Southern Alberta Resource Economics Centre

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Source water protection through regulation, best management practices, and economic instruments

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ABSTRACT

Managing the use of shared resources, such as water, is an extraordinarily complex issue. Non-point sources of pollution (NPS) are increasingly regarded as a dominant threat to source water supplies, and aquatic ecosystems. As these problems will only become exacerbated by climate change, finding unique and innovative solutions that are economically feasible represents a significant challenge to policy makers. "Command and control" legislation for NPS pollution is generally scanty, due to the intrinsic difficulties in designing effective policy, as the nature of NPS pollution makes it challenging to ascertain who and what should be targeted. Voluntary programs are often the dominant form of NPS pollution control, although this policy is generally criticized as being too soft. Most voluntary programs involve the implementation of best management practices (BMP), although some BMPs are enforced by provincial legislation. A wide range of BMPs exist, including those that function to reduce pollution inputs, control and manage soil runoff and erosion, and provide a barrier for pollutants to access watersheds. A wide variety of economic incentives exist to improve the participation in voluntary programs, including green payments, and financial government assistance. Other economic or market based instruments (MBI) that can play a role in source water protection include emission taxes, input taxes, ambient taxes, and water pricing. Market-based systems based on well-established property rights are generally regarded as integral to maintain the balance between the ecological and economical aspects of resource use. Market based instruments have been successfully established to reduce greenhouse gases and smog emissions into the air, and nutrients and salts into source waters. While the development of effective market systems has been at times a slow and arduous process. An examination of the successes and failures of existing MBIs may equip policy makers to better develop tailored solutions.

1. INTRODUCTION

The standard measures used to address point sources of pollution are largely rendered ineffective when attempting to translate these policies to the control and management of non-point sources of pollution, for a variety of reasons. The greatest challenge in addressing non-points sources of pollution lie in the difficulties of measuring actual emissions from individual dischargers; individual emissions can only be inferred based on ambient water quality, which includes the emissions from multiple sources within a sub-basin (Russell & Clark, 2006). Without the capacity to measure individual emissions, designing policy to charge or trade for emissions becomes exceedingly difficult and complex (Russell & Clark, 2006). Assessing non-point sources of pollution discharge is extraordinarily information intensive; emissions are dependant upon a wide variety of factors such as soil type, topography, climate, weather, crop patterns, etc. Thus, indirect approaches to approximate emissions through assessing inputs or production practices are criticized because these measures often fail to take into account such externalities (Russel & Clark, 2006). The intrinsic complexities of non-point sources of pollution have resulted in a diversity of potential strategies to control and manage this type of water contamination. Generally, management of non-point sources of pollution can be grouped in three categories; “command and control” regulation, voluntary adoption of land management practices that limit the introduction of pollutants into source waters, and economic instruments, including a wide variety of market based systems that are employed around the world. Elucidating the most politically, administrative and economically effective means to produce positive changes is an extraordinarily challenging task (for detailed review, see Shortle & Horan, 2001).

2. Command and Control

2.1. Developing effective legislation for non-point pollution

Shortle and Horan (2001) describe some fundamental questions that must be addressed in order to assess the most successful strategies that can be used to combat non-point sources of pollution, which largely stem from the agricultural sector. Firstly, it must be established whom policies, legislation, and regulations should target. Assigning the responsibility for non-point sources of pollution to individuals is no easy task (Shortle & Horan, 2001). Assessing pollution flows from individual farms is economically and technically impractical; the presence of pollutants in adjacent surface waters does not necessarily implicate the nearest farm, making it difficult to identify the source that should be targeted. As there exist both natural and anthropogenic sources of pollution, identifying individual operations responsible for pollution become increasingly complex. Non-point sources of pollution are largely unobservable, emissions can be highly random, and the transport and ultimate location of emissions is stochastic (Shortle & Horan, 2001). Thus, the nature of non-point sources of pollution poses significant problems for policy design. It is infinitely easier to target the manufacturers of products that contribute to non-point sources of pollution, such as commercial fertilizers and pesticides, than the use of such products (Shortle & Horan, 2001).

A second challenge is identifying *what* problems legislation, regulations, and policies can target. This type of command and control approach targets design standards, performance standards, or anything in between (Dowd et al., 2008). Regulations that dictate how pollution dischargers manage their facilities target design standards (Dowd et al., 2008). Examples of this type of targeting are legislations that require the adoption of best management practices, such as Alberta's Agricultural

Operations and Practices Act. Alternatively, regulations that target performance standards of a facility focus on levels of pollution discharges (Dowd et al., 2008). Shortle and Horan (2001) outline three categories of performance on which pollution control measures can focus. Most simply, legislation can target inputs, such as commercial fertilizer, and pesticides, and those practices, such as manure application and overgrazing, which are known to contribute to pollution. Emission proxies are also valuable tools that can be employed to assess the degree of pollution, and assign responsibility for pollution. A universal soil loss indicator on cropland, originally designed by Wischmeier and Smith (1987), or indicators developed to assess inefficient nutrient management, are examples of emission proxies. Finally, pollution control measures can potentially target ambient concentration of pollutants in water supplies (Shortle & Horan, 2001).

3. Voluntary Programs

The use of voluntary measures to improve economic outcomes is an alternative to forced participation through legislation. Generally, such volunteerism is celebrated for being democratic in nature, promoting public participation, allowing farmers to maintain control over the management of their operations, and incorporating industry expertise (Harrison, 1999 cited by Saunders). Pollution dischargers generally favour voluntary measures for the lax regulation and absence of compliance measures. Regulators favour the low costs and time commitment these voluntary programs require. Dowd et al. have identified three primary types of voluntary environmental programs (2008 p. 152):

> (1) Unilateral Action—a polluter or group of polluters voluntarily self-regulate and implement an environmental improvement program.
>
> (2) Negotiated Agreement—the government and stakeholders negotiate the terms of an environmental improvement program.
>
> (3) Voluntary Government Program—a government-sponsored program establishes

eligibility criteria, rewards, and requirements for participation.

Voluntary programs are only as successful as the level of participation; thus, the motivations to engage in voluntary programs are of great interest. Environmental stewardship embraces the notion of an ethical and moral responsibility to engage in those activities that contribute to environmental sustainability. The participation in certain programs is driven by economic gains that are either a direct result, or a by-product of participation, ie. market-based incentives. Finally, the participation in some programs is accomplished through government-based incentives, such as subsidies, or fines for non-compliance (Dowd et al., 2008). Some features of successful voluntary programs are the presence of complementary monitoring programs to gauge the environmental outcomes, and the ability to rely on threats of non-compliance if voluntary action fails (Dowd et al., 2008).

Best Management Practices

Measures of agricultural pollution control rely on land-use, production, and management decisions, among other things. Best Management Practices (BMP) play an instrumental role in reducing non-point sources of pollution. These practices are approaches based on science that help achieve a desired outcome, such as pollution management. A wide variety of BMP are available for farmers. Although some are binding, namely those outlined in the Agricultural Operations and Practices Act, the vast majority serve as voluntary recommendations. BMP vary in their costs, complexity, effectiveness, and in the amount of scientific support behind the practice. Outlined below are some of the most effective BMP for reducing agricultural pollution. Generally, BMPs can fall under three categories: those that reduce the input of pollutants, those that control runoff and erosion, and those that serve as barriers and buffers to source waters.

3.1 BMP that Reduce Inputs

3.1.1. Nutrient Management

Nutrient management planning is a practice in which only the necessary amount of nutrients are applied to a field to promote an optimal crop yield. This practice ensures that excess nutrients will not be applied, and thus not become available to source waters through soil erosion. Nutrient management planning first requires an assessment of the amount of nutrients required to produce the desired crop yield, which must additionally incorporate externalities such as climate and soil types (Hilliard & Reedyk, 2007). Soil testing is used to assess the levels of required nutrients already in the fields. The difference between these two values determines the proper quantities of nutrients that must be added to produce the target crop yield. Commercial fertilizers provide the opportunity to select specific types of fertilizer, which contains nutrients in plant-accessible forms; it is more difficult to determine these quantities in manure or organic fertilizers, as they vary greatly in specific nutrient content, and similarly vary in the rate at which the nutrients become accessible for plant uptake (Hilliard & Reedyk, 2007). Thus, consultation with specialists is recommended for those managers interested in using manure or organic fertilizers (Hilliard & Reedyk, 2007). Finally, nutrients should be added at the right time, to ensure they are available when crops most require them (Hilliard & Reedyk, 2007).

3.1.2. Manure Application

Manure is a valuable resource that contains plant essential nutrients that can be used to improve crop yields in place of fertilizers. Organic nitrogen in manure is more stable than nitrogen found in commercial fertilizers. These organic nutrients are released more slowly into the soil environment, which can be both an advantage and a disadvantage to crop growth. The nitrogen components in commercial fertilizers are highly water-soluble and mobile nitrates, or ammonium (which is readily converted to nitrate). Thus, commercial fertilizers present an increased risk of nitrate leaching into groundwater and transport into surface waters.

Manure is additionally valuable due to the benefits its application provides soils; manure application enhances soil structure and aggregation, increases water infiltration, and increases water holding capacity. These enhanced soil features result in reduced runoff and soil erosion (Gilley & Risse, 2000; Wortmann & Walters, 2006). Manure application has been demonstrated to cause greater microbial biomass increases than commercial fertilizers (Dormaar et al, 1988; Peacock et al., 2001). In addition, evidence suggests that manure application alters the microbial community structure, and effectively enhances both beneficial microbial populations and activity (Witter et al., 1993; Ritz et al., 1997; Peacock et al., 2001). Finally, the recycling of manure nutrients, rather than engaging in the manufacture and production of commercial fertilizers, has additional energy conservation benefits.

The benefits of manure application are contingent upon best management practices for manure storage and application, which are described below. Proper manure management reduces nitrate leaching, denitrification, phosphorus runoff, ammonia losses to the atmosphere, and erosion by wind and water (Alberta Agriculture and Rural Development (AARD), 2004). In addition, proper manure management ensures optimum crop yields. The Agricultural Operations Practices Act (AOPA) outlines the regulations for manure application, and manure storage; these regulations apply to all livestock operations, custom manure applicators, and anyone else that handles or applies manure. The Natural Resources Conservation Board monitors and enforces these regulations.

AOPA should be consulted to determine the location, amount, and timing of manure application in Alberta. AOPA regulations specifically dictate where manure can be applied relative to a common body of water and water wells. The surface application of manure on forages, direct seeded crops, snow or frozen grounds is not recommended, though if necessary, AOPA regulations outline the minimum distance manure applications can be from an open body of water. It is not recommended to apply manure in low, wet areas, or prior to predicted rainfalls. Incorporating manures after application, via tilling the manure into the lower soil profiles, reduces the potential for manure runoff into adjacent surface waters.

Many BMPs related to the timing of manure application have been developed. During winter, it is recommended that manure goes into storage, as manure applied to snow-covered or frozen grounds poorly infiltrates into soils and are prone to runoff. During the spring, it is recommended that manures be applied to lands prior to seeding, and applied early on pastures. Manure should be applied to well-drained soils, and incorporated into surface soil profiles within 48 hours. In the fall, manure should be applied before the ground freezes, and incorporated within 48 hours (AOPA, 2008)

3.1.3. Manure Storage

AOPA includes standards for the construction and location of solid and liquid manure storage facilities and manure collection areas (AARD, 2004). Manure storage facilities must be located 1m above the water table, above the 1 in 25 year floodplain, 100 meters from springs and well, and 30m from a common body of water. The construction of solid and liquid storage facilities is governed by AOPA regulation. Short-term (less than 7 months) solid manure storage must be located using similar guidelines as manure storage facilities, though they must additionally be located 150m away from neighboring residences, and at a minimum setback distance determined by the slope of the land towards a common body of water.

3.1.4. Integrated Pest Control

Integrated pest management is a BMP used to maximize the effectiveness of pesticide use, from an environmental standpoint as well as an economical one. This practice involves the integration of cultural, biological, chemical and mechanical methods to control pests (AARD, 2004). Thus, integrated pest management is founded on a holistic approach to manage pests, which results in fewer applications of pesticides, a reduced impact on soils and source waters, and a recognition of the complexities surrounding the advantages of beneficial and competitive organisms, the complexities of pest population dynamics, and the maintenance of crop quality and yield (AARD, 2004).

3.2. BMP that Control Runoff and Erosion

A wide range of natural and agricultural factors contributes to the risk of soil erosion and runoff. Soil erosion is a function of climate, soil types and topography, in addition to cropping practices, and is caused by wind or precipitation events. This process can result in the transportation of topsoils, which may contain a variety of pollutants, into surface waters. Some soil types are more prone to erosion than others; fine to medium textured soils are the most prone to erosion (Triplett & Dick, 2008). Clay soils or soils that develop shallow impermeable layers can form compacted regions that reduce water infiltration, which contributes to runoff (Triplett & Dick, 2008). Soils with high organic contents have a higher absorption capacity than those low in organic matter (Triplett & Dick, 2008). The topography contributes to the degree of soil erosion, as water runoff accelerates down unobstructed slopes, increasing erosive action.

Agricultural practices, especially tillage practices, play a significant role in soil erosion. Maintaining a high soil quality is a first step in reducing the risk of soil erosion by water.

Alberta Agriculture and Rural Development (2005) has developed an assessment of soil erosion risk across Alberta based on wind and water erosion risks, in addition to land management practices, specifically cultivation practices (Conservation tillage, conventional tillage, summerfallow, and no till). Identifying areas at high risk for soil erosion can help determine where BMP will be most effective.

3.2.1. Tillage Practices

Excessive tillage destroys soil structure, reduces soil porosity, and contributes to soil compaction of lower soil profiles. Conservation tillage is a practice that allows crop residues to remain, effectively protecting the soils surfaces after crops have been planted, through limiting the levels of tillage. It has been well illustrated that no-tillage or reduced tillage practices greatly reduce soil erosion (Fig. 1) under a variety of conditions including heavy precipitation, steep slopes, and erosion-prone soils (See Philips et al. 1980 and Rasmussen, 1999, for reviews). In some cases, no-tillage practices, also called direct seeding, reduce soil erosion to almost nothing (Philips et al., 1980). Schmidt *et al* provide compelling evidence that conservation tillage is an effective means in flood control, reduced tilling practices reduce soil crusting, reduce water erosion, and increase water infiltration (2001).

Generally, soil erosion decreases as crop residue and plant cover increases. Tillage tends to bury crop residues in deeper soil layers. Soils bearing crop residues and plants are less prone to water erosion than bare soils, for several reasons. Crop residues and plants absorb the impact of precipitation, which provides the impetus for soil particles to become displaced. Physically, more energy is required to dislodge soil particles from untilled soils than tilled soils. Additionally, plant root systems provide a stabilizing force that binds soil particles, providing increased resistance to precipitation-induced transport. For the same reasons, wind erosion is also reduced by the presence of plant cover and crop debris (Triplett & Dick, 2008).

