Creating a Market for Biodiversity: Applying Economics to Biodiversity Conservation

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ABSTRACT

Biodiversity fuels ecosystem functioning. The services provided by healthy ecosystems are the foundation for human wellbeing, but society faces a tradeoff between biodiversity protection and the externalities that result in biodiversity loss. So reaching equilibrium between the costs of this loss and the benefits of other economic choices is an important concept in strategies to improve biodiversity conservation. However when the benefits of biodiversity protection are not accurately valued, it becomes difficult to make appropriate choices about a sustainable level for the consumption of biodiversity. When the total value of biodiversity is not recognized, conservation incentives become weak, and an insufficient amount of biodiversity is protected compared to the social optimum. Developing market and government incentives for the total valuation of biodiversity can be achieved through economic valuation, which encompasses the significance of biodiversity protection and consumption in its entirety. While the overall cost of biodiversity loss is unknown, some parts of its value can be estimated by applying valuation tools to conservation incentives such as market-based instruments. Government and market strategies with the potential to reduce the loss of biodiversity can therefore be assessed on their ability to achieve conservation goals, and whether the benefits of the strategy outweigh the costs. This paper outlines how the role of economic valuation, combined with the use of the market, international co-operation and a strong economic framework, can help to create a market where biodiversity goods and services can be traded at prices that reflect their true value to society.

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I INTRODUCTION

Maintaining biodiversity requires more than just protecting wildlife and their habitats in nature conservation reserves. It also calls for improved practices in the management of a wide variety of natural resources. Society needs tools for understanding the trade-off between biodiversity protection and economic activities that result in biodiversity loss.

Biodiversity is often undervalued by modern economies because of market failures related to the production and consumption of goods. Society faces diminishing biodiversity, but in contrast a prospering economy, with limited market links between them. However, our quality of life depends not only on a strong economy, but also on a healthy environment. The functions performed by ecosystems, such as air and water purification, flood control and climate stabilization help improve the wellbeing of society. Biodiversity is a major contributor to economic well being through the supply of many ecosystem goods and services, but markets rarely exist for these ecosystem services. Typically, those who do supply ecosystem services are not rewarded for all the benefits they provide to others, and those who reduce ecosystem services do not bear all the costs they impose on others. This leads to reduced conservation incentives and insufficient biodiversity relative to the social optimum.

If the ecosystem and related services provided by biodiversity are not traded in the market, they are not directly quantifiable in monetary terms, and so do not have an obvious price or value. Consequently, when there is no explicit value for a good or service provided by biodiversity, conservation concerns are not included in society's decision-making process and biodiversity loss occurs. This problem arises because biodiversity loss reflects externalities and biodiversity conservation is a public good. Reversing biodiversity loss therefore requires dealing with these market failures. This problem is made more difficult by the fact that biodiversity has elements of a global public good so that effective decision-making requires international cooperation that is often difficult to organize.

Because policy makers may not have full awareness of the consequences of biodiversity loss, they need to be supported in increasing their understanding of the total

value that biodiversity offers and the valuation techniques and economic tools available to deliver the value of biodiversity to society. This paper develops a framework for incorporating economic instruments and theory into biodiversity conservation. The paper shows the potential of economic tools for policy development, and in essentially creating a market for biodiversity's goods and services. Through the use of economic valuation tools, policy makers will be able to incorporate economics into their decision-making processes more effectively. Increased use of economic techniques will improve their acceptance and help to correct flaws that remain in their application to biodiversity. Economic instruments and incentives that aim to overcome the market failure causes of biodiversity loss can provide a supportive economic environment for international biodiversity agreements and related policies to promote the conservation of biodiversity.

II BIODIVERSITY: AN OVERVIEW

An important step in incorporating economic instruments, theory and valuation into biodiversity preservation, is to formally define the term ‘biodiversity’. According to the United Nations Convention on Biological Diversity (CBD), biodiversity is the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the interactions between them, including diversity within species, between species and of ecosystems (Secretariat of the Convention on Biological Diversity (CBD), 2006). Essentially, biodiversity is the full variety of life on Earth, and is most often understood as the number of different species of plants, animals and microorganisms in existence on the planet. Biodiversity however also encompasses the specific genetic variation and traits found within a species, and the corresponding interactions with the ecosystems in which they live (CBD, 2006).

At the *genetic level*, differences in DNA codes within a species are responsible for uniqueness such as different varieties of crops and breeds of livestock. This refers specifically to the degree of variability in a species. The difficulty with biodiversity at the genetic level is that extinction of certain gene pools reduces their availability for future generations and the potential for still undeveloped medicines, crops, pharmaceuticals, petroleum substitutes and other products that may never be discovered. The variety of species in fact is the result of billions of years of evolution across species and within individuals of the same species leaving genes as the ultimate basis for biodiversity (UNDP, 2007).

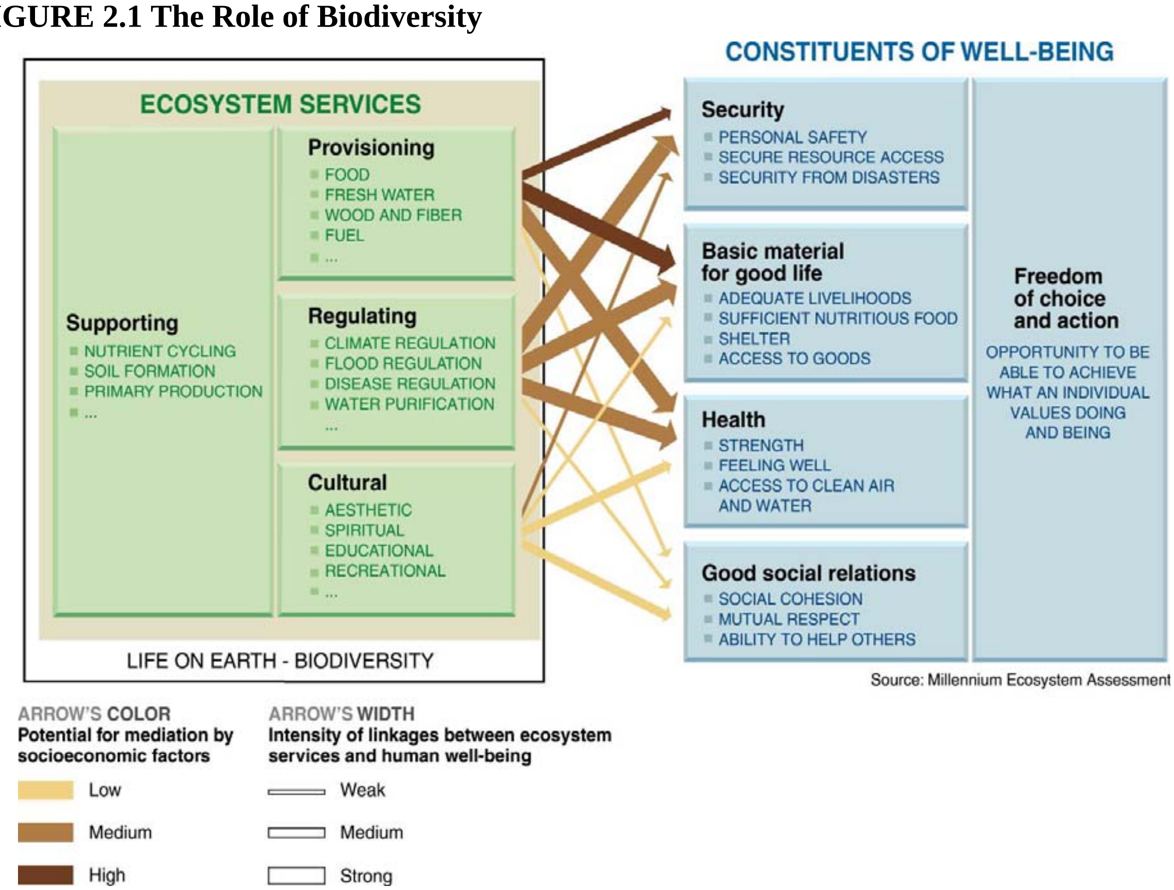
Species diversity refers to the variety of species on earth or in a given area. To date, around 1.7 million species have been discovered and described; which is only a fraction of the true number of species. For example, only around 275 000 marine species have been identified, even though in reality as many as 10 million more species may be undiscovered in the deep ocean basins. In fact, estimates of the total number of species on earth including those discovered ranges from 5 – 300 million, but only a small percentage of these species have been assessed for potential economic benefits (UNDP, 2007).

Ecosystems can be seen to encompass the species and their genetic diversity. An ecosystem is an array of living things and the physical and chemical environments with which they interact with (UNDP, 2007). Put more simply, ecosystems such as forests, wetlands, grasslands, lakes and rivers, consist of living creatures interacting with one another to form the variety of life which is biodiversity. Depending on the environment and characteristics of the ecosystem (area, latitude, etc.), an ecosystem may contain many millions of different species or subsequently not many at all (UNDP, 2007). It is clear that species and genetic diversity help to enhance the stability of an ecosystem, as each one consists of living creatures working together to create a basis of life from which humans are actively involved and dependent upon.

THE ROLE OF BIODIVERSITY

Biodiversity fuels ecosystem functioning and the services provided by healthy ecosystems are the foundation for human wellbeing. Ecosystems not only deliver the basic material needs for survival such as food, water and shelter, but the combination of species, genetic variations within the species, and their interaction with one another has also made the Earth habitable for humans (see Figure 2.1).

FIGURE 2.1 The Role of Biodiversity



Biodiversity is therefore an important factor in human existence and is particularly important in maintaining productivity and resilience. Photosynthetic species for example, purify the air and regulate the composition of the atmosphere by recycling oxygen. Wetland ecosystems absorb and recycle essential nutrients, treat sewage and cleanse industrial waste. As well, some 130 billion tons of organic waste are processed every year by Earth's decomposing organisms, not to mention industrial wastes such as detergents, oils and paper which are detoxified, decomposed and returned to the plants as nutrients (UNDP, 2007). Biodiversity is important in generating soil and maintaining soil quality to ensure an abundance of crops and plant fauna. An estimated 25-50% of the World's crops are destroyed every year by pests, but without biodiversity this number would be much larger, and would result in devastating impacts to countries that depend on strong agricultural systems. Around 99 percent of crop pests are currently controlled by a variety of insects, birds and fungi, and these natural pesticides are superior to chemical controls as pests can often develop resistance to artificial pesticides (UNDP, 2007). Biodiversity produces the majority of food and nourishment to the Earth, and when traditional food sources fail, a healthy biodiverse ecosystem can usually provide viable alternatives.

Biodiversity also helps to promote climate stabilization and reduces the effects of natural disasters. Plant tissues and other organic materials act as repositories of carbon, which slows the buildup of carbon in the atmosphere, thus contributing to climate stabilization. Ecosystems bordering regular flooding rivers help to absorb excess water and reduce the damage due to floods (UNDP 2007). The damage to New Orleans from the storm in recent years was increased dramatically by the loss of wetlands in the area, and resulted in dire consequences for the people and economy of New Orleans.

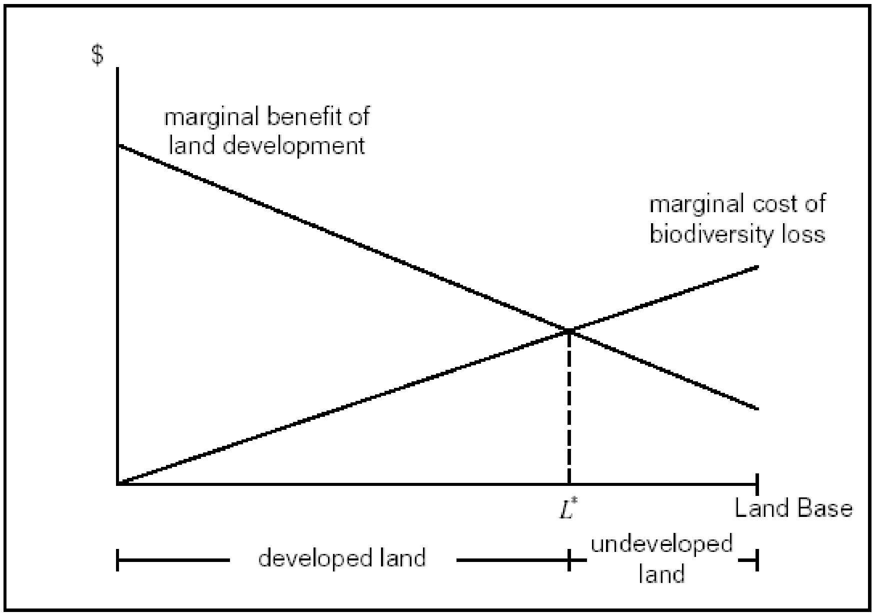
Because biodiversity's store of genetic material has so many applications, biodiversity contributes greatly to human knowledge and understanding, especially in the medical field. Examples of pharmaceuticals developed from prospects into biodiversity include *taxol*, a treatment for breast cancer found in willow bark; and the extraction of *vincristine* and *vinbalstine* in the treatment of childhood leukemia, extracted from rosy periwinkle flowers (Heal, 2005). Clearly biodiversity's contributions are enormous, and though generally overlooked it is clear that they are important because with greater biodiversity, society has increased insurance against the impacts of a future adverse event (Bennett, 2003).

BIODIVERSITY LOSS

As of 2005, thirty-five animal and plant species in Canada were either extinct or extirpated, while 184 were considered to be endangered and another 129 classified as threatened (Statistics Canada, 2006). However, the main threat to biodiversity is through the destruction of tropical rainforests which are believed to hold between 50 to 90 percent of all species on earth (Kennedy, 1999). With the current levels of deforestation, large numbers of species may be lost, and along with them potentially large economic values. However, the economic approach to biodiversity loss is to evaluate the costs and benefits associated with conservation. On one hand biodiversity loss negatively impacts society, so it is valuable to conserve. On the other hand, production and consumption activities such as agriculture and forestry, also provide value but tend to conflict with biodiversity conservation. So reaching equilibrium between the costs of biodiversity loss and the benefits of other economic choices is an important concept in any conservation strategy.

The main threat to biodiversity is land use, so addressing biodiversity conservation often involves the management and regulation of land. While other policies are important such as pollution controls and restrictions on agriculture and fishing practices, policy that maximizes conservation should target the destruction of biodiverse land. Figure 2.2 reflects the discounted long-term benefits and costs of land development (Kennedy, 1999). The graph illustrates the relationship between the marginal benefits of land conversion and the marginal costs associated with biodiversity loss. We see that marginal cost of biodiversity loss is upward sloping, which implies that as biodiversity becomes scarcer, the value of biodiversity increases. In contrast the marginal benefit of land conversion is downward sloping which reflects the diminishing returns to land development. In this case as more land is converted, each additional unit of land conversion contributes less to the benefits of conversion.

FIGURE 2.2 Optimal Land Development For Ecosystem Type



Source: Kennedy (1999) pg. 5

Therefore the equilibrium in Figure 2.2 at point L* gives the optimal balance between developed and undeveloped land, but will differ depending on the ecosystem in question. In particular, some ecosystems will have higher marginal costs to biodiversity loss, or perhaps have higher development potential than others, which may require more or less land to be converted. In addition to this intuition it is helpful to find the most efficient allocation of land conversion which also reflects marginal costs and benefits. Land therefore should be allocated to the land use type (agriculture, forestry, tourism etc.) with the highest marginal net benefit up to the point where the marginal net benefits of different land uses reach an equilibrium (Kennedy, 1999). However, since the marginal social cost of land development is greater than the marginal private costs of development; this implies that L > L*.

For example, Canada was the first nation to establish a clear commitment to the pursuit of the sustainable forest. The National Forest Strategy (2003-2008), has been applied across Canada and is aimed at the management of sustainable forests "so that its functions, biodiversity, resilience and productivity are maintained over the long term" (National Forest Strategy, 2003 pg 6). Canada's forest represents over 10 percent of the world's forest cover, 25 percent of the world's natural forest, 30 percent of the world's boreal forest and 20 percent of the world's temperate rainforest, and includes some of the world's largest intact forest ecosystems (National Forest Strategy, 2003). These forests are a significant asset, with the wood and paper product industries bringing in a net contribution of $34 billion annually (National Forest Strategy, 2003). Since 94 percent of Canada's forest is public land (6 percent private), Canadian forests are subject to public good characteristics. The National Forest Strategy helps to combat this issue by increasing awareness of the non-market value of Canada's forest, as well as an increased information and management focus, to ensure that decision makers and forest managers are held accountable for their actions. Canada's National Forest Strategy is a promising example of how together both environmental and economic sustainability can be secured.

