_1. This allows the regulator to decrease the incentive to behave strategically with respect to the exploitation of the adsorptive capacity of the aquifer by individual agents.

7.5 Numerical illustrations

This Section presents an illustration of the model, with the following parameters' values : $a = 0.05$, $b = 1.95$, $c = 0.80$, $d = 0.01$, $Xo = 0$, $r = 0.04$, $\delta = 0.01$, $n = 10$. The groundwater accumulation paths are given by :

$$X^{SO}(t) = 1.948441247(1 - e^{-0.7711384200t})$$
$$X^{OL}(t) = 19.34523809(1 - e^{-0.2317935662t})$$
$$X^{FB}(t) = 75.44920344(1 - e^{-0.1762655848t})$$

The stock paths are presented in Figure 7.1. The non-cooperative steady state stocks exceed the optimal one. Also, the feedback stock exceeds the open-loop one, at each point of time. In Figure 7.2, is illustrated the differentiated impacts of the various optimal steady state taxes on groundwater accumulation. It appears that $\hat{\tau}_1$ and $\hat{\tau}_3$ induce the same accumulation, lower than what is socially optimal, while $\hat{\tau}_2$ leads to an accumulation higher than what is socially optimal, except at the steady state. Figure 7.3 presents the evolution of the optimal tax rate $\tau_1 + \tau_2 X(t)$ through time, in both the open-loop and feedback cases.

7.6 Concluding remarks

This Chapter was devoted to the testing of Hypothesis 1, accepted in the case of a single stock accumulation, when efficiency is understood as inducing individual agents to make socially optimal decision along the whole time horizon. Indeed, when optimality is required at the steady state only, taxes based on one type of performance (either individual or group) are sufficient.

The modeling framework used in this Chapter captures the main features of agents' participation in the generation of irrigation-induced salinity. As pointed out in Chapter 6, the

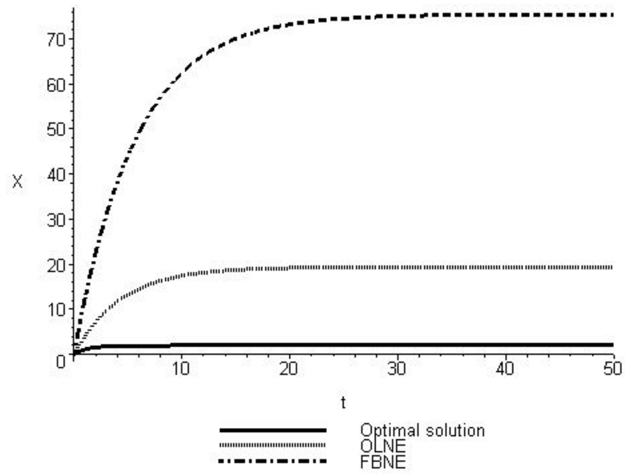

FIG. 7.1 – Comparison of the groundwater stock paths.

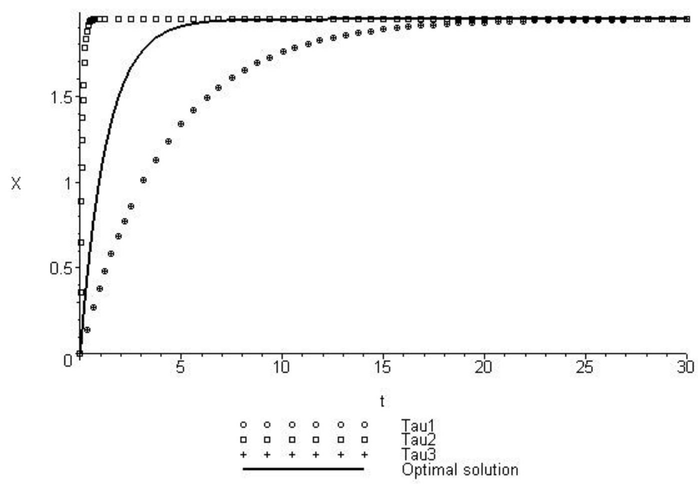

FIG. 7.2 – Impact of the steady state taxes on groundwater stock paths : open loop case

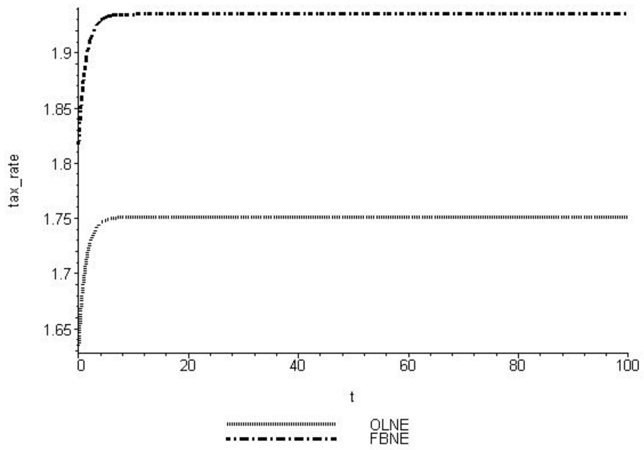

FIG. 7.3 – Optimal tax rate through time : open-loop and feedback cases.

model abstracts from heterogeneity and stochasticity issues. Considering heterogeneous agents would translate into the derivation of differentiated tax rates. The impact of heterogeneity on tax rates would depend on which factors are affected, and in which direction. For instance, an agent with a higher irrigation efficiency parameter would supposedly be subject to a lower tax rate than an highly inefficient agent. While heterogeneity would introduce a finer tax design, this would not change the qualitative results with respect to the implications of individual and collective performance measures in the design of policy instruments. Considering stochastic mechanisms, such as the contribution of rainfall to groundwater accumulation, would have necessitated the recourse to expected maximization techniques. A robust control analysis, that considers that some stochastic mechanisms cannot be modeled with certainty, was shown by Roseta-Palma and Xepapadeas (2004) to lead to a precautionary principle. This would lead in the present case to imposing higher tax rates to comply with a lower social optimum. A way to account for stochastic issues could be to carefully model agents' contributions, as opposed to rainfall contribution, to use this aggregate as the collective performance policy base. In other words, the strategy could consist in differentiating agents' from nature's contribution to groundwater accumulation.

The acceptance of Hypothesis 1 leads to advocate the use of a policy mix for the management of dynamic externalities. Policy mix here takes a somewhat different sense than usual, as it is understood as taxation schemes combining measures of individual and group performance.

The question of the implementation of such taxation schemes depends on the extent of the transaction costs associated with their design and implementation. The transaction costs associated with the design are addressed in Chapter 6 ; they concern the generation of information regarding individual agents' characteristic and the measurement of individual and collective performance. More importantly, at the implementation stage, are the transaction costs associated with the political acceptability of such policy instruments that are supposedly high in the Australian context. First, irrigation has been implicitly subsidized since its first years of development, in reason of the social features it acquired, as the key mechanism of development of the young colonies that formed Australia. Second, some irrigators may refuse to recognize the link between their actions and groundwater' evolution over time. Hence implementing such taxation schemes may necessitate investment in terms of information and education of the communities.

Another implication of this analysis is the importance of the formulation of the informational structure of the agents. The optimal tax, capable of inducing the agents to follow the optimal path along the whole time horizon, has a different ratio of individual vs group performance incentives according to the agents' strategy space. The regulator needs to decrease the group performance part of the taxation schemes, relative to the individual performance part, in the feedback case compared to the open-loop case. Indeed, introducing group performance in policy design may introduce negative incentives that counterbalance the positive interactions arising between the agents and the regulator. Consequently, inducing agents to use open-loop strategies may prove an efficient strategy for the regulator. Assuming that the agents make use of open-loop strategies may amount to (1) limiting their information about the state of the environment or (2) forcing them to commit to announced levels of input consumption. The latter is the way suggested by Amir and Nannerup (2006) in their analysis of dynamic resource extraction. Indeed, having shown that the open-loop equilibrium amounts to the optimal outcome, they argue that forcing the agents to submit a vector of consumption levels and to stick to these announcement may prove a first best strategy for the regulator. The former could be a way of institutionally inducing the agents to make their decisions without any knowledge of the state of the environment, or of their rivals' actions. Such a proposal is quite provocative, as argued by Amir and Nannerup (2006), as open-loop strategies are usually rejected in reason of their unrealistic nature. However, this analysis has pointed out the interest of considering open-loop strategies when GPBIs are applied.

Chapitre 8

Managing correlated externalities : water taxes with a pinch of salt

The aim of this Chapter is to extend the analysis of dynamic taxation schemes to the context of correlated externalities, and to assess the impact of the correlation on the optimal tax design. Indeed, a refinement of the differential game used to study irrigators' behavior includes the consideration of a second type of state variable, individual root zone salt stocks.

A number of the most stringent environmental issues are dynamic by nature as they involve the accumulation of pollution or the depletion of resources over time. Examples include acid rains, climate change, ozone depletion, groundwater depletion or biodiversity loss. Most economic analyses address the case of a unique stock (Mäler and de Zeeuw 1998) (Benchekroun and van Long 1998) (Farzin 1996) (Gisser and Sanchez 1980). Fewer papers recognise that in some settings it is more appropriate to consider multiple interrelated stocks. This is quite straightforward in the case of predator-prey interactions (Ragozin and Brown 1985). Forest ecosystems are also illustrative of complex interactions between multiple species. Crépin (2003) and Bergland, Ready and Romstad (2006) study the case of boreal forests. In particular, Crépin (2003) describes the interactions arising between moose, caduceus and conifers stocks. Bond and Farzin (2004) develop a multiple nutrient stocks model to highlight the differentiatied impacts of various decisions - fertilizer input, tillage - on the soil treated as a portfolio of nitrogen pools. Farzin and Tahvonen (1996) and more recently, Caplan and Silva (2005) or Yang (2006), have pointed out the need to model climate change more accurately by describing multiple interacting stocks. Caplan and Silva (2005) analyze pollution rights markets to manage carbon, a global pollutant, and smog, a local pollutant, emitted jointly by a set of firms. They show how a system combining a global market and regional markets is Pareto efficient. Yang (2006) focuses on negatively correlated externalities in the same context, stemming from the observation that primary energy consumption generates a global externality, global warming, and a local externality, SO2 emission, that alleviates global warming and in this sense is at the same time harmful to local air quality and beneficial at the global level through its cooling effect. Yang (2006) shows how the negative relation between the global and the local externalities impacts on the design of policy instruments, as it raises the need for a subsidy for local externalities when the global issue is accounted for.

This Chapter is organised as follows. Section 8.1 presents a general model of interrelated stocks, and highlights some properties of various taxation schemes. Section 8.2 presents the derivation of the social optimal and individual paths of irrigation water use and root zone and groundwater stocks' accumulation. The optimal tax parameters are derived in Section 8.3. Section 8.4 concludes.

8.1 A general model of interrelated stocks

Suppose the existence of a benevolent policymaker who maximises total benefits from the production activities of a set of n agents, denoted by i, $i \in \{1..n\}$. The policymaker also accounts for the existence of two sources of damage, a global stock X and multiple local stocks, Q_i, for $i \in \{1..n\}$. Let $B_i(u_i, Q_i, X)$ be agent i's benefit function, u_i his control variable, with $\partial B_i/\partial Q_i < 0$ and $\partial B_i/\partial X < 0$[1]. Social damage is denoted by $D(X)$ and r is the discount rate. The socially optimal maximisation program is :

$$\max_{\{u_i\}\geq 0} \int_0^{\infty} \left(\sum_{i=1}^{n} B_i(u_i, Q_i, X) - D(X) \right) e^{-rt}.dt \text{ , subject to :}$$

$$\dot{X} = F(\sum_{i=1}^{n} u_i, X), \tag{8.1}$$

$$\dot{Q}_i = G_i(u_i, Q_i, X), \forall i. \tag{8.2}$$

This model is close to the one developed by Yang (2006). Among the main differences, the collective stock impacts on the individual ones rather than the opposite. Hence in the present framework $\partial G_i/\partial X \neq 0$ and $\partial F/\partial Q_i = 0$, illustrating a type of feedback loop between the collective stock, an aggregation of individual decisions, and the individual stocks. Also, in this thesis, it is considered that the local stocks are managed by a unique agent, rather than by a subset of agents.

The Hamiltonian writes as follows :

$$H^{so} = \sum_{i=1}^{n} B_i(u_i, Q_i, X) - D(X) + \lambda^{so} F(\sum_{i=1}^{n} u_i, X) + \sum_{i=1}^{n} \mu_i^{so} G_i(u_i, Q_i, X),$$

where λ^{so} and μ_i^{so} are the costate variables associated with the collective and individual

[1]To come back to the standard case where the agents are not affected by the stocks, simply set $\partial B_i/\partial Q_i = \partial B_i/\partial X = 0$.

stocks, so that the first order conditions are given by equations (8.3)-(8.5) :

$$\frac{\partial H}{\partial u_i} = \frac{\partial B_i}{\partial u_i} + \lambda^{so}\frac{\partial F}{\partial u_i} + \mu_i^{so}\frac{\partial G_i}{\partial u_i} = 0, \tag{8.3}$$

$$-\frac{\partial H}{\partial X} = \dot{\lambda}^{so} - r\lambda^{so} = -\left[\sum_{i=1}^{n}\frac{\partial B_i}{\partial X} - D'(X) + \lambda^{so}\frac{\partial F}{\partial X} + \sum_{i=1}^{n}\mu_i^{so}\frac{\partial G_i}{\partial X}\right], \tag{8.4}$$

$$-\frac{\partial H}{\partial Q_i} = \dot{\mu_i}^{so} - r\mu_i^{so} = -\left[\frac{\partial B_i}{\partial Q_i} + \mu_i^{so}\frac{\partial G}{\partial Q_i}\right], \forall i. \tag{8.5}$$

Equation (8.3) can be rearranged to obtain expressions of the co-state variables :

$$\lambda^{so} = \frac{-\frac{\partial B_i}{\partial u_i} - \mu_i^{so}\frac{\partial G_i}{\partial u_i}}{\frac{\partial F}{\partial u_i}} \text{ and } \mu_i^{so} = \frac{-\frac{\partial B_i}{\partial u_i} - \lambda^{so}\frac{\partial F}{\partial u_i}}{\frac{\partial G_i}{\partial u_i}}.$$

Then equations (8.4) and (8.5) become[2] :

$$\left[\frac{\dot{\lambda}^{so}}{\lambda^{so}} - (r - \frac{\partial F}{\partial X})\right]\frac{1}{\frac{\partial F}{\partial u_i}} = \frac{\sum_i \frac{\partial B_i}{\partial X} - D'(X) + \sum_i \mu_i^{so}\frac{\partial G_i}{\partial X}}{\frac{\partial B_i}{\partial u_i} + \mu_i^{so}\frac{\partial G_i}{\partial u_i}}, \tag{8.6}$$

$$\left[\frac{\dot{\mu_i}^{so}}{\mu_i^{so}} - (r - \frac{\partial G_i}{\partial Q_i})\right]\frac{1}{\frac{\partial G_i}{\partial u_i}} = \frac{\frac{\partial B_i}{\partial Q_i}}{\frac{\partial B_i}{\partial u_i} + \lambda^{so}\frac{\partial F}{\partial u_i}}. \tag{8.7}$$

The RHS of equations (8.6) and (8.7) present the ratio between the marginal desutilities from the stocks and the marginal utilities from irrigating, in other words a marginal rate of substitution. The numerator of the RHS of equation (8.6), for instance, includes the sum of direct individual damage from the global stock, $\sum_i \frac{\partial B_i}{\partial X} < 0$, the social damage from the global stock, $-D'(X) < 0$, and the indirect impact of the global stock on the local stocks, $\sum_i \mu_i^{so}\frac{\partial G}{\partial X}$. The latter illustrates the interaction between the stocks, that can be positive or negative. Hence if the stocks are positively correlated, $\frac{\partial G_i}{\partial X} > 0$; as $\mu_i^{so} < 0$ the indirect impact of the global stock through the local stock acts as a damage. However, if the stocks are negatively correlated, this indirect impact is a benefit, as increasing the global stock tends to decrease the local one. Note that in equation (8.7) there is no interaction term, as it is the global stock that impacts on the local ones. The denominators of the RHS present the marginal utilities from using irrigation water, which is the marginal production plus a negative term accounting for the impact of increasing input use on the other stock.

The LHS of equations (8.6) and (8.7) are constituted of a dynamic term and of the marginal individual contribution to the (individual or collective) stock pollution. The dynamic term comprises the time path of the costate variables, for instance $\dot{\mu_i}^{so}/\mu_i^{so}$ and an environmental discount rate, $r - \partial G_i/\partial Q_i$, where $\partial G_i/\partial Q_i$ is the natural stock dissipation rate.

Now, consider the case of individual agents following open-loop strategies. The individual

[2] Assuming strict negativity of the co-state variables.

maximisation program is :

$$\max_{u_i \geq 0} \int_0^\infty B_i(u_i, Q_i, X)e^{-rt}.dt \text{ , subject to equations (8.1) and (8.2).}$$

The conditions driving the accumulation paths are :

$$\left[\frac{\dot{\lambda}^{ol}}{\lambda^{ol}} - (r - \frac{\partial F}{\partial X})\right] \frac{1}{\frac{\partial F}{\partial u_i}} = \frac{\frac{\partial B_i}{\partial X} + \mu_i^{ol}\frac{\partial G_i}{\partial X}}{\frac{\partial B_i}{\partial u_i} + \mu_i^{ol}\frac{\partial G_i}{\partial u_i}}, \tag{8.8}$$

$$\left[\frac{\dot{\mu_i}^{ol}}{\mu_i^{ol}} - (r - \frac{\partial G_i}{\partial Q_i})\right] \frac{1}{\frac{\partial G_i}{\partial u_i}} = \frac{\frac{\partial B_i}{\partial Q_i}}{\frac{\partial B_i}{\partial u_i} + \lambda^{ol}\frac{\partial F}{\partial u_i}}. \tag{8.9}$$

Comparing with the social optimum case, equations (8.8) and (8.9) highlight multiple sources of inefficiency. First, from equation (8.8), it appears that the social damage is not accounted for. Furthermore, individual agents account only for their individual damage, without taking into consideration the impact of a rising of the stock on the other agents. Also, in equation (8.8), μ_i^{ol} isn't valued at the same level as μ_i^{so} in equation (8.6). Consequently, none of the stocks is accumulated optimally.

Suppose that the policymaker wishes to induce the agents to act optimally by implementing a taxation scheme. He consider various taxation bases : individual input use, individual stock, global stock :

$$\tau(u_i, Q_i, X) = u_i(\tau_1 + \tau_2 X) + \tau_3 X + \tau_4 Q_i.$$

Then individual decisions are modified in such a way as to lead to the following conditions on stock accumulation :

$$\left[\frac{\dot{\lambda}^{\tau}}{\lambda^{\tau}} - (r - \frac{\partial F}{\partial X})\right] \frac{1}{\frac{\partial F}{\partial u_i}} = \frac{\frac{\partial B_i}{\partial X} + \mu_i^{\tau}\frac{\partial G_i}{\partial X} - (\tau_2 u_i + \tau_3)}{\frac{\partial B_i}{\partial u_i} + \mu_i^{\tau}\frac{\partial G_i}{\partial u_i} - (\tau_1 + \tau_2 X)}, \tag{8.10}$$

$$\left[\frac{\dot{\mu_i}^{\tau}}{\mu_i^{\tau}} - (r - \frac{\partial G_i}{\partial Q_i})\right] \frac{1}{\frac{\partial G_i}{\partial u_i}} = \frac{\frac{\partial B_i}{\partial Q_i} - \tau_4}{\frac{\partial B_i}{\partial u_i} + \lambda^{\tau}\frac{\partial F}{\partial u_i} - (\tau_1 + \tau_2 X)}. \tag{8.11}$$

Tax parameters (τ_1, τ_2, τ_3,τ_4) don't have the same impact on the optimal paths. τ_1 only appears in the expressions of marginal benefits from input use, τ_3 in the expression of marginal disutility from the global stock accumulation and τ_4 in the expression of marginal disutility from the local stock accumulation. τ_2 plays a particular role as it is the interaction term, attached to both input use and the global stock.

Finally, consider the case of individual agents following feedback strategies. Assuming the existence of a MPNE, the Hamiltonian is :

$$H^{fb} = B_i(u_i, Q_i, X) + \lambda^{fb} F(u_i, \{u_j(Q_j, X)\}_{j\neq i}, X) + \mu_i^{fb} G_i(u_i, Q_i, X).$$

Then the same type of conditions are derived as in the previous cases describing the co-state variables time paths :

$$\left[\frac{\dot{\lambda}^{fb}}{\lambda^{fb}} - (r - \frac{\partial F}{\partial X} - \sum_{j\neq i} \frac{\partial F}{\partial u_j}\frac{\partial u_j}{\partial X})\right] \frac{1}{\frac{\partial F}{\partial u_i}} = \frac{\frac{\partial B_i}{\partial X} + \mu_i^{fb}\frac{\partial G_i}{\partial X}}{\frac{\partial B_i}{\partial u_i} + \mu_i^{fb}\frac{\partial G_i}{\partial u_i}} \tag{8.12}$$

$$\left[\frac{\dot{\mu}_i^{fb}}{\mu_i^{fb}} - (r - \frac{\partial G_i}{\partial Q_i})\right] \frac{1}{\frac{\partial G_i}{\partial u_i}} = \frac{\frac{\partial B_i}{\partial Q_i}}{\frac{\partial B_i}{\partial u_i} + \lambda^{fb}\frac{\partial F}{\partial u_i}}. \tag{8.13}$$

In the LHS of equation (8.12), the summation term illustrates the difference between open-loop and feedback strategies ; indeed in the latter case, agents account for the impact of their decisions on the others' decisions. The direction of this impact depends on the sign of two terms. First, $\partial F/\partial u_j > 0$ in the case of stock accumulation. Second, the sign of $\partial u_j/\partial X$ depends on the complement - substitute nature of agents' strategies (Bulow et al. 1985). If they are strategic complements, $\partial u_j/\partial X > 0$, then the resulting environmental discount rate perceived by i is decreased. Consequently, agent i is induced to reduce his contribution to the stock. When $\partial u_j/\partial X < 0$, the perceived discount rate is increased : emission activities are shifted from the future to the present.