There is, however, a tradeoff for tillage. As reducing tillage increases water infiltraton, the risk of groundwater contamination also increases. Additionally, as more organic content accumulates in untilled soils and decays, nutrients become increasingly available for runoff. Incorporating manure applied to fields via tillage is a best management practice (see section 3.1.2), which is known to reduce the runoff of manures into water supplies. Additional disadvantages of tillage reduction are outlined in Table 1 (Triplett & Dick 2008). This is just one example that illustrates the potential complexities and tradeoffs involved in making environmentally sound land practice decisions.

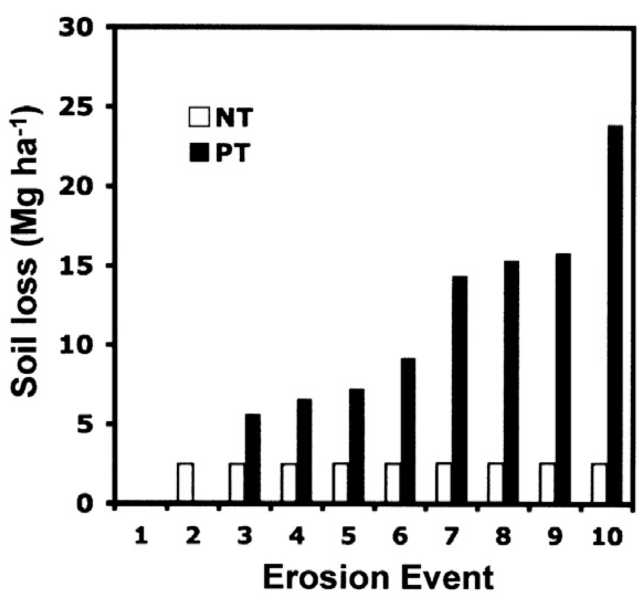

Figure 1. Cumulative soil loss from No Tillage (NT) and Plow Tillage (PT) watersheds at Coshocton, OH, for the years of 1970–1973. The erosion events are all rainfall events that produced runoff and erosion during this time period. To visualize the NT values,

they were multiplied by 10 before being plotted in the graph (Cited by Triplett & Dick, 2008, originally from Harrold & Edwards, 1974).

Table 1. Advantages and disadvantages of applying tillage for crop production.

Tillage advantages
1. Reduces competition from weeds
2. Creates a seedbed that promotes uniform and efficient crop establishment
3. Buries residues and reduces disease inoculums
4. Reduces extreme surface roughness and thus facilitates more efficient equipment use
5. Promotes water infiltration in loosened soil—at least initially
6. Stimulates organic matter mineralization and nutrient release
7. Leads to more rapid warming in the spring if tillage is applied before cold dormant period
8. Mixes fertilizer nutrients throughout the rooting depth of the crop
9. Disrupts habitat and life cycle of harmful pests
10. Breaks up surface soil crusts
11. Reduces surface and subsurface compaction
Tillage disadvantages
1. Disrupts aggregates and reduces soil structure
2. Supports greater extremes in soil temperatures
3. Exposes the soil to raindrop impact
4. Promotes soil erosion
5. Destroys macropores
6. Destroys cracking patterns in soils with vertic characteristics
7. Promotes increased soil drying and reduces water use efficiency
8. Causes compaction and tillage pan formation
9. Disrupts life cycles of beneficial organisms such as earthworms
10. Decreases the store of organic nutrients in the soil
11. Fragments fungal hyphal networks
12. Accelerates oxidation of labile organic matter and interferes with soil carbon sequestration
13. Causes increased fertilizer immobilization/fixation
14. Requires greater inputs of fuel and energy
15. Requires greater investment in equipment

(From Triplett and Dick, 2008)

3.2.2. Soil Compaction and Overgrazing

Compacted soils are prone to increasing runoff, as the movement of air, microbes, and water is impeded. Excessive grazing of cattle contributes greatly to soil compaction; one hoof of a mature cow is estimated to exert 20-30 pounds of pressure per square inch (AARD, 2004). Additionally, reducing soil porosity reduces the productive capacity of the forage pasture or crop. Uneven plant growth, nutrient deficiencies, plant water stress, and shallow root systems are evidence of soil compaction (AARD, 2004).

3.2.3. Cover Cropping

Cover crops, crops grown during periods when agricultural fields would otherwise be fallow, are valuable tools used for sustainable agriculture. Their presence greatly reduces soil erosion and agricultural runoff between growing seasons, while additionally reducing weed growth, diseases, and helping to manage excess nutrients in the soil. Multiple studies illustrate the benefits of cover cropping in reducing runoff, soil and phosphorus losses, and nitrate levels (Dabney et al., 2001).

3.2.4. Increasing the Organic Content in Soils

Both the chemical and physical properties of soil are improved by the presence of organic matter. Organic matter accumulates in soils when plant and crop residues are added faster than decomposition occurs. Organic matter serves to bind soil particles in aggregates, which reduces soil erosion. Additionally, organic matter improves the structure of soils, increases soil porosity and has an increased water holding capacity. The decomposition of organic matter releases stored nutrients essential to microorganisms and plants, increasing crop yields. Organic matter decomposes more rapidly in dry warmer conditions, and tends to accumulate in cooler, wetter conditions. Practices that improve crop yields, such as fertilizers and manures, increase the quantity of crop residues, which decompose into organic matter. Overall increasing the organic content in soils reduces soil erosion, and confers both conservation and economic benefits (Pimental, 1996).

3.2.5. Increased Irrigation efficiency

Irrigation accounts for 71% of consumptive water use in Alberta (Nicol et al., 2009). A variety of irrigation technologies are known to increase water use efficiency, such as pivot irrigation systems, low-pressure spray nozzles, low clearance sprinkler systems, and trickle and drip irrigation systems (Harker et al., 2008; CSFSP, 2009). Reducing the amount of runoff generated by irrigation systems is instrumental in reducing soil erosion events, especially in regions like the Prairies, which rely so heavily on irrigation for agriculture. Surface runoff results in major losses of irrigation water, as this water becomes unavailable for crops. Increasing the efficiency of on-farm water delivery systems, in addition to curbing water consumption, can play a role in reducing the transport of agricultural pollutants into source waters through soil erosion. On average, 15% of sprinkler irrigation, 69% of flood irrigation (Tate et al., 2000), and 18% of border dyke irrigation becomes runoff (AARD, 2001). The use of wetting agents and soil surfactants can also contribute to reducing soil erosion; water repellency of soils reduces water infiltration, and thus contributes to runoff (Mitra et al., 2006).

3.3 BMP that serve as Barriers and Buffers

3.3.1. Vegetative Filtration Strips and Constructed Wetlands

When methods to reduce soil erosion are insufficient, or fail, reducing the level of pollutants in agricultural runoff can be viewed as a last resort. The use of biofilters, such as vegetative filter strips or constructed wetlands, are capable of removing significant levels of a wide variety of agricultural pollutants before runoff reaches source waters.

Vegetative filter strips are an effective best management practice that can effectively reduce agricultural runoff, and control levels of pollutants contained in runoff. The presence of these buffers enhances the opportunity for pollutants in runoff to be infiltrated in soil profiles, absorbed by vegetation, and adsorbed by soil particles (Dillaha et al., 1988). Vegetative field strips also serve to decrease the capacity for sediment and pollution transport, as surface runoff is slowed down. These strips have been shown to reduce levels of nitrogen (Ikenberry & Mankin, 2000; Fajardo et al., 2001; Blanco-Canqui, 2004) phosphorus (Chaubey et al. 1995; Abu-Zreig et al., 2003; Blanco-Canqui, 2004), pesticides (Tingle et al., 1998; Krutz et al., 2005), sediment (Dillaha et al., 1988; Blanco-Canqui et al., 2004), fecal coliforms (Lim et al., 1998; Fajardo et al., 2001; Tate et al., 2006), Giardia cysts (Winkworth et al., 2008) and *C. parvum* oocysts (Atwill et al., 2002; Trask et al., 2004).

The effectiveness of vegetative filter strips (VFS) is a function of multiple factors. The amount of soil erosion occurring in the runoff area, which determines the amount of sediment reaching the buffer is important, as the over-accumulation of sediment reduces the efficiency of the filter. The water retention time of the VFS determines how long processes such as infiltration, absorption, and adsorption have to function; this is a function of the quality of the vegetation employed in the buffer, and the width of the buffer. The infiltration rate of soil determines the speed and effectiveness at which nutrients will be incorporated into lower soil profiles. The uniform velocity of water flow ensures constant volumes will be filtered; high volumes of water may not be effectively filtered. Thus, the construction design of VFS is integral to its efficiency. High quality of the VFS can be ensured through proper maintenance, which may include grading, reseeding, harvesting, and mowing (Dillaha et al., 1988).

Constructed wetlands also function as effective biofilters, as similar infiltration, adsorption and absorption processes result in the uptake, transformation, and/or breakdown of nutrients and microorganisms. Constructed wetlands are also used to treat industrial and municipal wastewaters, in addition to agriculture. Research indicates that with proper construction and maintenance, constructed wetlands are effective tools that can be used to reduce non-point sources of agricultural pollution (Hammer, 1992).

3.3.2. Fencing around watercourses

The unrestricted access of livestock to watercourses can cause severe damage to riparian areas. Livestock trample the banks of watercourses, which can remove riparian vegetaions, and contribute to soil compaction, both of which lead to increased soil erosion (Ontario Ministry of Agriculture and Rural Affairs, 2010). Additionally, livestock wastes deposited in source waters contributes to nutrient loading, and the loading of waterborne pathogens like *E. coli*, Cryptosporidium, and Giardia. The use of fencing is a simple way to restrict livestock access to watercourses.

4. Market-Based Instruments

Initially, many of societies' resources were common property; there was a free right to pollute since there was plenty of fresh air; many fisheries were open because there was plenty of fish, there was free access to take water out of rivers, since supply was plentiful, water was clean, and access was relatively limited. When the use of a resource increases, however, an open access common property approach generally results in resource overuse, and conflicts over resource access arise. The lack of individual property rights is an incentive for individuals to maximize their immediate gain in competition with, and at the expense of, other resource users. The resource and its users then suffer: air pollution becomes apparent and health problems increase; fish become scarce, and species are threatened; water flow declines, which causes water quality problems, with serious consequences for consumptive users and ecosystems. This predicament was first articulated by Hardin's essay, The Tragedy of the Commons, in 1968.

Thus, ethics and stewardship can only go so far to motivate voluntary behavioral changes necessary to reduce nonpoint pollution. Enforceable regulations or economic incentives are believed necessary to generate a significant decrease of agricultural water pollution (Shortle & Abler 2001). However, economists have criticized the use of regulatory mechanisms to address environmental pollution as economically infeasible (Pearce & Turner, 1990; Baumol & Oates, 1988). Additionally, the diffuse and immeasurable nature of non-point pollution makes for exceedingly complex policy design. Thus, increasingly policy makers are looking towards the role that economic incentives can play to address issues related to the overuse of a particular resource.

According to economists, the overuse of resources is caused by 'market failure', which in turn is mainly caused by externalities. Externalities are costs associated with using the resource, which are not payable by the user, and benefits derived from using resources, which cannot be gained by the user. Companies will continue to pollute because they do not have to pay the cost imposed on third parties such as the cost of treating people from diseases caused by air pollution, or repairing damage to buildings, machinery or equipment belonging to other companies. Water users causing water quality problems do not have to pay the costs of treating downstream waters, or pay the costs of increased wear and tear on water-using equipment in urban areas. The price of using the resource is therefore cheap, and users continue to consume more and more. Market-based or economic instruments simply use economic signals or incentives in the form of rewards or penalties to direct the behavior of resource users by changing the market signals that guide firms' decisions, and thereby internalize externalities.

Two main categories of market-based or economic instruments exist: 1) the use of taxes, levies, subsidies, tax concessions and prices; and 2) the use of property rights and markets. A third group of instruments is emerging, building on education and consumers' desire to be environmentally correct. This group includes instruments such as eco-labelling and certification and environmental management standards. It has been argued that economists expand the legal approach to property rights, which mainly looks at whether a system is fair, easily understood and enforceable, to give people incentives to make efficient use of the resource that they own (Scott, 1999).

The first set of instruments has been most widely used. The problem with this system is that the regulator will have to determine the correct level of taxes or subsidies to encourage companies to invest in improved technology for more efficient resource use. To do that successfully, the authority needs to know the cost structure of the individual industries, as well as the marginal cost of increased resource use. The disadvantage of this approach is that the authority has no real way of controlling the final level of resource use, and can only try to influence it by sending appropriate price signals. If use is still too high, the authority can increase the taxes or levies, or increase subsidies to invest in better technologies. These economic instruments also do not allow companies with different cost structures to make trade-offs to minimize their costs.

4.1. Emissions tax

Emissions taxes are based on the polluter pays principle; simply, producers pay for every unit of pollution that is discharged into the environment. Ideally, the tax is equal to the social costs of pollution. Emission taxes require the measurement of individual emissions, which must be accomplished at a reasonable cost for such taxes to be feasible (Weersink et al., 1998). An alternative to assessing actual non-point source emissions is to apply emission taxes to estimates, as developed by simulation models (Weersink et al., 1998). Arguably the biggest challenge is to develop accurate models that take into account the environmental fate and transport of specific agricultural pollutants, in addition to the management practices and physical land characteristics that influence the likelihood of specific pollutants contaminating source waters (Weersink et al., 1998). Thus, there is the potential that the costs of such complex model development and application may outweigh the benefits of developing accurate emissions taxes (Weersink et al., 1998).

4.2. Ambient Tax

Another approach to reducing non-point pollution is to apply an ambient tax or subsidy based on the ambient concentrations of pollutants in waters (Segerson, 1988) When pollution levels fall below a predetermined threshold, regulators pay dischargers a subsidy; when pollution levels exceed the threshold, dischargers are fined (Dowd et al., 2008). As Shortle and Hortle (2001) point out, this mechanism can be applied to both point and non-point sources of pollution. However, levels of agricultural non-point sources of pollution are highly contingent upon a variety of externalities, such as weather, that are beyond the control of dischargers. Thus, an ambient tax system could unfairly penalize dischargers who may be actively engaging in efforts to reduce non-point sources of pollution. Additionally, the nature of water pollution makes it difficult to assess and target those dischargers responsible for waterway contamination, or responsible for reductions waterway contamination. Xepapadeas (1997) proposed that by partitioning such pollution taxes into two parts, whereby measurable effluent pollution would be subject to an effluent tax, while pollution not measured by a monitoring program would be subject to an ambient tax, such ambient taxes might be more realistic. The success of such taxes is contingent on reliable, accurate, and well-developed monitoring programs, which continues to be a pervasive challenge in addressing non-point sources of pollution.

4.3. Input Tax

Input taxes are those fees attached to products that contribute to environmental pollution. Fees attached pesticides or commercial fertilizers are examples of some input taxes. The fees generated by input taxes in parts of the United States and the European Union largely function to fund pollution control programs, rather than to reduce the use of taxed inputs (Larson et al., 1996). While research indicates that increasing the levels of taxed inputs would be an effective way to reduce the consumption of pollutants (Dowd et al., 2008), the taxes would have to be unrealistically high (Shortle and Horan, 2001). Thus, designing effective input-based incentives is challenging, especially given the lack of studies that examine the failures and successes of these instruments (Dowd et al., 2008).