However, because land development and forest consumption is inevitable, it helps to provide models for land conversion and use as discussed earlier, to guide policy to the optimal conservation and development objectives. In some cases, the benefits of land use will outweigh the costs associated with biodiversity loss, but in other instances the opposite will occur and the costs of biodiversity loss will outweigh the benefits of land use. Therefore land conversion should only be allowed if the benefits outweigh the costs. Creating a monetary value for the benefits and costs of biodiversity conservation or loss is achieved through economic valuation and applying this value in resource allocation choices involves the use of economic tools.

THE ROLE OF LOCAL COMMUNITIES IN BIODIVERSITY CONSERVATION

When evaluating biodiversity conservation, especially for policy decisions, problems of poverty and environmental degradation must be considered as they are closely linked. The majority of the poor live in areas where the world's biodiversity is most threatened, and derive most of their income from these ecologically fragile environments. More than one billion people live within the world's 19 forest biodiversity 'hot spots' with population growth in these areas amounting to over twice the world's average rate of growth (Kaimowitz, 2003). Therefore, a particular policy concern is that the poorest segments of society tend to depend more on environmental income.

The apprehension behind this is that current forest management and protection schemes do not address this problem. For example, large-scale logging in commercial forests, industrial forest plantations and public protected areas all deprive poor communities of lands and biodiversity they traditionally controlled, while restricting their involvement in the commercial use of biodiversity (Kaimowitz, 2003). In particular, poorly defined property rights, a major source of environmental degradation, give few incentives to locals to invest in conservation because of uncertainty about the benefits they will derive from their investments. As a result changes in resources, markets and governance must be addressed to increase the role of local people in biodiversity conservation. As local communities use biodiversity as a source of subsistence, it seems fundamental to include them in conservation decisions.

Because the demand in developed countries for forest products cannot be fully supplied from within these countries, there opens up an opportunity for industry and investors to work together with local communities and producers. This in turn will contribute to employment and income for the rural poor, with economically valuable forests providing incentives for local people to protect environmental goods and services (Kaimowitz, 2003). Working together with local communities to create new market structures, new market instruments and new industries, may have the potential to enhance the economic value of biodiversity resources, and reduce the threat of extreme degradation. Therefore it seems that the problem of biodiversity loss is a global challenge that requires international co-operation to achieve substantial results. For example, Canada has participated in a number of international agreements that have a direct bearing on how Canadians manage the environment. These agreements include the Convention on Biological Diversity, Framework Convention on Climate Change, Convention on the International Trade in Endangered Species of Wild Fauna and Flora (CITES) and the Kyoto Protocol (National Forest Strategy, 2003). However, developing countries may not have the capacity to build biodiversity strategies, so it is important for capable countries to fill the information and perhaps financial gaps. Even so, achieving positive outcomes will require supportive governance and monitoring throughout, as well as an enabling market framework – this is where economics can be applied.

ECONOMICS AND BIODIVERSITY

An economist's approach to biodiversity involves the analysis of improving the wellbeing of people. In addition, the economic approach is *marginalist*, in that it involves the consideration of the impact on people that will occur when a change is implemented. The economic approach seeks to maximize utility from biodiversity conservation. This is an important distinction, because governments, contrary to economists, seek to maximize their probability of re-election, which may or may not involve the inclusion of environmental concerns in their platform. Therefore, the economics of biodiversity focuses not only on the value of biodiversity, but also the value of a change in biodiversity. So analysis requires an understanding of the connection between the choices people make, the changes in biodiversity resulting from these choices, and the changes in the wellbeing of people.

Biodiversity is clearly valuable to society, so the loss of biological resources and their diversity in turn will impact human wellbeing now and in the future, both directly and indirectly. Therefore an important point about biodiversity loss is that there is a causal relationship between its use and the choices people make, the status of the environment, and the wellbeing of people (Bennett, 2003). As a result the role of the economist involves an assessment of the impact of wellbeing that results from changes in biological diversity. This requires an analysis of the marginal utility of a person as the environment changes, which can then be applied in conservation goals and policy.

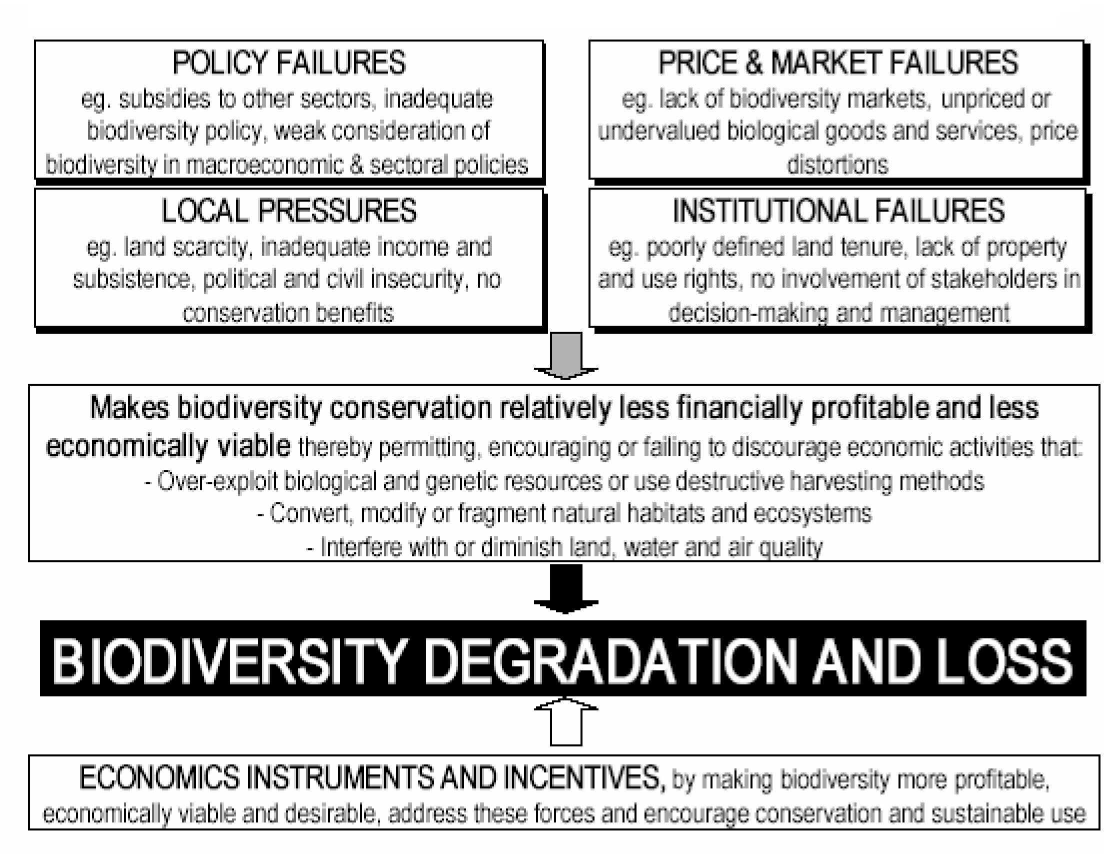
The implementation of economic instruments is expected to increase economic efficiency relative to the status quo, so they are of particular interest to biodiversity conservation policy. According to Hahn (2005), economists can influence environmental policy by advocating the use of economic tools, by analyzing the benefits and costs of regulations and standards and identifying the inefficiencies in the political economy of environmental regulation. Economists therefore help to deliver the value of biodiversity to society through the use of feasible economic instruments for promoting more efficient environmental policy.

III ECONOMIC INCENTIVES FOR BIODIVERSITY CONSERVATION

Conservation of biodiversity and natural resources in general provides a challenge due to the interaction of poorly defined property rights, externalities and market failure. Many of the benefits associated with biodiversity conservation outlined in Section II, are subject to public good characteristics, and as a consequence, the types of market forces we rely upon to supply society with goods and services, are unlikely to yield biodiversity benefits at the socially most desirable levels. This is the classical ‘market failure’ argument.

Price signals set by markets coordinate decisions about the quantity of goods and services produced and consumed. Economic activity responds to these signals because they reflect incentives of different production and consumption options. However, many economic and market decisions do not fully reflect the environmental costs associated with the use of environmental resources. Even when there is a market for biodiversity-based services such as bioprospecting, the market tends to only capture a portion of society’s willingness to pay for the service; thus biodiversity is undervalued even when it is in the market (Heal, 2005). Thus traditional reliance on market prices as scarcity indicators of resources, distorts the value of biodiversity to society, which in turn may cause people to over-consume and degrade the environment. As well, sometimes governments pursue policies that create or intensify negative externalities associated with natural resources, such as macroeconomic policies targeting industry protection, exchange rates, export taxes, perverse subsidies etc. that create feedback loops resulting in unfavourable consequences for ecosystems (Brown, 2000). It seems that in order for economic instruments and incentives to be effective they must address the distortions and failures in the ways in which markets, prices and policies operate (See Figure 3.1).

FIGURE 3.1: System Causes of Biodiversity Loss



Source: The Use of Economic Measures in National Biodiversity Strategies and Action Plans: IUCN

MARKET FAILURE

Markets usually evolve and are shaped by the interests of buyers and sellers, and allow for individuals and firms to engage in transactions that create value (Grafton, 2005). However, markets may not evolve in all areas of the economy and are missing for some goods and services such as those involved with biodiversity use. When markets do evolve for biodiversity, typically those who supply ecosystem services are not rewarded for all the benefits they provide to others, and those who reduce ecosystem services do not bear all the costs they impose on others, or in other words an *externality* is created. Protecting biodiversity is a public good, but the incentives to destroy it cause negative externalities. Presence of externalities in turn can lead to market failure, which may be one of the largest incentives for biodiversity conservation through the use of economic tools.

Markets in this area tend to fail because of the public good nature of biodiversity, in particular due to 'non-excludable' and 'non-rival" goods and services. *Non-excludability* refers to the fact that it is hard to prevent parties who do not pay for an ecosystem service from benefiting from it. It may not be possible to exclude them from enjoying these services, and more importantly from incurring costs due to their use (Aretino et al, 2002). Thus, non excludability gives people the opportunity to free ride by using ecosystem services purchased by others, and generally is due to the lack of an enforceable property right. People find it easier to free ride on those who pay to conserve biodiversity than to incur the costs themselves. The free rider problem associated with biodiversity conservation can seriously decrease conservation incentives. Additionally, many goods and services derived from the ecosystem are *non-rival*, so the enjoyment of the environment by one individual does not detract from the ability of another individual to also enjoy the environment, at least up to a certain level (Grafton, 2005).

Even though society is willing to pay for an increase in the supply of environmental goods, and that potential suppliers exist, markets do not necessarily develop. However, willingness to pay for environmental goods and the costs of supplying these goods varies; therefore more information must be provided to avoid the free-rider problem, and to increase investment in biodiversity. Given better incentives through the use of economic instruments, there will be more opportunities to invest in activities that supply environmental goods and services. However there exist a number of impediments to repairing market failure in relation to the environment. These include property right specification and asymmetric information.

Inappropriate property rights: Missing markets for environmental goods and services can be the result of a lack of or inappropriate specification of property rights. Coase (1960) suggests that if property rights were clearly specified, with little or no transaction costs, trade would occur and markets would arrive at efficient outcomes. However, often in the case of environmental goods, property rights are poorly defined and transaction costs can be quite large. In order for the exchange of property rights to be credible for people to be willing to participate, property rights must be very clear to specify the environmental outcomes sought as well as acceptance of the security of those rights (Grafton, 2005).

Asymmetric information: Asymmetry of information can create uncertainty for parties as it refers to a situation where one party has more relevant information than the other party. Consequently, there are instances where this can prevent markets from being formed, and potential gains from doing business may not be realized (Grafton, 2005).

GOVERNMENT AND MARKET FAILURE

The implication of market failure is that there is a role for economists to influence potential government decisions, which attempt to fill the gap left by market forces. However, governments should only become involved when their actions can be justified by an improvement in human wellbeing, or that the benefits of the intervention exceed the costs. As such there is always the idea of 'government failure' arising because of inadequacies in the bureaucratic/political processes involved in policy design and implementation (Bennett, 2003). For example, political decisions in South America have adversely affected rates of deforestation through: tax free agricultural income; subsidized land clearing for cattle ranchers; and public road construction for military purposes or to reach new mines, which all reduce the costs of obtaining access to forests (Deacon, 1994). Thus with government intervention, there is a danger of moral hazard and government decisions being misdirected through the influence of different interest groups.

In addition, there is discussion on whether environmental services are characterized by an income elasticity of demand and whether they can be classified as luxuries or not. But because of the nature of environmental goods, it is quite conceivable that some environmental goods may be considered luxuries while others may not (Hokby & Soderqvist, 2003). So an importance must be put on the extent of the income elasticity of willingness to pay for environmental services, or how willingness to pay (WTP) is affected by changes in income. Because willingness to pay reflects preferences, the income elasticity of willingness to pay depends upon the demand income elasticities of all other environmental goods, and the substitutability between these goods (Carson &

Flores, 1997). Part of the problem is that there is a strong income elasticity for biodiversity, but many of the biodiversity ‘hot spots’ are in low income countries where the demand for biodiversity is small. Plus it is difficult to find ways to translate high income demand into efficient mechanisms.

Economists however, have developed analytical tools to help guide policy towards the more sustainable use and allocation of natural resources for current and future generations, as well as helping society internalize the true value of biodiversity loss. The discussion will begin with economic valuation.

IV ECONOMIC VALUATION OF BIODIVERSITY

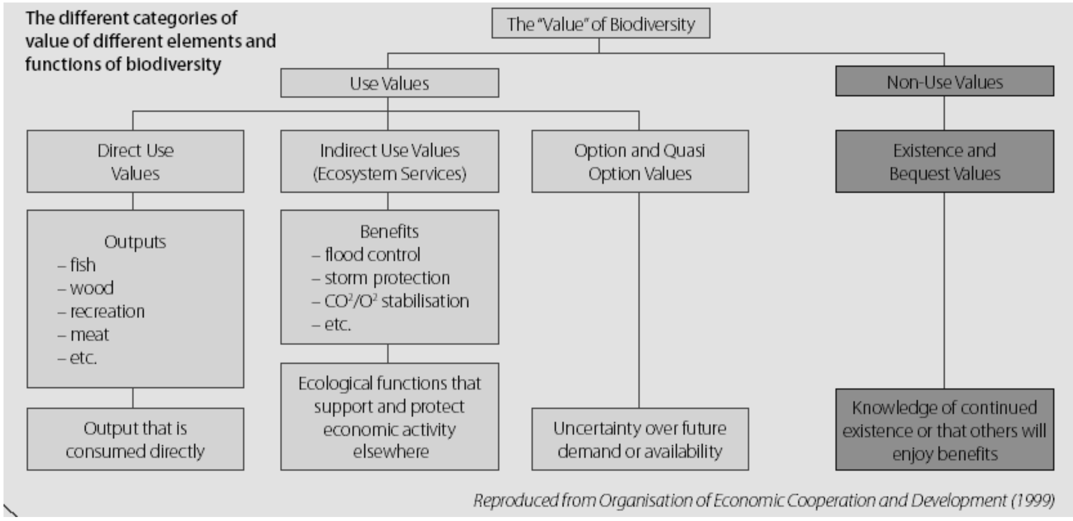
In economics, the importance of biodiversity stems from the relationship among the consumption choices people make, the resulting changes in biodiversity and changes in welfare. Biodiversity therefore serves as one indicator of human wellbeing because of its causal link to human wellbeing, and the relationship between human cause and human effect (Bennett, 2003). In order to predict what effect a choice will have on a person's utility, it is first necessary to be able to predict what is happening to the biological resource. Thus, a clear understanding of the impacts of human choices on the environment is a prerequisite to economic analysis of changes in the utility of people (Bennett, 2003). It is useful therefore to measure biodiversity-induced changes in human welfare using the different types of economic value that individuals take from biodiversity, or 'total economic value'. Developing a complete value for biodiversity and the goods and services it provides will help to encourage the formation of markets where biodiversity can be traded, and conservation incentives realized.