Note that the consideration of individual local stocks simplifies the setting ; indeed if the local stocks were managed by numerous agents, another layer of strategic interaction would exist.

Applying a tax $\tau(u_i, Q_i, X) = u_i(\tau_1 + \tau_2 X) + \tau_3 X + \tau_4 Q_i$ has the same effect as in the open-loop case. However the derivation of the optimal tax parameters needs to account for the altered 'environmental discount rate'. Consequently, in the remainder of this Chapter, individual agents are assumed to follow open-loop strategies.

The next Sections analyse irrigation-induced salinity as an illustrative example of such a setting combining interacting individual and collective stock pollutions. They address the design of optimal taxation schemes, such as the ones analysed in Chapter 7.

8.2 Stock accumulation and irrigation decisions in the context of irrigation-induced salinity

Consider the case of one catchment, understood as the recharge area of a specific aquifer. Above this aquifer, n agents divert water from a river to undertake irrigation. In doing so, they generate a certain amount of water percolating down to the aquifer, and a certain amount of water available for uptake by plants in the root zone. Each water reservoir (river, root zone, watertable) is also characterised by its salt content. To highlight the main mechanisms associated with irrigation-induced salinity, only the collective stock of groundwater and the individual root zone salt stocks are considered[3]. It is also assumed that the catchment under study is already affected by the various manifestations of irrigation-induced salinity.

8.2.1 The model

Watertable dynamics. A simple description of watertable dynamics is used that accounts for the main mechanisms at stake, highlighting the impact of agents' decisions on the state of the watertable :

$$\dot{X} = \sum_{i=1}^{n} P_i - (\delta + \gamma)X, \tag{8.14}$$

where X is the stock of groundwater, P_i is the water percolating from farm i, δ is a discharge fraction and γ is a global capillary rise parameter[4].

Root zone salts dynamics. As presented in Chapter 6, the following expression of the root zone salts dynamics equation is used :

$$\dot{Q}_i = S^W u_i + \gamma_i S^G X - \frac{1-\beta}{\beta} Q_i, \tag{8.15}$$

where γ_i is the capillarity rate for property i, defined so that $\sum_i \gamma_i = \gamma$. The quantity of groundwater X is a variable common to all the farmers, so is the concentration of salt S^G. Furthermore, S^W and S^G are treated as exogenous variables. However, water remaining in the root zone and percolating water are individual variables, as they depend on individual choices, regarding the quantity of water applied and the irrigation technology.

Different types of damages. Irrigation-induced salinity is a complex environmental issue, as it combines problems associated with both the quantity and the quality of water, at two levels, the surface system and the underground system. Three main sources of damage are considered in this study : soil salinity and waterlogging are treated as individual damage, affecting directly the agents' production function, while the discharge of salty

[3] Hence the interactions that exist between the river and the aquifer are not accounted for explicitely.

[4] With $0 \leq \beta \leq 1$ and $0 \leq \gamma \leq 1$

water is treated as a social damage that individual irrigators do not account for.

The primary effect of saline water in the root zone is the inability of the plant to compete for water with ions in the soil solution, a phenomenon known as physiological drought. The higher the salinity the less water available to plants. This has negative consequences in terms of crop yields. Crop yield response functions to salinity usually highlight thresholds within which the loss of production is linear in the salt content (Maas and Hoffman 1977). Assuming that irrigation salinity already affects the study area, we consider the following individual damage term : dQ_i.

Several authors (Kahlown and Azam 2002) stress that salinity usually happens in conjunction with waterlogging, and that these phenomena have an impact on crop yield both jointly and in isolation. In this study, it is assumed that the interactions between these two mechanisms are accommodated through the interacting stocks ; hence the 'isolated' damage from waterlogging is : $f\frac{X^2}{2}$.

Finally, increased discharge of salty water into the surface system results from increased inflows to the groundwater stock. They have consequences on a range of factors, ecological but also linked to human activities. They enter the policymaker's utility function, as a function of the salt content of the discharge : $D(\delta S^G X)^2/2$.

Benefit function. The agents under consideration are assumed to be profit-maximiser agents ; they use water for irrigation purposes and are affected by soil salinity and waterlogging. They maximise their utility with respect to the use of input (irrigation water) only. In other words, irrigation technology is not considered as a decision variable. Each technology is associated with an efficiency β. It impacts on the definition of water remaining in the root zone and percolating. The agents are assumed price-takers, with respect to the price of water p_e and the price of their production p. Their utility function is defined as follows :

$$B_i(u_i, Q_i, X) = p(a + bu_i - c\frac{u_i^2}{2} - dQ_i - f\frac{X^2}{2}) - p_e u_i.$$

8.2.2 Comparison of socially optimal and individual programs

The regulator's maximisation program is :

$$\max_{\{u_i\}\geq 0} \int_0^\infty \left(\sum_{i=1}^n B_i(u_i, Q_i, X) - D(S^G \delta X) \right) e^{-rt}.dt \text{ subject to (8.14) and, } \forall i \text{ , (8.15)}$$

The program of an individual agent following an open-loop strategy is :

$$\max_{u_i \geq 0} \int_0^\infty B_i(u_i, Q_i, X) e^{-rt}.dt \text{ subject to (8.14) and (8.15)}$$

Comparisons between the socially optimum and individual paths are summed up in Propositions 1, 2 and 3.

Proposition 1. *There exists an equilibrium to each program (policymaker, open-loop agents) with the saddle point property.*

Démonstration. The resolution process follows Dockner (1985) who provides an analysis of the local stability of two state variables optimal control problems. Indeed, as homogeneous agents are considered, and thus salt stocks are homogenous, the regulator *de facto* accounts for two state variables. Consequently, $\gamma_i = \gamma/n$, $\forall i$. The full derivation is provided for the social optimum case ; refer to Appendix B-2 for the open-loop case.

The Hamiltonian writes as follows :

$$H^{so} = \sum_{i=1}^{n} B_i(u_i, Q_i, X) - D(\delta S^G X)$$
$$+ \lambda^{so}[\sum_{i=1}^{n}(1-\beta)u_i - (\delta+\gamma)X] + \sum_{i=1}^{n} \mu_i^{so}[S^W u_i + \gamma_i S^G X - Q_i \frac{1-\beta}{\beta}],$$

where λ^{so} and μ_i^{so} are the costate variables associated with groundwater and salinity[5]. Equations (8.16) to (8.18) constitute the first-order conditions to the socially optimum program :

$$\frac{\partial H^{so}}{\partial u_i} = p\frac{\partial F}{\partial u_i} - p_e + \lambda^{so}(1-\beta) + \mu_i^{so} S^W, \tag{8.16}$$
$$\dot{\lambda^{so}} - r\lambda^{so} = -\frac{\partial H^{so}}{\partial X} = X(npf + D\delta S^G) - \mu_i^{so}\gamma S^G + \lambda^{so}(\delta+\gamma), \tag{8.17}$$
$$\dot{\mu_i^{so}} - r\mu_i^{so} = -\frac{\partial H^{so}}{\partial Q_i} = -p\frac{\partial F}{\partial Q_i} + \mu_i^{so}\frac{1-\beta}{\beta}. \tag{8.18}$$

Rearrange equations (8.16)-(8.17)-(8.18) to obtain equation (8.16.a)-(8.17.a)-(8.18.a). Equation (8.16.a) exhibits the standard equality of marginal benefits and costs associated with using one more unit of irrigation water :

$$p(b - cu_i) = p_e - \lambda^{so}(1-\beta) - \mu^i S^W. \tag{8.16.a}$$

The marginal costs consist of the direct cost of buying water, the indirect cost of an additional unit of water percolating and the indirect cost of salty water brought by irrigation and remaining in the root zone.

$$\dot{\lambda^{so}} = (r+\delta+\gamma)\lambda^{so} + (D\delta S_G + npf)X - \mu_i^{so}\gamma S^G. \tag{8.17.a}$$

[5] It is assumed that the conditions for an interior solution hold, so that the non negativity constraint on u_i doesn't bing. Refer to Appendix B-1 for an analysis of non negativity conditions.

Equation (8.17.a) simply illustrates that the costate variable associated with the watertable dynamics must increase at the rate of $(r+\delta+\gamma)$ and taking account of the downstream damage (in terms of discharge of salty water in the surface system) and of the salts brought upwards by capillary rise.

$$\dot{\mu_i^{so}} = (r + \frac{1-\beta}{\beta})\mu_i^{so} + dp. \tag{8.18.a}$$

In equation (8.18.a), the costate variable associated with the salt accumulation in the root zone increases at the rate $(r+\frac{1-\beta}{\beta})$ and taking account of the damage on production due to salts in the root zone. Eequation (8.16.a) leads to :

$$u_i = [\lambda^{so}(1-\beta) + \mu_i^{so} s^W + pb - p_e]/pc.$$

Replace u_i by this expression in equations (8.14) and (8.15) and add equations (8.17) and (8.18) to form the Modified Hamiltonian Dynamic System (MHDS) :

$$\begin{pmatrix} \dot{X} \\ \dot{\lambda^{so}} \\ \dot{Q_i} \\ \dot{\mu^{so}} \end{pmatrix} = \begin{pmatrix} -\delta-\gamma & \frac{n(1-\beta)^2}{pc} & 0 & \frac{nS^W}{pc}(1-\beta) \\ D\delta S^G + npf & r+\delta+\gamma & 0 & -\gamma S^G \\ \gamma_i S^G & \frac{S^W}{pc}(1-\beta) & \frac{\beta-1}{\beta} & \frac{s^{W2}}{pc} \\ 0 & 0 & 0 & r+\frac{1-\beta}{\beta} \end{pmatrix} \cdot \begin{pmatrix} X \\ \lambda^{so} \\ Q_i \\ \mu^{so} \end{pmatrix} + \begin{pmatrix} n(1-\beta)\frac{pb-p_e}{pc} \\ 0 \\ S^W \frac{pb-p_e}{pc} \\ dp \end{pmatrix}$$

Let J stand for the matrix with typical elements $\partial u/\partial v$ with $u, v \in \{X, Q_i, \lambda, \mu_i\}$ and C for the constants matrix. Define K as follows :

$$K = \begin{vmatrix} \frac{\partial \dot{X}}{\partial X} & \frac{\partial \dot{X}}{\partial \lambda} \\ \frac{\partial \dot{\lambda}}{\partial X} & \frac{\partial \dot{\lambda}}{\partial \lambda} \end{vmatrix} + \begin{vmatrix} \frac{\partial \dot{Q_i}}{\partial Q_i} & \frac{\partial \dot{Q_i}}{\partial \mu_i} \\ \frac{\partial \dot{\mu_i}}{\partial Q_i} & \frac{\partial \dot{\mu_i}}{\partial \mu_i} \end{vmatrix} + 2 \begin{vmatrix} \frac{\partial \dot{X}}{\partial Q_i} & \frac{\partial \dot{X}}{\partial \mu_i} \\ \frac{\partial \dot{\lambda}}{\partial Q_i} & \frac{\partial \dot{\lambda}}{\partial \mu_i}. \end{vmatrix}$$

The conditions $K < 0$ and $0 < detJ < (\frac{K}{2})^2$ are necessary and sufficient to ensure the saddlepoint property (Dockner 1985).

Let $z_1^{so} = \frac{1-\beta}{\beta}(r+\frac{1-\beta}{\beta}) > 0$ and $z_2^{so} = (\delta+\gamma)(r+\delta+\gamma) + \frac{n}{pc}(1-\beta)^2(D\delta S^G + npf) > 0$.

Then $K = -z_1^{so} - z_2^{so} < 0$, which ensures that the first condition is met. Also, $\det J = z_1^{so} z_2^{so} > 0$. Then $(\frac{K}{2})^2 - \det J = \frac{1}{4}(z_1^{so}+z_2^{so})^2 - z_1^{so}z_2^{so} = \frac{1}{4}(z_1^{so}-z_2^{so})^2 > 0$. Hence the second condition is met. □

Proposition 2. *The steady state groundwater and root zone salt stocks in the open-loop case exceed the socially optimum ones.*

Démonstration. The value of the steady state variables is obtained after recognizing that at the steady state, the following relation holds : $Z^\infty = -J^{-1}.C$ where Z^∞ stands for the matrix with typical elements v^∞, $v \in \{X, Q_i, \lambda, \mu_i\}$. Computations give the following steady state values of the groundwater (respectively root zone salt) stocks and input use paths, for $m = \{so, ol\}$:

$$X^m_\infty = \frac{1}{z^m_2}[M_1 + \mu_\infty M^m_2]\frac{n(1-\beta)}{pc}, \tag{8.19}$$

$$Q^m_\infty = \frac{\beta}{1-\beta}\left[\gamma_i S^G X^m_\infty + S^W u^m_\infty\right], \tag{8.20}$$

$$u^m_\infty = \frac{1}{pc}[pb - p_e + \mu^m_\infty S^W + \lambda^m_\infty(1-\beta)]. \tag{8.21}$$

Refer to Appendix B-3 for the values of the parameters. Then $z^{so}_2 > z^{ol}_2$ and $M^{so}_2 > M^{ol}_2$ imply $X^{so}_\infty < X^{ol}_\infty$. It is also straightforward to note that $|\lambda^{ol}_\infty| < |\lambda^{so}_\infty|$. Also, from equation (8.21), $u^{ol}_\infty > u^{so}_\infty$, consequently $Q^{ol}_\infty > Q^{so}_\infty$. □

The steady state value of the costate variable associated with individual salt stock, μ_∞, is negative, and has the same value in the open-loop and social optimum cases. It depends on the extent of individual damage and on an environmental discount rate, $r + \frac{(1-\beta)}{\beta}$. It is the sum of the usual discount rate and of the discharge rate associated with the stock of salt. A higher discount rate induces a lower value of $|\mu_\infty|$: when the future counts less, or when the natural discharge capacity is high, the stock's shadow price decreases. The contrary happens when the individual damage term increases.

The steady state groundwater stock is related to the extent of damage due to its accumulation. The higher the damage, the smaller the steady state groundwater stock. The steady state salt stock accounts for the various types of damages.

Proposition 3. *Open-loop agents induce a higher accumulation of groundwater than what is socially optimum, which indirectly leads to a higher accumulation of the root zone salts.*

Démonstration. To derive the stock and input paths, the resolution method is a follows. We find the eigenvalues of matrix J, and choose the negative ones w_j, $j \in \{1, 2\}$ to ensure stability. Then we compute the associated eigenvectors w_{jv}, $v \in \{X, Q_i, \lambda, \mu_i\}$. The solutions are of the following form (Farzin and Tahvonen 1996) :

$$\varphi(v, v_\infty, v_0, t) = v_\infty + c_1 w_{1v} e^{w_1 t} + c_2 w_{2v} e^{w_2 t} \text{ with } v \in \{X, Q_i, \lambda, \mu_i\}.$$

The groundwater, root zone salinity and water input paths are, for $m = \{so, ol\}$:

$$X^m(t) = X^m_\infty + (X_0 - X^m_\infty)e^{w^m_1 t}, \tag{8.22}$$

$$Q^m_i(t) = Q^m_\infty + (Q_0 - Q^m_\infty)e^{w^m_2 t} + \frac{X_0 - X^m_\infty}{w^m_{1X}}(e^{w^m_1 t} - e^{w^m_2 t}), \tag{8.23}$$

$$u^m(t) = \lambda^m(t)\frac{1-\beta}{pc} + \mu_\infty\frac{S^W}{pc} + \frac{pb - p_e}{pc}. \tag{8.24}$$

Assuming that the initial groundwater stock is below the steady state, so that the issue at stake is an accumulation problem, the groundwater stock path is monotonic, and increasing from X_0 to X_∞^m at the rate w_1^m. Then, as $w_1^{so} < w_1^{ol}$, it follows that open-loop agents not only accumulate more groundwater, they also accumulate faster than what is socially optimal[6].

The salt stock's behavior depends on two phenomena. The first part of the RHS of equation (8.23), $Q_\infty^m + (Q_0 - Q_\infty^m)e^{w_2^m t}$, is an autonomously increasing part, in the sense that it is not directly affected by the collective groundwater stock. The second part of the RHS shows the correlation between the individual and the collective stocks. The sign of the interaction depends on three terms. First, $X_0 - X\infty^m < 0$ in cases of accumulation. Then, $e^{w_1^m t} - e^{w_2^m t} > 0$, as the exponential function is monotonously increasing, and because the following applies :

$$w_1^m > w_2^m \Rightarrow 1-\beta > -\beta w_1^m \Rightarrow 1-\beta > -\beta r/2 \text{ , as } w_1^m < r/2$$

Third, the sign of w_{1X} is a priori undetermined. It can be shown that its numerator is positive, however its denominator is more difficult to assess. It comprises two terms, $n(1-\beta)\gamma S^G$ and $S^w(w_1^m + \delta + \gamma)$; the term $w_1^m + \delta + \gamma$ illustrates a tradeoff between dissipating and accumulating features of the groundwater stock, the greater the accumulation rate, the lower $w_1^m + \delta + \gamma$, hence the greater w_{1X}. □

8.3 Designing optimal water taxes

In this Section, the potential of various dynamic taxes to induce the agents to take optimal decisions are assessed. The taxation schemes considered are the same as in Chapter 7 :

$$\tau(U_i, X) = \tau_1 u_i + \tau_2 u_i X + \tau_3 X.$$

Implicit to this formulation is the consideration that individual salt stocks are not appropriate as policy bases. Consider an agent, following an open-loop strategy, subject to such a taxation scheme. His maximisation problem is :

$$\max_{u_i} B_i(u_i, Q_i, X) - \tau(u_i, X) \text{ subject to equations (8.14) and, } \forall i \text{ , (8.15).}$$

Proposition 4. *The introduction of taxes based on the individual use of water and/or the level of groundwater does not alter the saddle point property of the equilibrium.*

[6] Refer to Annex B-4 for the values of the parameters.

Démonstration. The MHDS writes $Z_\infty = J^\tau Z + C^\tau$ with[7] :

$$J^\tau = \begin{pmatrix} A(\tau_2) & B & 0 & C \\ D(\tau_2) & E(\tau_2) & 0 & F(\tau_2) \\ G(\tau_2) & H & I & J' \\ 0 & 0 & 0 & K' \end{pmatrix} \text{ and } C^\tau t = \begin{pmatrix} L(\tau_1) \\ M(\tau_1, \tau_2, \tau_3) \\ N(\tau_1) \\ P \end{pmatrix}$$

The necessary and sufficient conditions to ensure the saddle point property are $K < 0$ and $(K/2)^2 - \det J > 0$, K being defined as in the previous Sections. Define :

$$z_1^\tau t = -IK' > 0,$$
$$z_2^\tau = -A(\tau_2)E(\tau_2) + D(\tau_2)B > 0$$

Then $K = -z_1^\tau - z_2^\tau < 0$ and $(K/2)^2 - det(J^\tau) = z_1^\tau z_2^\tau > 0$. Both conditions are satisfied, ensuring the saddle point property. □

Proposition 5. *Each tax parameter, associated to individual input use (τ_1), collective groundwater stock (τ_3) or a mix of both (τ_2), impacts on the steady state values of the stocks and input choice. However, only τ_2 affects the groundwater stock accumulation rate, thus indirectly the root zone salt stock's speed of accumulation.*

Démonstration. See Appendix B-6. □

Consequently, taxing individual agents on the basis of their input use and the collective groundwater stock will have an indirect impact on their root zone salt stock accumulation. These derivations confirm the insights from the general case study, that is that the mixed tax parameter τ_2 has a special status in the sense that it impacts on both the steady state value and the accumulation rate of the groundwater stock.

Proposition 6. *In order to induce the agents following open-loop strategies to reach the optimal steady state, the regulator can either use a pure ambient tax $\tau_3^* X$ or a standard input tax, $\tau_1^* u$. To induce open-loop agents to take optimal decisions along the whole time horizon, the regulator needs to have recourse to a mixed tax, based on parameter τ_2, and one of the purely collective (τ_3) or individual (τ_1) ones.*

Démonstration. Refer to Appendix B-6. □

8.3.1 Numerical illustrations

The values of the parameters in use in the simulations are as follows : $\delta = 0.01$, $\gamma = 0.0001$, $X_0 = 0$, $S^G = 5EC$, $B_i(u_i, Q_i, X) = 0.05 + 1.95u_i - \frac{0.80}{2}u_i^2 - 0.01Q_i - \frac{0.0001}{2}X^2$, $\beta = 0.6$, $S^W = 1.5EC$, $Q_0 = 0$, $p = 5$, $p_e = 1$, $r = 0.04$ and $n = 10$. Figure 8.1 illustrates the

[7] Refer to Appendix B-5 for the detailed MHDS.

variation between individual and socially optimal decisions through its impact on individual and collective stock accumulation. Figure 8.2 illustrates the differentiated impact of various steady state taxes on the accumulation of the stocks.