4.4. Government Financial Assistance

Efforts to reduce non-point sources of pollution are most often accomplished through the government funding of various programs through grants and subsidies

(Johns, 2000). One of the most common grants employed are via cost-sharing programs that subsidize the costs of implementing BMPs. These programs are believed to be the most effective policy regimes regarding the challenges surrounding control and management of non-point pollution (Johns, 2000). As described in section 3, BMPs are an effective way to reduce the likelihood of contaminants entering source waters. In Alberta, the Environmental Farm Plan, which was developed and implemented by a partnership between the Prairie Farm Rehabilitation Administration, Alberta Agriculture and Food, and the Alberta Environmental Farm Plan Company, assessed the specific environmental risks that producers face on their farms. Assistance in implementing BMPs, through cost-sharing and technical assistance, was accomplished through the Canada-Alberta Farm Stewardship Program (CAFSP). Across Canada, 51,000 producers have participated in the Environmental Farm Plan, resulting in the implementation of 24,000 BMPs; producers and the federal government have invested over $400 million in the implementation of BMPs from 2003, to 2009, when the program ended (AAFC, 2008a).

Growing Forward represents the current Canadian agricultural policy framework, which offers a variety of subsidized programs that can be used to fund best management practices. In Alberta, available programs include the manure management plan, the integrated crop management plan, and the grazing and winter-feeding management plan; all of these programs are designed to help producers develop a work plan that maximizes the use of a particular resource while minimizing the environmental impact (AAFC, 2008b). The water management plan is designed to provide both technical assistance and incentives to create long-term water management plans for farms, which would presumably include water quality sustainability (AAFC, 2008b).

4.5. Green Payments

Green payments are those monies given to producers in exchange for taking steps towards the reduction of non-point pollution. The difficulties in measuring non-point source pollution, and the stochastic nature of non-point emissions, requires that green payments be awarded based on observable specific actions, rather than reductions in emissions (Horan et al., 1999). This type of economic incentive would likely attract those producers not involved in the subsidized cost-sharing programs described above. Payments could be structured in one of two ways; input subsidies, or contract subsidies. In the former, subsidies are based on reducing the use of inputs that contribute to pollution, or increasing the use of inputs that contribute to pollution reduction, or some combination of the two (Horan et al., 1999). In this way, producers would have the opportunity to participate, depending on the specific subsidized input. Contract green payments are awarded to those producers that successfully implement a specific set of actions designed to reduce non-point pollution (Horan et al., 1999).

4.6. Water Pricing

There exists a longstanding perception that Canada has an abundant, if not limitless, supply of freshwater; media reports have previously concluded that Canada boasts as much as 40% of the world's freshwater supply (Sprague, 2007). In reality, Canada possesses approximately 20% of the world's freshwater resources, and although seemingly substantial, this figure too is misleading. Most of this freshwater is non-renewable, as it is composed of "fossil waters" left over from melting continental ice sheets, tied up in glaciers, or as inaccessible groundwater (Boyd, 2003). The renewable freshwater resources of Canada are only approximately 6.5 percent of the global total (Sprague, 2007). While Canada's relatively sparse population affords a significant amount of renewable freshwater on a per-capita basis; 99,000m3/person/year, which is nearly an order of magnitude greater than the per person per year quantities afforded to citizens of the United States, Canada's spread-out population does present a problem regarding water distribution. Nearly 60% of Canada's renewable freshwater supply flows into the Atlantic Ocean and Hudson Bay, while 90% of Canada's population resides in the south (Environment Canada, 2009); similarly, approximately 80% of Alberta's freshwater is in the North, with 80% of its population in the south (Alberta Water Smart, 2006).

Access to water is considered a basic human right, a social necessity, and a vital environmental resource (Abu-Zeid, 2001). However, one cannot deny the economic characteristics associated with drinking water, given the costs of water treatment, distribution, monitoring, and wastewater treatment. In Canada, provincial governments control raw water pricing, while municipalities set consumer water rates (Renzetti, 2007). Effective water pricing is suggested to play an integral role in improving water conservation, water allocation, and water quality (Abu-Zeid, 2001). Renzetti (2007) describes four features that should guide water pricing; first, water prices should be high enough to provide enough revenue to cover costs related to water and sewage treatment services. Second, prices should reflect the "full social costs" of water, including costs associated with raw water, water storage and water distribution. Third, water prices should reflect the impact that our water use has on the environment, and public health; water prices should serve to protect aquatic ecosystems, and promote water conservation. Finally, water prices should be equitable, and reflect the fact that safe drinking water is a human right, and should be affordable to all Canadians, included low-income households (Renzetti, 2007).

Multiple water pricing structures exist (see Table 2). According to a 2004 report by Environment Canada, approximately 23% of Canadian households pay a flat rate for water, 46% are charged by a constant unit rate, 8% pay via declining block rates, and 23% pay via increasing block rates (Environment Canada, 2009) Those Canadian consumers who pay a flat rate for water consume 70% more water than consumers who pay by volume (Environment Canada, 2009). Canadian municipal water prices are relatively low compared to water prices by other OECD countries (Figure 2).

Table 2:Types of Rate Structures (From Tate et al., 2001)

Type	Description	Conservation Potential
Flat rates	Customer pays a fixed rate per time period for unlimited access to public water supply	None; encourages excessive use
Declining block rates	Water use is divided into two or more volume ranges or blocks. The rates decline progressively for water use in the larger blocks.	Progressively decreasing as water use increases
Constant unit rates	Charges per unit of water use (e.g. cubic meter) are constant through the range of usage.	Moderate to good
Increasing block rates	Similar to decreasing block rates except that rates increase progressively through the range of usage.	Progressively increasing as water use increases

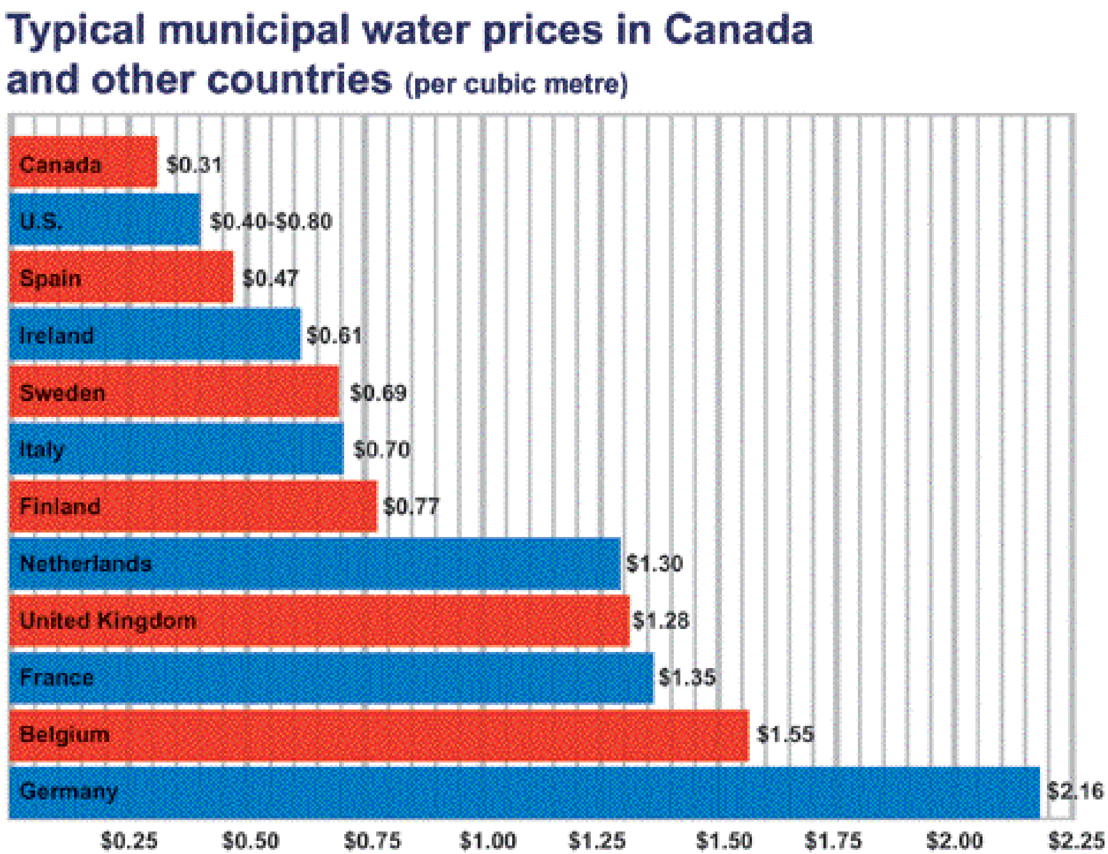

Figure 2. World Commission on Water for the 21st Century, 1999. "The Poor Pay Much More for Water... Use Much Less - Often Contaminated." (www.worldcouncil.org).

4.7. Virtual Water

It would be imprudent to discuss the value of water without addressing the value of Albertan commodities that require an abundant, reliable source of freshwater. Alberta boasts a strong agricultural export economy, which is unquestionably dependant upon waters provided by irrigation. The production of 1 kg of wheat requires approximately 1300L of water; the production of one kilogram of beef requires 15,500L (Chapagain & Hoekstram 2004). Similarly, the oilsands projects in Northern Alberta are also water-intensive, requiring an average of 2 to 4.5 barrels of freshwater to produce a single barrel of oil (Griffiths et al., 2006), of which 66% are exported in the United States (Tar Sands watch, xxxx). Over 1 million barrels of oil is extracted from the Athabasca oil sands/day, which means approximately 1.3-2.9 million barrels of freshwater are virtually exported to the United States every day. Thus, while Alberta currently does not engage in bulk water transfers to the United States, Alberta does exports large quantities of "virtual water". Virtual water represents the water required, and thus embedded, within a trade commodity (Allan, 1998). While it is out of the scope of this report to comment further on the economies of agriculture and oil/gas developments, the concept of virtual water does play a significant role in addressing the value of water in Alberta. The implications of virtual water in the South Saskatchewan river basin, given threats of water scarcity and the dominance of agriculture, are undeniable.

5. Market-Based Systems

Under a market-based system, the ecologically sustainable level of resource use is established, and tradable use rights or permits are issued to resource users. If existing resource use is in excess of the ecologically sustainable level, the total use needs to be reduced. With markets in place, those with the lowest cost of reducing resource use will do so, while those with higher cost will buy resources from those with lower cost. Alternatively, those with the most efficient and highest value use will buy resources from those with an inefficient low value use. In this way, the marginal cost of reducing resource use will be equalized across firms and, as a consequence, total cost will be minimized, and total economic benefits from using the resource will be maximized – at little or no political cost. The main issues are how to determine the ecologically sustainable level of use, and how to initially allocate this level to existing users. By examining the various market-based initiatives, it is possible to identify the factors that have facilitated the establishment of such markets, and also to evaluate the practical experiences with their operation, and the potential lessons that can be learned from such experiences for the next generation of water markets. This report will therefore discuss the establishment, program specifics, and progress of various selected markets including those related to air quality, water quality, and wetland habitat.

5.1. Examples of Market-based Trading Schemes: Air Pollution

In the late 1970s, the U.S. Environmental Protection Agency (EPA) allowed States to use variants of tradable permits to control localized air pollutants. This allowed companies with different cost of pollution abatement to make trade-offs to minimize the total abatement cost to industry, which had started to escalate in the early 1970s (Stavins, 1998). In 1986, new legislation provided the foundation for the introduction of quasi-tradable permit systems including netting, offsets, bubbles and banking provisions (Lotspeich, 1998). This emission trading policy is the oldest and, until the Acid Rain program was introduced in 1995 (see section 5.1.2), the most far-reaching attempt to implement a marketable permit system.

5.1.1. The Los Angeles Emission trading Programs

Although the emission-trading program operates nationally across the United States, the market for Los Angeles has been by far the most active (Foster & Hahn 1995). It covers the most significant stationary sources of pollution for the five major air pollutants: hydrocarbons, nitrogen oxides, particulate matter, sulphur oxide, and carbon monoxide. The products traded in the emission market are called ‘emission reduction credits’ (ERC). These credits can be generated in a number of ways: if a plant is shut down or a particular piece of equipment is no longer used, if a plant reduces its emission below the regulatory standard through changes to the production process, or through the introduction of more efficient equipment. Credits are normally issued in terms of a volume of pollutant reduced per period of time.

Netting enables a company to introduce a new source of emission within an existing plant, and avoid the stringent emission limits that would normally apply by reducing emission from another source within that plant. When using netting, a company can only use credits generated from its own sources; this approach therefore is also known as ‘internal trading’. Offset provisions are effective within so called ‘non-attainment areas’, areas that have not met specific ambient standards. Within such areas, new polluting developments are not permitted. Offset provisions enable such developments to take place by the developer reducing emissions from existing sources within the attainment area by an even larger amount. The ERCs for offsets could be generated either by ‘internal trading’ or by ‘external trading’. External trading is achieved by buying credits from another company. The bubble concept places an imaginary bubble over existing sources of pollutants, and considers all dischargers within that bubble to have a single point of emission. Dischargers within the bubble are allowed to trade ERCs between them, as long as the total emission from the bubble stays below the limit. Finally, banking enables companies to save ERCs for later use or sale.

5.1.1.1. Pre-1994 Emission Trading Program

Inter-company transfers dominated early markets, and very few arms-length transactions involving the exchange of money took place, due to significant uncertainty about future reductions in emission levels. In Los Angeles, market activities for offsets took off in 1991 when institutions were introduced to facilitate small transfers (Foster & Hahn 1995). Prices of ERC varied significantly, and this variation did not diminish over time, even as market players became more familiar with the system, as market information became more available, and when market activity was high (Foster and Hahn, 1995). Foster and Hahn (1995) attributed this to the heterogeneity of the ERC as a product. First, they observe that there are several ways in which an ERC can be created; through the introduction of new technology, through change in the fuel used, or via the shutdown of machinery. Each of these methods of ERC creation is associated with significantly different levels of cost, and the level of cost associated with creating ERC in turn influences the price at which companies are willing to sell. Second, if ERC had first been deposited under the banking scheme, the seller has incurred significant approval cost. Third, there can be significant geographic distance between the trading parties, which has regulatory implications for the terms of the transaction. Fourth, the contract under which the trade takes place can differ significantly, especially with regard to how the risk associated with the transfer is shared. These features, in addition to frequently changing regulations, attribute to highly fluctuating ERC prices. However, minimum prices in real terms remained rather constant over the entire period. This suggests a reservation price that does not vary with market conditions. While minimum prices were stable, maximum and mean prices fluctuated and were generally increasing. Prices increased significantly from 1990 to 1991 due to an increase in demand caused by significant rule changes.

The relatively low market activity, despite the significant pressure for development in the L.A. region, is associated with the transfer process and uncertainty. The transfer process has two distinct parts: finding a trading partner, and obtaining approval. Problems associated with identifying a suitable trading partner were found to be one of the major obstacles to trade. In the early market, more than 25% of all transfers fell through during the negotiation process. It was argued that one of the reasons for this was the diversity of industries involved in the process, including industries that normally do not deal with each other. To reduce this uncertainty, companies employed three strategies: 1) the use of internal trading, with more than 25% of all trades investigated being internal trades; 2) the use of intermediaries, and; 3) the use of the ERC Bank. Since the latter part of the 1980s, companies have been able to contact the ERC bank to access a list of companies that have deposited credits, allowing buyers a means to identify potential sellers.