TOTAL ECONOMIC VALUE

Because decision makers have traditionally seen the value of biological resources in terms of the direct uses they support, such as the raw materials provided for human production and consumption, it is important to use economic valuation to avoid the undervaluation of biodiversity, and help reflect the resultant changes in human welfare. Biological valuation also helps to provide a means by which the benefits of biodiversity conservation can be better taken into decision making processes to create more efficient decisions about biodiversity and its use (Johns et al, 2006).

Total economic value (TEV), reflects the broader benefits associated with biodiversity aside from the direct, commercial uses, including non-marketed values, ecological functions and non-use benefits. In other words, TEV consists of *use value* and *non-use value* (see Figure 4.1). Total economic value helps to create a more complete idea of the economic importance of biodiversity while also demonstrating the economic costs associated with its loss and/or degradation.

FIGURE 4.1 Total Economic Value



Source: Biological Diversity Advisory Committee (2005)

Use values involve interaction with the biological resource either indirectly or directly, including option and quasi-option value.

Direct use value: Individuals make use of a resource in a consumptive or non consumptive way (Johns et al, 2006). For example:

- Goods and services arising from marketed biological goods such as pharmaceuticals and agricultural products that are impacted by the health of that particular biological resource
- Benefits generated by eco-tourism and recreation

Indirect use value: Individuals benefit from biological services, supported by a resource rather than actually using it (Johns et al, 2006). For example:

- Nutrient removal, flood control, climate and weather stabilization etc.

Option value: an individual derives utility from having the option to make use of a natural resource in the future, even if there is no current plan to make such use. Option value exists through risk-averse consumers, because of the uncertainty concerning future preferences and/or availability of a particular good (Johns et al, 2006).

Quasi-option value: a value arising from avoiding or delaying irreversible decisions, where technological and knowledge improvements can alter the optimal management of a natural resource (Johns et al, 2006). For example:

- Potential for bio-prospecting - using genetic information to be used in pharmaceutical development

Non-use values are derived from the knowledge that ecosystems continue to exist and are being maintained. Non-use values are not associated with any use of the resource or tangible benefits derived from it, but users of a resource may attribute both use and non-use value to resources (Johns et al, 2006).

Existence value: derived from knowing that ecosystems continue to exist, regardless of use made by society (Johns et al, 2006). Existence value also includes ethical considerations such as the extinction of species and ecosystems in that they should exist for future generations.

Bequest value: value derived from this stems from the idea that there will be a continuing availability of biological resources passed on to future generations (Bennett, 2003).

Biodiversity therefore provides a wide range of values to society and many contributions to human wellbeing. Understanding the impact on wellbeing, or the change in value associated with biodiversity loss or conservation is of particular interest for economic incentives to be applied to the concept of biodiversity.

UNCERTAINTY

It is important to note that while the idea of total economic value is complete in theory, in reality it is often not easy to calculate these values. Uncertainty is first introduced as a problem because we cannot be completely sure of the consequences of human production and consumption decisions on the environment. We cannot predict exactly the impact of human choice on a particular species or ecosystem, as this is unlikely to be a black/white decision. That is, human choices will have an impact on the *probability* of a species becoming extinct, endangered etc, leaving the economics of biodiversity preservation as a matter of probabilities (Bennett, 2003).

EXAMPLE OF A VALUATION FRAMEWORK

The problem in biodiversity preservation lies in specifying the objective that we are trying to preserve. Like most of what society wants, biodiversity is subject to the problem of preserving the maximum degree of diversity subject to a budget constraint. Considerations of the cost effectiveness of a conservation incentive are of particular importance in order to maximize the allocation of the budget across different conservation goals. When there are not enough resources to protect every species, policy must choose which species, ecosystems etc. to invest in; investment in a conservation incentive will ultimately increase the survivability of a particular species (Metrick & Weitzman, 1998).

Benefits of a species can be measured by the value obtained by society through direct utility from each species, and indirect utility coming from the overall diversity of genes. Utility is measured through a combination of use and non-use values such as commercial, recreational and emotional reactions to a species. Species therefore should be ordered for protection, according to their gains in utility plus diversity, weighted by the increase in the probability of survival, per dollar of cost (Metrick & Weitzman, 1998). This framework incorporates the collection of economic value subject to a real world constraint such as a conservation budget through the government.

According to Metrick and Weitzman (1998), assuming the cost of saving a species is relatively small, so that many species can be saved, a priority based ranking system can be justified as providing an arbitrarily close first order approximation to optimal policy. The priority ranking for policy purposes is as follows:

Di = distinctiveness of the species i = how unique the species
Ui = direct utility of i = how much we value i per se

ΔPi = by how much can the survivability of i actually be improved
Ci = how much does it cost to improve the survivability of i by ΔPi

*Provided ΔPi is relatively small (for all i), then a priority ranking which gives an arbitrarily close first order approximation to optimal policy is:

$$Ri = [Di + Ui][\Delta Pi/Ci]$$

In practice, it will not be easy to quantify the four variables, due in part to uncertainty problems, but nevertheless, when making conservation decisions in the name of preserving diversity, it is helpful to at least consider the above four factors, especially in policy decisions that lack otherwise clear guidelines for endangered species protection (Metrick & Weitzman, 1998). By deciding on some kind of objective to preserve, the problem of biodiversity preservation will become clearer and more focused for decision making purposes. Because preservation, like everything else, is subject to some sort of budget constraint, it is important to set an objective for preserving the maximum degree of diversity.

In conclusion, the undervaluation of biodiversity in the market, as well as the externalities that are associated with its loss, requires first addressing how to value biodiversity correctly. This has been demonstrated through defining total value. Secondly, addressing market failure and externalities involves implementing measures to ensure these values are taken into account in biodiversity conservation choices. These measures involve the application of valuation, and are known as valuation tools.

V VALUATION TOOLS

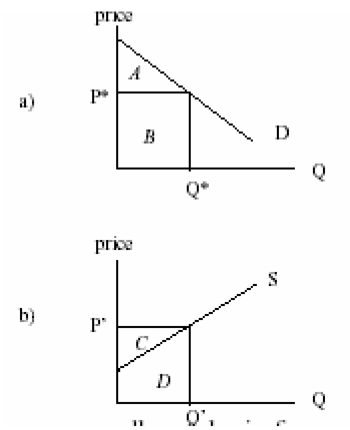
The next step in incorporating valuation into biodiversity and conservation policies is to apply a set of tools to encompass the total value. The problem faced by the government regarding biodiversity is to determine whether proposed interventions provide net benefits to its constituents. To a large degree this is no different than cases of intervention for other goods with public good characteristics such as defence, health services and education (Bennett, 2003). The economic framework used for these purposes is known as benefit cost analysis (BCA), where the benefits of an intervention are compared with the costs to see whether the policy produces a social gain relative to a specified 'base case' (Bennett, 2003).

However, the problem with government BCA is that the process requires estimation in monetary terms, which is problematic when many of the benefits associated with biodiversity are outside the market. This absence of market signals is related to the externality problem. Costs of biodiversity conservation on the other hand, are usually goods and services that are bought and sold in the market. These are predominantly the benefits of resource development that are foregone because of the restrictions imposed with biodiversity protection (Bennett, 2003). Benefit cost analysis is demanded when there is market failure, but requires estimation of the benefits and costs using market value. Therefore in the case of biodiversity focused interventions, it is necessary to have economic valuation tools that encompass the total economic values generated by biodiversity conservation (Bennett, 2003).

MARKET BASED TECHNIQUES

When biodiversity-related products are bought and sold directly in markets, standard economic techniques such as opportunity cost methods can be used to estimate values of benefits for both consumers of biodiversity benefits and their producers (Bennett, 2003). This technique is most suitable where direct use benefits are involved. Based on the quantity people purchase of a biodiversity good at different prices, and the quantity supplied at different prices, producer and consumer surplus can be calculated. Using market supply and price received through market transactions, the producer's profit can be estimated (producers' surplus) by the area C in Figure 5.1. Observations of market demand and price paid give estimations of the net benefit received by consumers when they purchase biodiversity goods and services, indicated by the area A in Figure 5.1, where A + B is the value obtained by consuming the good or service, at a cost of B. For marketed biodiversity goods, such as bio-prospecting, if there is sufficient market activity, then it is possible to use standard economic techniques to estimate values for buyers and sellers.

FIGURE 5.1



Where S = Supply of good/service
D = Demand of good/service
P = price of good/service
Q = quantity demanded/supplied

Source: http://www.bus.msu.edu/econ/brown/pim/pdffiles98/csps98.pdfht

Market based techniques however, can only be applied to environmental goods and services that are marketed, or have clear market-based substitutes (Johns et al, 2006). For this reason, market-based techniques are rarely used to value biodiversity because many of the benefits of biodiversity are yet to be valued in markets or subject to prices that reflect their true value. In these cases, alternative methods should be used.

REVEALED PREFERENCE TECHNIQUES

When market data are available for goods and services that are specifically related to the biodiversity value in question, the data can be used to infer values for the biodiversity goods and services (Bennett, 2005). These revealed preference techniques include three different approaches.

The first group of these techniques involves observing consumer behaviour (specifically purchasing behaviour) when biodiversity benefits are threatened. Values are calculated by the amount of money people are willing to spend to avoid the consequences of biodiversity loss. These are known as *preventative or mitigatory expenditure* and *averting behaviour approaches*. The second approach is known as replacement cost techniques, which generate value from how much it would cost to replace the loss of biodiversity benefits with a substitute (Bennett, 2003). However, the above approaches do not specifically estimate the value of biodiversity benefits, but rather they estimate costs associated with providing substitutes or avoiding biodiversity loss. Because of this, their accuracy may be affected if the substitutes are not perfect and if decisions are not made with consideration of the benefits provided, where the costs can begin to exceed the benefits (Bennett, 2003). Thus, these approaches are better for providing a first approximation of value, but whose values should be reinforced through other methods such as the production function technique or hedonic pricing.

More appropriate revealed preference techniques rely on the observations of market behaviour that are specifically related to the values impacted by biodiversity change. The first approach, *the production function technique*, focuses on the indirect relationship that may exist between a particular ecosystem good or service and the production of a marketed good, and captures the indirect use component of total economic value (Johns et al, 2006). The relationship between inputs and outputs in the production process, or production function, can be used to infer values for the inputs even when they are not marketed. Environmental goods and services are considered the inputs to the production process, and changes in the production process of market goods that result from an environmental change, allow for the value of the environmental good or service to be inferred (Johns et al, 2006). However, to apply this approach, good biophysical information on the production function is required before values can be estimated (Bennett, 2003). The approach requires a great deal of data regarding the final goods market and factor inputs, to identify and specify the relationship between different factors in the production and/or cost functions. So generally, the production function approach is used for environmental inputs such as water, soil, raw materials, air quality, and again is best used with other methods (Johns et al, 2006).

Hedonic property pricing techniques are based on the idea that the price at which property sells is determined in part by the environment and environmental conditions surrounding the location. The economic value is calculated using data on property sales and by regressing the sale price against all other factors thought to affect the price. However, like other revealed preference techniques, hedonic pricing requires a large amount of data on property prices and property characteristics and a high level of econometric analysis. However the biggest limitation of this method is it can only detect changes in welfare due to changes in the environment that are perceivable by individuals, such as smog or proximity to landfills (Johns et al, 2006).

Because biodiversity typically holds a high value as a recreational resource or destination, in many cases there are charges to view or enjoy natural ecosystems and species, and people spend time and money to reach these areas. The *travel cost method* is used to infer a demand curve for a non-marketed environmental asset that relates to recreation and tourism. The curve is derived from an estimation of the relationship between visitation rates and the costs of traveling to the site (Bennett, 2003). These costs reflect the value that people place on the leisure, recreation and tourism aspects of biodiversity. However, this approach is still limited to measuring direct use value, and while users may also hold non-use values, these must be estimated separately (Johns et al, 2006). As well, this method is limited to valuing environmental goods and services that have recreational uses such as woodlands, national parks, rivers and lakes and coastal areas (Johns et al, 2006).

STATED PREFERENCE TECHNIQUES

As we can see, there are limitations to the range of values that market based and revealed preference techniques can estimate. Stated preference techniques were developed in part to address this problem, and in particular to create a value for non-use environmental benefits. Since people do not reveal their entire willingness to pay for biodiversity goods and services through the purchases they make or through their behaviour, the only other option is to ask people directly about their willingness to pay. In stated preference techniques, people are asked about their willingness to pay for the services offered by biodiversity under a hypothetical set of circumstances (Bennett, 2003).

The first stated preference technique developed was the *Contingent Valuation Method* (CV), which is one of the few techniques that can be used to assess the option and existence values associated with biodiversity. Originally this method involved asking a sample of people their willingness to pay for an improvement in a particular aspect of biodiversity, but more recently has included a dichotomous choice version, asking participants if they would or would not support a proposal to improve biodiversity based on some personal monetary cost (Bennett, 2003). The method is therefore able to estimate the total economic (both use and non-use value) of an environmental good or service. However, reliable valuation methods are not simple to implement as time is required to develop the survey and to ensure that the survey is clear and explained properly. Also because CV methods are done through questionnaires, this method is prone to biases, such as when the participant provides a biased answer believing that it will influence policy or when participants are asked to value something for which they have little or no experience (Bennett 2005).

The most recent stated preference technique developed is *Choice Modeling*. Under this method a sample of people are asked to choose their most preferred alternatives from a number of grouped options that relate to different biodiversity conservation strategies (Bennett, 2003). Each option is described in terms of a personal monetary cost and in terms of different biodiversity outcomes. Because each combination has a 'price' associated with it, the participant's choice reveals their willingness to pay or to accept for each of the options presented to them, which also encompasses the total economic value for biodiversity changes (Johns et al, 2006). Choice modeling is more flexible than contingent valuation because it values a wide range of environmental goods and services. Choice modeling also includes changes that have not yet taken place, which allows for more incorporation of uncertainty surrounding environmental impacts than was possible with CV methods (Johns et al, 2006). Limitations of this approach are similar to CV methods, particularly as the complexity of the survey increases, leading to an increased degree of random error in the responses and potential biases. It should be expected that as the number of rankings increases, so does the likelihood of inconsistent responses due to limits on cognitive ability (Johns et al, 2006). Nonetheless, choice modeling can be used for benefit cost analysis of projects and policies as it reflects both use and non-use value components.

ISSUES

The main issue with valuation stems from the collection of relevant data. Unless the ecological system is understood, the role of economics can be very limited. In many circumstances, the ability of biophysical scientists to predict the outcomes of alternative interventions is very limited. As well, often the information that is provided about an ecological system is not of direct relevance to the types of policy issues facing society (Bennett, 2003). Therefore, before economic valuation can reach its potential, the capacity of science to provide more policy relevant results, must be expanded.

Another set of issues that emerges are ethical concerns to whether it is appropriate to put a dollar value on something like an endangered species or depleting ecosystem. Regardless of this debate, decisions are constantly made about the human choices that may affect biodiversity, so it seems that placing a value on biodiversity is necessary to preserve whatever value society places on it. Therefore, improved valuation studies of biodiversity will contribute to decision making practices (that would continue regardless of a valuation system); practices that through economics will consider the environmental, economic and social attributes of the outcomes (Bennett, 2005). The issue of placing a monetary value on biodiversity should therefore not be one of ethics, but more of a question of whether the values are made explicit or kept implicit in decisions and policy development (Bennett, 2003).

Another limit to the application of economic valuation tools to biodiversity is questions about the tools' abilities to deliver accurate estimates of biodiversity value. Issues arise because of the complexity of the biodiversity issue and the general public's ability to understand these complexities (Bennett, 2003). As well, there is a huge potential for biased results, when respondents deliberately misrepresent their true preferences in order to manipulate the results of policy decisions. Or they may bias their results because the questionnaire is hypothetical and they may not take it seriously, and in hypothetical situations people are not subject to a budget constraint. These debates regarding economic valuation tools have made policy makers reluctant to implement them or use their results in the policy making process.

Nonetheless, economic valuation techniques are helpful to capture the total value of biodiversity that otherwise would be ignored. If anything they are a step in the right direction towards helping society realize the potential gains from biodiversity conservation, by helping to internalize the costs of biodiversity loss. As well, economic valuation and tools give rise to greater interest in their role in policy development, and their ability to achieve conservation goals, often at lower costs than current regulatory measures. Valuation tools have the potential to achieve conservation goals more effectively and this gives policy-makers the incentive to support continued research into their use and effectiveness. This potential will be discussed in the next section by introducing the idea of market-based instruments to conserve biodiversity.