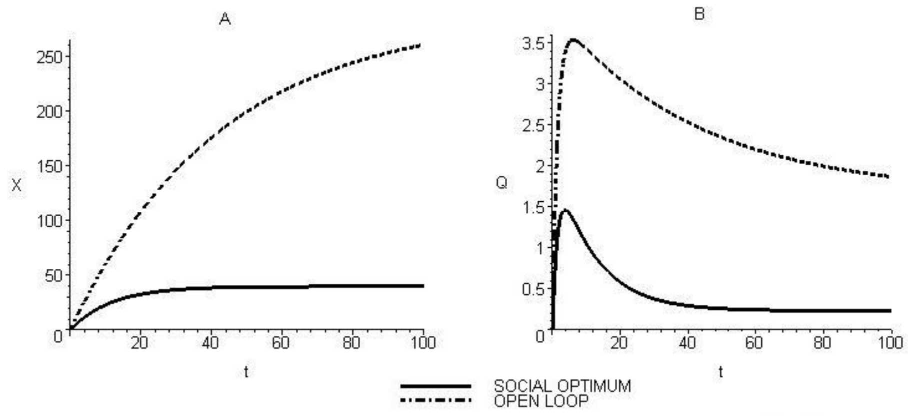

FIG. 8.1 – Comparison of the groundwater and root zone paths

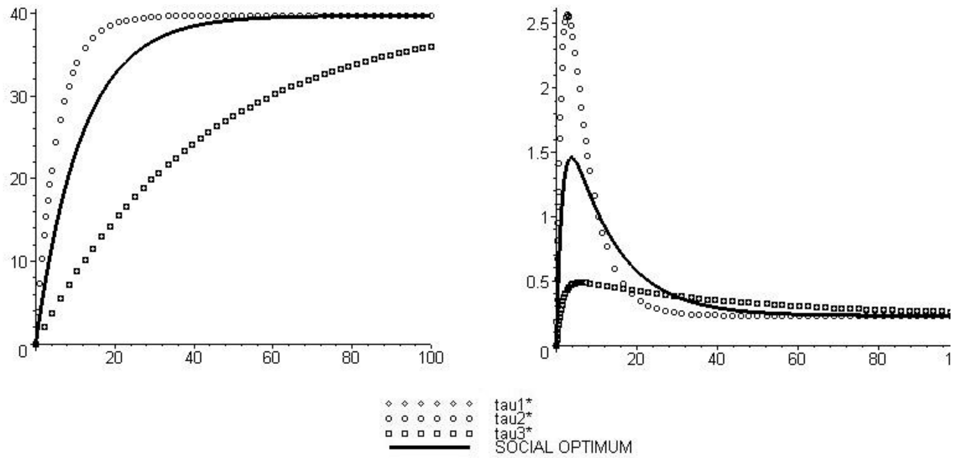

FIG. 8.2 – Impact of the steady state taxes on groundwater stock and root zone stock paths

8.4 Conclusion

This Chapter extends the analyses carried out in Chapter 7 to the case where the collective stock interacts with multiple individual stocks. It confirms Hypothesis 1 : the need for mixed taxation schemes to manage the stocks over the whole time horizon.

The interaction between the stocks is quite particular in this framework. First, the correlation is positive, so that an increase in the level of groundwater leads to an increase of root zone salts. Second, in this Chapter, two types of stocks are considered and the global stock impacts on the local ones. Indeed, in a more realistic, hence complex, model of irrigation-induced salinity, it would be necessary to describe a stock of salt and a stock of water for each reservoir at stake (river system, watertable, individual root zones). Then the individual salt stocks would impact on the global one. However, multiple state variables problems rapidly become not analytically tractable, hence the choice to focus on the main sources of damage : root zone salinity and groundwater level. Consequently, the taxes' impacts on the individual stocks is of the same order as their impacts on the collective stock.

Further research could include considering agents following feedback strategies. The analysis of the general model showed that the use of feedback strategies impacts on the value of what we refer to as the groundwater's 'environmental discount rate' which is the sum of the discount rate, the discharge rates and the manifestation of the strategic interactions with the other players.

An extension of this work could include a refinement of this model to provide a more precise model of irrigation-induced salinity. First, this could imply using more control variables, such as irrigation technology choice or the amount of water used for leaching, rather than irrigation, purposes. Second, this could consist in coupling adaptations of this model and the one developped in Chapter 7, in order to account for the fact that capillarity is activated when the watertable is within 3 meters of the surface. Indeed, in this Chapter, it was assumed that the study was set in the context of a salinity-affected area, where irrigation salinity mechanisms are already in place, so that both capillary rise and percolation happen at the same time. An alternative modeling strategy could consist in considering two 'root zone salinity regimes', the first one corresponding to a situation with percolation only and the second one describing what happens when capillarity is activated. The associated maximisation problem would then consist in :

$$\max_{u_i} \int_0^\infty e^{-rt} F(u_i, X, Q_i) \text{ , subject to :}$$

$$\dot{X} = \begin{cases} g_1(u_i, X) & \text{for } t < T \\ g_2(u_i, X) & \text{for } t > T \end{cases}$$

$$\dot{Q}_i = \begin{cases} f_1(u_i, Q_i) & \text{for } t < T \\ f_2(u_i, Q_i, X) & \text{for } t > T \end{cases}$$

where T stands for the time at which the root zone regime switch. The determination of T is directly linked to the state of the aquifer, as it is when it is within 3 to 2 meters of the surface that capillary rises start[8]. When $t < T$, the two stocks are isolated, and the resolution method is similar to the one used for Model 1 ; when $t > T$ it amounts to Model 2. The regulator's problem could then be approached as an optimal timing one, as in Farzin (Farzin 1996). A question could be the determination of the optimal time until the second root zone salinity regime is attained. Then the regulator's problem would be the following :

$$\max_{u_i, T} \int_0^T e^{-rt} F(u_i, X, Q_i) + \psi(T) \text{ subject to } \dot{X} = g_1(u_i, X) \text{ and } \dot{Q}_i = f_1(u_i, Q_i)$$
$$\psi(T) = \max_{u_i} \int_T^\infty e^{-rt} F(u_i, X, Q_i) \text{ subject to } \dot{X} = g_2(u_i, X) \text{ and } \dot{Q}_i = f_2(u_i, Q_i, X).$$

Other extensions include the consideration of imperfect information settings or of properties other than efficiency , such as their budget balance property.

[8]This denotes an homogeneous view of aquifers as 'bathtubes', in consistency with most economic studies of aquifers. See Brozovic (2002) for an alternative modeling strategy.

PART III : COUPLED MARKETS AND COUPLED EXTERNALITIES

The purpose of this Part is to assess the potential of various combinations of recharge rights and diversion rights markets, also referred to as cap and trade systems, to manage water scarcity and irrigation salinity. Hypotheses 2a-c translate into the following statements : in the absence of a specific instrument to manage the recharge, a series of diversion cap and trade systems at the catchment scale performs better than a regional diversion cap and trade to manage both externalities ; and when a system of recharge cap and trade is introduced, the regional diversion rights cap and trade is *de facto* constrained within catchments. In other (general) words, when externalities are coupled, and express at various geographical scales, how many policy instruments are needed to attain efficiency ?

In a general setting, according to the Tinbergen principle (Tinbergen 1950), to each issue should correspond a policy instrument. This may not apply when the issues at stakes are linked. Among the contexts exhibiting interrelated externalities that express at various scales, greenhouse gas emissions have recently received increasing attention. Within this context, Caplan and Silva (2005) and Yang (2006) address the interactions between global and local pollutants. Yang (2006) shows how the negative relation between the global externality (global warming) and the local externality (SO2 emissions) impacts on the design of policy instruments, as it raises the need for a subsidy for local externalities when accounting for the global issue. Caplan and Silva (2005) analyze pollution rights markets to manage carbon, a global pollutant, and smog, a local pollutant, positively correlated and emitted jointly by a set of firms. They demonstrate how a system combining a global market and a set of regional markets is Pareto efficient. Both studies highlight the necessity to account for the interactions arising between policy instruments implemented at various scales.

In the context of water management, as discussed in Chapter 4, Weinberg et al. (1993) and Dinar and Letey (1991) show how the introduction of a market for water rights has a negative impact on the generation of drainage, as a result of a general increase of water use efficiency. Hence managing water scarcity has an indirect impact on the management of drainage-related problems. However these studies don't capture the spatial problem characteristic of river systems. Another strand of literature has focused on the efficient spatial allocation of irrigation water accounting for conveyance issues (Chakravorty and Roumasset 1991) (Chakravorty, Hochman and Zilberman 1995) (Chakravorty and Umetsu 1998) (Chakravorty and Umetsu 2003). Chakravorty and Roumasset (1991) analyse the impact of conveyance losses on the optimal allocation rule ; they show that the quantity of water applied should fall with distance from the source, while the effective price should increase. Chakravorty et al. (1995) extend the previous analysis to endogenize distribution losses by determining optimal conveyance investment and on-farm conservation technology. They show that under optimal conveyance investment, water use is decreased, and investment in on-farm technology is increased. They also argue that the introduction of water markets without associated conveyance upgrade leads to reduced water rents in favors of agents located near the water source. Chakravorty and Umetsu (1998) and Chakravorty and Umetsu (2003) further extend the analysis to explicitely account for return flows and their impact on groundwater recharge. In particular, Chakravorty and Umetsu (2003) shows that accounting for return flows introduces a specialisation of production over space, as upstream agents use surface water while downstream agents rely on groundwater. However, in these studies, the spatial models consider agents located along a straight line from the source, who produce return flows that recharge a unique aquifer. The model used in this Part of the thesis accommodates multiple aquifers, linked through a surface water course, by extending Weber's (2001) framework so that each location along the river corresponds to a catchment, underlying an independant aquifer.

Chapter 9 presents the modeling framework and describes how the externalities are accounted for. Chapter 10 addresses the 'basic' problems : the regulator's program and individual agents' program subject to various sets of constraints. Then Chapter 11 focuses on cap and trade mechanisms as a way to manage environmental issues. It also provides some concluding remarks.

Chapitre 9

Water management in the context of irrigation-induced salinity : modeling framework

This Chapter details the modeling framework used to test Hypothesis 2. The model is developed in an idealistic framework, however it still captures the hydrologic and economic mechanisms at stake. Its main features are as follows.

First, the study is set in the context of irrigation districts. Water is provided to the irrigators by way of irrigation channels, so that they do not pump directly from the river. Hence there is a unique uptake point for each hydrologic zone (or irrigation district). It is also assumed that water is not pumped from the watertable, in reason of its existing very high salt content. Second, only consumptive users of water are considered, more precisely irrigators. Hence it is assumed that non consumptive users' interests are taken care of by the regulator by setting environmental constraints (Weber 2001). Third, the dynamics of groundwater accumulation are not specified. Aquifer management is accommodated through the setting of aggregate recharge targets that ensure that at each relevant time step, the aquifer is at the hydrological equilibrium. However, a spatially dynamic description of river flows is provided. Finally, the relations between irrigation water, percolation and discharge are assumed to be known with sufficient certainty[1].

Section 9.1 details the hydrologic and economic parts of the model. Then Section 9.2 introduces how concerns for the environment are accounted for.

[1]Refer to Chapter 6.

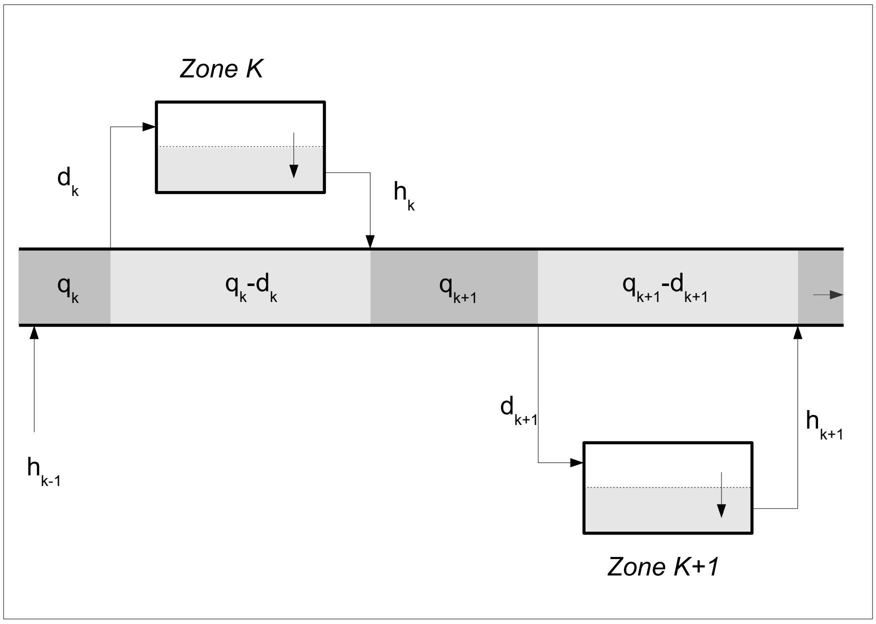

FIG. 9.1 – Hydrologic model.

9.1 Definitions and assumptions

Hydrologic component. Figure 9.1 illustrates the hydrologic component of the model. It is illustrative of a fully regulated river : a quantity q_0 of water is released from a dam located upstream, irrigation areas are then provided with irrigation water diverted at identified uptake points along the river. The analysis abstracts from the uncertainty of supply[2] and from any conveyance losses[3]. Between these uptake points, water uses are non-consumptive. Instream-users' interests are assumed to be accommodated by the regulator in defining and implementing a constraint on instream flows[4]. An aggregate quantity of water d_k is diverted from the river at one uptake point for each zone k, and an amount h_k of return flows goes back to the river from the underground system at zone k's outset point. Water available for diversion at point $k+1$ is described by the following equation :

$$q_{k+1} = q_k - d_k + h_k. \tag{9.1}$$

The assumption underlying these formulations is that only the actions undertaken at diversion point k have an impact along the segment $[k, k+1]$ of the river.

[2]Freebairn and Quiggin (2006) provide an analysis of different systems of property rights when variability of supply is important.

[3]Refer, for instance, to Chakravorty et al. (1995).

[4]See Griffin and Hsu (1993) for a model with consideration of instream users.

Economic component. In each of the m zones, n_k agents denoted by $i \in [1..n_k]$ undertake irrigation. Agent i's utility function is :

$$\pi_{ik}(u_{ik}, a_{ik}) = \rho^p f_{ik}(u_{ik}) - \rho^e u_{ik} - \frac{D_{ik}}{2} a_{ik}^2 - \varepsilon_k \sum_{i=1}^{n_k} p_{ik}(u_{ik}, a_{ik}),$$

where u_{ik} is the quantity of water applied for irrigation, a_{ik} denote abatement decisions, $f_{ik}(u_{ik})$ is the production function and $p_{ik}(u_{ik})$ the percolation function :

$$f_{ik}(u_{ik}) = A_{ik} + B_{ik} u_{ik} - \frac{C_{ik}}{2} u_{ik}^2,$$
$$p_{ik}(u_{ik}, a_{ik}) = \alpha_k u_{ik} - \delta_k a_{ik}.$$

α_k is a percolation rate, inversely related to the efficiency of irrigation technology supposed fixed for an agent, with $0 \le \alpha_k \le 1$. δ_k is an index of the efficiency of abatement actions, with $0 \le \delta_k \le 1$. The following relations apply :

$$d_k = \sum_{i=1}^{n_k} u_{ik} \quad \text{and} \quad h_k = \sigma_k \sum_{i=1}^{n_k} p_{ik}(u_{ik}, a_{ik}),$$

where σ_k is a return-flow parameter, $0 \le \sigma_k \le 1$. Abatement actions are costly to the irrigators, and do not provide any benefits apart from reduced percolation. ε_k is an individual damage term associated with irrigation-induced salinity ; it is the resulting effect of aggregate percolation in zone k. It is a translation of soil salinization and waterlogging in a static context. Parameters α_k, δ_k and ε_k are catchment-specific. Their respective values depend on pedological characteristics, which are considered more homogeneous among, than between, catchments. ρ^p and ρ^e are price terms of production and water.

9.2 Accounting for environmental concerns

The context of irrigation-induced salinity is complex, encompassing inter-relating issues on the quality and the quantity of the resource water, in both the surface and the underground systems. Consequently, environmental concerns in this context are numerous. Three main issues are addressed : water scarcity, rising watertable and instream salinity.

9.2.1 Social environmental flows constraint

The first objective of the regulator is to manage water scarcity, by guaranteeing a minimum level of instream flows at each point along the river, in order to satisfy the needs of the environment and of a range of other non -consumptive users. These uses include the provision of habitat for freshwater species, the maintenance of banks' soil-moisture for riparian vegetation, the maintenance of the appropriate balance between fresh and salty water in estuaries, aesthetic features, and recreational and cultural values (World Bank

2003). The definition given by the World Bank for environmental flows is the 'water left in a river ecosystem, or released into it, for the specific purpose of managing the condition of that ecosystem'(World Bank 2003, p.11). Various constraint formulations can be imagined in order to reflect the concern for environmental flows[5].

A unique constraint at the river mouth. Since the development of irrigation, the periods of low flows, which are a natural pattern of the Murray River, have become more frequent. This led, in 1981, to the temporary closure of the river mouth, which has been severely silted since then. This has been shown to have a detrimental impact on the Ramsar-listed Coorong area[6], located between the mouth and a series of barrages upstream, dedicated to the provision of water to the Lower Murray. Among the options contemplated to manage this issue is the securing of 2000 ML/day flowing over the barrages (Murray-Darling Basin Commission 2002). Consequently a first formulation of the regulator's concern for instream flows is the definition of a unique constraint requiring a minimum amount of flow to reach the river mouth[7]. When the river under study is a tributary, then this limit may reflect an agreement that has been signed between two jurisdictions. An Australian example is provided by the recommendation that the Cap be implemented as 'end-of-valley flow regimes' (Murray-Darling Basin Ministerial Council 1996).

Consequently, a social objective in terms of environmental flows could be that a minimum flow $\bar{Q}_m$ reaches the river termination point,

$$q_m = q_0 + \sum_{k=1}^{m-1} [h_k - d_k] \geq \bar{Q}_m.$$

This formulation imposes a greater burden on downstream catchments. Also, it doesn't guarantee that the flows will be sufficient at each point along the river. Next two ways of sharing the burden between the catchments are defined that ensure that environmental flows are supported along the river.

A constant constraint along the river. A second possible formulation is the requirement that a minimum flow $\bar{Q}$ remains in the river after each catchment's uptake point, ensuring that along the river a minimum flow is secured. Indeed, considering the structure of the hydrological model, the portion of the river between a zone's uptake and outset points is the most vulnerable with respect to flows; so that ensuring the constraint is met in this portion of the river is sufficient to ensure that it is met at each point along the

[5] As Coram (undated, p.11) puts it 'there is no single problem of optimal allocation, but a number of different problems that depend on the sort of constraints a community wishes to put on water allocation'.

[6] The Ramsar Convention is an international treaty for the conservation and sustainable utilization of wetlands, that is to stem the progressive encroachment on and loss of wetlands now and in the future, recognizing the fundamental ecological functions of wetlands and their economic, cultural, scientific, and recreational value.

[7] In a static setting, this means considering a unique value; extension to a dynamic setting would induce a limit evolving over time.

river. The fact that $\bar{Q}$ is the same for every catchment denotes a homogeneous view of the river : no particularly important ecological zones have been identified. Alternatively, this may denote a high commitment to environmental flows, so that the most constraining area imposes its limit to the whole river system :

$$q_k - d_k \geq \bar{Q}, \forall k.$$

The homogeneity of the constraint allows deriving some insights on the interaction between adjacent catchments, after some simple manipulations. Indeed, when the flow constraint binds in zone k, the following applies :

$$\begin{aligned} q_k - d_k &= \bar{Q}, \\ q_{k+1} &= q_k - d_k + h_k = \bar{Q} + h_k, \\ q_{k+1} - d_{k+1} &= \bar{Q} + h_k - d_{k+1} \geq \bar{Q}, \\ h_k - d_{k+1} &\geq 0. \end{aligned}$$

Water diverted in zone $k+1$ is less than the return flows from the upstream zone. When the flow constraint binds in zone $k+1$, then :

$$\begin{aligned} q_{k+1} - d_{k+1} &= \bar{Q} = q_k - d_k + h_k - d_{k+1}, \\ q_k - d_k &\geq \bar{Q}, \\ d_{k+1} &\geq h_k. \end{aligned}$$

The flow constraint binds in zone $k+1$ when diversions exceed the return flows from zone k. When both constraints bind, then $h_k = d_{k+1}$. Consequently, such a formulation of the environmental flow constraint explicitly introduces a dependency between the return flow from a zone and the water divertible by the adjacent downstream zone.

Constraints differentiated by catchment. The last formulation is a refinement of the previous one, as it introduces heterogeneity between the riparian zones : the minimum instream flow to be sustained differs from zone to zone. Such a management strategy may be explained by the fact that key ecological assets have been identified in certain areas that require more water than the others. This is illustrated by the MDBC's Living Murray Initiative (Murray-Darling Basin Commission 2007*a*). As of 2006, 6 key ecological assets have been identified, including the Murray Mouth and the Barmah-Willema forest. For each, there is a management plan that specifies the requirement of a minimum amount of flows[8] :

$$q_k - d_k \geq \bar{Q}_k, \forall k. \tag{9.2}$$

[8]This may also result from a negotiation process in each of the catchment, within stakeholders, and with the government, which resulted in different choices of instream flow constraints.

The same type of relation between return flows and diversion arise with this formulation : if the constraint binds in both k and $k+1$, then the sign of $d_{k+1} - h_k$ is the same as the sign of $\bar{Q}_k - \bar{Q}_{k-1}$. If the constraint is binding only in one of the zones, then there may be indetermination. However, in all cases, the relation between the water diverted and the return flows from the adjacent upstream catchment depends on the relative value of the flow targets in each of the zones.

Whatever the formulation of the environmental flows constraint, the m catchments are held collectively responsible for its management. While the first formulation imposes a greater burden on downstream catchments, the two other formulations are a way of sharing this burden. Note that with the homogeneous constraints, assuming that the water use in all the catchments is of the same extent, then downstream catchments are also subject to a harsher regulation, as there are no tributaries[9]. In the remainder of this thesis, only the third formulation, differentiated caps by catchment, is considered. Introducing cap and trade systems for water diversion will turn these constraints on the state of the river into constraints on water extractions.