The transfer approval process is the next hurdle. The process can be lengthy (between 5 and 12 months) and is associated with large administrative fees and levies. In addition to these, sellers must pay significant costs in providing the substantial documentation required to certify the reduction. In many instances, the magnitude of the transfer cost exceeded the market value of the credit exchange. The process is also associated with a high level of uncertainty about the outcome. Only 20% of applications reaching this stage were approved as proposed; half of the remaining 80% were rejected outright, and the other half were finally approved at a lower level of ERC. Given the substantial cost, the long transfer period, and the considerable uncertainty of the outcome, the transfer/approval process constitutes a significant impediment to trade.

The regulatory environment has also added to the uncertainty, with a significant impact on market activities. The regulatory rules change quite rapidly, with 59 rule changes over the twelve-year period from 1980–1992. A major review took place in 1990 with four major changes. Firstly, a new retrospective rule was implemented, requiring even small new emitters to purchase credits, which resulted in a pent up demand, as small emitters had been previously exempt. Secondly, there were changes to the way credits from equipment shutdowns were computed. Under the old rules, credits were measured against the current emission baseline; under the revised rules, credits were measured against the emission that would have resulted had best technology controls been employed. Thirdly, the region was divided into 38 zones, and restrictions were made on trade from east to west. Fourthly, under the old system, shutdown credits had to be traded within 90 days; at the end of that period, credits deposited in the ERC Bank would be cut down to 20% of their original value and could no longer be traded. Under the revised rules, shutdown credits can be banked and traded indefinitely. At the time the new rules were introduced, 11,500 tons per year of shutdown credits were lying dormant in the ERC Bank because they were un-tradable under the old rule. As a result of these changes, activities in the market increased significantly, as demand was high from the many small emitters now forced to buy credits, and supply was high due to the large volume of previously dormant credits.

Policy makers are often as concerned about the spatial distribution of pollution as they are with the total emission level. Thus, market mechanisms for air pollution must take this into account. The Los Angeles market did this in two different ways before and after 1990. Before 1990, trade was allowed across the region, but exchange rates were in place varying from 1.2:1 to 1.6:1, depending on location (with those in environmentally most sensitive regions having to buy 1.6 units in order to be allowed to emit one unit). After 1990, this was changed to a standard exchange rate of 1.2:1, although the market was segmented into 38 different regions with rules controlling trade between them. The overall rule was that trade was only allowed from east to west, or within regions. This change of rules caused an already thin market to be divided into a number of even smaller and thinner markets, with distinctly different demand and supply conditions, resulting in a sharp increase in price dispersion.

5.1.1.2. Post-1994 emission trading program: old vehicle scrapping program

With the Clean Air Act Amendments of 1990, states were given further authority to develop economic incentive programs. This enabled Los Angeles to introduce two new pollution-trading programs (Drury et al. 1999). Rule 1610 was introduced in 1993, allowing stationary polluters such as factories and electricity generators, to avoid installing expensive pollution control devices through the purchase of pollution credits created by scrapping old, high-polluting cars. Under this program, licensed car scrappers purchased and destroyed old cars. The car scrapper then acquired ERCs based on the predicted emissions of that car, had it not been destroyed, which can then be sold to stationary polluters. Under Rule 1610, producers have to buy 20% more credits than what would have been achieved through compliance with technology standards.

The effectiveness of such voluntary accelerated vehicle retirement (VAVR) programs to reduce air pollution is dependant upon several factors. Firstly, how much sooner the vehicle was retired due to the program; 2) how much distance the retired vehicle would have driven if not retired for the program; 3) what the emissions generated by the retired vehicle were; 4) how the owner of the retired vehicle subsequently traveled; 5) how many miles the replacement vehicle traveled, if one was purchased, and; 6) what the emissions generated by this replacement vehicle are (Hsu & Sperling, 1994; cited from Dill, 2004).

Drury et al. (1999) criticize this program, arguing that it over-allocates emission credits, and is prone to fraud. Anecdotal evidence suggests that many of the cars destroyed under the program in fact did not result in reduced emissions; the car body was destroyed, while the engine causing the pollution was often taken out and sold, and thereby continued to pollute. Drury et al. (1999) also criticize the fact that many of the cars scrapped for credits were at the end of their useful life, and therefore would have been destroyed anyway. However, current vehicle eligibility requirements include an evaluation to ensure that the vehicles possess three remaining years of useful life (South Coast Air Quality Management District, 2009). Further concerns are raised by the instances related by some inspectors of inoperable cars being towed to the scrapping facility with the sole purpose of obtaining the payment under the program (Drury et al., 1999).

In general, evidence suggests that while vehicle retirement programs are contributing to reducing air pollution, likely reductions are not as significant as predicted (Dill, 2004). Overall, it is believed that vehicle retirement programs are most successful at reducing reactive organic gas emissions, while reductions in NO_x emissions are virtually negligible (Dill, 2004). However, as Dill (2004) points out, the reduction of reactive organic gases is often the primary goal of these programs.

5.1.1.3. Smog trading program: Regional Clean Air Incentive Market (RECLAIM)

The second pollution-trading program is the Regional Clean Air Incentive Market (RECLAIM), which was introduced in 1994, and is claimed to be the world's first urban smog trading program. RECLAIM replaces many of the old technology-based regulations aimed at reducing emissions of SO_2 and NO_x. When RECLAIM was introduced, all existing users were issued with a tradable permit to pollute based on their history of pollution (Drury et al. 1999). These permits were explicitly stated not to constitute property rights (Murtough et al. 2002). Emission was thereby capped, and a 'declining cap-and-trade' program was initiated (Drury et al. 1999). Trade was also restricted by geographical region, and the number of permits required per unit of emission depends on the location of the emitter (Murtough et al. 2002). Each year until 2003, reductions were made to the quantity of units of pollution allowed per permit. Companies then have to reduce their emission by improving production techniques, or by purchasing additional permits from other companies. This allowed arbitration between companies with different abatement costs.

According to the US National Center for Environmental Economics (NCEE 2001, reported in Murtough et al. 2002), the RECLAIM project has been successful in achieving reductions in air pollutants at significantly lower costs than what would have been possible using traditional command-and-control instruments. Actual and forecast permit prices for all years from 1994 to 2010 were far lower than the marginal abatement costs that companies would have incurred under existing, or proposed regulation. Compliance with permit conditions is monitored and enforced by end-of-pipe measurements via computer link to the enforcing agency. Even though industry initially complained about the cost of this monitoring, the NCEE have estimated savings in compliance costs of US$580 million over a ten-year period compared to the use of regulatory emission limits.

Regardless of these economically beneficial outcomes, the program has been criticized on a number of accounts. Drury et al. (1999) argue that both the LA programs create environmentally unjust outcomes by concentrating a large proportion of pollution within certain areas (so called toxic hot-spots), with additional adverse health impacts. These areas are predominantly low-income communities with a population of ethnic and racial minorities. They argue that RECLAIM is an inefficient air quality policy in at least four different ways: 1) it does not significantly reduce air pollution; 2) it discourages technological innovation as a mean to reduce pollution; 3) it decreases public involvement in environmental decision making; and 4) it increases the difficulty of monitoring and enforcing emission reductions.

Since industry bodies were heavily involved in lobbying during the creation of the program, the initial setting of the baseline emission levels resulted in heavily inflated allocations well in excess of actual emissions. Consequently, even though allowable emissions have declined each year according to regulation, early ‘reductions’ in emissions were largely illusory. For the first two years of the program, initial allocations were as much as 40% greater than actual emissions; as a result, the prices of RTCs were lower than the cost of controlling emissions, thus facilities did not participate in the market, and price controls were not properly addressed by the market (California Air Resources Board (CARB), 2006). In the first three years of the program, actual industrial NO_x emissions had declined by at most 3%, compared to a projected 30% aggregate reduction (Drury et al. 1999). In comparison, NO_x emissions from industrial polluters declined by about 37% from 1989 to 1993 as a result of technology-based regulation. Not only does the trading in excess allocations result in illusory environmental gains, it drives down prices of credits and thereby reduces the incentive to invest in actual emission reductions or technological innovation (Drury et al. 1999). When RECLAIM was adopted, the average cost of reducing NO_x by the use of best available technology was about US$12,500 per ton, or about 50 times the price of NO_x credits in 1997. A company would place itself at a competitive disadvantage if it invested in innovative pollution control in such a market. The U.S. EPA has similarly acknowledged the failures of this program. In a 2002 review of the program, the EPA admitted that the high emission caps resulted in the failure to reduce emissions as much as initially desired, and the reductions of emissions gained through this program cost more per tonne than reductions would have cost through a command and control strategy (U.S. EPA, 2002).

Southern California's Air Quality Management District (SCAQMD) claim the program can be considered a success, as the aggregate emissions of NO_x and SO_2 are below allocations every year, and NO_x and SO_2 emissions have decreased from initial allocations by the target goal of 75% and 60%, respectively, by 2003 (CARB, 2006). Additionally, the SCAQMD boasts that compliance with the program is generally high; in 2004, all facilities complied with SO_2 Reclaim Trading Credits holdings, while 13 facilities exceeded their NO_X holdings (CARB, 2006). Thus, facilities have generally met target reductions, or exceeded them, although whether the reduction targets were sufficient is debated.

In 2005, 749 trades for over 17,000 tons of NO_x and SO_2 credits were approved by the AQMD (CARB, 2006). Since its inception in 1994, over 370,000 tons of NO_x and 130,000 tons of SO_2 have been traded; those trades involving RTCs with an assigned monetary value were worth over $780,0000 million (CARB, 2006). Generally the RECLAIM market is considered a thinly traded market, with active credit training only occurring among a small group of facilities. This is potentially due to the costs of RTC transfers, the high umber of low-emitting facilities involved in the program, and the uncertainty of the RECLAIM market (CARB, 2006). Although the market is not especially strong, it is considered diverse, as evidenced by the transfers, swaps, contracts, and participation by brokers, traders, and investors; in 2005, mutual funds entered the RTC market (CARB, 2006).

5.1.2. SO_2 Emission Trading: The Acid Rain Program

The U.S. nation-wide SO_2 cap-and-trade program was made possible with the 1990 Clean Air Act Amendments under the authority set out in Title IV. It is also called the Acid Rain Program, since one of the driving forces behind its introduction was to stop escalating problems with acid rain in the northeastern United States, and southern Canada. Acid rain occurs when SO_2 and NO_x react in the atmosphere to produce sulphuric and nitric acids. These acids then fall to earth, often hundreds of kilometers downwind from where they were emitted, and can have devastating consequences for aquatic life, and forests (Joscow & Schmalensee, 1998). The predominant source of acid rain in the U.S. is SO_2 emission from coal-fired power plants, especially in the mid-west, where plants are fueled by local coal with high sulfur content. The 1990 amendment ended more than 10 years of political arguments involving several presidents, conflicting interest of States, the involvement of unions, and a period where the environmental movement gained increased power. Joskow and Schnalensee (1998) provide a revealing discussion and analysis of this process.

The Acid Rain Program has set a permanent cap on SO_2 emissions from electric generating units in the United States, and has consisted of 2 phases. During Phase 1 from 1995 to 1999, all larger and dirty electricity-generating units were required to reduce their emission by about 3.5 million tons per year (Joskow & Schmalensee, 1998). During Phase 2, beginning in 2000, virtually all fossil-fueled power plants in the U.S. were included in the program. Emissions from all units under the program are capped at 8.9 million tons, or about half of the total emissions generated by electricity utilities in the early 1980s. The program was implemented by issuing tradable allowances to all affected units free of charge for a 30-year period, generally in proportion to each unit's average annual heat input during the three-year baseline period from 1985-87 (Stavins, 1998). Emission units not used at the end of the year can be banked for use in future years or for sale (Ellerman, 2003). The EPA held back 2.8% off allowances issued to each unit to be sold by auction every year, and the unit holders then receive a proportion of the total proceeds. This auction provision was included in response to concern that an active market would not emerge, and that existing utilities would hoard their allocations and refuse to sell in order to prevent new entrants into the market.

This cap-and-trade program is supplementary to an extensive set of command-and-control regulations that has been in place since the early 1970s. These regulations impose either a limitation on emission rates, or technology mandates on individual units. Due to this underlying and complex set of regulatory instruments, the trading program under Title IV does not have to meet all environmental objectives since other regulatory instruments are available to avoid negative local health effects, and ensure that other environmental values, such as visibility, are preserved. In effect, these other regulatory instruments can, and often do, limit individual power plant's ability to participate in emission trading (Ellerman, 2003).

Ellerman (2003) provides a thorough ex-post evaluation of the program, through analyzing prices and market activities since the first sales in 1992 to late 2002, based on publicly available data from the Emission Exchange, the annual auction, and two major brokers. Prices have fluctuated during the ten-year period, but apart from the very early period with few transfers, prices paid on the exchange, via brokers, and on the annual auction, were generally indistinguishable. It was also found that the proportion of trade taking place between economically distinct organizations increased significantly over the first ten years. During the first years of market trading, only 10%–15% of all transfers were between economically distinct organizations, while in the latter years the split has been about 50%. This indicates that market participants have become familiar with the way the market operates, and are taking advantage of the cost savings offered by emission trading. During the early years from 1990 to 1995, very little trade took place as companies relied on cost-saving opportunities within their own plants, such as switching from high to low sulfur coal. Not until Phase 1 of the Acid Rain Program did trade take off in large volumes and numbers, and trade between economically distinct companies became more common. This development was spurred on by more consistent rules between states, and the emergence of an emission exchange, private brokers and an EPA auction facilitating trade and providing market information (Colby, 2000).

The estimates of cost savings provided by the introduction of efficient markets are considerable. Initial ex ante analysis indicated that compliance costs ranged from US$3.5 to 7.5 billion. These fell to a range of US$2.3 to 6.0 billion at the time the final details of Title IV were proposed; current cost estimates vary from US$1.0 to 1.4 billion (Ellerman, 2003). The reason for the declining compliance costs are because the marginal cost of abatement has been halved, with 80% of this reduction attributed to falling prices of low-sulfur coal relative to high-sulfur coal, while the remaining 20% is caused by technological change (Carlson et al., 2000). The main reason for the reduction in low-sulfur coal prices is reduced freight cost following the deregulation of the rail market. Carlson et al. (2000) estimated that allowance trading has reduced compliance costs by 25–35%. They further estimated that this gap would widen to almost 43% by 2010, as technological advances continue. The reduction in costs has not only been generated by the ability to trade emission reductions among users; part of this reduction has been caused by the general dynamics of the market. The fact that utilities are now facing various options for reducing emissions has provided them with the flexibility to combine the use of these options, and benefit from incremental improvements. This has been an incentive for experimentation and innovation. It has led to competition between providers of different sources of emission reductions, putting a downward pressure on cost that would not have happened under command-and-control regulations (Ellerman, 2003).