VI THE USE OF MARKET-BASED INSTRUMENTS TO CONSERVE BIODIVERSITY

BACKGROUND

Biodiversity value estimation has struggled to reach a higher priority in policy development, but has been aided by the recent interest in economic analysis of biodiversity and its sustainable input practice. Efficient economic tools aim to provide the proper incentives to promote biodiversity conservation.

Cost effective policy measures and strategies are becoming more important in developing policies able to reach specific environmental goals. Thus far, the predominant response to market failures for public goods, including biodiversity, has been to provide the good through the public sector and to place limits on the amounts used. The most common tool to achieve this objective has been the use of government provision and regulation derived from environmental law, known as command and control (CAC) methods (Brauer et al, 2006). In the case of biodiversity, CAC methods take the form of parks and nature reserves, and as a result may fail to address all of the externalities associated with biodiversity loss. As such, more attention has been put on biodiversity protection for privately owned land. Although CAC and other government initiatives raise community awareness of environmental issues, they have not been sufficient to solve the problem.

The response to this led to the development of policy measures known as 'market based instruments' (MBIs), which were developed by economists to involve the use of financial incentives to promote biodiversity protection (Bennett, 2003). Because the cost of regulation can be quite high, market-based instruments are increasing in popularity due to evidence that they can achieve some environmental objectives at lower economic cost than conventional CAC approaches. Referring back to the priority ranking model as discussed in Section IV, the cost of improving the survivability of a species is very important when making conservation decisions. Because preservation is subject to a budget constraint, the more cost effective a policy incentive turns out to be, the larger is the number of species and ecosystems that can be protected. Market-based instruments therefore give biodiversity preservation a more favourable benefit-cost context.

The case for market-based biodiversity conservation is based on some of the failures of traditional approaches, and an early positive policy experience in addressing other environmental issues (Bishop et al, 2006). Other advantages include greater flexibility and innovation, and in some cases reduced enforcement costs due to better co-operation between private and public interests. The greatest potential however is persuading the private sector, through policy incentives, to consider improved management of both public and private natural resources (Bishop et al, 2006).

MARKET-BASED INSTRUMENTS

Market-based instruments are policy tools that influence behaviour that will ultimately lead to environmental benefits, through market signals rather than through direct regulation. MBI's attend to the causes of market failure for a good, and according to the European Environment Agency: *"Market-based instruments seek to address the market failure of 'environmental' externalities' either by incorporating the external cost of production or consumption activities through taxes or charges on processes or products, or by creating property rights and facilitating the establishment of a proxy market for the use of environmental services"* (EEA; as cited in Brauer et al, 2006). For environmental goods, addressing the market failure means addressing the lack of fully defined and enforceable property rights for these goods. By defining and enforcing clear property rights, MBIs use economic valuation to create a monetary measure which can be exchanged through the market (Coggan et al, 2005). When biodiversity conservation has the potential to be profitable, firms will begin to see incentives in its conservation and the sustainable use of environmental goods and services should follow.

When designed correctly, MBIs have the potential to deliver environmental protection outcomes at a lower cost and according to Coggan et al (2005) achieve these cost reductions in three ways:

1. Allowing flexibility with respect to the way participants respond to the instrument, and in turn encouraging innovation.

2. Encouraging change amongst more cost-effective parties who can achieve change most cheaply, as opposed to leveling change requirements on all.
3. Placing positive incentives on natural resource management, compared to negative incentives associated with regulatory approaches, helps to attract new sources of funding for a reputable under-funded activity.

When MBIs are used, these instruments should be more effective because participants are allowed free choice and the flexibility to act in the manner that benefits them the most. Firms that can achieve changes at lower costs will do so, leaving other options for those who cannot achieve low-cost change. However, MBIs may not always be the best policy response because their success depends on proper price signals being sent or that potential gains from trade exist, and that transaction costs involved in trade must be low enough for the mechanism to be cost effective (Brauer et al, 2006).

TRANSACTION COSTS, THE ALLOCATION OF PROPERTY RIGHTS AND MARKET CREATION

Despite the advantages of MBIs, there are a number of factors that may affect the performance of these instruments such as transaction costs and property rights. Transaction costs have the potential of adversely affecting MBIs with respect to the costs involved in implementing the instrument. As mentioned before, a market-based policy incentive must be cost effective to implement, and high transaction costs can burden this process. This is particularly a problem when property rights are not pre-existing, which is often the case for biological goods and services (Coggan et al. 2005).

Property rights in general, help to facilitate exchange in markets, to ensure that the market is working smoothly and efficiently. According to Coase (1960), if there were no costs of exchange, the final outcome from the exchange is not dependent on the initial allocation of property rights and full information will be revealed through trade. In the case of biodiversity, costs of exchange, or transaction costs do exist. These costs occur due to the coordination and activities of people in the market, and arise because of uncertainty associated with market exchange. To address this uncertainty, participants in the market have to find and test knowledge through actions such as finding contract partners, gaining knowledge about goods and services, and negotiating, monitoring and enforcing contracts - activities that bear a cost. When transaction costs are too high such as where property rights are not well-defined, the quality and quantity of market exchange will be reduced (Coggan et al, 2005). This is the problem faced by ecosystem goods and services because often property rights are not defined. MBIs aim to address a part of these costs so a better flow of exchange can be made in the market. The result is the MBI itself incurs a portion of the transaction costs through better definition and allocation of property rights, such as codifying, identifying and enforcing ownership over these rights (Coggan et al, 2005).

Property rights essentially are the basis for MBIs because they specify a benefit or entitlement, they specify who can access that benefit, and the associated obligations that access to the property would entail. This is known as *duty of care*, and is required in order to access part or all of the benefits available under the rights owned (Whitten et al, 2003). Defining the duty of care gives owners incentives to change their behaviour in a way that would protect and maximize the benefits received from access to the property. For biodiversity this implies sustainable use. Thus by constructing a new property right, MBIs will encourage participants to uphold their duty of care, and promote better management of the environments of which they have access to. However, there may exist other barriers to market creation, so only designating property rights may not result in market creation. Market creation is thus more likely to be effective when a number of criteria are met; most which can be addressed in relation to the newly defined property right (see Table 6.1).

As well, consideration must be given to problems arising from asymmetry of information, and a small number of buyers/sellers. Asymmetry can cause the market to not be formed in the first place, and a small number of buyers and sellers give opportunities for strategic behaviour which may manipulate the price of the new property rights. Unfortunately with biodiversity, it is not completely clear what outcome a market would deliver as there is yet to be a broad consensus on how biodiversity can or should be measured. It is therefore difficult to define a common unit of trade as it relates to biodiversity property rights, thus leading to uncertainty which as discussed earlier increases transaction costs (Aretino et al, 2002). This fact reinforces the importance of government research into economic valuation as a means of defining a unit of trade for biodiversity and its components to be traded, which usually involves providing a better value for biodiverse land. In summary this section has examined MBIs largely at a conceptual level; the next step will be to apply these concepts to practical examples.

TABLE 6.1 Desirable Property Right Characteristics For Creating Markets

Property right characteristic	Description
1. Clearly defined	Nature and extent of the property right is unambiguous.
2. Verifiable	Use of the property right can be measured at reasonable cost.
3. Enforceable	Ownership of the property right can be enforced at reasonable cost.
4. Valuable	There are parties who are willing to purchase the property right.
5. Transferable	Ownership of the property right can be transferred to another party at reasonable cost.
6. Low scientific uncertainty[a]	Use of the property right has a clear relationship with ecosystem services.
7. Low sovereign risk[a]	Future government decisions are unlikely to significantly reduce the property right's value.

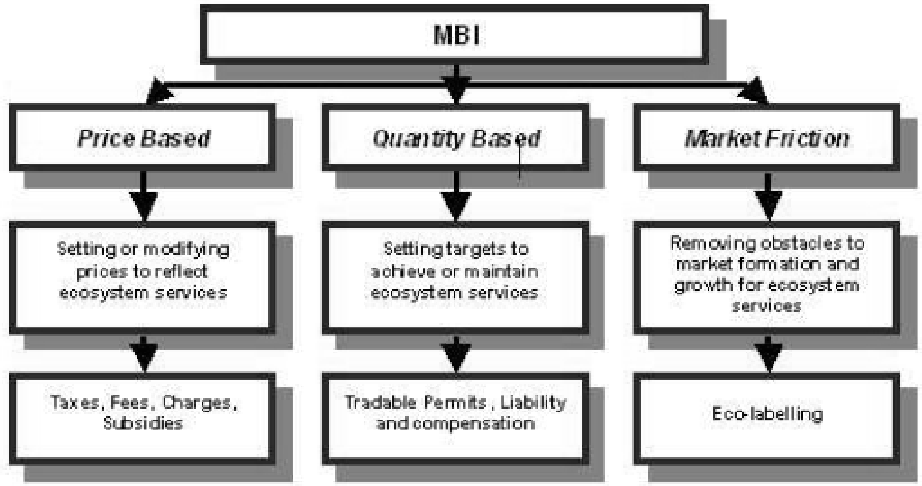
[a] Low in the sense that it does not prevent a market from forming. Moderate levels of risk and uncertainty are not necessarily insurmountable barriers to the operation of a market.

Source: Aretino et al (2002), p. 10

FUNCTIONAL MECHANISMS OF MBIs

MBIs are commonly categorized as either price or quantity based instruments, and in addition, instruments aimed at improving the operation of existing markets, known as 'market-friction' instruments, are sometimes included in MBI categorization. Each functional mechanism helps to address the specific market failure involved with biodiversity preservation. The categories of instruments are illustrated in Figure 6.2 below.

FIGURE 6.2 Functional Mechanisms of MBIs (Source: Brauer et al, 2006 p. 12)



Price based instruments

Priced based MBIs, also known as direct incentives, assign a price to environmental impacts (beneficial or damaging) within markets by using direct positive incentives such as subsidies, or negative incentives in the form of taxes, charges and fees (Brauer et al, 2006). This is an attempt to give these activities a price, thus internalizing any previous externalities, with firms responding to the new market signals by adopting the resource use or management practice that offers the greatest benefit and costs them the least. Effective policy leads to better resource management by relying primarily on price signals to create incentives for potential participants (Coggan et al, 2005). A disadvantage of price based instruments however, is that they cannot guarantee the extent of changes in behaviour, and as a result there is still a danger of overexploitation.

Quantity based instruments

These indirect incentives create ‘tradable rights’ to engage in an activity associated with a specific resource use or environmental damage, and create a market where permits are distributed and traded. By restricting the total level of environmental activity and allocating rights to participants, not only is an efficient allocation of rights achieved through market exchanges, but they also produce a certain environmental outcome (Coggan et al, 2005). Thus the total amount of damage can be controlled while allowing for more flexibility as those who can make changes to their behaviour the cheapest and easiest will make the biggest changes and then sell their permits to those who find change very expensive such as older, less innovative firms (Brauer et al, 2006). The limit on the number of permits is set by the Government or a designated authority, that also determines who can own the various rights, the initial allocation of rights, the conditions under which trade can take place, and how rights will be monitored and enforced (Aretino et al, 2002). Given this information, it is clear that quantity based instruments need greater administration and government backing in order for them to be successfully implemented and monitored.

Market friction

As mentioned before, market friction instruments work to improve the way a current market functions by reducing asymmetry of information and reducing transaction costs. For example, providing consumers with more information about the products that they buy gives them more choice about which products they might prefer. Through the use of certification and labeling schemes, producers of sustainable products may benefit from differentiating their product from similar products to create a competitive advantage which should allow them to gain higher revenues (if consumers value biodiversity conservation) (Brauer et al, 2006).

THEORETICAL CONSIDERATIONS

According to economic theory, the price elasticity associated with the benefits curve, gives information useful for deciding what type of economic instrument to use. If a small change in cost will result in a large amount of benefit (elasticity is high), then quantity based instruments should be used. If the benefits curve is shallow (inelastic) or unknown, then price based instruments may be more appropriate. (Brauer et al, 2006).

However, these considerations assume to a large extent a case of perfect information and low transaction costs, and for biodiversity this is rarely the case. As a result, according to Brauer et al. (2006), MBIs are usually best when:

- Applied specifically to the particularities of the biodiversity in question, and applying them at the local level
- Used accordingly with command and control regulation already in place

IMPLEMENTING MARKET BASED INSTRUMENTS

A. Taxes, fees and charges

Taxes, fees and charges are one of the most common instruments used to achieve biodiversity preservation, and are a relatively simple concept. They require a payment to the government in response to using relevant areas of the environment (Brauer et al, 2006). Most commonly they are applied in the case of biological diversity to extracting minerals from river beds, forestry activities (see Box 6.1), charges for hunting permits and agricultural use of pesticides and fertilizers (Brauer et al, 2006). Even though there are few assessments of how these instruments work, it is clear that it is important to set taxes at the right level in order to send the right behavioural signals. For example, Anderson et al. (2000) examined the application of fertilizer and pesticide taxes in Europe and found that they were not particularly effective and in fact the predicted increases in the tax triggered panic-buying which instead led to a temporary period of higher pesticide use.

Box 6.1 Taxes, fees and charges

In **Quebec (Canada)** the charge on the cutting of trees provides financial means for the sustainable maintenance of forests. In 1997, CAN$ 150 million was available for that purpose. In the Province of **Alberta** a reduction on the stumpage fee for timber that can only be harvested against high costs has contributed to an economic use of that timber.

In **British Columbia (Canada)** the permit fee system charges permit holders according to the amount of waste permitted. Some permit holders have changed their production processes in order to arrive at a lower permitted waste volume and consequently avoided high fee payments. A charge on leftover paint funds a program that resulted in the diversion from landfill of 6.5 million equivalent litre containers with leftovers.

Source: Economic Instruments for Pollution Control and Natural Resource Management in OECD Countries: A survey (2006)

While these instruments have the potential to create behavioural changes, taxes also aid in conservation policy by generating the necessary revenues needed to implement the policy, and are especially important in countries where public money for biodiversity conservation is quite limited, such as would be evident in developing countries (Brauer et al, 2006). An example would be setting fees for entering national parks, as tourism can be quite damaging to ecosystems.

Overall, taxes can be seen as more efficient than direct regulation because they help to minimize the cost of protecting an ecosystem service. Parties will involve themselves in conservation practices if the cost of doing so is below the tax that would have otherwise occurred had they not adopted conservation strategies. Thus the low opportunity cost of providing an ecosystem service and avoiding the tax, would be adopted instead of higher cost options.

To see that taxes are cost-effective, we can apply a cost minimizing pollution model to biodiversity, assuming that pollution represents any activity that reduces environmental quality, including biodiversity degradation such as deforestation or over-fishing, for example. According to the model laid out by Gayer and Horowitz (2006): let *rk* represent a vector of inputs used by producer *k* = 1,...,*K*; *w*, the price vector of inputs; *sk* , the pollution generated by plant *k*; and *fk* (*rk ,sk*) the plant's output. Let *p* represent the price of this output. Then the cost-effective choice of inputs *r* = *{r1 , . . . , rK}* to generate overall pollution no greater than *S* satisfies:

$$\max_{r} \sum_{k} pf^{k}(r_k, s_k) - wr_k \quad \text{subject to} \sum_{k} s_k \leq S$$

For a well-behaved problem, this gives first order conditions:

$$p\frac{\partial f^k}{\partial r_{ik}} - w = 0,\ p\frac{\partial f^k}{\partial s_k} - \lambda = 0$$

where *λ* is the Lagrange multiplier associated with the aggregate pollution equation. Of particular consideration in this model, is the production decisions of the individual polluters who face pollution tax *τ*. Each firm's objective function is to maximize *pfk* (*rk ,sk*) *wrk* − *τsk* . This gives first order conditions:

$$p\frac{\partial f^k}{\partial r_{ik}} - w = 0,\ p\frac{\partial f^k}{\partial s_k} - \tau = 0$$

and these are equivalent whenever *τ* = *λ*. Alternatively, for any given *τ* , there exists an *S** such that *λ*(*S**) = *τ* . In theory, a pollution tax or a tax on environment degradation should be cost-effective; whatever pollution amount it yields will have been generated at the lowest cost (measured in foregone profits) to the economy (Gayer & Horowitz, 2006).