9.2.2 Social recharge constraint

Another objective of the regulator is to maintain the level of the watertable below a critical point, above which salinization occurs. For this purpose, in this static framework, it is assumed that the regulator has set a limit on aggregate recharge to ensure that the aquifer is in the conditions of a satisfying equilibrium. It is accepted that a depth of three meters is the limit above which salinization mechanisms are enhanced (Whitten et al. 2003) ; monitoring watertable levels is then a way of identifying which zones are the most at risk from salinity. In what follows, it is assumed that the regulator has access to sufficient information to derive the aggregate recharge that ensures that the watertable remains stable or that its level doesn't exceed three meters-depth (Whitten et al. 2003) (Whitten et al. 2004). Consequently, the regulator seeks to enforce the following recharge constraint in each of the zones :

$$\sum_{i=1}^{n_k} p_{ik}(u_{ik}, a_{ik}) \leq \bar{R}_k, \forall k. \tag{9.3}$$

The constraint is zone-specific, in reason of the nature of hydrologic mechanisms. Indeed, watertables are considered disconnected from a zone to the other, so that recharge management is relevant at the catchment scale.

It is assumed that both constraints are optimally set by the regulator, in order to deal with values that are not captured by the model. If the environmental flow constraint captures non-consumptive values, the recharge constraint captures values which are inherently dy-

[9]Furthermore the analysis abstracts from conveyance losses.

namic and as such cannot be described by this model. In particular, this constraint allows taking account of the dynamic externalities arising between the irrigators[10].

9.2.3 Stream salinity damage

Besides these constraints, the regulator also accounts for the qualitative impact of water discharged into the river. Indeed, return flows have an ambiguous effect on the environment. In quantitative terms, they generate positive externalities by increasing river flows. In qualitative terms, however, they contribute to increasing salt concentration in river flows. Stream salinity causes various types of damage : to the environment, to irrigation activities and to infrastructure. Damage from instream salinity along the segment $[k, k+1]$ is expressed by : $\Gamma_k h_k$, Γ_k being the marginal damage from salts contained in return flows from zone k. It is not incorporated as a constraint set by the regulator. Instead, it is integrated in the regulator's program as a term of damage. This choice is consistent with the management of stream salinity in the MDB, where salt interception schemes are managed by the States (Murray-Darling Basin Commission 2007*b*). The marginal damage Γ_k reflects the cost of operating these salt interception schemes. Hence the cost of stream salinity from each catchment is directly proportional to the amount of water discharged into the river. While not entirely satifactory[11], this formulation has the advantage of highlighting the tradeoff that the regulator has to account for when managing recharge and surface water scarcity at the same time.

9.2.4 District flow constraint

Besides these constraints that the regulator will seek to enforce, there exists a type of irrigation community awareness of the need for minimum flows (Tisdell, Ward and Capon 2004). A simple interpretation of that is the need for a minimum level of instream flows to ensure the operations of water transmission to the irrigation districts. Hence, this district constraint could be approached as a physical constraint, the extreme case being that diversions for use in zone k should not induce the river to run dry. Consequently, in each catchment a minimum flow constraint can be enforced :

$$q_k - d_k \geq \underline{Q}_k. \tag{9.4}$$

The enforcement of this constraint, and its implication, will be analyzed in more details in Chapter 10.

9.3 Conclusion

Set in a static framework, the model used to test Hypothesis 2 captures the main mechanisms at stake in the management of surface and ground water resources in the context of

[10] Externalities that have been addressed in Part II.

[11] It doesn't account for the assimilative capacity of the river.

irrigation-induced salinity. In particular, it highlights a tradeoff with respect to the management of instream flows : they have both positive and negative impacts on the system, depending on the focus being placed on their qualitative or quantitative features.

The setting of the regulator's constraints is outside the scope of this thesis. Indeed, in the remainder of the analysis, it is assumed that the constraints are exogenously set. The definition of the constraints involves an estimation of the non-market values attached to, for instance, instream flows. Among the techniques currently used for this purpose, Morrison and Bennett (2004) and Bennett (2005) provide an illustrations of the use of Choice Modelling[12] to determine the demand for environmental flows for Australian rivers.

Absent from the formulation of the model are issues of stochasticity of the mechanisms and heterogeneity of the agents within a district. As in the previous Part, the model abstract from conjunctural and structural sources of stochasticity in terms of precipitations. Recourse to an expected maximization framework would amount to introducing the former, while an analysis set in the framework of robust control would be a way to integrate the latter. However, the static framework adopted in this Part isn't convenient to capture issues associated with long term changes in climate. Nevertheless, the conjunctural adaptation of individual irrigators to stochastic variations in precipitations once the diversion and recharge caps have been set could be addressed within this framework through the analysis of contingent rights, that allow a right-holder a share of an aggregate volume available (Freebairn and Quiggin 2006). The impact of the assumption of homogeneity of most individual parameters within districts will be discussed in the following chapters.

According to the maximization problem under study, the irrigators will be subject to various sets of constraints on water extraction and use. All the constraints that have been defined have the peculiarity that they concern a set of irrigators, those pertaining to a district, whose aggregate actions condition the respect or not of the constraint. These are coined coupling constraints (Krawczyk 2005). The next Chapter presents the resolution of the basic problems, the regulator's and the individual agents', subject to two different sets of constraints : the district flow constraint only and the regulator's constraints.

[12] Choice Modelling is a stated preference technique in which respondents choose their most preferred resource use option from a number of alternatives. Refer to Bennett and Blamey (2001) for more on this topic.

Chapitre 10

The basic programs : social optimum and individual choices

This Chapter presents the analyses of the benchmark cases constituted by the regulator's problem and an individual agent's problem. They differ with respect to the type of environmental constraints accounted for, and by the fact that the regulator operates a maximization of aggregate benefits. Social costs are identified that individual agents don't account for. Also, the resolution of the individual problems is done in the framework laid by Rosen (1965) to deal with coupling constraints. This methodology is based on two notions : a common burden if the constraint is violated and individual weights as a way of sharing this burden. The results derived in this Chapter highlight the necessity to implement a policy instrument to manage irrigation-induced salinity. Cap and trade systems will be the focus of Chapter 11.

10.1 The regulator's problem

The regulator maximises the social welfare with respect to the quantity of water applied and the abatement decisions, accounting for the social concerns described in Chapter 9 :

$$\max_{u_{ik},a_{ik}} \sum_{k=1}^{m} \left[\sum_{i=1}^{n_k} \pi_{ik}(u_{ik}, a_{ik}) - \Gamma_k h_k \right], \text{ subject to :}$$

$$q_{k+1} = q_k - d_k + h_k, \tag{9.1}$$

$$q_k - d_k \geq \bar{Q}_k, \forall k, \tag{9.2}$$

$$\sum_{i=1}^{n_k} p_{ik}(u_{ik}, a_{ik}) \leq \bar{R}_k, \forall k, \tag{9.3}$$

and a set of initial conditions.

10.1.1 The socially optimal allocation of water use and abatement levels

This is a general constrained control problem, with two controls u_{ik} and a_{ik} and a state variable q_k which spatial evolution is given in equation (9.1). The Lagrangian is :

$$L^*(u_{ik}, a_{ik}, \lambda_k, \mu_{1k}, \mu_{2k}) = \sum_{k=1}^{m}\sum_{i=1}^{n_k} \pi_{ik}(u_{ik}, a_{ik}) - \sum_{k=1}^{m} \Gamma_k h_k$$
$$+\sum_{k=1}^{m} \lambda_k[q_k - d_k + h_k - q_{k+1}] + \sum_{k=1}^{m} \mu_{1k}[q_k - d_k - \bar{Q}] + \sum_{k=1}^{m} \mu_{2k}[\bar{R}_k - \sum_{i=1}^{n_k} p_{ik}(u_{ik}, a_{ik})],$$

where λ_k is the costate variable, associated to the flow equation, μ_{1k} and μ_{2k} are the shadow costs associated with the regulatory constraints. The first order conditions for an interior solution are :

$$\frac{\partial L^*}{\partial u_{ik}} = \rho^p(B_{ik} - C_{ik}u_{ik}) - \rho^e - \alpha_k\varepsilon_k - \lambda_k - \mu_{1k} - \alpha_k\mu_{2k} + \alpha_k\sigma_k[\lambda_k - \Gamma_k] = 0, \tag{10.1}$$

$$\frac{\partial L^*}{\partial a_{ik}} = -D_{ik}a_{ik} + \delta_{ik}\varepsilon_k + \delta_{ik}\mu_{2k} - \delta_k\sigma_k[\lambda_k - \Gamma_k] = 0, \tag{10.2}$$

$$\frac{\partial L^*}{\partial q_k} = \lambda_k - \lambda_{k-1} + \mu_{1k} = 0,$$

$$q_k - d_k - \bar{Q} \geq 0 \ , \ \mu_{1k}[q_k - d_k - \bar{Q}] = 0,$$

$$\bar{R}_k - \sum_i p_{ik}(u_{ik}, a_{ik}) \geq 0 \ , \ \mu_{2k}[\bar{R}_k - \sum_i p_{ik}(u_{ik}, a_{ik})] = 0.$$

Rearranging the expressions :

$$\lambda_k = \frac{1}{1-\sigma_k\alpha_k}[\rho^p(B_{ik} - C_{ik}u_{ik}) - \rho^e - \mu_{1k}] - \frac{\alpha_k}{1-\sigma_k\alpha_k}[\varepsilon_k + \mu_{2k} + \sigma_k\Gamma_k], \tag{10.3}$$

$$\mu_{2k} = \frac{D_{ik}}{\delta_k}a_{ik} + \sigma_k[\lambda_k - \Gamma_k] - \varepsilon_k, \tag{10.4}$$

$$\lambda_k - \lambda_{k-1} = -\mu_{1k}. \tag{10.5}$$

Equation (10.3) describes the cost λ_k of reducing the water flow between zones k and $k+1$ by one unit. At the equilibrium, this cost equals the marginal benefit of allocating a extra unit of water to agent ik. It is expressed according to the the consumptive / percolating nature of water use. The first bracketed term on the RHS of equation (10.3) is the net benefit of consuming a extra unit of water for agent i : the marginal benefit of consuming a extra unit of water, minus the extra cost of meeting the environmental flows constraint in zone k by diverting water. The coefficient $1/(1-\sigma_k\alpha_k)$ renders this net benefit per unit of water consumed. The second bracketed term of the RHS of equation (10.3) is the marginal cost of percolating one unit of water : damage to user ik, extra cost of meeting the recharge constraint and damage of an increased stream salinity downstream. The coefficient $\alpha_k/(1-\sigma_k\alpha_k)$ renders this net cost per unit of water percolated.

Equation (10.4) shows the cost of meeting the recharge constraint in each zone. It is equal to the abatement cost, plus the cost of reducing the flow downstream, plus the benefits accruing from avoided damage : damage from waterlogging and downstream instream salinity damage.

Equation (10.5) illustrates the path of the cost of reducing water to downstream users. It depends on k, and not i, due to the structure of the model, with one uptake point for an irrigation area, rather than individual riparian diverters. As shown in equation (10.5), a shift of water diversion from $k-1$ to $k+1$ reduces the environmental flow constraint for zone k. As instream users are not considered in this model, this reduction of the environmental flow constraint is the only benefit accruing from changing the location of diversion. As the cost of meeting the flow constraint in zone k, μ_{1k}, is positive, $\triangle\lambda_k < 0$. Hence, as water goes downstream, less agents are affected by individual decisions regarding diversion or abatement (Weber 2001).

Equations (10.3)-(10.4)-(10.5) can be rearranged in order to show the optimal irrigation and abatement choices :

$$u^*_{ik} = \frac{1}{\rho^p C_{ik}}\left[\rho^p B_{ik} - \rho^e - \lambda_k - \mu_{1k} - \mu_{2k}\alpha_k - \alpha_k\varepsilon_k + \sigma_k\alpha_k(\lambda_k - \Gamma_k)\right], \tag{10.6}$$

$$a^*_{ik} = \frac{\delta_k}{D_{ik}}\left[\varepsilon_k + \mu_{2k} - \sigma_k(\lambda_k - \Gamma_k)\right]. \tag{10.7}$$

The optimal water use level is positively correlated to the marginal benefits from production, and negatively correlated to the shadow costs associated with the constraints, and to the individual damage. However, the impact of increased discharge on the level of optimal water use illustrates the tradeoff between the qualitative and quantitative features of discharged water. Indeed, if $(\lambda_k - \Gamma_k)$ is positive, so that the benefits from increasing the quantity exceeds the cost due to the decrease in quality, then the incentive is to use more water. This tradeoff also appears with the choice of the level of abatement, as when there is a benefit from the discharge, the incentive is to abate less. Equation (10.7) is analogous to a type of sharing rule of the social damage from abating, according to the individual value of D_{ik}.

Before addressing the question of whether individual agents subject to various sets of coupling constraints can be induced to undertake the optimal levels of irrigation and abatement, as derived in equations (10.6) and (10.7), the impact of the parameters' value on the binding nature of the constraints is further investigated.

10.1.2 Shadow cost analysis

The regulator's problem is a constrained control optimal problem. Consequently, most of the conclusions derived from its resolution will depend on the binding nature of the constraints. In this Section, the status of the environmental flow and recharge constraints in each of the zones is investigated in more details. For this purpose, the shadow costs associated with each constraint, and their impact on the value of the optimal water use and abatement levels, are analysed.

None of the constraints binds : $\mu_{1k} = \mu_{2k} = 0$. In this case, both shadow costs are set to zero, and the resulting irrigation water use and abatement effort levels are as follows :

$$u_{ik}^1 = \frac{1}{\rho_p C_{ik}} \left[\rho_p B_{ik} - \rho_E - \alpha_k \varepsilon_k + \sigma_k \alpha_k (\lambda_k - \Gamma_k)\right],$$
$$a_{ik}^1 = \frac{\delta_k}{D_{ik}} \left[\varepsilon_k - \sigma_k (\lambda_k - \Gamma_k)\right].$$

Furthermore, as $\mu_{1k} = 0$, $\lambda_k = \lambda_{k-1}$ from equation (10.5). The cost of shifting diversion from uptake point $k-1$ to $k+1$ is null as the environmental flow constraint does not bind in zone k.

Only the flow constraint binds : $\mu_{1k} > 0, \mu_{2k} = 0$. This illustrates the case of a catchment where instream flows are scarce, or where the environmental flow constraint is high - a zone of high ecological value for instance. Also, the recharge constraint is not binding, for instance because the catchment doesn't have a history of irrigation, so that it is not prone to salinity. In this case, the following relations hold :

$$\mu_{1k} > 0 \Rightarrow q_k - \sum_i u_{ik} = \bar{Q}_k,$$
$$a_{ik}^3 = a_{ik}^1 \text{ and } u_{ik}^3 = \frac{q_k - \bar{Q}_k}{n_k} < u_{ik}^1.$$

Quite intuitively, when agents are restricted with respect to diversions, they make use of less irrigation water, $u_{ik}^3 < u_{ik}^1$. However they don't change their abatement choices compared to the previous case $a_{ik}^3 = a_{ik}^1$.

Only the recharge constraint binds : $\mu_{1k} = 0, \mu_{2k} > 0$. This illustrates the case of a catchment highly prone to irrigation salinity, and where water resources are not scarce. Consider the case of a catchment with a long history of irrigation, located near the dam governing flows in the river system. When the recharge constraint binds :

$$\mu_{2k} > 0 \Rightarrow \sum_i p_{ik}(u_{ik}, a_{ik}) = \bar{R}_k,$$

and u_{ik} and a_{ik} are set at u^2_{ik} and a^2_{ik} (Refer to Appendix C-1 for the analytical value of u^2_{ik} and a^2_{ik}).

Both constraints bind : $\mu_{1k} > 0, \mu_{2k} > 0$. This is the extreme case where in the same catchment both diversion and recharge are constraining. This is illustrative of most catchments in the MDB.

$$\mu_{1k} > 0 \Rightarrow q_k - \sum_i (u_{ik}, a_{ik}) = \bar{Q}_k \Rightarrow u^4_{ik} = u^3_{ik},$$

$$\mu_{2k} > 0 \Rightarrow \sum_i p_{ik}(u_{ik}, a_{ik}) = \bar{R}_k \Rightarrow a^4_{ik} = \frac{\alpha_k(q_k - \bar{Q}_k) - \bar{R}_k}{\delta_k n_k}.$$

To sum up, the irrigation water and abatement levels compare as follows :

$$a^1_{ik} = a^3_{ik} < a^2_{ik} = a^4_{ik}, \tag{10.8}$$

$$u^1_{ik} > u^3_{ik} \lesseqgtr u^2_{ik} > u^4_{ik}. \tag{10.9}$$

Equation (10.8) simply states that abatement levels increase when the recharge constraint binds. Equation (10.9) states a less straightforward relation between the environmental flow constraint and the level of irrigation water used. Indeed, irrigation levels are lower when both constraints bind, and higher when nonebinds. However, the sign of the difference between u^3_{ik} and u^2_{ik} depends on the intensity of the binding of each constraint :

$$u^2_{ik} - u^3_{ik} = \frac{\mu_{1k} - \alpha_k \mu_{2k}}{\rho_p C_{ik}}.$$

A highly binding flow constraint translates into $u^2_{ik} > u^3_{ik}$, while a very high recharge constraint induces $u^3_{ik} > u^2_{ik}$.

In this Section the socially optimal level of abatement and irrigation water use were derived, highlighting a number of values that the regulator should induce the individual agents to account for in their decision-making problems. There exists a cost within each catchment of diverting water from the irrigation channel, as it renders the flow constraint more binding for this specific catchment. There exists a cost within each catchment of accumulating water underground. There exists a cost of reducing the amount of water available to downstream users. Indeed, there is a dynamic of water flowing from upstream catchments to downstream catchments, so that any action undertaken will have an impact on the agents located downstream. Hence there is a need to replace each individual catchment in the broader context of the river system : the overall state of the river depends on the bulk of individual catchments.

10.2 Individual agent's problem

Either subject to the district constraint on instream flows or to the combination of the two optimal constraints, individual agents face coupling constraints : there is a joint constraint on the combined strategy space of all the agents pertaining to the same zone. Coupling constraint problems have been analyzed in a number of settings - the concept proves very useful to study networks, applied to electricity production (Contreras, Klusch and Krawczyk 2004) or Internet transmission (Azouzi and Altman 2001). In the environmental context, it is common that standards or caps are enforced that concern the contributions of numerous agents. Then the concept of coupling constraint proves useful in understanding the issues associated with the multiplicity of equilibria that may arise when a group of agents is subject to a unique global constraint. In Section 10.2.1, the notion of coupling constraint is reviewed, and the concept of Normalized Nash equilibrium, introduced by Rosen in his seminal paper (Rosen 1965), is illustrated. Then Section 10.2.2 provides the analysis of the case of irrigators facing a district constraint on instream flows and illustrates Rosen's approach to induce the agents to reach an equilibrium. Finally, Section 10.2.3 addresses the enforcement of the regulator's constraints as a policy tool and discusses Rosen's approach with respect to its implementation.

10.2.1 Coupling constraints and environmental issues

Coupling constraints problems refer to settings where the set of options available to an agent depends on the other agent's choices. Hence, the strategy spaces of the agents are not independent. One of the difficulties associated with this particular type of game is the lack of uniqueness of the solution in a general context. Indeed, if all agents act simultaneously, no traditional solution is available (Krawczyk 2005). A number of operational research studies have been published that tackle coupling constraints problems (Krawczyk 2005). Rosen (1965) is the most appealing in the economics area as he provides a criterion that guarantees the uniqueness of the solution. The Normalized Nash equilibrium relies on the definition of a Lagrangian multiplier, common to all the agents, and individual weights that allow discriminatory treatments among players.

Borrowing from Tidball and Zaccour's (2005) approach, consider a two-player pollution control game. Each player seeks to maximise the difference between the net revenue from production and a damage cost due to aggregate pollution. Let $f_i(e_i)$ be a twice differentiable concave net revenue function and $d_i(e_1 + e_2)$ the convex damage cost function, with $i = 1, 2$. Player i's welfare function is given by : $w_i(e_1, e_2) = f_i(e_i) - d_i(e_1 + e_2)$. Let E_i be exogenous upper bounds on emissions. In the cooperative case, the two players jointly maximise their welfares :

$$\max_{e_1,e_2} \sum_{i=1}^{2} w_i(e_1, e_2) \text{ subject to : } e_1 + e_2 \leq E_1 + E_2.$$

Then the Lagrangian and first order conditions write :

$$L_i^C(e_i, \lambda^C) = \sum_i w_i(e_1, e_2) + \lambda^C[E_1 + E_2 - e_1 - e_2],$$
$$f_i'(e_i) - d_1'(e_1 + e_2) - d_2'(e_1 + e_2) - \lambda^C = 0.$$

The maximisation program in the non-cooperative case if agents are subject to a common constraint is :

$$\max_{e_i} w_i(e_1, e_2) \text{ subject to : } e_1 + e_2 \leq E_1 + E_2.$$

The Lagrangian associated to the standard Nash equilibrium is :

$$L_i^N(e_i, \lambda^i) = f_i(e_i) - d_i(e_1 + e_2) + \lambda^i[E_1 + E_2 - e_1 - e_2].$$

The multiple equilibria problem is apparent here : first, in the absence of coordination, each agent will choose a λ^i ; second, any combination of e_1+e_2 equal to E_1+E_2 renders the agents in compliance with the constraint. Now consider the Normalized Nash Equilibrium, characterised by the following Lagrangian :

$$L_i^R(e_i, \lambda^R, r_i) = f_i(e_i) - d_i(e_1 + e_2) + \frac{\lambda^R}{r_i}[E_1 + E_2 - e_1 - e_2],$$

where λ^R is the Lagrangian multiplier common to all the agents, and r_i is the (positive) weight assigned to player i. Recourse to this notion ensures that the problem of multiple equilibria is overcome for any given set of weights [1].