The combination of falling marginal compliance cost and relatively low allowance prices on the market has prompted earlier than expected cuts to SO_2 emissions, with firms that have generated early emission reductions banking the resulting credits against future periods where the cost of emission reduction might increase (Colby 2000). By the end of Phase 1, the level of actual accumulated reduction was twice that required. By 2001, some banked credits were activated, and the level of accumulated reduction was reduced to 29% more than that required; by the second half of this decade, accumulated reductions are expected to equal required reduction as the bank will be exhausted (Ellerman, 2003).

It is generally believed that the change from command-and-control regulation to a market approach under the Acid Rain Program has caused a reduction in administrative costs for both the regulator and the regulated, but no evidence exists to back up this claim (Ellerman, 2003). What is more profound is the significant change in the tasks required by the regulator and the regulated in order to administer the program. For the regulator, there has been a shift in enforcement from a labor-intensive intermittent inspection, which ensures compliance with technical standard, to data-intensive but continuous measurement and reporting of emissions, which requires the handling of more data and a greater emphasis on accounting.

With regard to those that are regulated, the system requires the installation of continuous monitoring equipment at a considerable cost to that party (estimated at 7% of direct compliance cost). However, these firms seem to be unanimously in favor of the new framework, presumably because the reduction in direct compliance more than offsets the additional cost of monitoring and allowance management (Ellerman, 2003). One major gain for the firms is a significant reduction in transaction costs. The creation of a standard unit of account in allowances and the lack of review of the individual transfer has avoided the very large transaction costs experienced under previous EPA trading programs. The right to emit has been converted into a readily tradable commodity, and as a result broker commissions are also low.

The Acid Rain Program has been heralded by many as a resounding success in reducing SO_2 emission at the lowest possible cost (Borough & Bourke, 1998; Arnold, 1999). The SO_2 cap for 2010 has been set at 8.95 million tones, which represents approximately half the emissions from electric generating units in 1980 (US EPA, 2009). In 2008, the combined SO_2 emissions were at 7.6 million tones, well below the cap of 9.5 million tones (US EPA, 2009). Reductions in NO_x emissions in 2008 were twice the reduction objective; while reductions are largely attributable to the Acid Rain Program, additional state trading programs have also contributed significantly. Data indicates that reductions in SO_2 emissions have resulted in improved air quality, with an overall 71%

reduction between 1980 and 2008 (US EPA, 2009). Between 2006 and 2008, the deposition of wet sulfate decreased by 30%, with additional reductions, though to a lesser extent, in the deposition of wet nitrogen (US EPA, 2009). Stream chemistry data also indicates an improvement in the quality of surface waters, as reductions in sulfate concentrations have been detected in sites in New England, the Adirondacks/Catskills, and Pennsylvania. However, several sites in the Central Appalachians indicate increases in sulfate concentrations (US EPA, 2009). Overall, stream ecosystems in the Northeastern United States appear to be recovering from acidification caused by acid rain pollutants, as evidenced by the increased acid neutralization capacity. The deleterious effects of acid rain on human health are well studied; acid rain is known to contribute to respiratory and cardiac ailments, the exacerbation of asthma, and premature death. Estimates of the health benefits of the acid rain program, based on a 2005 study conducted by Chesnut and Mills, suggest a value of US $170-410 billion dollars by 2010 (US EPA, 2009).

One of the major reasons for the environmental effectiveness of the program is a very high compliance rate, with virtually 100% of participating utilities complying every year. Such a level of compliance has never been achieved by any command-and-control regulation, under which utilities often get different forms of dispensation delaying or permanently relaxing the imposed standards. The reason for this is that a single standard imposes higher costs on some than on others, due to local conditions, and this has been used by some utilities to seek and get administrative relief. This plea for special circumstances cannot be presented under a cap-and-trade program since all sources face the same maximum burden – the price of an allowance (Ellerman, 2003).

5.1.3. Carbon Markets: The Kyoto Protocol

The discussion about carbon markets has been driven by a worldwide concern of climate change caused by the emission of greenhouse gasses to the atmosphere. This concern was first expressed on the international scene when the United Nations General Assembly in 1988 adopted resolution 43/53, which recognized that climate change is a common concern to the world community (Borough & Bourke, 1998). Following significant international debate from 1988 to 1997, a draft protocol was adopted in the meeting in Kyoto in 1997 known as the Kyoto Protocol. Under the protocol, the parties of developed countries, called “Annex 1” countries, agree to reduce their emission of greenhouse gasses (carbon dioxide, methane, nitrous oxide, sulfer hexafluoride, and all gases in the hydrofluoorocarbons and perfluorocarbons groups) to a proportion of their 1990 level. Under the protocol, 37 Annex 1 industrialized countries agreed to collectively reduce their greenhouse gas emissions by 5.2% from 1990 levels by 2008-2012. Emission reduction can be achieved through the introduction of cleaner production methods, or using cleaner fuels. Sequestration of carbon dioxide through planting of new forests or reforestation may also be used to meet the target. This provision acknowledges the fact that green vegetation absorbs and stores carbon through the photosynthesis process. Approximately 50% of the dry weight of a forest biomass is carbon (Murtough, 2002); these are known as ‘carbon sinks’.

The Protocol introduces three types of flexible mechanisms to assist countries to meet their stipulated targets: 1) international emission permit trading, where parties which have reduced their emission beyond their agreed target can sell credits to parties who need additional credits to stay within their target; 2) joint implementation, which allows countries to earn emission reduction units by providing financial support to another industrialized country for project level activities that reduce emissions or that sequester carbon in the host country; and 3) clean development mechanisms, which allow industrialized countries to invest in emission reduction projects or sinks in developing countries in exchange for ‘certified emission reductions’ (van Bueren, 2002).

Many researchers have argued against carbon trading, and claim that it is a cheap fix that will discourage innovation in emission reduction technology. There are significant concerns over the serious risks in international carbon trading, especially for developing countries where no cap exists on carbon emission. The third mechanism listed above has serious deficiencies that can be exploited, which will effectually increase rather than decrease carbon emission. An industrial country may get rid of an unclean burner in the home country and gain credits there while transporting the unclean burner to a developing country where it replaces an even more unclean burner and gets credits again under mechanism 3. Industrialized countries can move whole dirty processing plants to developing countries which have no cap, and can thereby continue to emit carbon gasses while helping the industrialized country meet its goal. Special attention has been paid to the old Soviet Union and some former ‘Eastern block’ countries, where economic stagnation has led to the closure of many dirty industries. As a result, Russia and some other countries have emission credits, which can be sold to industrialized countries so that they can meet their target, but without generating any real reduction. For a further discussion of these international implications see Driesen, 1998.

Canada’s Kyoto target was to reduce greenhouse gas emissions by 6% from 1990 levels, for the period of 2008-2012. As of 2006, Canada exceeded Kyoto targets by 29% (Environment Canada, 2008). When the Conservative government was elected into power in 2006, the Kyoto Protocol received little attention, which was alternatively shifted towards a “Made in Canada” solution to climate change. Part of this policy includes Canada’s Clean Air Act, which has intensity-based targets. Intensity-based targets are highly criticized as a means to sanction increased growth and pollution of heavily polluting, economically viable industries (CBC news, 2007a). In 2007, Environment Minister John Baird announced that Canada will not attempt to meet its Kyoto targets (CBC, 2007b).

5.2. Examples of Market-Based Trading Schemes: Water Pollution Markets

Overall, market-based approaches to improve water quality are not as frequent, or evolved, as market-based approaches for air quality. Such approaches can be used to address both point sources, and non-point sources of pollution. The following sections provide examples of the introduction of markets for both point source and non-point source polluters, as well as the ability to trade between them.

5.2.1. Point-Point and Non-point-Point Pollution Trading

There has been a strong push by U.S. State and Federal governments to develop formal management plans for watersheds where pollution exceeds acceptable limits. These plans set total maximum daily load limits (TMDL) for each watershed, at a level that ensures that water quality standards are met. The plans also set load limits for each point source and for non-point sources. Initially only point sources will be enforced, but the Clean Water Act gives the EPA the authority to make the limits on the agricultural sector enforceable as well. The development of the TMDL has been a significant incentive to develop water quality trading programs, because existing point sources are facing costly reductions in discharge in order to stay within the new quality standards. The EPA is actively encouraging the development of such programs, and there are now approximately 35 programs in various stages of development and implementation including both point to point, and non-point to point trading. Point-point trading allows firms with high costs of reducing discharge to buy credits from companies with low costs of reducing discharge.

Tradable point-non-point source credit programs are receiving increasing attention due to their low costs, and success these programs have in improving certain parameters of water quality. In these trading programs, dischargers can reduce the source outputs of a given pollutant to levels below present requirements to generate credits. In turn these credits can be sold to dischargers that may experience greater difficulties, particularly associated with high costs of treatment, in reaching effluent thresholds. This is termed the Emissions for Loadings (E-LO) method of point-nonpoint trading (Dowd et al., 2008), and involves estimates of emissions rather than actual emissions to assess credit values, since point and non-point source do not pollute equally (Horan & Shortle, 2002). Thus, these programs are cost-effective, flexible, and result in the desired environmental outcome (Obropta, 2004). These programs also have the capacity to provide support for projects, such as BMPs, which can contribute to improving water quality, and they can provide incentives to reduce levels of pollution beyond current regulated limits, and incentives for technological innovation (Obropta, 2004). Because these programs are based on environmental outcomes, there remains a focus on water quality. Another route of point-non-point source trading involves the generation of credits through Emission for Inputs (E-I) trading. In this method, farmers can generate credits by making a contract to implement approved Best Management Practices (BMP) (van Bueren, 2001). Dischargers trade changes in point-source emissions for changes in management practices, such as fertilizer use (Dowd et al., 2008).

5.2.2. Hunter River Salinity Trading Scheme (New South Wales, Australia)

The Hunter River, in New South Wales, Australia, and many of its tributaries possess a natural high level of salinity. This problem is significantly exacerbated by discharge of saline water into the river from 25 coalmines, 2 coal-fired power stations, irrigation, and other industries. Increased salinization imposes external environmental costs on other users within the catchment. It has been estimated that each one EC (electrical conductivity units) increase in river salinity causes a $10,000 per annum loss due to reduced agricultural yields, and the increased cost of supply and treatment (James, 1997). By 1992, salinity levels in the Hunter River reached 1,800 EC and exceeded the benchmark of 900 EC, 49% of days. In 1992 stakeholders started to work together to find a solution to the problem. Initially, a trial of managed discharge was introduced which showed that if discharge was restricted to high flow, the impacts on the river would be greatly reduced (O'Shea et al., 1997). Following this collaborative process and the development of real-time salinity monitoring along the length of the river, a pilot project commenced on 1 January1995 (New South Wales Environmental Protection Agency (NSW EPA), 2001). After a thorough review and economic evaluation of numerous other salt disposal measures (such as desalinisation, piping to the sea, variation of discharge scheduling etc.), the Salinity Trading Schemes emerged as the most cost effective way of meeting the environmental objectives of the catchment (NSW EPA, 2001).

The program is unique in the sense that it is the only scheme in the world based on real-time environmental conditions rather than modelled predictions. Initially 1,000 credits were issued, each allowing 0.1% of the maximum allowable salt discharge for each day. The credits were originally allocated by a hybrid grandfathering process. All mines were allocated credits according to a multicriteria formula designed to reward good environmental performance, which allocated 50% of all mine credits to operations that did not currently have a need to discharge. The power stations were allocated a fixed share based on their existing discharge entitlements (O'Shea et al., 1997).

River flows are distinguished either as "low", "high" or "flood" flows. At low flow no discharge is permitted, during periods of high flow discharge is permitted up to the maximum daily allowable level, and during flood flows no volume discharge limit applies.

The total allowable salt discharge into a given block of water as it passes through the river is set in real-time to ensure that the salinity levels do not exceed a critical limit at three monitored points along the river, and the total allowable discharge is divided among credit holders according to their proportion of total credits. The credits are tradable, giving them a monetary value. Companies now have an incentive to improve their on-site storage, or treatment facilities to allow them to store water on-site during low flows, and discharge during high flow. Better on-site treatment will reduce the need to discharge. In both instances companies will be able to sell their excess credits (NSW EPA, 2001). A 'sector credit discount factor' may be applied if necessary to protect water quality if participants in a particular sector acquire too many credits (Murtough et al., 2002).

During the first three years of the scheme very little trade took place, and mining operations had little need to discharge due to drought conditions (the most need to discharge from mines is caused by rainwater that has dissolved salt in disturbed soils around active mines (Murtough et al., 2002)). Also, most mines were not ready to make long-term trades because they had not confirmed their future needs under the new regime, and credit holders were uncertain about the financial value of credits (O'Shea et al., 1997). However since 1998 trade has picked up, with 57 trades occurring in 2001. Credit trading has also increased as operators became more familiar with the system and were further encouraged by the introduction of an online trading facility in August,

2000. The introduction of on-line trading removed the need of manual processing by the EPA, and thereby reduced cost. Additionally, on-line trading ensured that participants can trade at short notice whenever opportunities arise due to changes in river flow. (Murtough et al., 2002).

In 2002, the Hunter River Salinity Trading Scheme was made a permanent institution. As regulated, the Department of Environment, Climate Change and Water of New South Wales must auction 200 salinity credits every two years (20% of the total active credits), each credit bearing a lifespan of 10 years (NSW Environmental Climate Change and Water, 2010a). The auction ensures that new industries have an opportunity to participate in the trading program, while the long lifespan provides credit holders with a degree of certainty. In 2008-09, the scheme was considered highly successful; the upper sector of the river boasted a 100% compliance with the salinity goal, while the middle and lower sectors of the river only exceeded the salinity goal for one, 9-hour discharge period (NSW Environment Climate Change and Water, 2010b). Thus, throughout the prescribed discharge periods, average salinity levels remained below maximum salt levels, thereby ensuring the health of the Hunter River for this water quality parameter, and ensuring diverted water from the Hunter River was suitable for irrigation (NSW Environment Climate Change and Water, 2010b).

As in many markets, the size of the market is an issue for efficiency. The Hunter River Scheme has a limited number of participants. It has therefore been considered to include non-point sources as well, but so far it has not been possible to adequately measure, quantify and monitor the effect of such arrangements (Murtough et al., 2002).

5.2.3. South Creek Bubble Licensing Scheme (New South Wales, Australia)

This scheme was introduced in 1996 to reduce phosphorus and nitrogen levels in the South Creek area of the Hawkesbury-Nepean River. The river demonstrated significant environmental stress from high nutrient levels, leading to algal blooms and eutrophication (James, 1997). It was estimated that three sewage treatment plants contributed 60% of the phosphorous load, and 75% of the nitrogen load to the Hawkesbury-Nepean River. Under this licensing scheme, the New South Wales Environmental Protection Agency sets a limit for the aggregate discharge from an imaginary bubble placed over the three participating sewage treatment plants, and allowed free choice on how the limit is met. This enables individual dischargers within the bubble to share the burden of staying within the limit, and negotiate the most cost-effective solution. For example, investment could be made in upgrading one or two of the three sewage facilities to reduce discharges, rather than all three, which may not be economically feasible. These arrangements also seek to reduce total pollution; load targets for 2004 mandated a reduction of 50% and 83% in nitrogen and phosphorus loads, respectively. While load limits are specified on an annual basis, additional regulatory controls are in place to control maximum concentration levels at each plant at any given time. Cost savings of using this more flexible instrument, compared to traditional command-and-control regulations, are estimated around 10% to 20% (James 1997). The program also includes requirements such as installation of equipment to better remove nutrients, and improvements in the reliability of treatment processes. Compliance is established through discharge and water quality monitoring (Murtough et al., 2002).