Despite this however, there remains a huge informational demand for government as the case for these instruments assumes that the government has sufficient information to set the tax at appropriate levels. Ideally a tax should be set so that the cost of supplying the last unit of an ecosystem service equals its benefit to society, and if set at inappropriate levels, the tax may not be cost effective (Aretino et al, 2002). Another issue arises because benefits of an ecosystem service vary across regions – essentially due to the localized nature of ecosystems and species. It would therefore seem appropriate to set different taxes and fees depending on the biological concern at hand, which would result in more transaction costs and greater administrative complexity (Aretino et al, 2002). While this type of instrument is fairly simple to set up, if the demand is inelastic, then the instrument will be ineffective.

B. Environmental subsidies/support

After taxes, the most frequently used MBIs are in the form of subsidies and support due to their simplicity, political acceptability and ability to work with existing property rights (Brauer et al, 2006). They help by influencing changes in consumer behaviour and to help create new markets for environmental goods (see Box 6.2). Because a priced based subsidy is essentially a 'negative tax', it has the potential to lead to the same efficient outcome as an environmental tax such as was discussed in the previous section (Gayer & Horowitz, 2006).

Box 6.2 Subsidies for environmental protection

One such case is the subsidy program for the purification of municipal wastewater in **Quebec (Canada)**. It is expected that the wastewater of 98% of the residents connected to the sewer system will be treated in 1999. The equally successful Clean Water Grant Program in **New Brunswick** has supported the connection of residents in municipalities to sewer systems. Virtually all municipalities have state-of-the-art sewer systems and sewer overflow in fresh water bodies is drastically reduced. The financial aid for reuse and recycling of used car tires in **Quebec** was also reported to be successful, since the objectives set for 1997 were met.

Source: Economic Instruments for Pollution Control and Natural Resource Management in OECD Countries: A survey (2006)

Forestry management for example is dominated by subsidies as far as economic instruments are concerned. The purpose of these subsidies is for extending or maintaining forestry areas and forest quality through environmentally sustainable management and cutting practices. Agriculture as well shows extensive use of subsidies/support such as in Agri-environmental Schemes (AES) to pay farmers to carry out environmentally beneficial actions. These encourage farmers to produce more goods but also more recently to reach environmental goals such as protection or improvement of farmland species or to reduce pollution (Brauer et al, 2006). Studies that have analyzed the success or failures of AES concluded in general that AES are ineffective in protecting farmland biodiversity. However, the problems are more to do with the design of the schemes rather than with the theoretical considerations of how a subsidy should work. Countries that achieved their aims well generally have clear objectives, targeting that is area specific, realistic, quantitative and time delimited (Kleijn; cited in Brauer et al, 2006). Similar to taxes, an AES may need adjustment to local conditions in order to implement a proper subsidy scheme.

Funds and grants are similar to subsidies, though the latter are more likely to be paid by the government. Funds may be used to target a particular species, ecosystem or environmental cause by paying communities within the targeted area in order to help reach preservation goals (see Box 6.3). In general payments, through subsidies, grants and funds are fairly simple MBIs which establish the link between an environmental outcome and an economic incentive. However, it is important that what is done with the subsidy/fund translates into improvements of biodiversity and not conflicts of interest (Brauer et al, 2006). That is, some subsidy schemes, reflecting political economy approaches, may have the potential to make the problem worse.

Box 6.3 The Water Fund

WWF, in collaboration with local partners, is developing a water fund to finance responsible watershed management in Guatemala's Sierra de las Minas Biosphere. Under this initiative a range of water users – including bottling companies, distilleries, hydroelectric plants and paper processing mills – are making significant financial contributions towards environmental services in the region.

According to Carlos Morales, Freshwater Officer for WWF Central America, "This Fund will encourage short-term investments to optimize water use in the industries as a means of reducing effluents to the Motagua and Polochic Rivers, as well as the vulnerability of the soils. Investments will also encourage better management of watersheds and water recharge zones in the upper reaches of the watershed to ensure a permanent water supply". In the future, WWF intends to work with agro-industry and household users of freshwater.

Source: Bishop et al. 2006

Unfortunately there is no long term guarantee for biodiversity gains and attitudes may not change (e.g. when payments stop participants may return to their destructive behaviour). As well, continuous funding and monitoring raises concerns as in these situations 100% must be paid, while other MBIs use the power of the market to multiply their investments. Additionally, instruments must be adequately targeted, and though the idea may seem simple, distribution of subsidies/support may be relatively complex and bureaucratic (Brauer et al, 2006). However subsidies and support work well to encourage the participation of private participants such as farmers and landowners in the protection of biodiversity, as taxes and other instruments may not be able to achieve such cooperation.

C. Tradable Permits

Tradable permits, quantity based MBIs, address the public good nature of biodiversity by limiting access to the resource and privatizing the resulting access rights (Tietenberg, 2002). This involves activities that provide a market in which rights to pollute, develop or use environmental resources are traded. Their most common use internationally seems to be for tradable fishing quotas, which involves total allowable catch (Brauer et al, 2006). Tradable permits are designed to achieve a fixed amount of environmental protection and because they are traded through the market, conservation goals will be achieved in the most cost efficient way.

The first step in tradable permits involves setting the limit on user access to the resource, which defines the aggregate amount of access to the resource that is authorized. This is important because unlike some other instruments, a specific amount of conservation can be reached. The access rights are then allocated on some basis to potential individual users, and depending on the system, these rights may be transferable to other users and/or bankable for future use. Users who exceed limits imposed by the rights they hold face penalties including the loss of the right to participate (Tietenberg, 2002).

There are two broad types of tradable permit systems in operation: those based on emission reduction credits (ERCs), and those based on *ex ante* allocations (cap-and-trade) (OECD, 1999). The emission reduction approach takes a baseline scenario and compares this with the actual performance of participants. If a user performs better than the baseline, they earn a 'credit' which can then either be used by the user himself, either at the current location or elsewhere, or sold to some other participant whose use is higher than the accepted baseline (OECD, 1999). The 'cap-and-trade' approach sets an overall use limit or the 'cap' and requires all participants to acquire a share in this total before they can emit, either free-of-charge by a relevant environmental authority, or the shares may be auctioned. As mentioned above, regardless of how the initial allocation of shares is determined, the owners can either use permits in current production, save them for future use, or trade them with other users which creates a market in which biodiversity use can be traded and valued.

A good example of a system for tradable fishing quotas is the system developed in New Zealand. Starting off as a fairly small-scale scheme it has managed to expand to include 93 different species. This inclusion of a large number of species is thought to be successful in reducing by-catch as fishermen have to take out quotas for each species that they choose to fish (Brauer et al, 2006). Adding to its success are well defined rules to promote stability, stakeholder trust, as well as a high level of NGO and political support, and proper monitoring through satellite systems. However, a system such as this may become problematic when the agendas of different groups begin to conflict, so tradable permits may therefore be more applicable where the interests of different groups are more similar. Tradable permits have also been suggested for sport hunting and fishing – an application that may be very appropriate for Canada, as well as permits aimed at wetland preservation in the United States (Brauer et al, 2006). These approaches have been controversial however, especially concerning the allocation of the wealth associated with the resources. Because the access rights can be very valuable when the resource is managed efficiently, the owners of these rights may benefit substantially which raises controversy about the resulting distribution of benefits among competing users (Tietenberg, 2002). As well depending on their design and the environmental resource in question, they may have high transaction costs, inactive markets and high administrative and compliance costs in registering users and keeping track of trades (Brauer et al, 2006).

D. Liability and compensation schemes

Liability and compensation schemes are quantity based MBIs which open up the possibility for a number of markets. While they are not strictly market based instruments, liability regimes result in the internalization of externalities associated with biodiversity use and degradation by holding parties liable for environmental damage that they cause (see Box 6.4). Essentially under this type of scheme, companies have to pay when they cause environmental damage, so have an incentive to reduce risk, so firms should take care to not cause damage to the environment. Another advantage of liability and compensation schemes is that in the event that the environment has already been damaged, the environment can still be compensated through restoration projects at the expense of the firm (Brauer et al, 2006). Liability is also economically equivalent to a tax, so a liability scheme potentially faces the same cost minimization problem as discussed in part A, but its ability to achieve cost minimization or efficiency depends on the specific externality at stake and the effectiveness of the court system (Gayer & Horowitz, 2006). However, liability schemes may result in high transaction costs when taking companies to court is costly and risky. However, using the legal system to control external costs may not be recognized by all analysts as a market-based instrument.

Box 6.4 Liability Directive (2004/35/EC)

The liability directive is an ex post scheme aimed at holding those causing environmental damage responsible for their actions, and includes land and water damage, protected species and habitats.

It requires that an operator should take preventative action when environmental damage is likely to occur, and when damage does occur, the operator must take remedial action. In both cases the operator must bear all costs.

Source: Brauer et al. 2006

POLICY RECOMMENDATIONS

To implement market based instruments successfully it is important to consider impeding features related to their use and development (See Table 6.2). In particular, MBIs need to be designed specifically for the characteristics and needs of each individual community and ecosystem.

TABLE 6.2 Are MBIs an Appropriate Policy Response?

Community values and potential gains from trade	A market mechanism can only function where there are potential "gains from trade" –where participants to the trade all end up "better off" after the trade. In simple terms, if there is no potential value to the community, then no trades will take place and an MBI will not work. Gains from trade are primarily realised because of heterogeneity. Heterogeneity exists within the landscape (environmental goods may be located across the landscape or located in particular areas 'hotspots'), between different management actions (different landowners can undertake different actions to address a NRM issue, for example planting trees or lucerne to manage salinity), and where social and economic variation exists between landholders (landholders experience different cost structures and have different preferences). Heterogeneity will lead to trade.
Transaction costs and potential gains from trade	Trades will only occur where the value of the relevant environmental good outweighs the sum of production and transaction costs[1] incurred in the market process. Thus MBIs are only a practical option where the good generates sufficient value to encourage trade and where transaction costs can be sufficiently minimised to facilitate market exchange.
Existing policies and schemes	No policy instrument or reform is truly 'new' since it must be superimposed over existing rules, regulations and customs. Thus in crafting policy instruments, it is helpful to think of them as complementing the *status quo*. One must consider not only the proposed policy instrument but the current institutions and operating frameworks and whether they need to or can change.
Community capacity	Policy is not only generated within existing rules, regulations and customs but also within constraints and opportunities provided through existing political structures, biophysical constraints and physical, human, financial and social capitals. One must consider these contextual attributes in assessing whether the policy opportunity and the policy instrument can be adapted to achieve the desired outcome.

Source: Coggan et al, 2005

Incentives must be clearly defined in order to achieve their desired target. Participants must be clear on the aims of the incentive, and must coordinate to achieve the same goal. This includes defining what the exact aim is to be, to avoid any confusion or problems in the future. Policies should also take into consideration the environmental, social and economic situation, as it is possible for an instrument to be cost effective and accepted by the local population, but does not actually deliver the desired environmental goal (Brauer et al, 2006). As well, early and continuous involvement of the community and stakeholders is important to help clarify the best possible approach of an environmental goal. The time over which an instrument is carried out is important for reaching biodiversity goals, and for biodiversity it is generally seen as desirable to increase the flexibility of incentives – a major advantage over CAC methods, but instruments should be implemented with particular attention paid to the way in which different types of incentives combine (Brauer et al, 2006).

It is essential that funding is set up for the managing and monitoring of instruments and for the introduction of pilot studies. This will allow for a more credible incentive, and for evaluation to be thorough and effective. In order for instrument effectiveness to be assessed, adequate information collection is needed from the beginning about biodiversity and the potential influence the instrument may have, including its environmental impact and cost-effectiveness (Brauer et al, 2006).

MARKET-BASED INSTRUMENTS IN DEVELOPING COUNTRIES

Market-based instruments for biodiversity conservation have the potential to contribute to the reduction of biodiversity loss, but face barriers through poverty and property right issues in developing countries. Because most of the demand for ecosystem services comes from the urban population, while the supply of ecosystem goods and services comes mainly from rural areas, at aggregate levels, biodiversity markets are likely to involve transfers from richer to poorer (Bishop et al, 2006). This case is even stronger where biodiversity business is based on exporting goods and services produced in developing countries to consumers in rich countries, with rich countries consuming biodiversity and poor countries facing biodiversity loss. Therefore it is important in policy decisions when implementing market based instruments that consideration is taken of all distributional impacts

MARKET-BASED INSTRUMENTS VS COMMAND AND CONTROL

MBIs minimize the aggregate cost of achieving a given level of environmental protection and provide incentives for the adoption of cheaper and better control technologies. But despite this, MBIs are used less frequently than command and controls methods (Keohane, Revesz & Stavins, 2005). When CAC methods are used, the required level of environmental damage abatement has been stricter on new damage sources than for existing ones – possibly worsening the situation by encouraging older, dirtier producers to operate. However, when MBIs are adopted, they usually take the form of tradable permits instead of other instruments, when economic theory tells us that the optimal choice is dependent on the particular situation at hand. The restricted use of MBIs for biodiversity conservation might be due to the fact that biodiversity is a much more complex and multi-faceted issue because of the many perspectives on how to tackle it. Not only that, but the idea of giving living things a market price raises ethical concerns among conservationists who may be reluctant to support the use of MBIs. What conservationist may not realize is that the majority of the population makes decisions based on costs and benefits. Biodiversity valued at a zero price implies its loss comes at no cost, and its benefits will not be fully realized by most of the population. So biodiversity will continue to be exploited if value cannot be attached to its use.

Command and control policies are likely more common because they are relatively simple to achieve, and typically require firms to limit their activity in a certain sensitive area, adopt performance standards or use particular technologies. Market based instruments in contrast seek to make biodiversity conservation more desirable and profitable (Bishop et al, 2006). MBIs however, according to Brauer et al. 2006, have the following advantages over CAC:

- They allow a flexible response to price signals and encourage innovation
- They are cost effective and encourage improvements to be achieved in a cost minimizing way
- They should avoid some negative incentives such as protected species being a liability, which may be present with regulatory approaches

Because they incorporate marginal costs and use market forces, MBIs can be more cost effective than traditional CAC, which means that either more ambitious conservation goals can be reached under a particular budget constraint or that considerable cost savings can be achieved (Brauer et al, 2006).

The growing interest in the use of MBIs can be explained by the increased familiarity with MBIs as a possible solution to environmental degradation, increased environmental control costs which create a greater demand for cost effective instruments, and a general shift in policy towards using the market to solve social problems (Keohane, Revesz & Stavins, 2005). However, many examples of MBIs show that they may work best as complements rather than substitutes to current CAC approaches, so it is worth considering pairing some combination of MBIs and regulatory approaches to achieve environmental targets (Brauer et al, 2006). Current preferences for CAC approaches reflect a desire to maintain the status quo of environmental. So in this case it would seem that the demand for market based approaches will likely be the greatest when the environmental issue has not yet been regulated (Keohane, Revesz & Stavins, 2005).

In recent years MBIs have been applied to biodiversity conservation, however have been very limited in their application and only applied to specific areas or local needs. MBIs are limited in their application to biodiversity because the various components of biodiversity are very heterogeneous in their value, not to mention the lack of knowledge and experience with successful examples (Brauer et al, 2006). The biggest problem stems from the fact that biodiversity conservation still requires a great deal of research and information for a more complete understanding of the interaction between human behaviour and biodiversity.

CONCLUSIONS

The potential benefits of using MBIs to assist in environmental protection and natural resource management are large given the proper design and implementation of these tools. If new markets can be created or current markets adjusted, individual firms will be able to make decisions with a broader range of information about the costs and benefits of their investments into biodiversity protection. Not only would firms be aware of cost minimizing options for preservation, but also would be given incentives to innovate and discover even cheaper and more efficient ways of achieving environmental outcomes. In other words, biodiversity would become a part of the economic system, and incentives would be provided for investments in biodiversity based goods and services in the same way as they are in the rest of the economy.

Despite the potential for good MBI design to provide a market for biodiversity, there are still many issues to be solved. Particularly, good MBI design involves setting appropriate property right structures and incorporating flexibility in order to maximize the gains from trade (Whitten et al, 2003). Finally, good MBI design accounts for the risk of an asymmetry in information and consideration of the number of views and strategies for addressing biodiversity loss.