Applications of this concept to environmental settings include (Krawczyk 2005) (Haurie and Krawczyk 1997) (Tidball and Zaccour 2005) (Haurie, Moresino, Vielle and Viguier 2005). Most apply to the contexts of nonpoint (or point) source pollution where it is common that the regulator defines a pollution standard that numerous agents have to comply with : in a general setting (Tidball and Zaccour 2005), in the case of effluent emitters located along a river (Haurie and Krawczyk 1997) (Krawczyk 2005). Haurie et al. (2005) consider international climate change policy setting as a non-cooperative game between parties that are collectively committed to a target for total cumulative emissions.

An illustrative explanation of the concept of Normalized Nash equilibrium is that the La-

[1] Details of the assumptions that guarantee existence and unicity of the equilibrium are given below.

grangian multiplier is a tax announced by the regulator that will be applied in case the constraint is violated. In this respect, the tax will not be collected if the agent comply with the standard[2] and it only requires the observation of physical attributes rather than economic variables[3]. The weights are not usually addressed in more details than the fact that they are a way to assign more or less responsibility to the agents in the generation of the externality. Tidball and Zaccour (2005) treat them as a policy tool to induce the agents not only to comply with the constraint but also to take socially optimal decisions. They also compare them with the 'political weights' as usually understood - when the cooperative solution depends on the optimization of a weighted sum of the individual objectives (Tidball and Zaccour 2005).

In the next Section the use of the Normalized Nash Equilibrium in the resolution of a coupling constraint game is illustrated through the analysis of irrigators only subject to the district constraint on instream flows.

10.2.2 District constraint on instream flow : a 'community-based' penalty

Here, recourse to the Normalized Nash equilibrium is a way of considering a type of community-based penalty. Consider a Rosen Lagrangian multiplier associated to the district flow constraint as a penalty that will be applied to all the irrigators pertaining to the district if they aren't in compliance. In this respect, this mechanism is close to the one tested experimentally by Tisdell et al. (2004). Among various mechanisms to ensure the optimal management of a river as a common pool resource, they consider the implementation of an environmental levy to 'socialize the cost to the community of altering natural flow regime' (Tisdell et al. 2004, p.1) when diverting water for irrigation purposes. A key difference is that the penalty considered here will be applied if the constraint is violated only. Another way to grasp this notion is to envisage it as a type of peer-sanction, a phenomenon that has been observed and documented in agricultural contexts[4].

An agent's problem is expressed as follows :

$$\max_{u_{ik},a_{ik}} \pi_{ik}(u_{ik}, a_{ik}), \text{ subject to } :$$
$$q_k - d_k \geq \underline{Q}_k. \tag{9.4}$$

Existence and uniqueness of the equilibrium. It can be verified that agents' strategies u_{ik} and a_{ik} are selected from convex, closed and bounded sets. Furthermore the utility function is continuous and concave in each control. Then from Theorem 1 from

[2]In this regard, it is similar to an ambient tax.

[3]However in this respect, is differs from ambient tax.

[4]Marshall (2001) documents the case of peer pressure exerted 'at the pub'(Marshall 2001, p.24) by irrigators to induce other farmers to uptake Land and Water management plans.

Rosen (1965) this game admits an equilibrium point. Uniqueness of the equilibrium relies on the concept of diagonal strict concavity of the joint payoff function[5] (Krawczyk 2005) :

$$f(\mathbf{u}, \mathbf{a}, \mathbf{r}) = \sum_{k=1}^{m} \sum_{i=1}^{n_k} r_{ik} \pi_{ik}(u_{ik}, a_{ik})$$

The diagonal strict concavity assumption ensures that 'each player has more control over his payoff than the other players have over it' (Krawczyk 2005, p.162).

Individual water use and abatement levels. Consider the following vector of district Rosen weights, $\mathbf{r^d} > 0$, and a set of Lagrangian multipliers η_k^d associated with the district constraints. Formulating the Lagrangian :

$$L^d = \pi_{ik}(u_{ik}, a_{ik}) + \frac{\eta_k^d}{r_{ik}^d}[q_k - d_k - \underline{Q}_k],$$

the irrigation and abatement decisions are :

$$u_{ik}^d = \frac{1}{\rho^p C_{ik}}[\rho^p B_{ik} - \rho^e - \varepsilon_k \alpha_k - \frac{\eta_k^d}{r_{ik}^d}],$$
$$a_{ik}^d = \frac{\delta_k}{D_{ik}} \varepsilon_k.$$

It is straightforward to notice that the district constraint has no impact on individual agents' abatement decisions. However, as expected, it impacts negatively on the choice of irrigation level. The extent to which individual agents modify their decisions when the penalty is enforced depends on the value of the Lagrangian multiplier but also on their individual Rosen weights :

$$\frac{\partial u_{ik}^d}{\partial r_{ik}^d} = \frac{\eta_k^d}{r_{ik}^{d2} \rho^p C_{ik}} > 0$$

A higher weight allows using more water : it constitutes a concession gained by an agent to produce more of the externality (Krawczyk 2005).

A district flow constraint as a safeguard mechanism. The aim of this Section was to illustrate the concept of the Normalized Nash equilibrium in the context of the management of instream flows in a regulated system of irrigation districts. The enforcement of the district flow constraint alone can't guarantee that the optimal levels of abatement and irrigation are met ; however it constitutes a safeguard mechanism that the districts can enforce in extreme cases. The question remains as to the implementation of such a policy tool. Indeed, implicit to the analyzes of coupling constraint is the assumption that there

[5] Refer to Appendix C-2.

exists a local government with legislative powers to collect taxes in case the constraint is not respected. If the associations governing irrigation management in the districts are assumed able to do so, then this analysis *à la* Rosen is relevant. The next Section addresses the use of coupling constraints as a policy tool that the regulator uses in order to induce the agents to make optimal decisions.

10.2.3 Coupling constraints as a policy instrument : enforcement of the social constraints

This Section asesses the use of coupling regulatory constraints as a way to enforce socially optimal decisions by individual decision makers. When a district is in breach of a constraint, all the irrigators in the district are subject to a penalty. This penalty is defined as the combination of a Lagrangian multiplier, characteristic of a district, and an individual Rosen weight[6]. Note η_k and γ_k the multipliers associated with, respectively, the optimal environmental flow and recharge constraints. The vector of Rosen weights assigned by the regulators is $\mathbf{r} > 0$. An individual agent's maximization problem is now :

$$\max_{u_{ik}, a_{ik}} \pi_{ik}(u_{ik}, a_{ik}) \text{ subject to } :$$

$$q_k - d_k \geq \bar{Q}_k, \tag{9.2}$$

$$\sum_i p_{ik}(u_{ik}, a_{ik}) \leq \bar{R}_k. \tag{9.3}$$

The resolution is the same as in the previous case, consequently only the main results are provided.

Individual water use and abatement levels. After formulating the Lagrangian, the derivation of the first-order conditions leads to the following expressions of irrigation and abatement levels :

$$u_{ik}^R = \frac{1}{\rho^p C_{ik}}[\rho^p B_{ik} - \rho^e - \varepsilon_k \alpha_k - \frac{1}{r_{ik}}(\eta_k + \alpha_k \gamma_k)], \tag{10.10}$$

$$a_{ik}^R = \frac{\delta_k}{D_{ik}}[\varepsilon_k + \frac{\gamma_k}{r_{ik}}]. \tag{10.11}$$

Enforcing the recharge constraint modifies the abatement level choice. However, the extent to which the agents account for the constraints vary. The impact of the Rosen weights is given by the following relations :

$$\frac{\partial u_{ik}^R}{\partial r_{ik}} = \frac{\alpha_k \gamma_k + \eta_k}{r_{ik}^2 \rho^p C_{ik}} > 0 \quad \text{and} \quad \frac{\partial a_{ik}^R}{\partial r_{ik}} = -\frac{\gamma_k \delta_k}{D_{ik} r_{ik}^2} < 0.$$

[6]The Rosen weights are different from the ones studied in the previous Section, as they are now assigned by the regulator in charge of the management of water resources on the whole river system.

Results are consistent with the previous case : a higher weight induces a higher amount of water used and a lower level of abatement from an individual agent. Hence an agent with a higher weight has the possibility to use more water, while producing a lower abatement effort ; in other words to produce a greater amount of the environmental externality.

Comparison with the socially optimal allocation of water. Compatibility of the Normalized Nash Equilibrium with the optimal solution requires the following conditions, obtained by comparing the social optimum and the individual first order conditions :

$$\frac{\eta_k + \alpha_k \gamma_k}{r_{ik}} = \mu_{1k} + \alpha_k \mu_{2k} + \sigma_k \alpha_k [\Gamma_k - \lambda_k] + \lambda_k, \tag{10.12}$$

$$\frac{\gamma_k}{r_{ik}} = \mu_{2k} + \sigma_k [\Gamma_k - \lambda_k]. \tag{10.13}$$

Replace the value of γ_k / r_{ik} obtained from equation (10.13) in equation (10.12) :

$$\frac{\eta_k}{r_{ik}} = \mu_{1k} - \lambda_k. \tag{10.14}$$

Equation (10.13) states that to induce the agents to make socially optimal decisions, the ratio γ_k / r_{ik} has to be set at the social cost of groundwater accumulation, μ_{2k}, plus the increased cost of discharge, $\sigma_k [\Gamma_k - \lambda_k]$. Note that, if the social marginal cost of stream salinity is lower that the social marginal benefits from increased stream flow, then the latter term may be negative. Also, equation 10.14 shows that η_k / r_{ik} has to incorporate the cost of increased diversion, μ_{1k} and the cost of reduced water flowing to downstream users, λ_k.

Another result is that all the weights need to be zone-specific, rather than individually specific. Without loss of generality, set $\sum_i r_{ik} = 1$. Then it is straightforward that all the weights within zone k will be set at $1/n_k$.

10.3 Concluding remarks

This Chapter provides the analysis of the basic problems constituted by the regulator's optimal control problem and individual agents' maximization problem. The socially optimal irrigation use and abatement level derived in this Chapter will serve as benchmarks against which the subsequent results will be compared. Also, as a constrained program, it is highly dependant on the binding/non binding nature of the constraints enforced. An analysis of the associated shadow costs has provided some insights into the consequences of the degree of bindness on the resulting irrigation and abatement choices.

To solve the individual problems, this Chapter introduces the notion of coupling constraint. An analysis of the case where a district flow constraint constitutes a discussion of the notion of a community-based penalty. This will constitute a safeguard mechanism to preserve instream flows in the remainder of this analysis.

The enforcement of coupling constraints according to Rosen's methodology poses a number of questions. First, there needs to be some instance with legislative powers to enforce the constraint, to collect the tax if necessary. Related to this issue is the question of how the information, about the Lagrangian multiplier and the individual weights, is transmitted to each individual agents. In the next Chapter, a more politically soft process to enforce the regulator's constraints is analyzed, through the implementation of cap and trade mechanisms. In the process of the analysis hypotheses 2a, 2b and 2c will be tested.

Chapitre 11

Water allocation under various cap and trade systems

This Chapter addresses the optimal set of markets (nature of the rights, spatial scale, number of markets) to manage two interdependent environmental issues that can be approached at various scales.

In Chapter 10, the socially optimal levels of abatement and irrigation water use were derived. In this process a number of values that the regulator should induce the individual agents to account for in their decision-making programs were derived. Indeed, there exists a specific cost within each catchment of diverting water from the irrigation channel, as it renders the flow constraint more binding for this catchment. Also, there exists a cost within each catchment of accumulating water underground. In conjunction with this, an important trade-off faced by the regulator in the consideration of return flows was illustrated. Finally, there exists a cost of reducing the amount of water available to downstream users. Hence the need to replace each individual catchment in the broader context of the river system. Indeed a bulk of localized situations leads to the overall state of the river system.

In what follows, the socially optimum solution is decentralized by means of a series of cap and trade systems. In such a setting, the decisions of diverting water and producing percolation water are conditioned upon the holding of associated rights. The aim of this Chapter is to appraise various types of cap and trades with respect to their effectiveness (potential to induce the agents to respect both environmental constraints) and their efficiency (potential to induce the agents to respect the constraints by taking the socially optimal decisions).

The cap and trade systems and constraints under study are presented, respectively, in Sections 11.1 and 11.2. Then Section 11.3 addresses the case of zonal cap and trades. The analysis is extended to the case where barriers to trade are removed in Section 11.4.

Section 11.5 sums up the results to test hypotheses 2a-c and provides some concluding remarks.

11.1 Cap and trade systems under study

The regulator issues a total number of diversion rights, W_k or W according to the market design. Each agent benefits from an initial allocation w^0_{ik} so that $\sum_i w^0_{ik} = W_k$ when the market is zonal and $\sum_k \sum_i w^0_{ik} = W$ when the market is regional. An irrigator can increase or decrease his allocation by purchasing/selling rights, w^{jh}_{ik}, from agents located in the same catchment ($h = k$) only in the zonal case, and also in upstream ($h < k$) or downstream catchments ($h > k$) in the regional case. A positive w^{jh}_{ik} indicates a purchase by ik from jh, a negative w^{jh}_{ik} a sale to jh. For each irrigator, the aggregate number of rights after exchange is w_{ik} :

$$w_{ik} = w^0_{ik} + \sum_h \sum_j w^{jh}_{ik} \qquad \text{for a regional cap and trade,}$$
$$w_{ik} = w^0_{ik} + \sum_j w^{jk}_{ik} \qquad \text{for a zonal cap and trade.}$$

Each participant is subject to an individual compliance constraint that states that an agent will not make use of more of an input that what he owns the right to :

$$w_{ik} - u_{ik} \geq 0. \tag{11.1}$$

It can be shown that as long as prices are positive, this constraint holds with equality (Montgomery 1972). There is also a market clearing condition (MCC) that states that in equilibrium, supply equals demand. This translates into the following statement : 'when all licences are allocated to firms, [it] implies that any expenditure on licences by one firm is a revenue to another firm' (Montgomery 1972, p.402). Hence the following relations apply :

$$0 = \rho^w \sum_k \sum_i \left[w_{ik} - w^0_{ik}\right] \qquad \text{for a regional cap and trade,}$$
$$0 = \rho^w_k \sum_i \left[w_{ik} - w^0_{ik}\right] \qquad \text{for a zonal cap and trade.}$$

which implies that each purchase by an agent is balanced by a sale by another agent :

$$\exists j, h \text{ such that } \frac{\partial u_{ik}}{\partial w_{ik}} = -\frac{\partial u_{jh}}{\partial w_{jh}}.$$

The presentation is analogous in the case of recharge rights markets : g_{ik} is agent ik's recharge rights endowment after exchange, the aggregate allocation on zone k is $G_k = \sum_i g^0_{ik}$ and the market clearing price in zone k is ρ^g_k. Consequently the individual compliance constraint with respect to recharge rights is :

$$g_{ik} - p_{ik}(u_{ik}, a_{ik}) \geq 0. \tag{11.2}$$

Note that this type of market is zonal by definition; indeed watertable management is relevant on a particular scale, the catchment itself.

With such a model formulation, the essential feature of agents' participation in the market is captured, their responsiveness to the price system. This builds on the assumption of simultaneous trading procedures[1]. Five combinations of cap and trade systems are analysed, varying according to the type of rights being exchanged (recharge or diversion) and the spatial scale of authorized exchanges. In the *'status quo'* case, a series of zonal cap and trades for diversion are implemented. This corresponds roughly to the current situation in the case study area, as only a few examples of inter-zone exchanges have been documented, in particular in Turral et al. (2005). The *'recharge only'* case corresponds to a hypothetical situation where only the recharge would be the base for implementation of cap and trade mechanisms. Both types of zonal mechanisms are considered together in the *'two markets'* case, which is illustrative of the pilot scheme that has been implemented in the Colleambally Irrigation Area (Whitten et al. 2003). Regional cap and trade mechanisms for diversion rights are analyzed in the remaining cases, as the only policy being implemented in *'no barriers to trade'* and in conjunction with zonal cap and trades for recharge in the *'two markets, no barriers'* case.

11.2 Constraints on diversion and recharge under study

This Section sums up the various constraints that an irrigator may be subject to within the framework of this model. In Chapter 10, the impact of enforcing the regulator's constraints on environmental flows and recharge was analysed and the idea of a district flow constraint was introduced. By focusing on the cap and trade type of policy instruments, another set of constraints is introduced, the caps.

Table 11.1 presents the constraints introduced in the model, expressed in terms of maximum aggregate diversion[2] and recharge allowed :

- $\bar{W}_k$ and $\bar{R}_k$ are the socially optimum constraints,
- W_k and G_k are the constraints introduced by the zonal cap and trade systems : they correspond to the aggregate initial allocation of rights,
- $W(G_k)$ and $G(W_k)$ are *de facto* induced caps : they capture the impact of capping one item on the other item. For instance, $G(W_k)$ is the maximum amount of recharge that

[1] See Weber (2001) for a water market modeled as a series of non-cooperative games between subsets of water users.

[2] Consequently, the constraint : $q_k - d_k \geq \bar{Q}_k$ is turned into $d_k \leq \bar{W}_k = q_k - \bar{Q}_k$.

is generated when a cap W_k is imposed on the amount of water diverted for use in zone k,

– $\underline{W_k}$ is the district diversion constraint.

Consequently, all the constraints are defined at the catchment scale, apart from the regional cap on diversion, W.

	Flow	***Recharge***
Cap - zonal	$W_k = \sum_i w_{ik}^0$	$G_k = \sum_i g_{ik}^0$
Cap - regional	$W = \sum_k \sum_i w_{ik}^0$	
De facto induced cap	$W(G_k)$	$G(W_k)$
Optimal constraint	$\bar{W_k} = q_k - \bar{Q_k}$	$\bar{R_k} = \sum_i p_k(u_{ik}, a_{ik})$
District constraint	$\underline{W_k} = q_k - \underline{Q_k}$	*none*

TAB. 11.1 – Limits on water diversion and recharge.

It is assumed that $\underline{W_k} > \bar{W_k}$: the district constraint isn't harsher than the socially optimum one, so that the aggregate amount of water that can be diverted is greater than under the optimal constraint. This reinforces the connotation of 'safeguard' mechanism attached to the district constraint. It is also assumed that the caps are optimally set[3], so that $\bar{W_k} = W_k$[4]. A consequence is that when a diversion cap and trade is introduced, $\underline{W_k} > W_k$, the district constraint isn't binding.

11.3 Zonal cap and trades

Consider that in each catchment in the river system, barriers to trade are implemented to prevent exchanges of rights with other catchments. While it is a straightforward feature of recharge rights markets, it results from institutional mechanisms in the case of diversion rights (Turral et al. 2005). First each type of market is analysed separately, in order to derive their main features. It is shown that the use of one type of market is sufficient to manage both externalities only in particular cases. Then a combination of the two instruments is addressed ; it allows an optimal management of water along the system.

[3]For an analysis of the strategic setting of regional targets in the implementation of global markets, see Caplan and Silva (2005). In the event of $\bar{W_k} < W_k$ the regulator is in the position to withdraw rights from the market. Such problems of over-allocation do occur. See Thoyer et al. (2004) for a discussion on this issue of over-allocation of diversion rights, and an illustration of reallocation strategies in various countries.

[4]The same applies to the recharge.

11.3.1 The status quo

The *status quo* is as a system of regional cap and trades for diversion. This case is a useful benchmark to which open markets will be compared. The program of an agent ik is as follows :

$$\max_{u_{ik},a_{ik},w_{ik}^{jk}} \pi_{ik}(u_{ik},a_{ik}) - \rho_k^w \sum_k w_{ik}^{jk} \quad \text{subject to :}$$
$$w_{ik} - u_{ik} \geq 0, \tag{11.1}$$
$$q_k - d_k \geq \underline{Q_k}. \tag{9.4}$$

Remark 1. *A cap and trade on diversion does not impact directly on abatement decisions.*

Démonstration. First order conditions are derived from the Lagrangian :

$$L_{ik}^A = \pi_{ik}(u_{ik},a_{ik}) - \rho_k^w[w_{ik} - w_{ik}^0] + \beta_{ik}[w_{ik} - u_{ik}] + \eta_k^d[q_k - d_k - \underline{Q_k}],$$

where β_{ik} is the shadow cost of holding a right and η_k^d is the cost of respecting the district diversion constraint[5], and remembering that $w_{ik} = \sum_j w_{ik}^{jk}$:

$$\frac{L_{ik}^A}{u_{ik}} = \frac{\partial \pi_{ik}}{\partial u_{ik}} - \beta_{ik} - \eta_k^d = 0, \tag{11.3}$$
$$\frac{L_{ik}^A}{a_{ik}} = \frac{\partial \pi_{ik}}{\partial a_{ik}} = 0, \tag{11.4}$$
$$\frac{L_{ik}^A}{w_{ik}^{jk}} = -\rho_k^w + \beta_{ik} + \eta_k^d = 0. \tag{11.5}$$

Equation (11.4) states that the marginal benefit from abating equates zero at the equilibrium. Hence the market for diversion rights has no direct impact on the abatement decisions of agent ik through its market clearing price. □

Introducing a cap and trade mechanism on diversion rights has the following impact on irrigation decisions. Equation (11.3) shows the marginal benefit from diverting an extra unit of water from the river, accounting for the shadow price of holding the associated permit and the increased costs of binding the minimum diversion constraint. In equation (11.5) is illustrated the fact that the agents perceive the consequences of trading a diversion right with another agent from the same zone on the diversion constraint. Combining equations (11.3) and (11.5), the following relation is obtained :

$$\frac{\partial \pi_{ik}}{\partial u_{ik}} - \rho_k^w = 0, \tag{11.6}$$

which simply states that agent ik equals the diversion permit price to the marginal benefit

[5]For expositional purposes, in the remainder of this Chapter all the agents from a zone are subject to the same Rosen weight which, without loss of generality, is set at $r_{ik} = 1$.

from diverting water as an input to production. Hence an intra-zone trade is neutral with respect to the district diversion constraint.