5.2.4. Lower Boise River Trading Program (Idaho, United States)

The lower Boise River is located in southwestern Idaho, and covers some 3,340 km^2. Within the basin sewage treatment plants, factories, and agricultural producers all contribute to phosphorus loading into the river. In 1997, the US EPA and local stakeholders decided to develop a program to trade in phosphorus reduction credits as a means of reducing the costs of meeting new water quality standards to be introduced in 2001 (van Bueren, 2001). It was expected that a trading program would yield economic benefits, since the cost of nutrient reduction varies significantly between sources in the basin (van Bueren, 2001). Under the new total daily maximum loads (TDML), each discharger is set a stringent discharge limit. Companies that reduce their discharge below this limit will be issued discharge credits, specified by the amount of phosphorus reduced per unit of time; these credits can then be sold to those companies that choose to operate above their discharge limit (van Bueren, 2001). However, credits can only be used during the month the underlying reduction in discharge takes place. Trade takes place between private parties, and does not require individual EPA approval. Credits will be registered on a trading database by submitting a trade notification form signed by both the buyer and the seller. A local trading association manages the system and does the day-to-day management of trading and provides crucial market information about prices and volumes traded to potential market participants.

Non-point sources are able to generate tradable credits by implementing BMPs included on a list of approved practices. Examples of such BMPs are buffer strips, wetland constructions, irrigation control systems, and tillage systems. Each BMP includes detailed descriptions of required design, monitoring and maintenance. Credits will only be used if the BMP is a change to existing practices, that is, credits will not be issued retrospectively (van Bueren, 2001). Two types of non-point credits are issued: 1) ‘measured credits’, which are issued for BMPs where it is possible to readily and reliably measure the discharge reduction and record it by an independent assessor; and 2) ‘calculated credits’, for those BMPs in which it is not possibly to reliably measure the discharge reduction (van Bueren, 2001). For calculated credits, the reduction will be calculated using some kind of a model. In these instances the number of calculated credits are adjusted by an ‘uncertainty discount’, which accounts for the variability in effectiveness of the practice (van Bueren, 2001). Both credits are specified as the quantity of phosphorus reduced per unit of time in a given month. When a company buys such credits, the company is required to retire a proportion of the credits in order to reduce the total amount of phosphorus discharged from the agricultural sector; this is known as a ‘water quality contribution’. The system is enforced by requiring the buyer to certify that the BMP is properly installed, and performs according to specification; further, the buyer is held liable for the failure of the BMP to deliver the anticipated reduction (van Bueren, 2001). The buyer is therefore also responsible for ongoing monitoring of the BMP, which must be carried out by an accredited third party.

The TDML sets a water quality target measured at the mouth of the Boise River. However, sources of pollution are distributed unevenly throughout the catchment. Marginal increases or reductions in discharge by a particular source will have a different impact in the target region depending on its location along the river. To account for this differential impact, three trading ratios have been introduced: 1) river location ratios for different regions along the river, which will account for the estimated unit change in phosphorus loading at the river mouth caused by a unit change in discharge at the source location; 2) drainage delivery ratios to allow for the fact that some sources do not discharge directly to the river but first into a drainage canal or tributary. A discharge credit generated some distance from the river will have lower offsetting effect than discharge credits created by sources discharging directly to the river; and 3) site location ratios to account for the possible diversion and reuse of the discharged water below the point of discharge into the drain or tributary (van Bueren, 2001). If the discharged water is pumped out of the drain or tributary for use in irrigation before it reaches the river, the impact of the discharge will be lessened. These ratios are used as multipliers to convert all discharge credits to a single unit, namely the 'Parma Pound', which is the amount of phosphorus reduction at Parma, the mouth of the Boise River (van Bueren, 2001). Potential buyers will then have to compute how many 'Parma Pounds' they will need in order to offset their excess discharge at that particular location. Once a buyer has purchased a number of 'Parma Pounds', they will be converted back to local 'currency' and the buyer's discharge limit will be adjusted accordingly (van Bueren, 2001).

Point source polluters in the Lower Boise River catchment can only purchase credits for the purpose of offsetting discharges. Thus, it is not possible for non-government organizations to purchase credits for the purpose of lowering the total level of emission by retiring credits. In order to prevent the development of hotspots, areas with a high level of pollution due to a single discharger buying up large volumes of credits, there might need to be a limit on the extent to which an individual source can use credits as offsets. This is particularly of concern if trade moves large volumes of credits upstream. The final approach for program implementation was published in July, 2008, so the success of the program has yet to be evaluated.

The World Resources Institute (a Washington NGO) is developing an inter net site called Nutrientnet, which will provide online life trading in nutrients and will be a kind of one stop shop for interested buyers and sellers in the nutrients market. It will also provide advice and services for farmers who consider creating non-point credits. Farmers can receive advice on how many credits different BMPs would generate on their property and what the cost would be, allowing them to find the most cost effective way of generating credits.

5.2.5. Tar Pamlico River Basin (North Carolina, United States)

High nitrogen and phosphorus levels within the Tar Pamlico river basin has led to eutrophication and fish kills. In response to this, a permit-trading program was introduced to reduce nitrogen and phosphorus loadings at low cost. The point source polluters consisted of 14 dischargers, which contributed approximately 90% of all point sources of nutrients into the river (North Carolina Department of Environment and Natural Resources (NCDENR)). These entities organized themselves into a single group known as the 'Association' (Randall & Taylor, 2000). The basin also has a large number of non-point polluters, predominantly cropping and livestock farmers. The existing point sources are all placed under one bubble, and if the total nitrogen and phosphorus loadings from the bubble exceed the allowable limit, members must purchase offsetting non-point credits (Randall & Taylor, 2000). The market is not a traditional market in which buyers and sellers set prices by negotiation; rather a fixed price is set based on computer simulations of potential trades, and is set at $29 (USD)/kg of nitrogen/year (Randall & Taylor, 2000). Agricultural users can generate credits with a ten-year lifespan through the introduction of best management practices (Randall & Taylor, 2000). Based on the estimates of the performance, and cost of specific best management practices, trading ratios are set at 3:1 for cropping farm operations, and 2:1 for livestock operations (Randall & Taylor, 2000). Thus, to offset 1 kg of point sources of nitrogen, a BMP on a cropping farm operation must reduce nitrogen loading by 3kg/year, and best management practices on livestock operations must reduce nitrogen inputs by 2 kg/year. Via the North Carolina Agricultural Cost Share Program, the authority arranges these trades.

The association and its members are not responsible for ensuring compliance by the non-point trading partner; rather, the authority bears the cost of inspection and enforcement of compliance (Randall & Taylor, 2000). This relieves the point source buyer of bearing excessive risk through trade. This is a different process in the Lower Boise, as discussed above in section 5.2.4. Trading schemes in which the buyers are responsible for non-compliance of the non-point sellers, such as within the Cherry Creek Reservoir in Colorado, have been cited as one of the reasons for relatively few transfers (van Bueren, 2001).

The first phase of the Tar-Palmico Nutrient Trading program, which ran from 1990-1994, boasted an overall reduction of nitrogen and phosphorus loads by approximately 20%, despite increases in growth (NCDENR). The association funded nearly $1 million worth of agricultural best management practices (NCDENR). In the second phase of this program, which ran from 1995-2004, the Association expanded to 16 members, and efforts focused on establishing in-stream reduction goals for both point and non-point sources of nutrient loading (Environmental Management Commission (EMC), 2005). Due to concerns that the cap limit was set too high for nutrients, Environmental Defense and the Pamlico Tar River Foundation, environmental groups who were signatories for Phase I, did not support, or sign onto the Phase II agreement (EMC, 2005). Goals for Phase II included a 30% reduction in total nitrogen loads from 1991 levels, and a zero increase in phosphorus loads (EMC, 2005). There was also a call for a separate non-point source strategy, which initially began as a voluntary program. Due to inefficient progress in this strategy, the Environmental Management Commission of North Carolina developed rules that came into effect in 2000-01, which addressed agriculture, fertilizer management, riparian buffer protection, and urban stormwater non-point source loading (EMC, 2005). Collectively, the agricultural community was required to reduce nitrogen loads by 30% in 5 years, and ensure phosphorus loading did not increase within the subsequent 4 years (EMC, 2005). The non-point strategy and reduction target goals were maintained and supported in the

Phase III, which runs from 2004-2014, and earned the support of previous environmental groups who did not sign onto the Phase II (EMC, 2005). As of 2008, the nitrogen and phosphorus load for the Association was 63% and 60% of the cap, respectively (Annual Nutrient Loads and Caps, Tar-Pamlico Basin Association, 2009).

5.2.6. South Nation River Watershed Trading Program (Ontario, Canada)

The South Nation River Watershed is 4000 km^2, and is host to a population of approximately 125,000 people, in addition to a variety of mixed farming operations (O'Grady, 2008). Total phosphorus loading into this watershed exceeded provincial guidelines for water quality, and studies showed that 90% of phosphorus loading can be attributed to non-point sources of pollution (O'Grady, 2008). Thus, in 1998, the Ministry of the Environment stopped issuing permits for phosphorus discharges, and required a zero phosphorus discharge from newly constructed plants. The ability of plants or municipalities to meet these standards is hampered by technical, economic, and physical limitations (O'Grady, 2008). These difficulties, in concert with the large contribution of non-point sources of phosphorus loading, led to the implementation of the Total Phosphorus Management Program. In this program, dischargers are permitted to contribute phosphorus from their treatment plants provided the loads are offset by reductions in non-point sources of phosphorus within the watershed. Dischargers looking to offset their phosphorus loads fund the projects implemented to reduce non-point sources of pollution through the Clean Water Program. Thus, farmers receive financial and technical assistance to implement BMPs that will contribute to reducing phosphorus loading in the watershed.

The Ministry of the Environment requires that for every 1 kg of phosphorus dischargers contribute to the watershed, 4kg of phosphorus must be removed from non-point sources (O'Grady, 2008). Although this ratio appears highly skewed, it is in place to account for the lack of knowledge on the amounts of phosphorus transported, and the uncertainty as to the amount of phosphorus that is soluble, or remains in particulate form, which impacts the ongoing monitoring processes that dictate maximum acceptable levels of phosphorus in the water (O'Grady, 2008).

In 2007, the Canadian Council of Ministers of the Environment awarded the South River Nation's Total Phosphorus Management strategy an honourable mention for the Environmental Pollution Prevention Awards (Environment Canada, 2007). Through the South Nation's Clean Water Program, 212 phosphorus-reducing projects have been realized; a net reduction of over 6900 kg/year has been estimated as a result of this program. Additionally, cost-benefit studies indicate that this program has leveraged over $100,000 in savings since 2000, due to the high costs of water treatment methods compared to the Total Phosphorus Management Program (Environment Canada, 2007).

5.2.7. Dutch Nutrient Markets: Mineral Accounting System (MINAS)

Relatively few market-based systems have been introduced to control environmental and pollution problems associated with agricultural activities. In the previous water market examples, non-point agricultural polluters were included in credit trading programs based on the introduction of best management practices. But these aforementioned markets are relatively recent, and apart from them, the Dutch system of phosphate quota in animal production, termed the mineral accounting systems (MINAS), is the only example in the world of a pollution permit-trading program.

Through an increasing number of confined swine, poultry, and dairy operations in the Netherlands during the 1960's and 1970's, problems with surplus nutrients emerged. Growing international demand for animal products, and EU policies that favour the import of feed, fueled this development. Between the early 1960s and mid 1980s, the number of swine increased by 450%, and poultry by 125% (Wossink, 2003). This resulted in an excess of manure production, and an excess manure application on land, which consequently leached into the groundwater. By the mid 1980s, it was estimated that 60% of land in the Netherlands exceeded the EU standard for nitrate content of groundwater (Wossink, 2003). The Manure Act and the Soil Protection Act were introduced in 1987, and included manure quota and manure bookkeeping systems.

Since the Manure Act 1987, total manure production from animal sources has been capped at 125 kg of phosphate per hectare (Wossink, 2003). Farmers producing more manure must acquire more 'registered animal-based manure production rights'. From 1987 to 1993, this could only happen through the purchase of additional farmland. Each farm was grandfathered a 'reference amount' based on annual manure production; the reference amount was computed through an inventory of animals and standards for manure production for each animal category, measured in kilogram of phosphorus per year (Wossink, 2003). The animal-specific standards were calculated as the difference between phosphate supply in the form of feed, fertilizer etc., and phosphate removal in the form of meet, milk, eggs, etc.; the difference is assumed to represent the phosphate content of manure (Wossink, 2003). The difference between a farm's reference amount and the hectare-based phosphate cap was used to distinguish between manure-surplus and manure-deficit farms. A deficit farm could still increase production based on its per hectare phosphate right; while a surplus farm would have to acquire additional rights by buying more land to expand animal production. However, such additional land purchases would first go to fill the gap between the reference amount and the area-based right before any expansion could take place. Transfer of rights was not possible, except under very specific circumstances to do with change to ownership of land. These restrictions caused a virtual freeze on structural change in the regions with predominantly phosphate-surplus farms, and prevented investments in appropriate technology to help alleviate the manure problem (Wossink, 2003).

The rules were subsequently relaxed in 1994 to allow some trading in manure rights. The old reference amount for each farm was renamed as manure quotas. The quotas were made more diversified so as to reflect the differences between animal types, and to restrict trade. The farm's manure quota was divided into two distinct parts: a land based part, and a non-land based part. The non-land based part varied from animal type to animal type, thus reflecting the different intensity of manure production (Wossink, 2003). When quotas were traded, 25% of the traded quota had to be retired for environmental benefits. Additionally, the buyer had to certify that they either had sufficient farmland to dispose of the total manure for the next two years, or a manure disposal contract with another farm (Wossink, 2003). Spatially, the country was divided into two regions: the manure-surplus region, where the average manure production exceed 125 kg of phosphate per hectare, and the manure-deficit region, where the average is below 125 kg (Wossink, 2003). Trade could take place within both regions, from manure-surplus to manure-deficit regions.

Initially, the future of the quota system remained uncertain. Originally the quota system was to be terminated and replaced with an accounting based program in 1998. This was eventually enacted, but the quota system was never abolished. In 1995, existing quotas for pork and poultry were reduced by 30% in response to the development of low nutrient feed, and by mid 1997 the pork quotas were cut by another 25%, and then another 10% in 1998 (Wossink, 2003). A quota buy back program was initiated specifically for the pork sector. The ongoing uncertainty of the long-term fate of the quotas restricted market activities. A survey of Dutch farmers in 1997 showed that farmers rated policy uncertainty a par with uncertainty related to production and markets (Wossink, 2003).