MBIs have several advantages over CAC instruments used in conservation. In particular when designed properly, they can achieve results beyond those of regulations, or at least achieve the same results at a lower cost. Generally however, it is not clear when to recommend the use of MBIs instead of CAC methods because biodiversity is such a heterogeneous good, so instruments and policy decisions must be tailored to specific circumstances. That said, many examples of MBIs show that they work best not as a substitute to regulatory approaches but as a complement to them, so using some combination of MBIs and CAC methods to achieve desired conservation goals should be considered. Even so, there are a number of examples of MBIs that work well in achieving desired biodiversity conservation, as MBIs can offer policy makers new options to achieve conservation in an effective and efficient way (Brauer et al, 2006). Therefore there is a significant potential for a wider application of MBIs to reach policy objectives – especially in combination with traditional regulations already in place.

While market based instruments deliver biodiversity values to society at the lowest cost, what they do not ensure is that the level of biodiversity protection delivered to society is at the most efficient level – either too little or too great. Therefore, MBIs must be reinforced by the input of valuation information. The investment in valuation techniques and actual estimates will assure society that the additional benefits of biodiversity protection are worth the additional costs. Therefore, not only is it important for good MBI design, but also for the application of economic valuation to encompass the total value of biodiversity goods and services traded in the market. Thus an evaluation of market-based instruments and their ability to achieve conservation goals is needed.

VII CASE STUDY: ECO-LABELING AND CERTIFICATION

As noted earlier, the externalities involved in the use and loss of biodiversity have resulted in an absence of a market for biodiversity goods and services along with their under-valuation. The following two chapters are devoted to examining the role of economic incentives in internalizing the economic value of biodiversity into resource allocation choices. This will be shown specifically through the emerging markets for eco-labeling (certification) and bioprospecting and whether they have the ability to achieve substantial environmental benefits that outweigh the costs of conservation to society. The discussion begins by graphically analyzing the willingness to pay for eco-labeled products.

INTRODUCTION

In the context of biodiversity, certification and eco-labeling inform the consumer about biodiversity product characteristics that are not directly related to its consumption or use, but do provide environmental benefits, including the conservation of biodiversity. They are included in the 'market-friction' function of MBIs and help to remove obstacles for market growth and creation for biodiversity. Eco-labeling is characterized by the evaluation of a product, or its characteristics, with respect to particular biodiversity specifications (Nunes & Riyanto, 2005). Information specific to the product's characteristics such as ecological, social and economic specifications, along with the product, are provided to the consumer. Certification on the other hand, is an evaluation of the product's underlying management system and how they compare against certain management criteria. This includes the identification and monitoring of the supply chain including the transport and processing of raw materials, secondary manufacturing and retail distribution of biodiversity; information that is then provided to the consumer (Nunes & Riyanto, 2005).

Forest certification for example is a well established method of labeling wood products so that in theory, the consumer can choose to purchase those which come from sustainably managed forests (Brauer et al, 2006). In the US there exist many certification schemes including Tree Farm (American Forest Foundation), Forestry Stewardship Council (FSC), Sustainable Forestry Initiative (SFI) and the National Forestry Association Green Tag (Fletcher; cited in Brauer et al, 2006). However, with such a multitude of labels, consumers may not know how to distinguish between legitimate schemes and may decide it is not worth paying attention to any of them. As well, forest owners may be discouraged themselves (especially small scale operations) by the cost of certification (Brauer et al, 2006).

Even so, the overall objective of eco-labeling and certification schemes is to link the supply and demand respectively, of producers of these products with consumers who value biodiversity (Nunes & Riyanto, 2005). Consumer preference to purchase these products is revealed through an increased willingness to pay for eco-friendly products, when consumers are aware of the non-market biodiversity values. This helps to create a separate market for these products and establishes an advantage for producers who choose to adopt biodiversity preserving schemes because it may improve their public image as well as differentiating their products. The resulting market price will therefore incorporate biodiversity benefits thus creating a measurement closer to what would be measured through total economic value (see Section IV), which will lead to a better allocation of biodiversity. The participation of consumers in markets for these differentiated products, will internalize the market values for biodiversity. If consumers internalize the non-market benefits of biodiversity, they must be willing to pay for these benefits.

The success of certification and eco-labeling as economic instruments to combat biodiversity loss is dependent on the ability of the instrument to internalize externalities caused by the lack of market prices for biodiversity. Because of the public good characteristics of most biodiversity goods and services, these types of tools first require the application of appropriate economic valuation methods to assess the potential magnitude of non-market biodiversity benefits. As well, the success or failure of certification and eco-labeling depends on the characteristics of the market's supply and demand; for example the level of consumer awareness of 'green' products and the ability of producers to successfully implement certification and eco-labeling schemes (Nunes &

Riyanto, 2005). In the next section the potential for certification schemes to achieve biodiversity conservation is examined.

THE MARKET IMPLICATIONS OF CERTIFICATION

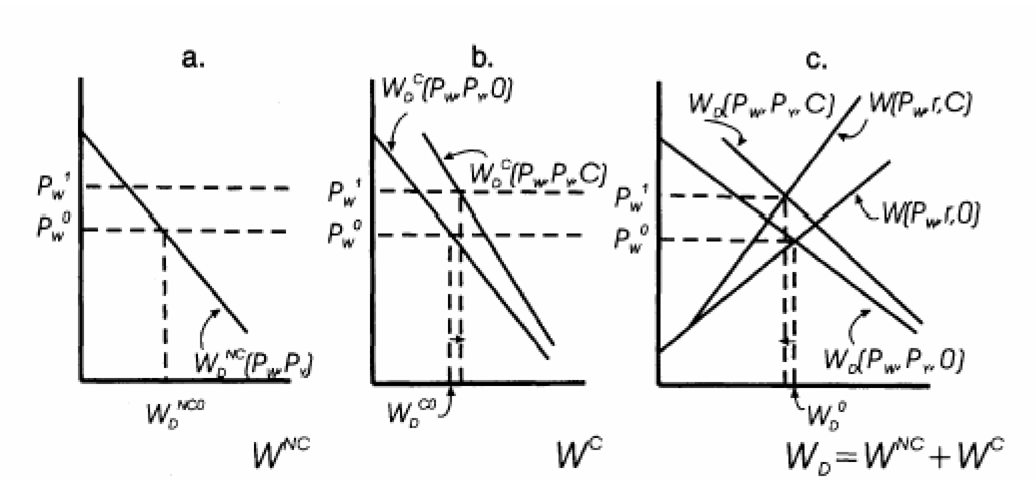
The following model developed by Sedjo and Swallow (2000) involves analyzing the market implications of certification for wood products, by incorporating analytical content through graphical tools within a general equilibrium analysis. However, the analysis can be expanded to include other products that are 'green' meaning that the raw material comes from a source that has been managed according to particular environmental or ecological standards. The model focuses on eco-labeling goals that aim to promote eco-system health or biodiversity protection from within forests producing wood products. While certification may increase the weight that firms place on environmental conservation in production, this analysis examines the possibility of market feedbacks with additional positive or negative consequences for ecosystem health.

CERTIFICATION WITHIN A SIMPLIFIED CLOSED ECONOMY

The main concept developed by Sedjo and Swallow (2000) uses a competitive economy with two goods, wood products (*W*) and all other goods (*Y*) which are produced through the use of a single input, land (*L*). In the economy producing *W* and *Y*, the market demands depend on the prices of both goods P_w and P_y, and also depend on the consumers' response to the certification scheme. Consumers who prefer certified products will therefore be willing to pay a premium for certified wood, so labeling will shift their demand for wood products to the right reflecting their willingness to pay (WTP) for wood plus biodiversity. However, the market still includes the demand of consumers not concerned with certification, who are not affected by labeling accept through the indirect effects on price. As well it is assumed that consumers face a diminishing marginal willingness to pay for certified wood, contrary to what their WTP would be if the wood was not certified.

Therefore the aggregate demand of eco consumers and indifferent consumers can be segregated into $W^C{}_D(P_w, P_y, C)$ and $W^{NC}{}_D(P_w, P_y)$ respectively. Within this framework, consumers' demand for wood products depends on the price of wood, P_w, and the price of all other substitutes, P_y, but only eco consumers' demand is dependent on the certification program, C. The pre-certification demand from eco consumers is therefore $W^C{}_D(P_w, P_{y,} 0)$ where $C = 0$, or no certification.

FIGURE 7.1 Certification with Eco-consumers



d. Demand for non-certified wood, W^{NC}
e. Demand for certified wood, W^C
f. Aggregate supply and Demand, W and W^D, respectively

Source: Sedjo and Swallow (2000), pg. 30

Figure 7.1 illustrates the two markets for non-certified and certified wood, and the aggregate demand and supply for wood. Prior to certification, aggregate demand and supply are given by:

Demand: $W_D(P_w, P^0{}_{y,} 0) = W^{NC}{}_D(P_w, P^0{}_y) + W^C{}_D(P_w, P_{y,} 0)$
Supply: $W(P_w, r, 0)$ *where r is the rental price of land*

This leads to an initial equilibrium involving demand in the two market segments $W^{NC}0_D$, and W^C0_D with aggregate quantity demanded of $W^0{}_D = W^{NC}0_D + W^C0_D$ at price $P^0{}_{w.}$ With certification, the costs of supply increase along with the demand for certified wood by eco-consumers, but in partial equilibrium the demand for certified wood by non-eco-consumers does not change. Thus continuing with the Sedjo and Swallow model, aggregate demand and supply become:

$$Demand: \quad W_D(P_w, P_y, \mathrm{C}) = W^{NC}{}_D(P_w, P_y) + W^C{}_D(P_w, P_y, \mathrm{C})$$
$$Supply: \quad W(P_w, r, \mathrm{C})$$

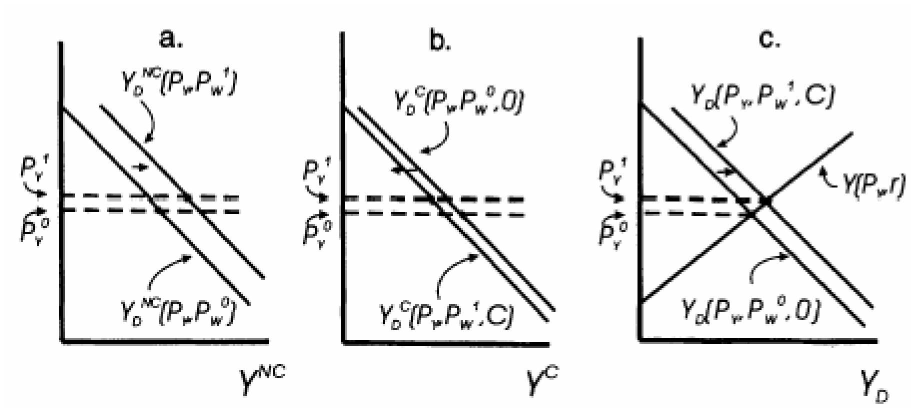
where P_y would take the initial value $P^0{}_y$ in a partial equilibrium setting and P_y would take its general equilibrium value after prices adjust.

Referring to Figure 7.1, the partial equilibrium effects cause a decrease in the quantity of wood products consumed from non-eco-consumers, while the consumption from eco-consumers increases. It must be noted however that the consumption of eco-consumers may either increase or decrease. Consumption will increase if the consumers' willingness to pay for certified wood is able to offset the decrease in consumption from the rise in the supplier price. Thus the aggregate quantity consumed will only increase if eco-consumers increase consumption, and only if this consumption is enough to offset the reduction in consumption from non-eco-consumers. Figure 7.1 illustrates a decrease in aggregate consumption even though eco-consumers' increased their consumption.

While the decrease in aggregate consumption may appear to reflect a decrease in the use of existing forest land, in fact it should be cautioned that a decline in aggregate consumption may be harmful to the environment. With less demand for forest products, the land may be reallocated for different production purposes such as agriculture, which would in fact generate negative impacts on the ecosystem. To assess this effect Sedjo and Swallow (2000) consider the demand for all other goods; for eco-consumers, $Y^C{}_D(P_y, P_w, C)$; for non-eco-consumers, $Y^{NC}{}_D(P_y, P_w)$ and aggregate demand, $Y_D(P_y, P_w, C)$.

Figure 7.2 illustrates the initial demand for other goods when $C = 0$ and the initial price of wood at $P^0{}_w$. Because mandatory certification causes a rise in the price of wood, non-eco-consumers will decrease consumption of wood and substitute towards the consumption of other goods (7.2a). Eco-consumers on the other hand, will allocate their expenditures towards wood and away from other goods because certification gives them higher utility. But because the price of wood is increasing, like non-eco-consumers, they will have an additional incentive to substitute back towards other goods. If the net effect increases the demand for *Y*, then either both types of consumers substitute towards the consumption of other goods, or the increase in consumption of wood products by eco-consumers is not sufficient to offset the decrease in wood consumption by non-eco-consumers. If the demand for *Y* increases, this gives the outward shift in demand for other goods (7.2c), creating upward pressure on the price of *Y*, P_y.

FIGURE 7.2 Certification Effects on Demand for All Other Goods



a. Non-eco-consumers, Y^{NC}
b. Eco-consumers, Y^C
c. Aggregate supply and Demand, *Y* and Y^D, respectively

Source: Sedjo and Swallow (2000), pg. 32

So the demand for land as an input to production is altered in both or either sectors, *W* and *Y*. Because certification increases the costs to producers, there is a decreasing marginal return to owning land for the purposes of wood production. However this decrease in demand is partially offset by the increase in the price for certified wood products. In the market for other goods, the increase in P_y will potentially create upward pressure on the derived demand for land in production of other goods. So depending on the exact situation, it may be hard to determine the effect of land allocation between forestry and other goods, but the implications of land conversion to other products can discourage the incentives for certification. Therefore implementing certification schemes requires considering the possibility of possible side effects from market feedback.

IMPLICATIONS

Under voluntary labeling, the average price for certified wood products cannot decrease (in partial equilibrium), and will increase if costs to certify are high, or if eco-consumers generate a large enough change in demand. If the price change is significant, consumers' net change in demand will result in an increase in demand for wood products *or* other goods. So in general equilibrium price changes may result in the reallocation of land from wood production to other good production with associated ecosystem impacts (Sedjo & Swallow, 1999). This example gives caution to the use of eco-labeling to produce large-scale environmental improvement. The market feedback illustrated in Figures 7.1 and 7.2 can cause the economy to reallocate land use to production of other goods, production that may harm the environment. In order to guarantee results from certification schemes, the improved quality of land experienced from certification must exceed the potential costs to environmental quality due to land reallocation (Sedjo & Swallow, 2000). However, an advantage of voluntary labeling is that producers, who find it too costly to convert to certified wood production, may continue to use the land for forestry, instead of converting it to other possible uses in the economy (Sedjo & Swallow, 1999). As a result, voluntary certification may have the advantage of addressing the market feedback assumption of mandatory certification. Even so, there is no guarantee that market feedback will occur, but that eco-labeling and certification schemes may need to prepare for additional market effects. Certification, even with the demand from eco-consumers, faces uncertainty that can only be addressed through continued research into its theory and application.

CONCLUSIONS

The above discussion implies that the success of certification policy is highly dependent on consumer demand. If consumer demand is sufficiently elastic, then an increase in the price of a labeled product will result in some decrease in consumption. The elasticity is a reflection of their willingness to pay for environmental protection and depends on the consumers' preference towards eco-friendly goods, as well as their awareness of the non-market values of biodiversity (Nunes & Riyanto, 2005).

Nevertheless, the product differentiation of certification and eco-labeling, can under certain conditions, lead to an increase in demand for these types of goods. Therefore it is likely that an efficient labeling strategy will involve additional government intervention to reduce asymmetry of information between consumers and biodiversity loss. Interventions include information and certification campaigns aimed at rising consumer awareness of the benefits of biodiversity conservation and can also extend to subsidies that encourage producers to certify and label their products. Additionally, an advantage to eco-labeling and certification schemes is that they can be used together with other methods to achieve policy goals (Nunes & Riyanto, 2005). So the important implications for eco-labeling and certification is to continue research into their development and application by economists and policy makers, as they show great potential for internalizing the externalities associated with biodiversity, and creating a greater willingness to pay for conservation by society.