Proposition 1. *A cap and trade on diversion is effective in inducing agents to constrain the aggregate recharge under restrictive conditions only.*

Démonstration. The impact of a diversion cap on the aggregate amount of recharge produced in zone k is as follows. If the market for diversion rights is enforced, and that agents do comply, then $\sum_i u_{ik} = W_k$. The resulting aggregate percolation is : $\sum_i p_{ik}(u_{ik}, a_{ik}) = \alpha_k W_k - \delta_k \sum_i a_{ik}$. From equation (11.4), $a_{ik} = \delta_k \varepsilon_k / D_{ik}$. Then the aggregate recharge is :

$$G(W_k) = \alpha_k W_k - \delta_k^2 \varepsilon_k \sum_i \frac{1}{D_{ik}}.$$

Consequently, a diversion cap is able to cope with the two social constraints under the following condition :

$$G(W_k) \leq \bar{R}_k \Rightarrow W_k \leq \hat{W}_k = \frac{1}{\alpha_k} \left[\bar{R}_k + \delta_k^2 \varepsilon_k \sum_i \frac{1}{D_{ik}} \right].$$

A sufficiently low diversion cap generates an aggregate recharge level in compliance with the optimal constraint. However, if $W_k \geq \hat{W}_k$ the diversion cap is not sufficient to constrain the aggregate recharge. □

This result is fairly intuitive, considering the type of dependency existing between the externalities under study. Indeed, increasing diversion impacts positively on the amount of recharge produced. The consistency between $G(W_k)$ and $\bar{R}_k$ is of empirical matter, as it depends on hydrological parameters. The partial derivatives of this limit value, $\hat{W}_k$ with respect to the model's parameters are as follows :

$$\frac{\partial \hat{W}_k}{\partial \bar{R}_k} > 0, \ \frac{\partial \hat{W}_k}{\partial \alpha_k} < 0, \ \frac{\partial \hat{W}_k}{\partial \delta_k} > 0, \ \frac{\partial \hat{W}_k}{\partial \varepsilon_k} > 0, \ \text{and} \ \frac{\partial \hat{W}_k}{\partial D_{ik}} < 0.$$

A higher $\hat{W}_k$ is consistent with the following features : a less stringent recharge constraint, that allows more water being percolated, more efficient irrigation or abatement, a higher individual damage term, higher cost of respecting the recharge constraint, and a lower cost of abatement. Any feature that induces a higher abatement by an individual irrigator translates into a less stringent diversion cap needed.

Next the potential of a zonal cap and trade for diversion to induce the agents to take optimal decisions is assessed. This feature is stronger than the effectiveness criterion assessed in the previous result.

Proposition 2. *A series of zonal cap and trades for diversion leads to the optimal allocation of water under special features of the trading price and of the socially optimum state of the system only.*

Démonstration. Remember the first-order conditions in the social optimum case :

$$\frac{\partial \pi_{ik}}{\partial u_{ik}} - \alpha_k \sigma_k [\Gamma_k - \lambda_k] - \lambda_k - \mu_{1k} - \alpha_k \mu_{2k} = 0, \tag{10.1}$$

$$\frac{\partial \pi_{ik}}{\partial a_{ik}} + \delta_k \sigma_k [\Gamma_k - \lambda_k] + \delta_k \mu_{2k} = 0. \tag{10.2}$$

Comparing equations (10.1)- (10.2) and (11.4)-(11.6) :

$$0 = \sigma_k [\Gamma_k - \lambda_k] + \mu_{2k}, \tag{11.7}$$

$$\rho_k^w = \alpha_k \sigma_k [\Gamma_k - \lambda_k] + \lambda_k + \mu_{1k} + \alpha_k \mu_{2k}$$

$$\Rightarrow \rho_k^w = \lambda_k + \mu_{1k}. \tag{11.8}$$

From equation (11.8), irrigation decisions are socially optimal if the market clearing price is set at the optimal shadow cost from diverting, which accounts for the environmental constraint and the impact on downstream users. From equation (11.7), individual abatement decisions are socially optimal if the social cost of recharging is null. This happens either if the recharge constraint isn't binding at the social optimum in zone k, and if the cost of discharging is null ; or if the cost of recharging is just equal to the benefit from discharging. □

A zonal cap and trade system for diversion may be efficient in managing the environmental flow constraint in an optimal way. However it has no direct impact on abatement decisions and as such can only lead to optimal abatement decisions if the socially optimal abatement level is null. Nonetheless, if the diversion cap is low enough, the aggregate recharge may be satisfactory.

11.3.2 Waterlogging first : managing the recharge

The purpose of analyzing this case is to clarify the functioning of a recharge cap and trade, and highlight its impact on the choice of irrigation and abatement levels. This will be useful when analyzing the case of an open market for diversion combined with a series of recharge cap and trades. The program of an agent ik is as follows :

$$\max_{u_{ik}, a_{ik}, g_{ik}^{jk}} \pi_{ik}(u_{ik}, a_{ik}) - \rho_k^g \sum_j g_{ik}^{jk} \text{ subject to :}$$

$$q_k - d_k \geq \underline{Q_k}, \tag{9.4}$$

$$p(u_{ik}, a_{ik}) \leq g_{ik}. \tag{11.2}$$

Remark 2. *A recharge rights markets impact on both diversion and abatement decisions.*

Démonstration. First order conditions as follows :

$$\frac{\partial \pi_{ik}}{\partial u_{ik}} - \alpha_k \beta_{ik}^g - \eta_k^d = 0, \tag{11.9}$$

$$\frac{\partial \pi_{ik}}{\partial a_{ik}} + \delta_k \beta_{ik}^g = 0, \tag{11.10}$$

$$-\rho_k^g + \beta_{ik}^g + \frac{\eta_k^d}{\alpha_k} = 0. \tag{11.11}$$

Equation (11.9) showing the marginal benefit from using a extra unit of water is similar to the case of diversion rights market, except that it accounts for $\alpha_k \beta_{ik}^g$ instead of ρ_k^w. Hence the recharge market clearing price impacts on the agents' choice of water input. The marginal benefit from abating is also affected by the shadow cost of holding a recharge right. In equation (11.11) is illustrated the fact that the agents perceive the effect of trading a recharge right on the respect of the minimum diversion constraint. Then equations (11.9) and (11.10) become :

$$\frac{\partial \pi_{ik}}{\partial u_{ik}} - \alpha_k \rho_k^g = 0, \tag{11.12}$$

$$\frac{\partial \pi_{ik}}{\partial a_{ik}} + \delta_k \rho_k^g - \eta_k^d \frac{\delta_k}{\alpha_k} = 0. \tag{11.13}$$

At the equilibrium, agent ik equates the marginal benefit from water use to the recharge market price weighted by a factor α_k. He also equates the marginal benefit from abating to the market price weighted by a factor δ_k, and accounts for the impact of abating on the environmental flow constraint if this constraint is binding. □

Consequently, a recharge rights market impacts on both decisions of irrigating and abating. Every agents from a zone are affected in the same manner, as the price is weighted by parameters assumed homogenous within a given zone, namely irrigation and abatement efficiencies.

Proposition 3. *A cap and trade on recharge is effective in inducing agents to constrain aggregate diversion under restrictive conditions only.*

Démonstration. Coming back to table 11.1, focus is on $W(G_k)$. As in the status quo case, the regulator sets the market caps G_k optimally. If the market is enforced and agents comply with it, then at the aggregate level $\sum_i p_{ik}(u_{ik}, a_{ik}) = G_k$. Also, from equation (11.13), $a_{ik} = \dfrac{\delta_k}{D_{ik}} \left[\varepsilon_k + \rho_k^g - \dfrac{\eta_k}{\alpha_k} \right]$. Then,

$$W(G_k) = \frac{1}{\alpha_k} \left[\bar{G}_k + \delta_k^2 (\varepsilon_k + \rho_k^g - \frac{\eta_k^d}{\alpha_k}) \sum_i \frac{1}{D_{ik}} \right].$$

Consequently, a unique recharge cap is sufficient to manage two externalities under the following condition :

$$W(G_k) \leq \bar{W}_k \Rightarrow \eta_k^d = 0 \quad \text{and} \quad G_k \leq \alpha_k \bar{W}_k - \delta_k^2(\varepsilon_k + \rho_k^g) \sum_i \frac{1}{\sum_i D_{ik}}.$$

Any feature that induces a higher abatement by an agent tends to increase the value of the recharge cap necessary to induce an optimal aggregate diversion. Indeed, when agents become more efficient in their irrigation practices (in the sense that they reduce the amount of water percolating from the same amount of diverted water) the recharge constraint needs to be more constraining to induce a decrease in water use. □

This statement illustrates a debate currently taking place concerning the environmental impacts of the development of increasingly efficient irrigation systems. Indeed, by allowing to produce more with less water, they tend to have induced more diversion and less recharge, leading to scarcer water resources and an increasing problem of soil salinity[6].

Proposition 4. *A series of recharge rights market leads to the optimal solution under a set of conditions on the pricing system and the social optimum.*

Démonstration. Compare equations (10.1)- (10.2) with (11.12)-(11.13) :

$$\eta_k^d = \lambda_k + \mu_{1k}, \tag{11.14}$$

$$\rho_k^g = \sigma_k[\Gamma_k - \lambda_k] + \mu_{2k} + \frac{\eta_k^d}{\alpha_k}. \tag{11.15}$$

If the district constraint isn't binding, $\eta_k = 0$. Then, irrespective of whether the induced diversion is above or below the optimal constraint, optimality conditions are :

$$\rho_k^g = \sigma_k[\Gamma_k - \lambda_k] + \mu_{2k},$$
$$0 = \lambda_k + \mu_{1k}.$$

Optimality of abatement decisions depends on the recharge market clearing price being set at the socially optimal cost of recharge. Optimality of irrigation decisions is ensured only if the social cost of diverting is null. This result is symmetric to that obtained in the status quo case.

If the district constraint binds, so that $\eta_k > 0$, then a somewhat surprising result is obtained. Indeed, optimality conditions become :

$$\rho_k^g = \sigma_k[\Gamma_k - \lambda_k] + \mu_{2k} + \frac{\eta_k^d}{\alpha_k},$$
$$\eta_k^d = \lambda_k + \mu_{1k}.$$

[6] Indeed, a minimum amount of water is needed to flush down the salts from the root zone to prevent their accumulation (Wichelns 1999).

In this case, it is the district constraint, through its shadow price, that constrains diversion decisions, instead of the recharge cap. Consequently, if the shadow cost associated to the minimum diversion constraint is set at the optimal cost of diversion, agents can be induced to take optimal decisions, assuming that the irrigation districts have the legislative powers to enforce a taxation scheme. □

A zonal cap and trade system for recharge may be efficient in managing the recharge constraint in an optimal way. Furthermore, it impacts on both control variables. If the recharge cap is low enough, the aggregate diversion can be satisfactory from the regulator's viewpoint. However, optimality of irrigation decisions is ensured under restrictive conditions only.

11.3.3 A system combining two types of zonal cap and trades

Here the combined effect of the two types of cap and trades on individual agents' decision-making process are investigated. Considering the correlation existing between the externalities 'water scarcity' and 'waterlogging' the markets are expected to be linked in some way. The program of an agent subject to both types of cap and trade systems is as follows :

$$\max_{u_{ik},a_{ik},w_{ik}^{jh},g_{ik}^{jh}} \pi_{ik}(u_{ik},a_{ik}) - \rho_k^w \sum_{j=1,\neq i}^{n_k} w_{ik}^{jh} - \rho_k^g \sum_{j=1,\neq i}^{n_k} g_{ik}^{jh} \text{ subject to equations (11.1)-(9.4)-(11.2).}$$

Assuming that the caps are optimally set, this diversion and recharge zonal cap and trade system automatically satisfies the effectiveness criterion. The following result addresses its efficiency.

Proposition 5. *A scheme combining zonal diversion and recharge cap and trade systems is efficient in managing both environmental flows and recharge constraints.*

Démonstration. The following first order conditions are obtained :

$$\frac{\partial \pi_{ik}}{\partial u_{ik}} - \rho_k^w - \alpha_k \rho_k^g + \gamma_k + \frac{\eta_k^d}{\alpha_k} = 0, \tag{11.16}$$

$$\frac{\partial \pi_{ik}}{\partial a_{ik}} + \delta_k \rho_k^g - \eta_k^d \frac{\delta_k}{\alpha_k} = 0. \tag{11.17}$$

This cap and trade system is compatible with the optimal allocation of water under the following conditions on the market clearing price :

$$\rho_k^g = \sigma_k[\gamma_k - \lambda_k] + \mu_{2k},$$
$$\rho_k^w = \mu_k + \lambda_{1k}.$$

This system corresponds to sharing the costs of diverting water, and thus percolating water, into the two policy instruments. □

Remark 3. *According to the state of the system under the optimal allocation of water, the efficient policy instrument may include implementing only one type of markets in some identified catchments.*

Indeed, refereing to the study of the status quo and recharge only cases, if one of the constraint is not binding at the optimum, then the associated cap and trade is not necessary to ensure efficiency. For instance, upstream catchments may be less constrained by the environmental flow constraint, as they are located near the outset point of the dam providing the river system with most of its flows. At the same time, downstream catchments are supposedly more subject to water scarcity. Then zonal cap and trade systems on diversion may appear more needed downstream than upstream.

Focusing on within-catchment trades, the differentiated impacts of the two types of cap and trade systems under study were highlighted. While a combination of the two is efficient in inducing the agents to take optimal decisions, the binding nature of the optimal constraint has been shown to condition the necessity to implement the two policy instruments in each catchment.

11.4 Removing barriers to trade

This Section investigates the impact of removing barriers to trade to allow inter-district trades of diversion rights. It is shown that such a trading mechanism can't support the optimal solution, unless the agents are induced to account for the asymmetric impact of their trades, according to the location of the seller / purchaser of diversion rights along the river.

The program of an agent ik is as follows :

$$\max_{u_{ik}, a_{ik}, w_{ik}^{jk}} \pi_{ik}(u_{ik}, a_{ik}) - \rho \sum_{h} \sum_{j} w_{ik}^{jh} \quad \text{subject to (11.1)-(9.4).}$$

Proposition 6. *A regional cap and trade for diversion alone can't lead to the optimal allocation of water and abatement efforts by individual irrigators.*

Démonstration. The following first order conditions are derived from the Lagrangian :

$$\frac{\partial L^D}{\partial u_{ik}} = \frac{\partial \pi_{ik}}{\partial u_{ik}} - \beta_{ik} - \eta_k^d = 0, \tag{11.18}$$

$$\frac{\partial L^D}{\partial a_{ik}} = \frac{\partial \pi_{ik}}{\partial a_{ik}} = 0, \tag{11.19}$$

$$\frac{\partial L^D}{\partial w_{ik}^{jh}} = -\rho + \beta_{ik} + AS_{kh} = 0. \tag{11.20}$$

In equation (11.20), the term AS_{kh} illustrates agent ik's awareness that there is an asymmetry in payoffs when he trades with an agent from upstream ($h < k$), from the same zone ($h = k$) and from downstream ($h > k$). Indeed, agent ik perceives the flow of water entering his zone as :

$$q_k = q_0 - \sum_{h=1}^{k-1} d_h + \sum_{h=1}^{k-1} h_h, \tag{11.21}$$

which is derived from equation (9.1)[7]. The differentiated impacts of trades carried out by agent ik with agents from various location depends on the effect of purchases (or sales) on the amount of water left after uptake in zone k, namely the amount of water entering the zone or the aggregate amount of water diverted. Consequently, AS_{kh} takes the following values :

$$AS_{kh} = \eta_k^d(1 - \sigma_h\alpha_h) \text{ if } h < k,$$
$$AS_{kh} = \eta_k^d \text{ if } h = k,$$
$$AS_{kh} = 0 \text{ if } h > k.$$

Purchasing a right from an upstream agent relaxes the environmental flow constraint by a factor $1 - \sigma_h\alpha_h$, which is the amount of water that the upstream agent renounces to consume by selling his right[8]. Symmetrically, selling a right to an upstream agent binds the flow constraint by a factor less than 1 as return flows are produced upstream. Trading with a downstream agent has no impact in itself on the flow constraint. Trades with agents from the same zones were analyzed in Section 11.3.1. Consequently the marginal benefit from using water depends on the origin of the right :

$$\frac{\partial \pi_{ik}}{\partial u_{ik}} = \rho + \eta_k^d\sigma_h\alpha_h \text{ if } h < k,$$
$$\frac{\partial \pi_{ik}}{\partial u_{ik}} = \rho \text{ if } h = k,$$
$$\frac{\partial \pi_{ik}}{\partial u_{ik}} = \rho + \eta_k^d \text{ if } h > k.$$

Compatibility with the optimal allocation requires :

$$0 = \sigma_k[\Gamma_k - \lambda_k] - \mu_{2k}, \tag{11.22}$$
$$\rho + \eta_k^d - AS_{kh} = \lambda_k + \mu_{1k}. \tag{11.23}$$

As in the *status quo* case, from equation (11.22) it appears that a regional cap and trade for diversion has no impact on abatement decisions and as such cannot induce the agents to take optimal abatement decisions, unless the social cost of recharging is null. From equation (11.23), it also appears that the spatially differentiated price system needed to

[7]Weber (2001) derives the same type of relations in the case of unique agents at each point along the river.

[8]More precisely, it is the difference between the water not diverted, passing directly into the river, and the amount of return flows not produced as a consequence of water not being applied.

induce an optimal level of irrigation[9] can't be supported by a regional market clearing price, unless the district constraint binds. In this case, a strictly positive η_k introduces a possibility of spatialization of the regional cap and trade for diversion. $\square$

Proposition 7. *When a recharge cap and trade system is associated to a regional diversion rights market, then (a) recharge and abatement decisions can be optimal; (b) constraining the recharge introduces a spatialization of the regional cap and trade on diversion, which may lead to the optimal aggregate diversion to be reached by introducing a type of barriers to trade; (c) however irrigation decisions are optimal only under very restrictive conditions.*

Démonstration. The program of an agent ik is as follows :

$$\max_{u_{ik},a_{ik},w_{ik}^{jk},g_{ik}^{jk}} \pi_{ik}(u_{ik},a_{ik}) - \rho_k^g \sum_{j=1,\neq i}^{n_k} g_{ik}^{jk} - \rho \sum_{h=1}^{m} \sum_{j=1,\neq i}^{n_k} w_{ik}^{jh} \text{ subject to (9.4)-(11.1)-(11.2).}$$

Individual irrigation and abatement levels are as follows :

$$\frac{\partial \pi_{ik}}{\partial u_{ik}} = \rho + \alpha_k \rho_k^g - AS_{kh}, \tag{11.24}$$

$$\frac{\partial \pi_{ik}}{\partial a_{ik}} = \delta_k \left[\frac{\eta_k^d}{\alpha_k} - \rho_k^g \right]. \tag{11.25}$$

Compatibility with the optimal solution requires :

$$\rho_k^g = \mu_{2k} + \sigma_k[\Gamma_k - \lambda_k] + \frac{\eta_k^d}{\alpha_k}, \tag{11.26}$$

$$\rho - AS_{kh} = \lambda_k + \mu_{1k}. \tag{11.27}$$

As in previous cases, assuming that the recharge caps are optimally set, then the aggregate recharge will be capped at the optimal level. Also, from equation (11.26), the recharge market clearing price is able to induce the agents to take optimal abatement decisions. Hence such a combination of cap and trades has the potential to efficiently manage the recharge and associated abatement decisions, explaining the first part of result 7.

In consistency with the analyses carried out in Section 11.3.2 it is possible to compute the diversion cap induced by the implementation of the recharge cap and trade systems in each zone. This means that once this diversion limit has been reached, the agents in possession of unused diversion rights would not be able to use them for irrigation on the zone. This sort of environmental justification to barriers to trade is an important feature of such a combination of policy instruments that can't be ignored when analyzing one of them. Hence the second part of Result 7.

[9] Unless the zones are homogeneous with respect to the flow constraint, which would constitute a very particular case.

Finally, as in the preceding case, if the district constraint is binding, then irrigation decisions could be optimal by the specialization of price introduced by a strictly positive η_k. □

11.5 Concluding remarks

In this Chapter, the conditions under which various combinations of cap and trade systems for diversion or recharge rights induce individual irrigators to efficiently manage surface and underground water resources were derived. In this process, hydrological links have been highlighted as having the consequence of coupling the associated policy instruments. In particular, in the context of irrigation-induced salinity, implementing recharge market will have the consequence of constraining the exchange of diversion rights within the regional cap and trade system for diversion. From this analysis it appears that a mix of two instruments is more efficient than a unique instruments. However, the importance of carefully assessing the nature of the constraints at stake was also emphasized. Indeed, according to the combination of constraints in each catchment, a unique type of instrument is not necessarily sub-optimal. Also, this analysis puts in perspective the 'community' feature of numerous environmental issues, by explicitly considering irrigation districts, rather than individual agents, located along the river.

11.5.1 Testing the Hypotheses

> **Hypothesis 2a : spatial extent. Where coupled externalities exist, a series of zonal cap and trades is more efficient than a regional cap and trade.**

Testing this hypothesis amounts to comparing the '*no barriers*' case with either the '*status quo*' or the '*waterlogging first*' cases.