Analysis of market outcomes showed great variation in quota prices both between regions, between animal types, and over time. Initially, quite low volumes of quota were sold. The first year saw approximately 1.5% of total quota traded and by 1997, it had increased to 8.1% and 9.5%, for the surplus and deficit regions respectively (Wossink, 2003). A number of reasons for low volumes were found. First, administrative procedures impeded trade, as farmers buying extra quota had to certify that they had an adequate manure disposal plan in place. During the first year, 37% of all applications had to be resubmitted due to shortcomings in the manure disposal plan. Additionally, the associated transfer costs have been found to be as high as 17% of average quota price (Wossink, 2003). Second, many farmers were issued with excess quota. When animal inventory information was being collected, many farmers gave their maximum capacity of animals rather than the actual number of animals on the farm; it has been estimated that initial quotas were over-allocated by 10%–25% (Wossink, 2003). Such an over-allocation would reduce the need for trade. The limitation of transfer between regions and animal categories served as an additional impediment to market activity. Finally, the 25% retirement of quota upon sale increased the cost of buying additional quota, reduced the future resale value of the quota, and constituted a disincentive to trade (Wossink, 2003).

The quota system, the spatial restrictions on trade, and the price difference in quotas between regions have encouraged farmers in the surplus region to sell their farm and quota and buy land with quota in the rest of the country. The quota system has encouraged exit adjustment, and the total policy mix has been an incentive to innovative approaches to reduce emission. A significant driver of reduction has been in the area of nutritional development in the pork industry, where the development of modified feeding regimes has reduced the nitrogen and phosphorus intake, thereby reducing the nitrogen and phosphorus surplus while maintaining daily weight gains. A major impact on the success and participation in the program is the farmer's uncertainty of the environmental benefits from the program. This has particularly been the case among pork producers who have born the brunt of the economic impact of the policies, which has resulted in very active and militant farm organizations in that sector.

In late 2003, the European Court of Justice ruled that the Netherlands Action Programme, which encompasses MINAS, violated certain portions of the European Union Nitrate Directive (Schroder & Neeteson, 2008). Briefly, the court decreed that manure application rates and surpluses were set too high, exceeding 170kg/ha/year, and essential manure policy regulation was not implemented soon enough (Oenema et al., 2005). Additionally, there were instances in which farmers were permitted to ignore certain fertilizer inputs, and further inputs were sanctioned to encourage manure purchase; thus, actual nutrient surpluses were greater than on-paper ones (Schroder & Neeteson, 2008). Further, there was no concrete evidence that water quality had improved as a result of the Netherlands Action Programme/MINAS (Schroder & Neeteson, 2008).

The new Netherlands Action Programme emerged as a means to reconcile the MINAS program with the EU Nitrates Directive. Alterations include application standards and rates calculated for a variety of crops and soil types, nitrogen fertilizer replacement values of various manure types, and regulations that outline the times of year in which fertilizer and manure application is prohibited (Schroder & Neeteson, 2008). Despite the shortcomings of the original MINAS program, it is believed that surplus reductions in agriculture is an integral step towards improving the ecological status of source waters surrounding agricultural fields (Oenema et al., 2005).

5.3. Habitats and biodiversity

Natural habitats and biodiversity provide a number of services imperative for sustainable agricultural productions of essential food and fibers, as well as the production of pharmaceutical products. Biodiversity is unique in the sense that if wiped out, it cannot be recreated. Concern about the growing rate of species extinction due to human activity is increasing; many of these species have not yet been fully explored for their potential agricultural and pharmaceutical benefits, and an unknown number of them play key roles in the ecological system; the extinction of these could result in major changes to ecosystem functions, and the consequence of such extinctions are unknown.

Three types of policy instruments are emerging to deal with this concern. The commercialization or privatization of species by pharmaceutical and seed producing companies, which may patent certain genes and knowledge, can play a role in protecting biodiversity, while ecotourism operators such as Earth Sanctuaries Limited function to preserve habitats to create wildlife sanctuaries on a commercial basis (Aretino et al., 2001). Government authorities can also protect biodiversity through the purchase of environmental services from private individuals and companies. Examples of these are the Conservation Reserve Program in the U.S., and the Bush Tender program in Australia, under which farmers are paid to take land out of production and convert it to natural habitats. The use of mitigation banks and markets for mitigation credits is a third policy instrument that can protect and enhance biodiversity. Many economically developed countries are introducing caps on the level of habitat degradation. For example, the US Clean Water Act protects against the net loss of wetlands as important natural habitat, and the Endangered Species Act prohibits actions that would threaten endangered species. (van Bueren, 2001).

5.3.1. Wetland Mitigation

The preservation of wetlands has gained increased attention in the U.S. since the 1970s, when analyses revealed that 50% of all US wetland had been converted to agricultural and urban uses (Shabman, 2003). There are two main types of wetland protection policies in the US: the Wetland Reserve Program, which pays farmers to preserve and enhance wetland (policy instrument 2); and section 404 of the Clean Water Act, which introduces a regulatory approach mandating a ‘no net loss’ of remaining natural wetland (Randall & Taylor, 2000). This section allows individuals who wish to drain wetlands in one location to mitigate the loss by enhancing wetlands in other areas (policy instrument 3); the way is then open for trading in wetland services, and the creation of wetland credits. Under this framework a developer must apply to the authority for a permit to alter any existing wetland. The authority will evaluate the physical qualities of the affected wetland, and determine whether any offset needs to be made due to the impact of the development. The authority can require the developer to make onsite offsets such as setbacks from the wetland, and filter strips to minimize the impact of runoffs from the development. In some instances it might be more beneficial to require the offset to be implemented offsite. In this instance the developer pays to create a new wetland, or improve another wetland in a different location. The authority will try to ensure that the offset is made in a wetland of higher ecological quality within the same hydrological and ecological region (Randal & Taylor, 2000).

It has been argued that the effectiveness of constructed wetlands is inferior to natural wetlands, and that the outcome and long-term sustainability of the constructed wetland is uncertain. The authority therefore applies a mitigation ratio between natural wetland destroyed and new wetland. The most common mitigation ratio is 1.5:1; that is the developer must create 50% more wetland than the new development destroys, but ratios of 3:1 have been applied for high quality or very sensitive wetlands. The final ratio is determined on a case-by-case basis as part of the permit process (Randall & Taylor, 2000).

Offsetting existing wetlands with new constructed wetlands is complicated due to the complex ecological interactions that take place within them. It is difficult to measure and monitor this interaction in existing wetlands, to predict the interaction that will take place within a constructed wetland, and the timeline for achieving this. The approach used by the authority has therefore been to set design standards, approve the wetland design, set measurable implementation goals, and require that the mitigation credits generated be certified before they can be used for offsets.

As the program under section 404 matured, critical reviews found that the policy objective of ‘no net loss’ was not being achieved for several reasons. First, the avoided wetland (the remaining wetland left after the permitted development was completed) was compromised by polluted runoff and changes to the hydrologic regime from the new development. The avoided wetland was transferred into a storm water pond that provided hydrologic and water quality services, but the original habitat functions were compromised. Also, where mitigation was made, nearby smaller wetlands often delivered little more than storm water benefits. These outcomes suggest the potential benefits of larger offsite developments. Secondly, when wetland credits were required, many problems with the credits were identified. In some instances the credits were not provided at all, but the agency did not have resources to enforce the requirement. This suggests that enforcement would be more efficient if multiple and dispersed mitigation projects were consolidated into one large and more easily monitored area. Sometimes mitigation projects were attempted without the necessary knowledge and advice, resulting in open water pond, which as mentioned previously, provide hydrologic and water quality services, but not habitat services. Even when created using sound mitigation methods, there was often a significant time lag between the loss of habitat and its replacement. A third problem identified with the 'no net loss' policy is that in some instances, rules did not require small fills to provide compensation credits, thus creating a net loss in wetland habitat. If large areas of mitigation wetland were developed it would be possible to require small fill developers to buy small volumes of credits from such large projects. (Shabman, 2003). Additionally, while wetland offsets may be successful in the short term, there are high risks of failure in the long term. Research in the US suggests that the failure rate of wetland offsets may be as high as 50%; in Virginia it has been found that only nine wetland projects out of 32 were successful. It has been estimated that the Wetland Reserve Program was 47,000 acres per year short of achieving 'no net loss', and that 80% of gross wetland losses occur without permit. This means there is a serious risk of significant environmental harm (Murtough et al., 2002).

Another problem associated with individual offset mitigation is that the quality of wetlands, and the diversity and productivity of the ecosystem they support often are a function of the size of the wetland, with larger wetlands supporting more diversified and productive habitats. Seen in isolation, a single development could call for a mitigation effort of a size that is unable to sustain a viable ecosystem. Thus, to achieve economies of scale and secure sustainable outcomes, it is necessary for multiple landowners or developers to collaborate and involve public agencies. Wetland Mitigation Banks have been introduced in the US to achieve just this objective. Mitigation Banks are large wetlands constructed for the sole purpose of providing wetland credits for future offsets for wetland loss due to new developments. If a developer creates a large-scale ecologically viable wetland, it can be sold in smaller sections to developers who are required to offset wetland conversion (Randal & Taylor, 2000).

Different types of wetland banking have been developed. Initially a mitigation bank was created by a single entity, creating a large and readily monitored wetland away from the area it is going to fill. The entity is then awarded credits for the wetland, which they can use for new developments as required. However, this approach only works for those entities that expect a large number of developments within the same basin. Recognizing this problem, some authorities began to charge a fee in-lieu of onsite mitigation before issuing a permit to develop. These fees were then accumulated until enough funds were available to construct a large single wetland. The problems with this approach are a large time lag between fill and mitigation, and that in some instances the fee did not properly reflect the mitigation costs. The fee system however did provide precedence for transferring the legal and financial responsibility for creating the credit from the developer to another party. It was this precedence that helped motivate private investors to produce credits for sale. To encourage this development, the U.S. Federal Government issued mitigation banking guidelines in 1995. The purpose was to satisfy both skeptical environmentalists that the 'no net loss' policy would be achieved, and to reduce the uncertainty that investors might face when assessing the financial viability of creating credits. Created credits are subject to certification and achievement of success criteria, or provision of financial assurance before they can be sold. Monitoring and reporting of credit performance is the responsibility of the seller, and the authority can exercise random inspection and audit of the site. Also, transfer to public ownership of permanent easement rights is required to insure that sites retain their wetland status in perpetuity (Shabman, 2003).

Based on a 2005 inventory, there exist a total of 450 mitigation banks, with an additional 198 submitted for approval (US EPA). The cost of wetland credits varies from US$4,000 to $20,000, and US$ 3,000-$10,000 for coastal and non-coastal mitigation credits, respectively (Caffey et al., 2009). Florida has 18 mitigations banks with some 20,000 acres of wetlands constituting a US$750 million industry (Randall & Taylor, 2000). The Ohio Wetland Foundations is a non-profit organization that creates wetland banks and sells acreage to developers. Since 1993, the Foundation has sold out three wetland banks of between 33 and 330 acres (Randall & Taylor, 2000). Mitigation banks are created through a Memorandum of Understanding between federal and state officials, and the creator of the bank (van Bueren, 2001).

Market activities for offsets have been limited, while environmental performance of mitigation offsets has been spotty and not ecologically acceptable; this has generated significant skepticism among environmentalists (Shabman, 2003; Randall & Taylor, 2000). The main impediments to market adoption among both buyers and sellers are high transaction and approval costs, and a high level of uncertainty about the outcome of the process. First, it is not uncommon for credit sales approvals to take several years and to require expensive legal council. This is among other things caused by the regulatory review process of the techniques used to create the wetland credits. Land must be acquired, and the construction plan for the wetland thoroughly reviewed and critiqued in order to assure that the project will be ecologically successful as measured against the performance criteria. However, at the same time the seller must also place a bond in case the performance criteria are not met. Second, there is great uncertainty about the future demand for credits. This is partly a result of normal uncertainty about demand and supply in the real estate market, and partly due to the greater demand uncertainty that is associated with the regulatory process. The process of approving the fill permit first tries to minimize the fill and avoid the need for mitigating activities. If mitigation is required, regulators prefer on-site mitigation, despite the fact that regulators have been increasingly concerned about the aforementioned problems with on-site mitigation. If regulators require or allow the use of offsite credits, the kind and location of the required wetland are determined specifically for that project, and therefore there may not exist a location or type of wetland available by a seller. This uncertainty has been amplified by ambiguity of legislative intent at the time the Act was created. There has consequently been persistent policy disagreement over many matters affecting the creation of and demand for wetland credits. In the absence of legislative clarity, the goals and structures of the program have been defined by executive orders, administrative rules, and rulings by the United States Supreme Court as well as lower courts. This policy limbo creates uncertainty about the future demand for wetland credits and what will create an allowable credit, which causes potential creators of wetland credits to further discount the value of future credits.

The fact that the program revolves around the wetland asset itself, and not the three types of services provided by it (water quality, storm water mitigation and habitat services), constitutes an additional impediment to the program. By nature, the first two services are best mitigated by the credit being created at or near the fill site, while the third service might be best mitigated at some distance from the fill site. Dealing with the three services together has limited the area within which the offsetting activity is carried out. In some areas, there have been very few suitable sites for mitigating wetland development. Landowners have become aware of the unique quality of their land and have asked for higher prices than estimated, which has resulted in increased cost of mitigation. This problem could have been avoided, and the whole process made more flexible, if the three services were separated with the ability to mitigate them separately. The habitat service is the only one of the three services that has no alternative solutions to wetlands. However, this is not site specific and can therefore be mitigated in a wider area providing a better supply of suitable land and also a larger demand area, which makes large-scale mitigation projects less risky. Storm water mitigation and water quality services are site specific, but have numerous other mitigating potentials. The mitigating credit markets should therefore concentrate on habitat services (Shabman, 2003).

Despite these impediments, there has been growth in the number of private firms dealing in wetland credits within some catchments. This is because of innovation and market knowledge of both the potential buyers and sellers. Sellers have some knowledge about potential demand in the area through their contacts in the property industry. Potential sellers and buyers then make a tentative agreement with the authority that the fill applicant (the buyer) will be allowed to purchase credits from the potential seller at the intended location. The seller then ensures an informal agreement with the authority that a limited share of the created credits may be sold before replacement wetlands have been certified, often in return for a performance bond. The seller and the buyer negotiate a credit price that is high enough for the seller to recover the total development cost from the entire proposed wetland, even though the buyer only requires a fraction of the created credits. In this way the agreed price is based on the buyers' avoided cost from buying credits, rather than the cost of producing the credits in a competitive environment. The performance bond together with other quality controls reassures the authority that the project will be ecologically successful, and the seller earns an acceptable return on its investment from the immediate sale of just a proportion of the created credits. This removes the risk of future demand uncertainty, while holding a possibility for further profits. Thus the buyer reduces the cost of compliance, although they pay a higher economic price than they would within a more competitive regulatory environment (Shabman, 2003).