VIII CASE STUDY: BIOPROSPECTING

INTRODUCTION

Bioprospecting refers to the potential of naturally occurring substances to contain pharmacologically active and valuable chemical compounds (biochemical leads) as well as helping to protect the environment (Heal, 2005). Genetic codes produced by natural organisms can be used for commercial purposes by acquiring the breeding stock of the organism that produces the desired compound, transplanting genes, or using the compound as a model for synthesizing new chemicals (Simpson et al, 1996). Bioprospecting can be of considerable value by adapting the new chemical knowledge to industrial, agricultural and pharmaceutical applications. Conservation incentives will emerge as landowners begin to be compensated for the values of the products generated from the species used in new product research. As a result, they will value the biodiversity of their land more (Simpson et al, 1996).

Biologists estimate that there are between 10 million and 100 million living species, and of these only 1.4 million have been researched, and an even smaller amount have been subjected to genetic or chemical testing (Simpson et al, 1996). Therefore much biodiversity research emphasizes the importance of saving unknown species as potential insurance against unknown, unidentified diseases. Over one-third by value of prescription drugs originated as naturally occurring compounds, so bioprospecting has been an important source of biochemical leads for drug development (Heal, 2005). For example, several drug companies have made agreements with tropical countries under which they will investigate the potential of pharmaceutical leads from compounds found in plants, insects, and sometimes from vertebrates such as snakes (Heal, 2000). Many millions of hectares of tropical rainforests are lost each year – losses that could be halted by policy interventions which properly value the biodiversity destruction. It is possible that the annual loss of biodiversity due to deforestation is costing billions of dollars annually from potential new drug development (Heal, 2005)

Bioprospecting has been deemed a potential source of finance for biodiversity conservation. However, will the revenue generated by bioprospecting be large enough to offset the opportunity costs faced by participants in order to preserve genetic resources? The emerging market for bioprospecting gives an opportunity to apply economic valuation to decide whether the bioprospecting value of certain genetic resources as a source of demand, could be large enough to support market-based conservation of biodiversity.

CONFLICTING STUDIES

Recent studies try to estimate the economic value of genetic material as a value derived from the *marginal species*, in contrast to previous estimates that considered estimates of average values or which attributed all drug value to the genetic resource (Pearce, 2004). The marginal species in terms of pharmaceuticals refers to the contribution that one more species makes to the development of new products, which concerns the costs and benefits of the change in the total stock, not the value of the total stock overall (Pearce, 2004).

However, discrepancies in the final estimates of value can result in controversy for biodiversity preservation through bioprospecting. If valuation information changes conservation incentives, then re-allocating scientific resources to provide such information may be the most efficient way to achieve private conservation (Costello & Ward, 2006). But, problems arise when valuation information varies. Simpson et al. (1996), suggest that the bioprospecting value of the “marginal unit” of genetic resources is likely to be quite small, less than \$21/ha, creating essentially no conservation incentive. Rausser and Small (2000) on the other hand claim that bioprospecting incentives are actually quite large, perhaps up to \$9177/ha, which seems large enough to motivate conservation. So the model that values bioprospecting more accurately can have a huge effect on conservation strategies.

THE MODEL

The fundamental equation by Simpson et al. (1996) is given by:

$$WTP = \frac{\lambda.(R-c)e^{\frac{-R}{R-K}}}{r(n+1)} \qquad (1)$$

where WTP is the maximum willingness to pay; λ is the number of independent tests per year, (10.52); n is the number of species that could be sampled (250,000); c is the search cost of determining whether a species will yield a successful product ($3,600); r is the discount rate (10% = 0.1); e is the natural logarithm (2.718); K is the expected research and development cost per new product successfully produced ($300 million); and R are the revenues from new product net of costs of sales but gross of research and development costs ($450 million).

Substituting the above values into equation (1) gives a maximum willingness to pay of $9410 for the marginal species. However, the literature explains that the WTP for the marginal species is not an easy concept to identify, and continues to transform these values into willingness to pay for land that is subject to the risk of conversion. First the 'species-area' relation estimates the number of species likely to be present on a given area of land (Pearce, 2004), and is given by:

$$n = \alpha A^Z \qquad (2)$$

where n is the number of species; A is area; α is a constant reflecting the species richness potential of the area; and Z is a constant equal to 0.25. Second, the economic value V of land area A is given by:

$$V[n(A)] \qquad (3)$$

Expression (3) refers to the value (V) of a collection of species (n) likely to be found in area A. Third, the value of a change in land area A is given by differentiating (3) with respect to A:

$$\frac{\partial V}{\partial A} = \frac{\partial V.\partial n}{\partial n.\partial A} \qquad (4)$$

The expression $\partial V/\partial n$ is the marginal value of the species, which is what we calculated to begin with or for the value of $9140 from above. The expression $\partial n/\partial A$ is the change in the number of species brought about by a small change in the land area. Differentiating (2) gives:

$$\frac{\partial n}{\partial A} = Z\alpha A^{Z-1} = \frac{Z.n}{A} = Z.D \qquad (5)$$

where $D = n/A$ is the density of species. Therefore, the bioprospecting value of marginal land (see Table 8.1) is given by Simpson et al. (1996) as:

$$\frac{\partial V}{\partial A} = \frac{\partial V}{\partial n}.Z.\frac{n}{A} \qquad (6)$$

TABLE 8.1 Estimates of Pharmaceutical 'hot spot' Areas

Area	Simpson et al. (1996) WTP of pharmaceutical companies per ha	Simpson and Craft (1996) 'Social value' of genetic material per ha	Rausser and Small (2000) WTP of pharmaceutical companies per ha
Western Ecuador	20.6	2,888	9,177
Southwestern Sri Lanka	16.8	2,357	7,463
New Caledonia	12.4	1,739	5,473
Madagascar	6.9	961	2,961
Western Ghats of India	4.8	668	2,026
Philippines	4.7	652	1,973
Atlantic Coast Brazil	4.4	619	1,867
Uplands of western Amazonia	2.6	363	1,043
Tanzania	2.1	290	811
Cape Floristic Province, S. Africa	1.7	233	632
Peninsular Malaysia	1.5	206	539
Southwestern Australia	1.2	171	435
Ivory Coast	1.1	160	394
Northern Borneo	1.0	138	332
Eastern Himalayas	1.0	137	332
Colombian Choco	0.8	106	231
Central Chile	0.7	104	231
California Floristic Province	0.2	29	0

(maximum WTP $ per hectare)
Sources: Simpson et al. (1996); Simpson and Craft (1996); Rausser and Small (2000)
Notes:
ha: hectare; WTP: willingness to pay

Source: Pearce (2005)

The resulting values derived by Simpson et al. (1996) along with an additional two studies are given in Table 8.1. The results show that valuation of bioprospecting using the criteria described above, gives very low values, which implies bioprospecting incentives are likely small. Reasons for this may be due to the fact that biodiversity is abundant and therefore it is uncertain, prior to testing whether a given species will have economic value. As well, there is a potential for redundancy in the leads, in that once a compound is found, finding that same compound in another species is essentially useless and redundant (Rausser & Small, 2000). Either way, low economic values for bioprospecting is the result.

The third column of Table 8.1 gives alternate estimates proposed by Simpson and Craft (1996). These new estimates relate to ‘social surplus’ by differentiating between species such that one is not a perfect substitute for the other – which is contrary to Simpson et al. (1996), whose estimates assumed either perfect substitutability or no relationship between species. By assuming a 25% loss in the number of species, according to Simpson and Craft (1996), the result is a social loss of around $111 billion in net present value terms. Comparing this to earlier work, economic values of around zero to $20/ha, are very unlikely to affect land conversion decisions, but the larger ‘social’ values could be sufficient to motivate conservation in some areas (Simpson & Craft, 1996). But still it seems as though the ‘social value’ of genetic material per hectare is quite modest, which reinforces the original model, and these social values may or may not be significant enough to provide conservation incentives.

However, these results have been challenged by Rausser and Small (2000) who have a more optimistic view of bioprospecting as a market-based conservation option. Their estimates are shown in the fourth column of Table 8.1. Rausser and Small argue that the differences in estimates are due to the fact that Simpson studies characterize pharmaceutical companies’ search programs as a random, ‘brute-force search’ (pg. 175), and unaided by an organizing scientific framework. In other words searching is based on the sequential testing of large numbers of leads, in no particular order, and does not in fact describe a cost-minimizing approach to selection. Rather, because researchers have access to a rich base of publicly available data describing the location and properties of plants, animals, microbes and their evolutionary history and survival, they provide ‘clues’ about their potential economic value (Rausser & Small, 2000). This suggests that attempts to value genetic resources should focus on how biodiversity research is formed. Therefore in a sense, scientific understanding creates information rents, because if ‘leads’ have potential to aid in research discovery, interested parties will show a willingness to pay for the natural resource. Rents build up for the owners of leads as they absorb part of the information generated by publicly available science (Rausser & Small, 2000). In effect, species will no longer be substitutes for each other, with some having a much higher demand because of their information value. The authors argue that the information value attached to a lead comes from search costs involved in bioprospecting, and the probability of a success, and bioprospecting information rents could in some cases, be large enough to finance meaningful biodiversity conservation. The effect of having different probabilities of success means that an equation like (1) will no longer apply to this type of valuation (Pearce, 2005). But Rausser and Small (2000) conclude that there can be a huge motivation for conservation, particularly with values associated with the highest quality sites – around $9000 in the above table. The basic difference between the Simpson models and the Rausser-Small model is that the former is based on a random search while the latter focuses on a more informed search whose information rents are reliant on the probability of success and the avoidance of search costs.

Perhaps however, these results are too optimistic. First, the Simpson et al (1996) estimates were set up to be upper bound estimates, therefore actual marginal values would be considerably smaller. Second, in reality there is much uncertainty to what kind and how many species are living on Earth – most which have not been identified or researched, which means it is more likely that the assumption of random sampling has more external validity (Pearce, 2005). Likewise, the Rausser-Small model considers searches based on species information already known, so it therefore implies nothing about those species for which nothing is known. Costello and Ward (2003) test for differences in value from informed search as compared to random search by conducting a numerical example using the Rausser-Small model. The example compared expected marginal values under efficient search (in which leads are ordered in descending probability of success), random search (in which the search order is a random permutation of research leads), and maximally inefficient search (in which leads are ordered in ascending probability of success).

Table 8.2 reports the expected marginal values from optimal, random, and backwards searches in each of the 18 biodiversity hotspots considered in Simpson et al. (1996) and Rausser and Small (2000). The optimal search column reproduces the results reported in Rausser and Small. The backwards search column simply reverses the order in which research leads are searched. The random search column represents the expected marginal value over all permutations of the search order. The table however reveals that the search order does not seem to have much of an impact on marginal values. For example, searching in random order reduces the marginal value of a hectare in Western Ecuador only 3.7%, from $9177 to $8836 (not $21 as suggested in Simpson et al.), which reinforces the Rausser and Small conclusions that these estimates are probably sufficient to justify private-sector conservation (Costello & Ward, 2003).

TABLE 8.2 Marginal value of land in biodiversity hotspots under different assumptions about search order

Biodiversity hotspot	Incremental value ($/ha)		
	Optimal	Random	Backwards
Western Ecuador	9177	8836	8455
Southwest Sri Lanka	7463	7190	6882
New Caledonia	5473	5277	5056
Madagascar	2961	2863	2751
W. Ghats of India	2026	1963	1890
Phillipines	1973	1912	1841
Atlantic Coast Brazil	1867	1809	1744
W. Amazonia	1043	1012	977
Tanzania	811	787	760
C. Floristic S. Africa	632	614	593
Peninsular Malaysia	539	522	503
SW Australia	435	420	402
Ivory Coast	394	379	362
North Borneo	332	317	301
Eastern Himalaya	332	317	301
Colombian Choco	231	215	198
Central Chile	231	215	198
CA Floristic Province	0	−20	−43

Source: Costello and Ward (2003)

Therefore Costello and Ward find that the differences between the Rausser-Small and Simpson et al. estimates have very little to do with the search assumption, but rather it is parameter differences that explain the differences. For example, Equation (4) above set $Z = 0.25$ in the Simpson et al. (1996) model, where Z is the exponent in the species-area relationship. Rausser-Small in contrast have an implicit assumption that $Z = 1$. Also, the value of n (the number of species) is far higher in Simpson et al. (1996) than in Rausser and Small (2000), which lowers the values in the former case and raises them in the latter. Essentially the Rausser-Small values multiply the Simpson et al. values by a factor of 344 (Costello & Ward, 2003). All together the Costello and Ward analysis creates a model to correct for parameter differences, which includes changes in the number of species, ecological model parameter (Z), the number of tests per year (λ), search costs (c), probabilities (p) and other parameters (see Table 8.3).

TABLE 8.3 Comparisons of Parameter Values

Parameter	Simpson et al. [28]	Rausser and Small [26]	Range of new estimates
N	2.5E5 species	74,640 kha	[2.5E5,5E5] species
z	0.25	1	[0.17,0.43]
λ	10.52 tests/yr	26.43 tests/yr	[3,15.03] tests/yr
c	$3600 per species	$485 per kha	[$4000,$18,000] per species
r	10%	10%	[1%,10%]
p	1.2E−5 per species	[1.1E−6, 1.1E−4] per kha	[2.5E−5,1.0E−3] per species
R	$4.5E8	$4.5E8	[$3.4E8,$2.2E9]
e	[0.09, 8.75] species/kha	[0.09, 8.75] species/kha	[0.1,4.7] species/kha

Source: Costello and Ward (2003)

Costello and Ward do find that the marginal value of the most promising hotspots does increase under ordered search of heterogeneous leads, to $14/ha (mean estimate) and to $65/ha (upper 5% quartile estimate). Still, these estimates do not provide significant incentives for large-scale private-sector conservation via bioprospecting. Now that the difference can be explained mainly by parameter values, it now appears that private values are parameter dependent (Pearce, 2004).

It appears that depending on the selected parameters, conservation incentives can be either extremely small or quite large. Costello and Ward (2003), by providing a range of defensible estimates for each of the parameters in the model, and recalculating the marginal value of land for bioprospecting, found that for most parameter combinations within that range, marginal land values from bioprospecting are far too small to provide plausible conservation incentives. In summary, these results support those of the pessimistic Simpson et al. (1996) estimates, that the bioprospecting conservation incentive is insufficient to offset land conversion. In general the private-sector cannot be expected to efficiently provide public goods such as biodiversity, which is supported by these results, so other incentive mechanisms besides bioprospecting are necessary for its protection.

CONCLUSIONS

These results have many implications for policy incentives that target biodiversity preservation. If private prospecting values are high, such as Rausser and Small (2000) suggest, then there may be no need for a policy instrument to encourage prospecting. If values are small, such as suggested by Simpson et al. (1996), then expected prospecting activities should be low, and there would be no rationale for encouraging prospecting because the values captured would be small. The role for instruments to encourage prospecting will come however, when private and social values diverge significantly (Pearce, 2004). It appears that the emerging and optimistic market for bioprospecting cannot be sustained without more development into valuation models, and in this case better developed parameter values. However, as noted earlier, there is more to the value of biodiversity than commercial benefits such as those from bioprospecting.

Moreover what can be taken from this argument is the importance of clear and properly designed valuation methods for biodiversity conservation incentives. This will give decision-makers a stronger foundation from which to build biodiversity policy that includes valuation and cost-effective behaviour. While market-based instruments have the potential to achieve a given level of environmental protection while minimizing cost, the only way to validate these claims is to implement more valuation studies on actual cases of market-based instruments. What can be said is that economic instruments are becoming more popular because of growing disappointment with the regulatory status quo for conservation. These instruments therefore deserve to be expanded, varied and validated so that incentives for conservation can be found.

IX RECOMMENDATIONS AND WAYS FORWARD

Economists have addressed the problem of biodiversity loss with the identification of policy tools and instruments that would allow biodiversity preservation to be delivered to society in a cost minimizing way. The increasing interest in incentive based mechanisms such as MBIs and the increase in analytical tools such as economic valuation in regulatory decision making have helped to increase economic efficiency in biodiversity conservation. However promising these tools seem to be, there are a number of conditions that have to be considered in order to provide society with the greatest net gain from policy decisions based on economics.

IMPROVED IMPLEMENTATION OF ECONOMIC TOOLS AND VALUATION

In order to recognize the importance of valuation as a tool for designing appropriate incentives, the methodologies need to be developed further as they play a very important part in the creation of incentives for biodiversity conservation and policy development. “The fact that biodiversity issues often receive low priority in policy decisions is at least in part due to problems involved in assessing its contribution to society – these values defy easy description and quantification” (OECD, 2002). Addressing this problem implies (see UNEP 2004):

- continued research of methodologies for biodiversity and its resources;
- developing and refining non-market methods of valuation; and
- circulating information on existing techniques for valuation.