Compare equations (11.6)-(11.4) and (11.18)-(11.19) : cap and trade for diversion at either the zonal scale or the regional scale have no impact on abatement decisions. In this respect, they lead to optimal abatement decisions only under the special conditions where it is optimal not to abate, which is a very restrictive case. However, their impact is different concerning irrigation decisions. Indeed, a zonal cap and trade effectively reduces input use in such a way as to induce the agents to comply with the optimal instream flow constraint, while a regional diversion rights market can't ensure that the agents will be in compliance with the target. Indeed, they are induced to account for the impact of their trades on the state of the river, a crucial condition of success of a market for surface water rights in a dendritic system such as a river, only if the district constraint binds. Furthermore, it is unlikely that the district be sufficiently committed to instream flows so as to induce optimal irrigation decisions, by enforcing a sufficiently large penalty. Hence, a single instrument at the zonal scale proves more efficient that a single instrument with a wider spatial extent to manage two environmental externalities expressed at the zonal scale, with the difference that the instream flow constraints are interdependent and jointly lead to the optimal management of river flows.

Compare equations (11.12)-(11.13) and (11.18)-(11.19) : a series of cap and trades for recharge has an impact on both decisions of irrigating and abating. Hence, it has the potential to induce socially optimal abatement decisions ; and it implicitly constitutes a constraint to water diversions on a zonal extent. In these respects, a series of zonal cap and trades for recharge is more efficient to induce agents to comply with the optimal constraints.

In a framework governed by the need to comply with constraints on water use (diversion and percolation) defined at the zonal scale[10], Hypothesis 2a is validated. This confirms that the regional scale is inappropriate to account for zonal externalities.

> **Hypothesis 2b : number of instruments. Where coupled externalities exist, combining two types of instruments is more efficient than implementing only one type of instruments.**

To test this hypothesis, in a first step the '*two zonal markets*' case is compared with the '*status quo*' and the '*waterlogging first*' case, focusing on the zonal scale. Then the regional scale is addressed by comparing the '*no barriers, two markets*' case with the '*no barriers*' case.

Compare equations (11.16)-(11.17) with equations (11.6)-(11.4) or (11.12)-(11.13) : optimality of the combination of diversion and recharge cap and trades has been established, while it is not ensured in the unique instrument case. However, according to the state of the river system at the optimum, the enforcement of the two constraints may not be needed ; in this case applying two types of instruments is not needed in the zone under stake.

Compare equations (11.24)-(11.25) with equations (11.18)-(11.19) : implementing recharge cap and trade systems constitutes a *de facto* environmental barrier to trade, introducing a spatialization of the diversion rights market necessary to ensure that optimal diversion decisions are taken.

Whatever the scale of the diversion rights market, combining it with a series of recharge cap and trade proves more efficient than implementing it in isolation. Hence, even if the externalities are linked, two instruments are preferable to only one instrument. Indeed, each instrument plays its role of sending specific incentives to the agents, that participate

[10] With the same restriction that instream flows are managed along the river.

differently in the generation of the externalities.

> **Hypothesis 2c : spatial extent x number of instruments. Where coupled externalities exist, a regional diversion market associated to a series of zonal recharge markets is more efficient than a series of zonal markets (either on diversion or on recharge).**

Compare equations (11.24)-(11.25) with equations (11.6)-(11.4) : both regional diversion rights market in association with recharge markets and zonal diversion markets induce a reduction in water use, to an extent that will depend on the relative value of ρ_k and $\rho + AS_{hk} + \alpha_k \rho_k^g$. However, as already addressed, a diversion rights market can't ensure that optimal abatement decisions are taken. Hence a regional diversion rights market in association with recharge markets is preferable to zonal diversion markets only, in this respect.

Compare equations (11.24)-(11.25) with equations (11.12)-(11.13) : both systems induce the same abatement effort, which can be set at the optimal level ; however the regional market for diversion associated to the recharge markets effectively induces a decrease in water use, as diversion becomes more costly ; in contrast, the recharge markets has a lower impact on diversion decrease.

Consequently, this confirms the need to carefully address the consequences of opening water markets in the absence of appropriate institutions to manage third party impacts, in a wider sense than the usually accepted one of 'instream flows constraints'.

11.5.2 Limits and extensions.

A note on heterogeneity. The model used in this Part abstracts from heterogeneity of most individual parameters at the district scale, such as the percolation and abatement efficiency rates, as well as the individual damage parameter. The rationale behind this modelling choice was that it was assumed that irrigators are more homogeneous within that between catchments on those features. Indeed, these parameters are highly linked to the pedological state of the district. However, assuming heterogeneity on these parameters would alter the subsequent analyses as follows. Considering within-catchment markets, the direction of trades would be affected, but not the aggregate compliance to the cap. Consequently, this would not alter the effectiveness of the markets, while the derivation of the *de facto* induced caps would be complicated. Considering inter district exchanges, heterogeneity of the parameters would alter the asymmetric nature of exchanges as follows. The variable AS_{kh} would have to be individual-specific, exacerbating, on on the reverse, diluting, the extent of the asymmetry. Nevertheless, this would not alter the fact that asymmetries would not be accounted for by the irrigators unless the district constraint binds.

Precising the socially optimum allocation of water. The analysis was voluntarily set in a general setting, in order to highlight the importance of considering the binding nature of the environmental constraints. Consequently, an immediate extension of this work could consist in developing a resolution algorithm for the socially optimal allocation of water use and abatement effort. This would provide the basis for illustrating selective water management settings.

Other trading procedures. The model is developed in a somewhat restrictive framework, however it still captures the main hydrological and economic features at stake. Perhaps most restrictive is the assumption of simultaneous bargaining, following previous works (Griffin and Hsu 1993) (Dinar and Letey 1991) (Hung and Shaw 2005). Future research might therefore investigate other trading settings, such as sequential bilateral trading procedures (Weber 2001) (Hung and Shaw 2005). This would necessitate extending the number of users by location from a representative one, as in (Weber 2001) or (Hung and Shaw 2005), to a finite number, which is not without posing methodological issues.

Transaction costs. The existence of transaction costs associated with the design and implementation of (water) markets (McCann and Easter 2004) is one of the rationale for analyzing the efficiency of a single instruments to manage coupled externalities. The objective of this study was not to assess these costs ; however a discussion on their potential extent is of interest to assess the operability of the proposed mechanisms. *Ex-ante* costs include those associated with research and information, enactment or litigation, and design and implementation (McCann and Easter 2004). *Ex-post* costs are associated with support and administration, monitoring/detection and prosecution/enforcement (McCann and Easter 2004). In the Australian context, most of these costs have already been borne for diversion rights, as described in Part I. The transaction costs associated with the design of recharge cap and trade are potentially high is non pilot catchments, not only due to the modeling effort, but also to farmers' reluctance to engage in yet another type of market.

An instream salinity indicator. In this model instream salinity was incorporated into the regulator's program as a damage arising from each district's return flows. An extension of the model could consist in explicitly defining an instream salinity index.

Exchange rates. The analysis was restricted to various combinations of cap and trade systems without any other link than the hydrological ones. The use of trading ratios to introduce a spatialization of markets for rights was analyzed by Hung and Shaw (2005). In this thesis, the use of recharge cap and trade was highlighted as an alternative to approaches based on trading ratios in the particular setting of irrigation-induced salinity. However the consideration of exchange rate constitute a possible an extension of this work.

Chapitre 12

Concluding remarks

The goal in this thesis has been to advance the design of policy instruments to manage environmental issues, guided by the particular setting of irrigation-induced salinity in Australia. The Australian policy context led to the adoption of a double analysis of cap and trade systems and dynamic taxation schemes. The peculiarities of irrigation-induced salinity (including the correlation between surface and underground water, the rising of both quantitative and qualitative issues and the particular nature of the pollutant 'salt') allowed various modeling strategies. However, the analyses carried out in this thesis have general applications that exceed the particular setting of irrigation-induced salinity.

Summary of approaches and findings

The first Part of this thesis laid the conceptual framework withing which each subsequent Part developed.

Chapter 2 was devoted to an analysis of the Australian water management context. Its aim was twofold : to identify the constraints that the history of water management and irrigation development have left to current policy makers, and to highlight the recent policy trends that are being implemented to manage water and salinity-related issues. In particular, it has been pointed out that Australian water-related initiatives are characterised by a double discourse of rising environmental concerns and economic purposes. In this Chapter, the recourse to price-based policies together with the consideration of the notion of collective responsibility, as well as the need to refine water markets to account for external impacts, were identifed as key issues in the development of current policies. Chapter 3 consisted in a literature review of the use of collective, or group, performance in the design of policy instruments, with a particular interest in dynamic taxation schemes. After showing that group performance based instruments extend beyond the standard ambient taxes that have been developed to manage nonpoint source pollution, this Chapter specifically addressed the main characteristics of group performance based instruments, the interdependency they introduce between the agents subject to the scheme. First, they introduce a strategic interaction between the agents and the policy maker, in the sense that the agents are induced to account for the impact they have on the environment, and thus the future level of the policy instrument. Second, they also introduce strategic interactions between the agents, as all the agents supposedly have an impact on the level of the tax, and strategically respond to this level. As a consequence, Chapter 3 posed the question of the optimal mix of individual and collective performance in the design of policy instruments. Chapter 4 laid the theoretical framework to address the second trend identified in current Australian environmental policy making. It reviewed the literature on water markets when environmental concerns, focused on the quantitative or qualitative aspects of water, are accounted for. In particular, the reallocation of water allowed by the introduction of water markets has the potential to create, or enhance, external effects. Applying these theoretical insights to the context of irrigation-induced salinity, this Chapter put in perspective the interest for recharge rights markets, and posed the question of their integration within the existing system of diversion rights market. Chapter 5 developed the Hypotheses by drawing Chapter 2,3 and 5 together.

The second Part of this thesis investigated the interest for mixed taxation schemes (based on both individual and collective performance). It focuses on the catchment as the relevant decision-making scale.

Chapter 6 introduced the modeling framework Differential Games. After presenting the adaptation of standard groundwater models to the setting of irrigation-induced salinity, it highlighted the importance of the choice of the agents' strategy space. With reference to the Australian context, it showed that both 'standard' strategies, open-loop and feedback, are relevant. The Australian context also led to some assumption, in particular concerning the confidence in the models and the resulting reliance on water inputs as a policy basis. In Chapter 7, irrigation-induced salinity was approached as a single stock pollution problem. This Chapter provided a validation of Hypothesis 1 under certain conditions. Indeed, when the regulator wants to induce individual agents to perform optimally along the whole time horizon, mixed instruments are necessary, while when optimality is required at the steady state only, taxes based on one type of performance only are sufficient. Chapter 8 offered an extension of this analysis to a setting exhibiting multiple state variables.

The third Part of this thesis was devoted to the analysis of cap and trade systems to manage surface water scarcity and salinity-related issues in various catchments located along the river. Consequently, focus was placed on the spatial dynamics of river flows, rather than on groundwater accumulation.

Chapter 9 developed the model and addressed the setting of various types of environmental constraints. The resolution of the basic problems (regulator and individual agents in the absence of cap and trade systems) led to the introduction of the coupling constraint framework in Chapter 10. Finally, the analysis of cap and trade systems in Chapter 11 put in perspective the importance of identifying correlated externalities and assessing their impact on the design of policy instruments. In the particular case of water markets in the presence of irrigation-induced salinity, the existence of a type of environmental justification to barriers to trade was pointed out. Hence, particular attention has to be given to the external effects of the implementation of water markets, on items other than water scarcity. The main result of this analysis is that the correlation existing between the externalities doesn't rule out the need for a policy instrument to manage each externality.

Key policy implications

The key policy implications derived from this thesis may be summed up in 5 key messages.

Identify the correlations. Environmental issues may involve more or less complex mechanisms. Water management in the context of IIS is compounded by the interactions existing between the various components of the system - surface and underground water, quality and quantity features. This complexity is also true in other contexts. It is then crucial to : identify all the externalities at stake ; identify the way they interact, the sign and extent of the correlation ; assess whether the management objectives are compatible ; if necessary, establish trade-offs. In the case of irrigation-induced salinity, for instance, it is necessary to assess whether return-flows produced from groundwater discharge should be addressed as net benefits, or net costs.

Assess the strategic nature of the agents. Economic agents interact in various manners. The extent to which an irrigator accounts for the other agents' behavior and decision has an impact on policy design. An instrument based on the collective contribution of a community, for instance, may suffer from free-riding behavior because the pursuit of individual interest may override the collective incentive. Hence, collective taxes have been shown to potentially induce irrigator to over-produce percolation water. This has to be accounted for in order to adjust the design of the instruments. In the spatial analysis of water markets, it was shown that under circumstances, individual irrigators may be induced to act strategically with respect to irrigators from other irrigation districts. This translates into the realization that water rights exchanges don't have the same features in terms of instream flows according to the partner' localisation along the river, which goes in favor of an optimal spatialisation of the market. A good knowledge of the agents' decision making process is crucial : the regulator even has the possibility to change this process by providing information - that has the potential to turn open-loop agents into feedback loop ones.

Think collective dynamic taxes. The analyzes developed in this thesis show that, once defined and accepted by the stakeholders, collective responsibility for an environmental issue proves an interesting policy basis in a dynamic framework. In other words, future policy developments should aim at rendering irrigation management more contingent on the state of the environment for each individual decision-maker. In this respect, initiatives such as the Sunraysia salinity levy should be extended to all the catchments, not only those that are already severely affected by salinity.

Spatialize the markets. It is necessary to accompany the removal of barriers to trade with an instrument to deal with each associated externality. In the setting of irrigation-induced salinity, a two tiered system of exchange rates should be introduced, to account for both within-zone salinity and between zones discharge impacts.

Mix policies. It was shown in Part II that mixing collective and individual incentives was necessary to ensure dynamic optimality. Adopting a more standard understanding of policy mix, quantity-based and price-based instruments were addressed in isolation in this thesis. However, the set of dynamic taxes that were analysed in Part II could be introduced in the system of water markets - in order to generate the spatialization to account for within-zones salinity. Mixing policies has the potential achieve multiple objectives, not only in terms of environmental issues, but also in terms of efficiency and political acceptability.

Opportunities for future research

The review on the use of group performance based instruments provided in Chapter 3 highlighted that group performance is a notion that has been used in various theoretical contexts and to answer various problems. Future research could include the consideration of a more general framework, including positive externalities and, in the multiple stocks model case, positively correlated stocks. In the vein of Jun and Vives (2004) [1], this would lead to a typology of the strategic interactions that arise from the implementation of mixed instruments. Another extension could consist in relaxing the assumption of non-cooperative agents. Millock and Salanié (2005) study the implementation of Segerson's ambient tax (1988) when some degree of cooperation, which extent is unknown to the regulator, develop between the agents. They show that the policy should consist in regulating cooperative agents only, through an ambient tax with a rate set lower than in the standard case of non-cooperative agents. A similar type of analysis could be applied to state-dependant input taxes such as the ones analysed in this thesis. Considering the possibility of agents cooperating when subject to a group performance based instrument

[1] Jun and Vives (2004) characterise the strategic incentives arising in a dynamic duopoly and show that it is the presence of production cost adjustments, rather that the nature of the competition (Bertrand or Cournot) that drives the competitiveness of a market in a dynamic framework.

is motivated, firstly, by the notion of group that the implementation introduces *de facto*. Also, when the policy instrument is implemented at the catchment scale, it concerns agents that are used to cooperating on other matters, or meeting in various social events. Thus an extension of this work could be to analyze the impact of various degrees of cooperation on the efficiency of mixed individual and collective instruments.

Part II showed that dynamic strategic interactions have an impact on the design of efficient policy instruments, and that the nature of the impact depends on the informational structures that the agents are assumed to have. Hence the importance of defining the informational structure with care. This provides a rationale for a careful analysis and determination of the appropriate informational structure to be used in the formulation of a game for the analysis of a particular situation. This could be approached by way of experimental economics, in various ways, including the identification of the strategies used by the agents or the comparison of the outcomes when agents make use of different strategies. To the best of our knowledge, few authors have tackled this type of problem. Sterman (1989) (1993) has documented the phenomenon of 'misperception of feedback' by decision makers, in the context of managerial economics. He shows that when agents are subjected to a stock management problem - where a manager seeks to maintain a quantity of goods within an acceptable range - most attribute the dynamics they perceive to external events, without recognizing the impact of their decisions on the environment. As such, they act in an open-loop way, rather than accounting for the feedbacks that arise between their actions, their environment and the perceptions they have of their environment. However the stock management problem is typically an individual problem - each type of agent[2] has 'total' control over his stock, so that the inter-agent interactions are absent from the understanding of feedback strategies. In an environmental context, Keser and Gardner (1999) provide an analysis of the various strategies developed by experienced subject[3] in a common pool resource game. Using the strategy method[4], they identify the strategies of their pool of subject as either open loop (decisions not contingent on previous outcomes) or closed loop (decisions depend on previous outcomes), the latter constituting the great majority of submitted strategies[5]. They note the great stability of

[2]Factory, distributor, wholesaler, retailer in the case of the Beer Distribution Game used in the experiments (Sterman 1989).

[3]Undergraduate or graduate students with at least a semester of Game Theory, supposedly aware of the concept of Nash Equilibrium

[4]The basics of the strategy method is that instead of playing the game, the subjects are asked to indicate an action for each information set. Advantages of this method include the generation of more information, about the motivation of the players and their behavior off equilibrium. The main drawback is that the information provided by the subject is based on 'cold emotions' while an actual play induces 'hot emotions'; hence agents might act differently when faced with a precise situation rather than when asked to imagine this situation.

[5]They further differentiate the agents according to the commentaries written by the subject at the end of the experiments, between 'proactive' and 'reactive' subjects. While the former indicate the will to dominate the common pool resource by making large investments and forcing the other to adjust to that, the later indicate that they will adjust to the history of the game. These agents' strategies are either strategic complements or substitutes.

the strategies, as only 4 subjects out of 16 changed strategies between the three rounds of the strategy method experiment. This analysis provides some insights into the nature of the strategies formulated by experimental subjects competing for the exploitation of a common pool resource over multiple periods. Future research might consist in providing experimental subjects with various levels of information, and identify if disclosing updated information about the state of the common pool resource, for instance, is conducive to the use of feedback-type strategies, and under which conditions ; or on the contrary, what type of strategies agents develop when they don't have access to this information - and if they comply with the level of commitment implied by the definition of open loop strategies.

An immediate extension of this work could be the integration of taxation schemes with water markets to accommodate salinity-related issues. Indeed, the analysis of dynamic taxation schemes to manage irrigation-induced salinity was set in the framework of independent catchments. Further research could be carried out on the integration of mixed taxation schemes within the broader context of markets for water rights implemented in the MDB. As such, this could provide some insights into ways to improve the system of exchange rates currently in place in some areas, such as the Sunraysia salinity levy described in Chapter 2. Applying a state-dependant tax, with a rate directly linked to the state of the aquifer, and thus the sensitivity to salinity, to every purchase of water for use on a catchment could be a way to overcome a type of asymmetry currently present in the Sunraysia system, as some types of exchanges are simply forbidden rather than subject to differentiated exchange rates. This would constitute a *de facto* dynamic exchange rate system, dynamic understood as state-contingent.

More generally, accounting for coupled externalities, either positively or negatively correlated, in the analysis and design of policy instruments, constitutes an opportunity for future research. For instance, the analysis of cap and trades for correlated externalities could be applied to other settings. The species-specific system of Individual Transferable Quotas (ITQs), currently implemented in New-Zealand (Kerr, Newell and Sanchirico 2003), constitutes in this respect an interesting case study of the lack of accounting for inter-species interactions, that may have consequences on the efficiency of the ITQs policy package. Also, the analysis of dynamic taxation schemes could be extended to the case of negatively correlated environmental issues.

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Appendix A

Proof of Proposition 10

Let $R^{OL} = \tau_1^{OL}/\tau_2^{OL}$, $R^{FB} = \tau_1^{FB}/\tau_2^{FB}$ and $z = nr + \delta + \rho(1-n) > 0$.

The objective is to show that $R^{FB} - R^{OL} > 0$. For this purpose, let $D(R^{FB} - R^{OL})$ be the denominator, and $N(R^{FB} - R^{OL})$ the numerator of the expression under study.

$$D(R^{FB} - R^{OL}) = nz(r+\delta)[z - \delta + \rho(1-n)](-c)(\delta+\rho)[(1-n)2\rho + r(n-1)] > 0,$$
$$N(R^{FB} - R^{OL}) = F_b + \rho F_{X_\infty},$$
with $F_b = n^2 bz(r+\delta)[z + \delta - \rho(n-2)] > 0$,
and $F_{X_\infty} = \rho^2 cnz(1-n) + \rho c[z^2(n-1) + nz(\delta + rn)] - c(r^2\delta + nz^2 r)$
$+c\delta(r+\delta)[n^2(\delta+\rho) - cnz]$.

Appendix B

B-1 : Non negativity constraint of the optimal input use.