Shabman (2003) proposed a system that would transfer the demand uncertainty risk from the creator of the credits to the public sector. Under this system it would be up to the wetland mitigation agency to estimate the number and types of credits they anticipate being required over some period to meet the net wetland loss incurred by new developments within their area. Based on this estimate, the agency issues a request for proposals from potential suppliers of the required credits. The lowest cost bidder, who will also provide ecological success assurances as under the present system, would receive the credit supply contract. Such a system is presently working for the Colorado

River Salinity Control Program. The price charged for the credits by the mitigating agency might be similar to the in-lieu fee, but in this instance the fee is based on real projects and real costs of producing ecologically successful credits, elicited from a market bidding process. The winning bidder would immediately start the construction process and payment would be made according to a performance schedule for completion. If the authority overestimates demand, public funds would have been used to contribute to wetland restoration; if demand is underestimated, the agency would simply repeat the process (Shabman, 2003).

The experiences with the US wetland reserve program have highlighted the fact that biodiversity is a far more complex and unique commodity than $SO_{2,}$ forest and water, and therefore not as readily tradable, for once lost it cannot be replaced (Murtough et al., 2002).

6. Property Rights

As illustrated in Hardin's Tragedy of the Commons, property rights play a significant role in pollution of shared resources. The right of ownership in an asset can be defined in three elements: the right to use the asset (*usus*); the right to the return from the asset (*usus fructus)*, and the right to change the asset's form, substance and location (*abusus*) (Kemper, 2001). The third right is fundamental because it confers on the holder the right to sell the asset to another user, and it should therefore encourage the holder to use and allocate the resource efficiently over time. The success of this depends on the strength of the property right. Scott (1999) suggested six characteristics of a property right as being important in determining its strength:

1. Duration: the time period over which the right holder can expect to have the power to control and benefit from the resource. This is important for many resource users

since considerable capital expenditures are required to utilize the resource and benefit from its use. The magnitude of such investments can often only be justified if the time period over which the resource can be put to beneficial use is of a certain length, and if continued access over the specified period is sufficiently secured. Long duration promotes good husbandry in resource use, and thereby promotes a more sustainable resource use. However the need for long tenure must be balanced with the need to regulate for adaptive management.

2. Flexibility: the flexibility of the right will determine how well the use of the resource can be adapted to changing conditions. What can be done without consulting others? If changes require consultation with others, it can result in delay and costly procedures and the operator can suffer losses (Johnson, 1992).

3. Exclusivity: the right holder must have the exclusive right to use the resource and the means to exclude others from using it, or to determine who can use it. This ensures that all benefits derived from the resource are returned to the rights holder; it thereby encourages him or her to reduce costs, improve efficiency and increase yield, and thereby improve economic efficiency. The right to exclude others from using the resource is a precondition for the right to trade.

4. Security or quality of title: how vulnerable to challenge is the right? Can a third party claim the right or do government authorities have the power to change the conditions of the right or take it away (with or without compensation) and under what circumstances? A set of legal rights reinforced by a registration system must secure the property right, and must be enforceable against third parties, and be enforced by the proper authorities. Lack of security produces uncertainty, which is an impediment

to good long-term husbandry, just as short duration is, as right holders will be reluctant to make long-term investments in good resource use. This impact is amplified by the fact that when selling the right, the new buyer will take this uncertainty into account and therefore pay a lower price for the right.

5. Transferability: it must be possible to transfer the right to other parties so that it can move to the most efficient and highest value users, and thereby improve economic efficiency. It provides the right holders with an incentive to build up the value of the resource, as it enables them to benefit from the increased value the day they want to stop using the resource either by selling it, or transferring it to the next generation. The strength of this characteristic is determined by:

- to whom the right can be transferred;
- which geographical restrictions limit the transfer;
- to which uses the right can be transferred;
- how many parties must be consulted prior to transfer; and,
- environmental limitations.

6. Divisibility: the strength of the property right will also be determined by the ability to subdivide or aggregate property rights to accommodate changing industry structures and benefit from economies of scale.

Scott argues that the strength of a property rights system, and the potential for a market based approach to be successful in achieving economically efficient outcomes, can be assessed by evaluating its strength on each of these six characteristics. He illustrates this by a system of six vectors reflecting each of the characteristics (Fig. 3).

The strength of each characteristic for a given property right system can then be measured on the respective vector, and a star-like shape can be produced by connecting the measure on each vector. Comparisons of property rights systems can then be made by comparing the shape of their respective stars.

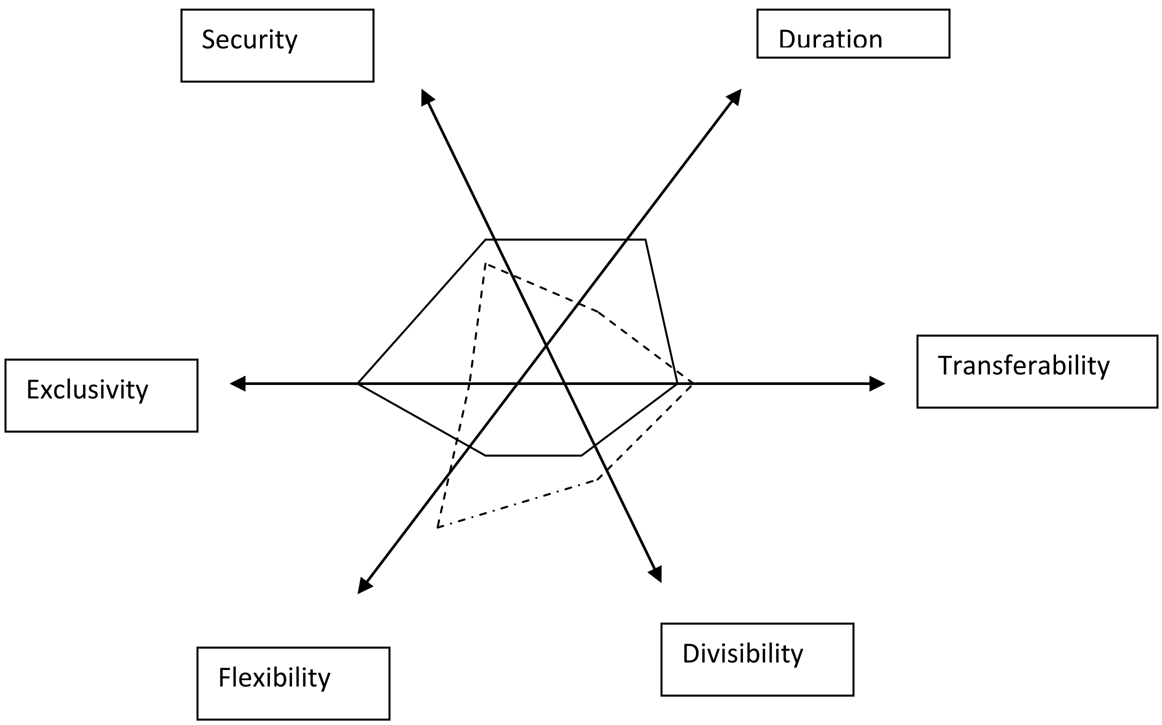

Figure 3: from Scott, 1999.

This discussion has concentrated on the necessary characteristics of a natural resources property right to send the appropriate economic signals to resource users that invest in efficient resource use, and adopt efficient long-term management strategies in pursuit of economic efficiency. An equally important discussion is based on which responsibilities should be embedded in the property right. It could be argued that the resource belongs to the State, and the benefits from using it should be shared with the wider community. Following that logic, individuals or companies granted a property right in a resource should be under an obligation to use it as efficiently as possible to ensure that the economic flow-on effect in the wider community is maximized, resulting in jobs and economic activity within the area. To ensure maximum community benefits, it will also be necessary to balance economic benefits with social, cultural and environmental benefits and costs. Markets in resource property rights, even if associated with perfect characteristics along each of the six vectors in figure 1, are unlikely to achieve such community outcomes. For this to happen, market-based approaches have to merge with traditional command-and-control instruments such as regulatory and planning frameworks (Bjornlund, 2000; Bjornlund & McKay, 2003; Young, 1995). This process often results in the attenuation of the right by reducing the strength of the characteristics discussed above and increasing transaction costs. Neoclassical economists will argue that any such restrictions on the market will reduce economic efficiency and that 'the invisible hand of the market' will provide equitable outcomes, while others argue that such restrictions and costs are necessary to direct the 'invisible hand of the market' to achieve social, cultural and environmental outcomes (Colby, 1990; Bjornlund, 2000; Young, 1999).

7. Economic Instruments in Alberta

Under Alberta's water for Life Strategy, the government of Alberta has examined and evaluated the use of some tools that can be used to address the issues surrounding water management. The government recognizes the use of four types of tools including economic instruments, cooperative management agreements, information disclosure, and voluntary stewardship. Some examples of the potential tools that can be used to address water quality and non-point sources of pollution include a cap-and-trade for nutrient trading in Albertan Watersheds, and an offset program in which polluters can fund improvements in nutrient management to offset nutrient loading into the watershed.

The Alberta government also intends to use some market-based instruments in its implementation of the recently passed Alberta Land Use Act (ALU) and Alberta Land Stewardship Act (ALSA). Under these acts, 7 land-use regions are established, in congruence with Alberta's seven watersheds. Regional plans are to be developed for each region, under the direction of a Land-use secretariat and a regional advisory council. These plans will integrate provincial policy, outline regional land-use objectives, employ cumulative effects management for the impact of developments on air, water, and land, and evolve and encourage a strategy for conservation and stewardship via the use of new policy instruments. Some of these market-based instruments include stewardship units, conservation easements, a conservation off-set program, and the transfer of development credits. Stewardship units would be used to provide a unit of measurement for stewardship of some area of native vegetation or wetland, for example. This unit could form the basis for a variety of other market-based incentives, and could be used to establish common property rights across various land-use categories (Kerr & Bjornlund, 2010). Conservation easements are a payment given to landowners to encourage the preservation of an ecosystem service, such as habitat protection, for example (Kerr & Bjornlund, 2010). The ALSA would enable the conserved land to be registered in the Land Titles Office, thereby providing some security to the property right. A conservation off-set program would provide a compensatory action when some ecosystem service is lost on private or public lands (Kerr & Bjornlund, 2010). Finally, the transfer of development credits would be a mechanism to allow for economic development on private lands, with projects being diverted from specific landscapes to maintain ecosystem services (Kerr & Bjornlund, 2010). Of these instruments, presently only conservation easements are functioning.

Selecting the most appropriate economic instrument is no simple task. Some economic instruments may be ideal for application in the industrial sector, while similar instruments may function poorly in the agricultural sector. Additionally, it can be extremely challenging to determine the effects and impacts of one economic instrument, because these tools are often implemented simultaneously with others. Thus, there is some support for a “one objective-one instrument” (Government of Canada, 2005, p.11). The use of multiple instruments, however, may be useful to serve as checks and balances of potential environmental effects (Government of Canada, 2005).

Some more specific examples of economic incentives are described by British Columbia’s action plan, first implemented in 1999, which was designed to specifically address NPS pollution. One of the mandates of this action plan is to assess the most effective economic instruments to encourage reductions and prevention of in NPS pollution. Some incentives include credit support, property tax breaks, density bonuses, buy-back and retrofitting incentives, “green” recognition, grants, environmental user fees, performance bonds, and tickets of fines (Government of British Columbia, 1999).

8. CONCLUSIONS

Managing the use of shared resources is an increasingly challenging task, especially as resource scarcity and pollution become exacerbated by climate change. Thus, there exists a range of potential solutions to meet these challenges, some of which may be extremely complex. Legislation designed specifically to manage and control non-point sources of pollution is largely absent in Canada, due to the intrinsic complexities involved in the policy design. The diffuse, stochastic, and heterogeneous nature of agricultural pollutants (the dominant source of non-point pollutants) presents a unique challenge to policy makers. Specifically, policy must address some complex and fundamental issues surrounding legislation related to non-point sources of pollution, including whom, and what should be targeted for regulation.

Alternative to forced participation through legislation is the use of voluntary programs. Generally, these programs involve the implementation of best management practices (BMP). A wide variety of BMPs exist, and they vary greatly in cost implementation, effectiveness, and the scientific research evaluating their effectiveness. Some BMPs reduce the input of pollutants, such as effective nutrient management, the use of manure to improve soil conditions, manure storage, manure application, and integrated pest control. Other BMPs are focused on reducing water and soil erosion. Such practices include reducing tillage, controlling soil compaction and overgrazing of pastures, cover cropping, increasing the organic content in soils, and increased irrigation efficiency. A third set of BMPs function as barriers to the entry of pollutants into watercourses; such practices include vegetative buffer strips, and fencing around watercourses to prevent livestock access. The successes of such programs are limited to their level of participation. Thus, the use of economic instruments is employed as an incentive to increase participation in voluntary programs. Emissions tax, ambient pollution concentration tax, and input tax are some measures employed that can be used to deter the introduction of pollution into aquatic environments, and the atmosphere. The assistance from governments and green payments can help to alleviate the costs of implementing BMPs. Ascertaining the social, cultural and economic value of water through water pricing is considered to be a useful tool to encourage the conservation and protection of source waters.

Market-based systems are increasing in popularity as a means to balance the costs associated with emission reduction, and the economic viability associated with resource use. Air pollution markets were the first kind of trading schemes developed, starting with a variety of cap and trade programs in Los Angeles. The first such trading schemes involved the trading of emission reduction credits to ensure the total pollution discharge among a group of polluters did not exceed a predetermined cap. A similar cap and trade style program was implemented to reduce smog in L.A. Additional programs in L.A. include the old vehicle swapping program. The Acid Rain Program, implemented nationwide across the United States, is considered to be very successful in reducing the deposition of wet sulfate and nitrogen deposits caused by high sulfer coal-fired plants. The Kyoto protocol represented the first international trading program that aimed to reduce the emission of greenhouse gases into the atmosphere. Under this program, industrialized countries must reduce emission levels to a specified level. Meeting reduction targets can be accomplished through the international trading of carbon credits, providing financial support to other industrialized countries for emission-reducing projects, or investing in clean development projects in developing countries.

Several market-based systems have also emerged to combat various types of water pollution. In Australia, a saline credit-trading scheme was developed to reduce the impact of saline discharges from salt mines into naturally high-salinity rivers. Another program in Australia placed three large point-source dischargers under a cap, which allowed them to work together to collectively reduce their emissions through the implementation of costly technologies. Most commonly, markets aimed at improving water quality involve the trading of point and non-point credits to achieve a desired level of nutrient discharge. Such programs have emerged in Ontario, Idaho, and North Carolina. Dutch nutrient markets have focused on individual farm inputs, in a goaloriented regulatory program. Other markets have surfaced to accommodate for wetland habitat loss in the United States. Although these schemes have met with varying degrees of success and criticism, it is certain that through evaluating the successes and failures of these programs, policy designers will be better equipped to produce programs with an increased market efficiency, and achieve meaningful environmental outcomes.

Property rights play a significant role in the way in which resources are exploited. The security, duration, transferability, divisibility, flexibility and exclusivity of particular property rights are believed integral in producing efficient and effective markets for resources, and the responsible and long-term management of these resources.

Clearly, a wide variety of strategies exist that can be used to control and manage point and non point sources pollution. As there exist a multitude of regional, geographic, economic, and political components that play a relevant role in any given aspect of pollution, policy designers, governments, economists and scientists must work together to address these extremely complex issues to achieve desirable economic, and environmental outcomes.

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