According to the Convention on Biological Diversity, CBD (see UNEP 2004), enhancing the use of economic instruments requires a greater effort to understand the potential of various economic instruments and tools. This will contribute to the more widespread use of these tools to conserve biodiversity and implement biodiversity-related policy more efficiently and effectively. As well as enhanced understanding, the successful use of economic tools also requires renewed efforts to support and guide their introduction into policy. Recommendations include:

Valuation: valuation is an important tool for designing appropriate policy incentives; therefore it is important to build on this knowledge and to pursue ways of creating market signals for the social, cultural and economic values of biodiversity.

Information: the effective design and implementation of incentive measures involves a large body of knowledge and information, and the cooperation of participants to ensure the availability of required information and to reduce transaction costs associated with market exchange.

Capacity building: the existence of appropriate legal and policy frameworks to help identify property rights, problems with existing conservation efforts, and opportunities to strengthen cooperation in order to support developing countries in making appropriate use of economic instruments and other incentive measures.

Pilot and case studies: in order to strengthen the understanding and capacity to design, implement and assess economic incentive measures, there is a need to launch pilot studies to provide more data. Case studies reflecting the experiences of different developed and developing countries provide a good basis from which to evaluate the strengths and weaknesses of specific incentive measures.

Also according to the CBD (see UNEP 2004), incentive measures should be designed in a way that addresses the conservation and sustainable use of biodiversity, while taking into account:

- local and regional knowledge, geography, circumstances and institutions;
- policy measures and structures already in place, including perverse incentives;
- the need to match the scale of the instrument to the scale of the problem; and
- the measures' relationship to existing international agreements.

Therefore cooperation in these areas may contribute to the more widespread use of economic instruments to conserve biodiversity as well as cooperation between interest groups and local communities.

BIODIVERSITY CONSERVATION IN LOCAL COMMUNITIES

In many cases economic instruments have resulted in the improvement of local standards of living, including behavioural changes towards the protection of biodiversity, such as through the creation of markets for biodiversity services or products which can provide financial returns to local communities. In addition, the development of property rights can improve asset ownership and investment, and cooperation with local communities in biodiversity management.

However, because of the dependence of locals on wildlife and other biological goods and services, policy decisions directly affect their livelihood. In particular when conservation limits the economic activity of the locals, the residents have to give up the outputs that they have been accustomed to receiving, which may be significant to their livelihood. Without any compensation for the economic loss they have experienced, and often with few alternative ways of making a living, there may not be any other choice but to continue using the resources to be protected. As such there becomes a need to either provide compensation for local residents if they have to give up products from a protected area or develop alternative income sources to replace them. For example, Debt for Nature Swaps provide environmental protection by forgiving debt owing from developing countries and in turn reinvesting the resulting savings from the cancelled debt payments back into the environment (see Box 9.1). From 1988 – 2003, the World Wildlife Fund, was able to generate $46,243,460 US dollars in conservation funds through commercial debt-for-nature swaps in seven participating countries (WWF Centre for Conservation Finance, 2003). But, without the participation of locals in conservation goals, local communities may be very reluctant to participate or to give up their traditional use of resources, and as a result the environment suffers.

Therefore considering local impacts are crucial when laying out incentive goals and discussing the distribution of the wealth generated by biodiversity resources. Benefit sharing therefore depends on the early identification of stakeholders and beneficiaries, including the creation of legal structures to protect these groups (UNEP, 2004). It is also important that governments provide information to increase local awareness and knowledge of resource conservation and environmental protection, including efforts to build up support for these conservation goals. So before applying economic instruments, it is important to clarify the role of local communities in the implementation of economic instruments.

Box 9.1 Philippines Tropical Forest Conservation Act Debt Reduction

PURPOSE: To finance the conservation, protection, restoration and sustainable use and management of tropical forests in the Philippines under the United States Tropical Forest Conservation Act (TFCA).

RESULTS: Under the agreement, the U.S. provided $5.5 million to cancel part of the Philippines' debt to the U.S. As a result, the Philippines will save $8.2 million in U.S. dollar debt payments over the next 14 years. In return, the Philippines will set aside the local currency equivalent of $8.2 million toward conservation activities over the next 14 years. The Government of the Philippines will pay this $8.2 million into a Tropical Forest Fund, which will be used to finance tropical forest conservation activities through restoring and maintaining parks, protected areas and reserves, as well as training scientists, technicians, and managers involved in conservation.

Source: http://www.worldwildlife.org/conservationfinance/swaps.cfm

SUSTAINABILITY AND INTERGENERATIONAL EQUITY

One may say that every life form has an intrinsic right to exist and deserves protection. This includes the right for future generations to inherit a thriving planet and continue to enjoy the benefits that nature has to offer. Sustainability generally refers to giving every generation an undiminished supply of water, air, soil, animal species and resources as was given to past generations (Solow, 2005). The problem with this definition, like many concepts of nature and preservation is that it is vague and difficult to implement. It is important, for the purposes of this paper to define more formally and more economically a definition of conservation and the sustainability of an ecosystem, its species and genetic variations.

The idea of sustainability must therefore be about the obligation of current generations to generations of the future, and more importantly the idea of sustainability as leaving the Earth untouched, is both economically and socially undesirable. Sustainability therefore should be the obligation to leave future generations the option or capacity to be as well off as current generations (Solow, 2005). This introduces the idea of substitutes, so that we are able to consume and meet our needs and wants as long as we leave behind a capacity to create well being for the future. However, under the definition of sustainability, this does not mean that any particular species and ecosystems in themselves should not be protected, if they have an independent value and no good substitutes. What is important is investment, such as investing the rents generated by non-renewable resources, and a reliance of renewable resources as a substitute for non-renewable ones. Knowledge such as research in science and technology is an investment into the future that is environmentally neutral and encourages better use of today's resources; ultimately reinforcing a sustainable environment for the future (Solow, 2005).

Although the idea of sustainability is still vague, this definition provides a deeper understanding of the concept for policy makers. The environment needs protection by public policy because of the externalities associated with its use. Sustainability is a problem because like biodiversity it becomes a market failure as people free ride and profit at the expense of each other and the future. Clear goals and incentives for environmental policy are important because as members of society continue to free ride on the future, they continue to burden the environment (Solow, 2005).

POLICY RECOMMENDATIONS

When developing biodiversity policy and implementing instruments successfully, it is important to consider the following key features (see Brauer et al, 2006):

Clear Objectives: In order for policy incentives to reach their objectives; they need to be clearly defined. All participants must know what the goals of the policy are, and all participants should be working towards reaching the same goal. In the case of biodiversity this requires incentives that work towards maximizing the marginal utility of parties involved when a change in environmental quality is achieved.

Consideration of Social and Economic Effects: policies should take into consideration the environmental, social and economic criteria surrounding conservation decisions. For biodiversity, this requires cooperation with local communities to encourage involvement in markets for biodiversity goods and services in order to provide local conservation incentives. For example, restricting access to forests or ecosystems can have negative consequences for local communities if the forest is one of their main sources of subsistence. In addition the lack of property rights for these ecosystems gives little incentive for locals to invest in conservation activities without clear compensation for their efforts.

Unexpected environmental effects: evaluation of a policy instrument should take into account not only the environmental impact of the natural resource in question, but also other aspects of the environment. If incentives are used to prevent the depletion of one natural resource, it is possible that this may have negative effects on something else. For example, a tax on a particular economic good or service may shift demand to goods and services which are close substitutes and not subject to the tax, which results in little or no change in the conservation of the taxed good, and increases in the loss of the other.

Timing: in order to reach biodiversity objectives, the time over which an instrument is carried out is critical. The problem stems from the long timescale needed to improve habitats and increase biodiversity when government instruments often have short timescales. This makes it difficult to persuade participants to enter into the agreement when they have to make long-term commitments. A balance must be created to make the instrument attractive to participants while being effective in achieving its goals.

Flexibility: Generally, increasing the flexibility of incentives for biodiversity is seen as desirable because the most cost effective firms will achieve the most change and as a result compensate for the small changes achieved by firms who find it more costly to adopt conservation practices. . For example, this advantage can be seen when comparing the use of market-based instruments over traditional CAC methods, which are less flexible (see Section VI).

Combination: Attention must be paid to the way in which different incentives are combined. If market forces and instruments work in the same direction then it is unlikely that problems will be experienced, but if instruments are to work against the market, which the majority of them have to, they must have enough finances to do so successfully. Care must also be taken that different incentives work in the same direction as often one government incentive will encourage negative behaviour at the same time as another incentive is used to try to correct it, such as perverse subsidies in the form of tax breaks for logging companies.

Managing and Monitoring: Of particular importance for biodiversity conservation is the efficiency and effectiveness of policy instruments achieved through managing and monitoring. For example, monitoring for cheating and providing advice to participants as a means of evaluating and encouraging the incentive are recommended. Having a group responsible for managing the instrument should allow it to be adjusted if it is not currently meeting the requirements of the policy objective, as well as to check for mismanagement incentives. Because of asymmetric information, firms do not have an incentive to truthfully reveal their own information, so the government is needed to regulate and monitor a firms' performance. It is important that these activities are funded as part of the set up of an instrument.

Pilot studies: Pilot studies are related to managing and monitoring, and are especially important before the introduction of a totally new instrument. Pilot studies must be carried out to see how stakeholders react and whether it is worth introducing the scheme on a wider scale. Most importantly, pilot studies are needed to fill informational gaps that may arise from the limited application of economic tools in policy.

Information and Effectiveness: As with most aspects of biodiversity, the assessment of policy effectiveness as well, requires adequate information from the start about biodiversity related resources. This includes relevant information about specific causes of the biodiversity loss in question, and information relating to those most affected by the instrument. Large amounts of data must be collected in order to determine what type of instrument is likely to be most successful.

Governments thus play an important role in the design and implementation of economic instruments. The demand for environmental policy comes from firms, individuals, and interest groups who create demands for particular instruments, with legislatures providing the political support for these instruments (see Keohane et al; in Stavins, 2005). In some cases, the government is seen as mainly responsible for providing biodiversity conservation such as creating national systems of protected areas, establishing national funds for preservation, applying environmental taxes, and working with institutions responsible for biodiversity conservation. However, an alternate view for the role of government in conservation is to focus on securing property rights, creating clear legal frameworks, but allowing for private actors and market mechanisms to help to coordinate conservation objectives. In reality the role of government can and should vary when implementing different policy instruments, taking special consideration of each situation.

Therefore there remains an uncertainty about when and to what extent to implement government intervention to achieve conservation goals. Discussions on using economic instruments must go beyond simply sharing experiences to a more focused approach of introducing economic instruments, and insuring they are successful in addressing their goals. Economists have helped influence the debate over environmental policy through research, teaching, and interest in conservation (Hahn; in Stavins, 2005). What economists have contributed is the use of incentive based instruments (such as MBIs) which help achieve goals at a lower cost than other instruments, and the use of benefit-cost analysis to provide a useful framework for decision making. In addition, economists help to educate those entering the world of policy and business by becoming advocates for efficiency through policy outreach and issues such as using economic valuation and the market to achieve biodiversity conservation goals (Hahn; in Stavins, 2005). By continuing to improve their knowledge of the political system, economists are able to design more efficient policies based on economic theory, which have a higher probability of being implemented. Because all policies and regulations have opportunity costs and are subject to some sort of budget constraint, the demand for environmental policy is high, but so are the costs (Hahn; in Stavins, 2005). Economists therefore can help by building more cost effective mechanisms for achieving policy goals.

INTERNATIONAL BIODIVERSITY CONVENTIONS

Over the past twenty years a number of international treaties have been agreed upon to ensure the ongoing sustainability of the biodiversity on Earth. The international co-operation between countries has led to the development of five treaties completely or closely related to biodiversity (Collins & Crain, 1999): The Convention on Biological Diversity (CBD); The Convention on International Trade in Endangered Species (CITES); The Convention on Migratory Species (CMS or the "Bonn" Convention); The Ramsar Convention on Wetlands of International Importance; and The World Heritage Convention (WHC).

Canada in particular has generated its own response to the Convention on Biological Diversity by creating The Canadian Biodiversity Strategy (2005). The CBD was signed by 150 government leaders at the 1992 Rio Earth Summit, and is dedicated to promoting sustainable development. The CBD recognizes that biological diversity is about more than plants, animals and micro organisms and their ecosystems – it is about people, their actions and how their actions effect the environment (Convention on Biological Diversity (CBD), 2000). The main issues addressed in the convention include:

- Measures and incentives for the conservation and sustainable use of biological diversity.
- The sustainable use of biological resources and biodiversity conservation
- Regulated access to genetic resources, and fair and equitable sharing of benefits generated
- Access to and transfer of technology, including biotechnology.
- Technical and scientific cooperation.
- Impact assessment.
- Education and public awareness.
- Provision of financial resources.

- National reporting on efforts to implement treaty commitments.

Canada's biodiversity strategy was developed to provide a framework of action to address the goals of the CBD in a Canadian context by enhancing Canada's ability to ensure the productivity, diversity and integrity of Canada's natural systems while continuing to support its sustainability (Canadian Biodiversity Strategy, 2005). The strategy's five goals are:

- Conserve biodiversity and use biological resources in a sustainable manner;
- Improve our understanding of ecosystems and increase our resource management capability;
- Promote an understanding of the need to conserve biodiversity and use biological resources in a sustainable manner;
- Maintain or develop incentives and legislation that support the conservation of biodiversity and the sustainable use of biological resources; and
- Work with other countries to conserve biodiversity, use biological resources in a sustainable manner and share equitably the benefits that arise from the utilization of genetic resources.

These conventions have recognized the need for co-operation at an international level, as well as greater information reporting and exchange requirements. Although not all countries have joined, participation is extensive in that it includes a broad range of industrialized, emerging and developing nations (Collins & Crain, 1999). However, it seems that current treaties to promote biodiversity conservation are beginning to take into consideration some of the economic causes of biodiversity loss such as the under-valuation of biodiversity, property rights and the tradeoffs between human consumption and biodiversity loss. This creates great optimism for policy to include further economic analysis into their decisions, and perhaps a greater economic focus has helped to contribute to the success of these treaties.

X CONCLUSIONS

Economic valuation techniques and tools provide policy-makers with better information and improved policy instruments to promote more sustainable use of environmental resources. In the case of biodiversity, economics provides decision makers with better options to deliver conservation goals to society more effectively. By providing these alternatives to traditional regulatory approaches, policy decisions can be based more on an understanding of the total value that biodiversity offers as well as addressing the market failure associated with biodiversity resources. Biodiversity offers much to the wellbeing of society. Through economic valuation and tools, the connections between the choices people make and the changes in biodiversity resulting from these choices can be made more direct. Valuation works to provide a framework from which economic tools can deliver the complete value for biodiversity and the goods and services it provides. This helps to encourage the formation of markets where biodiversity can be traded and to promote more effective international agreements to promote the achievement of conservation goals.

The undervaluation of biodiversity in the market first requires addressing how to value biodiversity correctly. Secondly, this requires implementing tools to ensure these values are taken into account in biodiversity conservation choices. Valuation tools have the potential to achieve conservation goals more effectively and this gives policy-makers the incentive to support continued research into their use and effectiveness. The investment in valuation techniques and actual estimates will assure society that the additional benefits of biodiversity protection incentives are worth the additional costs. For example, emerging markets such as those for eco-labeling and bioprospecting, when evaluated on the basis of economic criteria, have proven to not be fully successful in their ability to deal with all the externalities related to biodiversity conservation. Therefore, what must be stressed is the development of good instrument design that will maximize the value of conservation incentives.

The most important point to realize about biodiversity is that the *status quo* of government regulation is perhaps not the most desirable approach, and certainly not the most cost effective. Continued interest in economics as a viable source of information for biodiversity conservation is essential to maximize the potential of these tools developed from economic theory. Many barriers remain to using economic tools and valuation to promote biodiversity, but the theories behind them show great potential, and it is important to consider the various ways in which economics can aid in the process of biodiversity conservation. The biggest problem with biodiversity conservation in general is the lack of research and information for a more complete understanding of the interaction between human behaviour and biodiversity. To fight biodiversity loss, society needs to invest more in biodiversity research – with economics helping to uncover and demonstrate the total value of its loss.

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