The condition for an interior solution is to have, $\forall t$, $u(t) > 0$. Remember the social optimal irrigation path :

$$u_i^{so} = \lambda \frac{1-\beta}{pc} + \mu_\infty \frac{S^W}{pc} + \frac{pb - p_e}{pc},$$

As λ^{so} is a strictly decreasing function of time, so is $u^{so}(t)$. To ensure that $\forall t$, $u^{so} > 0$ it is sufficient to show that $u_\infty^{so} > 0$.

$$\begin{aligned}
&u_\infty^{so} = \frac{1}{pc}[pb - p_e + \mu_\infty^{so} S^W + \lambda^{so}(1-\beta)] = \frac{N(u_\infty^{so})}{D(u_\infty^{so})},\\
&D(u_\infty^{so}) = [r\beta + 1 - \beta]\left[pc(r+\delta+\gamma)(\delta+\gamma) + n^2 pf(1-\beta)^2 + nDS^G\delta(1-\beta)^2\right] > 0,\\
&N(u_\infty^{so}) = (pb - p_e)(r+\delta+\gamma)(\delta+\gamma)(r\beta+1-\beta)\\
&\quad -dp\beta(\delta+\gamma)\left[nS^G\gamma(1-\beta) + S^W(r+\delta+\gamma)\right].
\end{aligned}$$

Consequently, $u_\infty^{so} > 0$ if its numerator is positive :

$$\frac{pb - p_e}{dp}(r + \frac{1-\beta}{\beta}) > \frac{nS^G\gamma(1-\beta)}{r+\delta+\gamma}.$$

B-2 : Open-loop case : saddle point property and non negativity constraint

The individual maximisation program is now :

$$\max B_i(u_i, Q_i, X) \text{ subject to (8.14), (8.15) and } u_i \geq 0.$$

Supposing the conditions of an interior solution hold, the MHDS is :

$$\begin{pmatrix} \dot{X} \\ \dot{\lambda} \\ \dot{Q}_i \\ \dot{\mu} \end{pmatrix} = \begin{pmatrix} -\delta-\gamma & \frac{n(1-\beta)^2}{pc} & 0 & \frac{S^W}{pc}(1-\beta) \\ pf & r+\delta+\gamma & 0 & -\gamma S^G \\ \gamma_i S^G & \frac{S^W}{pc}(1-\beta) & \frac{\beta-1}{\beta} & \frac{S^{W2}}{pc} \\ 0 & 0 & 0 & r+\frac{1-\beta}{\beta} \end{pmatrix} \cdot \begin{pmatrix} X \\ \lambda \\ Q_i \\ \mu \end{pmatrix} + \begin{pmatrix} n(1-\beta)\frac{pb-p_e}{pc} \\ 0 \\ S^W\frac{pb-p_e}{pc} \\ dp \end{pmatrix}$$

The resolution process is the same as in the previous case. Let $z'_1 = \frac{1-\beta}{\beta}(r+\frac{1-\beta}{\beta}) > 0$ and $z'_2 = (\delta+\gamma)(r+\delta+\gamma) + \frac{n}{pc}fp(1-\beta)^2 > 0$. The rest of the demonstration follows.

To derive the conditions for the nonnegativity of the control variables, the same methodology as in the optimal case.

$$\begin{aligned}
&D(U^{ol}_\infty) = [r\beta + 1 - \beta]\left[pc(r+\delta+\gamma)(\delta+\gamma) + npf(1-\beta)^2\right] > 0, \\
&N(U^{ol}_\infty) = (pb-p_e)(r+\delta+\gamma)(\delta+\gamma)(r\beta+1-\beta) \\
&-dp\beta\left[S^G c\gamma(1-\beta)(\delta+\gamma) + S^W(f(n-1)(1-\beta)^2 + c(r+\delta+\gamma)(\delta+\gamma)\right], \\
&u^{ol}_\infty > 0 \Rightarrow \frac{pb-p_e}{dp}(r+\frac{1-\beta}{\beta}) > S^W\left[1+\frac{f(n+1)(1-\beta)^2}{c(r+\delta+\gamma)(\delta+\gamma)}\right] + S^G\left[\frac{\gamma(1-\beta)}{r+\delta+\gamma}\right].
\end{aligned}$$

B-3 : Steady state stocks parameters - social optimum and open loop case

$$M_1 = (r + \delta + \gamma)(pb - p_e) > 0,$$
$$\mu_\infty = -dp/[r + (1-\beta)/\beta] < 0,$$
$$M_2^{so} = S^W(r + \delta + \gamma) + \gamma S^G(1-\beta) > 0,$$
$$M_2^{ol} = S^W(r + \delta + \gamma) + \gamma_i S^G(1-\beta) > 0,$$
$$\lambda_\infty^{so} = [-X_\infty^{so}(npf + D\delta S^G) + \mu_\infty \gamma S^G]/[r + \delta + \gamma],$$
$$\lambda_\infty^{ol} = [-X_\infty^{ol} pf + \mu_\infty \gamma_i S^G]/[r + \delta + \gamma],$$
$$z_2^{ol} = (\delta + \gamma)(r + \delta + \gamma) + \frac{n}{pc}(1-\beta)^2(pf).$$

B-4 : Stock paths parameters - social optimum and open loop case

$$\mu^m(t) = \mu_\infty^m$$
$$\lambda^m(t) = \lambda_\infty^m + (\lambda_0 - \lambda_\infty^m)e^{w_1^m t} = \lambda_\infty^m + (X_0 - X_\infty^m)\frac{w_{1\lambda}^m}{w_{1X}^m}e^{w_1^m t}$$
$$D^{so} = npf + D\delta S^G,\ D^{ol} = pf$$
$$w_2 = \frac{\beta - 1}{\beta} < 0$$
$$w_1^m = \frac{1}{2}[r - \sqrt{[r + 2(\delta+\gamma)]^2 + 4n(1-\beta)^2 D^m/pc}] < 0$$
$$w_{1X}^m = \frac{n(1-\beta)(\frac{1-\beta}{\beta} + w_1^m)}{n(1-\beta)\gamma_i S^G + (w_1^m + \delta + \gamma)S^W}$$
$$w_{1\lambda}^m = \frac{(w_1^m - w_2)pcD^m}{S^W(1-\beta)D^m - \gamma_i S^G pc(r + \delta + \gamma - w_1^m)}$$
$$w_{1Q}^m = 1\ ,\ w_{1\mu}^m = 0\ ,\ w_{2X}^m = w_{2\lambda}^m = w_{2\mu}^m = 0$$
$$w_{2Q}^m = 1$$

B-5 : General model with tax

$$\begin{pmatrix} \dot{X} \\ \dot{\lambda} \\ \dot{Q}_i \\ \dot{\mu} \end{pmatrix} = \begin{pmatrix} \frac{n(\beta-1)}{pc}\tau_2 - \delta - \gamma & \frac{n(1-\beta)^2}{pc} & 0 & \frac{nS^W}{pc}(1-\beta) \\ pf - \frac{\tau_2^2}{pc} & r + \delta + \gamma + \frac{\tau_2(1-\beta)}{pc} & 0 & -\gamma S^G + \frac{S^W}{pc}\tau_2 \\ \gamma S^G - \frac{S^W}{pc}\tau_2 & \frac{1-\beta}{pc}S^W & \frac{\beta-1}{\beta} & \frac{S^{W2}}{pc} \\ 0 & 0 & 0 & r + \frac{1-\beta}{\beta} \end{pmatrix} \cdot \begin{pmatrix} X \\ \lambda \\ Q_i \\ \mu \end{pmatrix} +$$

$$\begin{pmatrix} \frac{n(1-\beta)(pb-p_e-\tau_1)}{pc} \\ \tau_3 + \frac{\tau_2(pb-p_e-\tau_1)}{pc} \\ \frac{S^W(pb-p_e-\tau_1)}{pc} \\ dp \end{pmatrix}$$

$$z_2^t = \tau_2\left[(\delta+\gamma)(1-\beta)/pc + n(1-\beta)^2(r+\delta+\gamma) + (\delta+\gamma)(r+\delta+\gamma) + n(1-\beta)^2 f/c\right] > 0$$

Applying the same method as in the previous cases, the steady-state values of the variables are obtained :

$$\begin{aligned}
X_\infty^\tau &= \frac{1}{z_2^\tau}[M_1^\tau(\tau_1,\tau_3) + M_2^\tau \mu_\infty], \\
Q_\infty^\tau &= \frac{\beta}{\beta-1}\left[X_\infty^\tau \gamma S^G + S^W u_\infty^\tau\right], \\
u_\infty^\tau &= \frac{1}{pc}\left[pb - p_e - \tau_1 - \tau_2 + \mu_\infty S^W + \lambda_\infty^\tau(1-\beta)\right], \\
\lambda_\infty^\tau &= \frac{-1}{z_2^\tau}[M_3^\tau(\tau_1,\tau_2,\tau_3) - \mu_\infty M_4^\tau(\tau_2)], \\
\mu_\infty^\tau &= \mu_\infty^{ol} = \mu_\infty^{so}.
\end{aligned}$$

Furthermore, the paths are of the same form as in the previous case :

$$\begin{aligned}
X^\tau(t) &= X_\infty^\tau + (X_0 - X_\infty^\tau)e^{w_1^\tau t}, \\
\lambda^\tau(t) &= \lambda_\infty^\tau + (X_0 - X_\infty^\tau)\frac{w_{12}^\tau}{w_{11}^\tau}e^{w_1^\tau t}, \\
Q_i^\tau(t) &= Q_\infty^\tau + (Q_0 - Q_\infty^\tau)e^{w_2^\tau t} + \frac{X_0 - X_\infty^\tau}{w_{11}^\tau}(e^{w_1^\tau t} - e^{w_2^\tau t}), \\
u_i^\tau(t) &= \lambda^\tau(t)\frac{1-\beta}{pc} - X^\tau(t)\frac{\tau_2}{pc} + \mu_\infty\frac{S^W}{pc} + \frac{pb - p_e - \tau_1}{pc}.
\end{aligned}$$

$$M_2^t = M_2^{ol} \ , \ M_1^t = \frac{n(1-\beta)}{pc}\left[(r+\delta+\gamma)(pb-p_e-\tau_1)-(1-\beta)\tau_3\right]$$

$$M_3^t = -\frac{n(1-\beta)}{pc}[\tau_2\tau_3 + pf(pb-p_e-\tau_1)] - (\delta+\gamma)[\tau_3 + \frac{\tau_2(pb-p_e-\tau_1)}{pc}]$$

$$M_4^t = \frac{n(1-\beta)}{pc}[S^W pf - \gamma S^G \tau_2] + (\delta+\gamma)[\frac{S^W \tau_2}{pc} - \gamma S^G]$$

To assess the impact of the various tax parameters on the steady state values, we derive the following expressions : $\partial M_1^\tau/\partial\tau_1 < 0 \ \partial M_1^\tau/\partial\tau_3 < 0$. As $z_2^t > 0$ and $\partial z_2^t/\partial\tau_2 > 0$, the steady state groundwater stock is clearly negatively affected by the individual, collective and mixed tax parameters. However the impact on the associated co-state variable is not analytically tractable. Indeed, the following relationships apply :

$$\frac{\partial M_3^\tau}{\partial\tau_1} > 0 \ , \ \frac{\partial M_3^\tau}{\partial\tau_2} < 0 \ , \ \frac{\partial M_3^\tau}{\partial\tau_3} < 0 \text{ and } \frac{\partial M_4^\tau}{\partial\tau_2} \text{ undet.}$$

The eigenvalues, which dictate the speed of accumulation, are :

$$w_1^\tau = [A(\tau_2)+E(\tau_2)]/2 - \sqrt{(A(\tau_2)-E(\tau_2))^2 + 4D(\tau_2)B}/2 < 0 \text{ and } w_2^\tau = I < 0$$

It is straightforward to notice that none of the taxes under study has an impact on the value of w_2^τ. However, τ_2 does appear in w_1^τ. The first order derivative of w_1^τ with respect to the relevant tax parameter are as follows :

$$\frac{\partial w_1^\tau}{\partial\tau_2} = 1/2[(1-\beta)(1-n) - 1/2\frac{M_6^{\tau\prime}(\tau_2)}{\sqrt{M_6^\tau(\tau_2)}}] < 0,$$

$$\text{with } M_6^\tau(\tau_2) = \tau_2^2\frac{(n-1)^2(1-\beta)^2}{2pc} + \tau_2(n+1)(1-\beta)(r+2\delta+2\gamma) + CTE.$$

B-6 : Optimal tax parameters

The system $\{X_\infty^{SO} = X_\infty^T \ , \ Q_\infty^{SO} = Q_\infty^T\}$ is solved with respect to different combinations of the tax parameters, as shown in the following table.

Case	**Solve w.r.t.**	**Opt. Par.**	**Case**	**Solve w.r.t.**	**Opt. Par.**
a	$\tau_1,\ \tau_2$,τ_3	$\tau_3(\tau_1,\tau_2)$	**e**	$\tau_3\ [\tau_1=\tau_2=0]$	$\hat{\tau_3}$
b	$\tau_1,\ \tau_2\ [\tau_3=0]$	$\tau_1(\tau_2)$	**f**	$\tau_2\ [\tau_1=\tau_3=0]$	$\hat{\tau_2}$
c	$\tau_1,\ \tau_3\ [\tau_2=0]$	$\tau_3(\tau_1)$	**g**	$\tau_1\ [\tau_2=\tau_3=0]$	$\hat{\tau_1}$
d	$\tau_2,\ \tau_3\ [\tau_1=0]$	$\tau_3(\tau_2)$			

TAB. B·1 – Optimal steady state tax rates

When any of the parameters associated with individual performance is equal to zero (cases **c** and **d**), then the optimal tax takes the form of a pure ambient tax, with a tax rate equal to $\hat{\tau_3}$. Case **a** constitutes a case of over-information. Indeed, when the three parameters are available, any choice of τ_1 and τ_2 and associated $\tau_3(\tau_1,\tau_2)$ allows attaining the optimal steady state (in particular when τ_1 and τ_2 are set equal to zero). When the pure collective parameter τ_3 is not available, the optimal tax is an input tax (cases **b** and **g**).

To derive the path tax parameters, the methodology used by (Benchekroun and van Long 1998) is used. The objective is to have $u^{so}(X)$ and $\tau^*(\tau_1,\tau_2,\tau_3)$ satisfy the first order conditions of open loop agents subject to the general tax scheme. In other words, it is to induce open loop agents, subject to the optimal tax scheme $\tau^*(\tau_1,\tau_2,\tau_3)$, to use the optimal reaction function $u^{so}(X)$. Remember the following socially optimum first order condition on input use and collective stock co-state variable path :

$$p(b-cu_i) = p_e - \lambda^{so}(1-\beta) - \mu^i S^W, \tag{8.16}$$

$$\lambda^{so} = \lambda_\infty^{so} + (X_0 - X_\infty^{so})\frac{w_{1\lambda}^{so}}{w_{1X}^{so}}e^{w_1^{so}t} = \lambda_\infty^{so} + \frac{w_{1\lambda}^{so}}{w_{1X}^{so}}\left[X - X_\infty^{so}\right]. \tag{8.17}$$

Replace λ^{so} with its value taken from (8.17), and derive (8.16) with respect to time to obtain :

$$pcu'(X)\dot{X} = -\tau_2\dot{X} + \dot{\lambda^\tau}(1-\beta). \tag{1}$$

Knowing that :

$$\dot{\lambda^\tau} = (r+\delta+\gamma)\lambda^t + pfX + \tau_2 u_i + \tau_3 - \mu_\infty\gamma SG,$$

$$\lambda^\tau = \lambda_\infty^\tau + \frac{w_{1\lambda}^\tau}{w_{1X}^\tau}\left[X - X_\infty^\tau\right],$$

and rearranging (1), a first order polynom of X is obtained :

$$C_1(\tau_2)X + C_0(\tau_1, \tau_2, \tau_3) = 0 \ , \ \forall X \Rightarrow C_1(\tau_2) = 0 \text{ and } C_0(\tau_1, \tau_2, \tau_3) = 0.$$

Solutions are obtained for the following combinations of parameters : $\{\tau_1, \tau_2\}$ and $\{\tau_2, \tau_3\}$. This confirms the needs for a stock-dependant mixed tax to induce optimality along the whole time horizon. A standard input tax, or a pure ambient tax, is not sufficient to induce the agents to irrigate in a optimal way, and consequently accumulate optimaly both individual root zone salt stocks, and the collective groundwater stock.

Appendix C

C-1 : Shadow cost analysis

$$
\begin{aligned}
u_{ik}^2 &= \frac{\bar{R}_k\alpha_k D_{ik} + n_k\delta_k^2[\rho_p B_{ik} - \rho_E - \lambda_k - 2\alpha_k\sigma_k(\lambda_k - \Gamma_k)]}{n_k(\alpha_k^2 D_{ik} + \delta_k^2\rho^p C_{ik})},\\
a_{ik}^2 &= \frac{-\bar{R}_k\delta_k\rho_p C_{ik} + n_k\alpha_k[\rho_p B_{ik} - \rho_E - \lambda_k - 2\alpha_k\sigma_k(\lambda_k - \Gamma_k)]}{n_k(\alpha_k^2 D_{ik} + \delta_k^2\rho^p C_{ik})}.
\end{aligned}
$$

C-2 : Diagonal strict concavity of the payoff function

Following Krawczyk (2005), the joint payoff function $f(\mathbf{u},\mathbf{a},\mathbf{r}) = \sum_k \sum_i r_{ik} B^i(u_{ik}, a_{ik})$ is diagonally strictly concave for fixed $\mathbf{r} > 0$ if the pseudo Hessian of f is negative definite.

$$
\frac{\partial f(\mathbf{u},\mathbf{a},\mathbf{r})}{\partial u_{ik}} = r_{ik}\left[\rho_p B_{ik} - \rho_p C_{ik} u_{ik} - \rho_E - \alpha_k\varepsilon_k\right] + \sum_j r_{jk}(-\varepsilon_k\alpha_k),
$$

$$
\begin{aligned}
\frac{\partial^2 f(\mathbf{u},\mathbf{a},\mathbf{r})}{\partial u_{ik}^2} &= r_{ik}\left[-\rho_p C_{ik}\right] \leq 0,\\
\frac{\partial^2 f(\mathbf{u},\mathbf{a},\mathbf{r})}{\partial u_{ik}\partial u_{jh}} &= 0.
\end{aligned}
$$

$$
\frac{\partial f(\mathbf{u},\mathbf{a},\mathbf{r})}{\partial a_{ik}} = r_{ik}\left[-D_{ik} a_{ik} + \varepsilon_k\delta_k\right] + \sum_j r_{jk}(\varepsilon_k\delta_k),
$$

$$
\begin{aligned}
\frac{\partial^2 f(\mathbf{u},\mathbf{a},\mathbf{r})}{\partial a_{ik}^2} &= r_{ik} D_{ik} \leq 0,\\
\frac{\partial^2 f(\mathbf{u},\mathbf{a},\mathbf{r})}{\partial a_{ik}\partial a_{jh}} &= 0.
\end{aligned}
$$

Hence the pseudo Hessian of f is a diagonal matrix, and its eigenvalues are negative : it is negative definite.

VU et PERMIS D'IMPRIMER

Montpellier, le

La présidente de l'Université Montpellier I

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DOCTORAT DE L'UNIVERSITE MONTPELLIER I

FACULTE DES SCIENCES ECONOMIQUES ARRETE DU 25 AVRIL 2002

RESUME : Cette thèse est consacrée à l'élaboration d'instruments de politique environnementale. Le premier objectif de ce travail est d'analyser l'utilisation de taxes dynamiques basées sur une mesure de la performance de groupe. L'analyse porte sur la caractéristique principale de ces instruments, l'interdépendance qu'ils introduisent entre les agents. En effet, leurs fonctions de gain dépendent alors de l'effort total procuré par le groupe. L'échelle d'analyse est le district d'irrigation et la performance du groupe se réfère au stock d'eau souterraine. Différentes taxes sont analysées : une taxe sur les intrants, une taxe ambiante et une taxe sur les intrants dépendant du stock. Ces taxes illustrent différentes combinaisons de prise en compte de la performance individuelle et de la performance collective dans l'élaboration d'instruments de politique publique. Les résultats montrent qu'introduire une composante basée sur la performance de groupe est nécessaire pour inciter les agents à adopter un comportement optimal sur l'ensemble du sentier d'accumulation. D'où l'intérêt de combiner des incitations individuelles et collectives. Le second objectif de ce travail est d'analyser l'élaboration de marchés de droits pour gérer des externalités couplées. Les districts d'irrigation sont alors placés dans le contexte plus général du système rivière, de sorte que l'analyse porte sur les interactions qui se développent entre les initiatives de gestion de l'eau à différentes échelles (district et rivière). Afin de gérer à la fois la pénurie d'eau et la salinité d'irrigation dans chaque district, deux types de droits sont considérés : des droits d'extraction et des droits de recharge. Ces derniers autorisent la production d'une certaine quantité de percolation qui recharge la nappe. Le principal résultat de cette analyse est que l'existence de corrélations entre les externalités ne réduit pas le nombre d'instruments nécessaire pour les gérer.

TITLE : CONTRIBUTIONS TO THE DESIGN OF POLICY INSTRUMENTS : an application to irrigation-induced salinity in Australia.

ABSTRACT : The focus of this thesis is on the design of policy instruments to manage environmental issues, with a particular interest in dynamic taxation schemes and cap and trade systems in the context of irrigation-induced salinity. A first goal of the thesis is to investigate the use of dynamic taxation schemes based on a measure of group performance, as a way of implementing the notion of collective responsibility. The analysis focuses on the main characteristics of group performance based instruments, the interdependence they introduce among the agents. Indeed, when subject to group performance based instruments, agents' payoffs result from the effort provided by the group. The analysis is set at the catchment scale, and the collective result is the groundwater stock. Various taxes are investigated, including a time-independent standard input tax, a state-dependent ambient tax and a stock-dependent input tax. These taxes are illustrative of various ratios of individual performance and collective performance in the design of the policy instrument. One of the results of this analysis is that including a group performance component in the policy instrument is necessary to induce the agents to behave optimally along the whole time horizon. Hence the interest for mixing individual and collective incentives in the design of policy instruments. A second aim of the thesis is to address the design of cap and trade systems to manage multiple coupled externalities. The catchments are replaced within the broader context of the river system, and the analyses relate to the interactions developing between water management initiatives at the catchment and the river scales. To manage both water scarcity along the river and irrigation-induced salinity in each catchment, two types of water rights are considered : standard diversion rights and recharge rights that allow right-holders to produce a certain amount of percolation. This analysis poses the question of the number of policy instruments needed to manage correlated externalities. It also raises issues associated with the implementation of cap and trade systems at different scales. The main result of this analysis is that the correlation existing between the externalities doesn't rule out the need for a policy instrument to manage each externality.

DISCIPLINE : Sciences économiques (section 05)

MOTS-CLES : taxes dynamiques, performance de groupe, cap and trade, externalités couplées, théorie des jeux non-coopératifs, jeux différentiels, marchés d'eau, salinité d'irrigation, Australie.

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