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INCENTIVE MEASURES: PROPOSALS FOR THE APPLICATION OF WAYS AND MEANS TO REMOVE OR MITIGATE PERVERSE INCENTIVES

Perverse incentives in biodiversity loss: submission by the Organization for Economic Co-operation and Development (OECD)

Note by the Executive Secretary

1. At the request of the Organization for Economic Co-operation and Development (OECD), the Executive Secretary is circulating herewith, for the information of participants in the ninth meeting of the Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA), a document entitled "Perverse incentives in biodiversity loss", prepared under the guidance of the OECD Working Group on Economic Aspects of Biodiversity. A preliminary version of the document was distributed to participants in the Workshop on Incentive Measures for the Conservation and Sustainable Use of the Components of Biological Diversity, which was held was held in Montreal from 3 to 5 June 2003, with financial support from the Government of the Netherlands, in response to decision VI/15 of the Conference of the Parties to the Convention on Biological Diversity. The report of that Workshop is available as an information document for the ninth meeting of SBSTTA (UNEP/CBD/SBSTTA/9/INF/10).

2. The document is being circulated in the language and the form in which it was received by the Secretariat of the Convention on Biological Diversity.

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PERVERSE INCENTIVES IN BIODIVERSITY LOSS

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FOREWORD

Governments use incentive measures in a variety of public policy contexts to achieve socially desirable outcomes as efficiently as possible. In many instances, however, those incentives will have unforeseen consequences — some of which may be harmful; i.e. are “perverse”. Policies that have perverse consequences — whether at their inception or subsequently — are generally welfare reducing and should be reformed.

The policy context in which to undertake the reform poses some analytical challenges. Since all economies have numerous policy instruments already in place, the discussion of perverse incentives must occur within a “second-best” policy setting. This document looks at some areas where policies have perverse impacts on biodiversity and suggests approaches for how a reform may be undertaken with the goal of ensuring welfare improvements. In so doing, it focuses on government subsidies that cause damage to biodiversity. The messages, however, can also be applied to non-subsidy incentives that affect biodiversity.

This document was drafted by Philip Bagnoli under the guidance of the OECD Working Group on Economic Aspects of Biodiversity. This report is published under the responsibility of the Secretary-General.

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PERVERSE INCENTIVES IN BIODIVERSITY LOSS

1. Introduction

OECD (1996) defines incentives to broadly include those measures that make use of the price system and market forces to achieve their objectives. Governments use incentive measures in a variety of public policy contexts to achieve socially desirable outcomes as efficiently as possible. In many instances, those incentives will have unforeseen consequences — some of which may be harmful. For such cases, the incentive can be considered “perverse”. For biodiversity, perverse incentives are important issues that have been identified as being particularly relevant to its conservation and sustainable use. More specifically, the Convention on Biological Diversity (CBD) describes perverse incentives as:[1]

...a policy or practice that encourages, either directly or indirectly, resource uses leading to the degradation of biological diversity. Hence, such policies or practices induce unsustainable behavior that reduces biodiversity, often as unanticipated side effects as they were initially designed to attain other objectives. Several common types of perverse incentives are usually identified as: environmentally perverse government subsidies; persistence of environmental externalities; and, laws or customary practices governing resource use.

So far, attention on perverse incentives has tended to be focused on subsidies. The CBD has encouraged this development through its view that considerable progress can, and should, be made in this area:[2]

Government subsidies that encourage biodiversity decline can be quantified financially, and represent a clear opportunity for policy reform to promote the objectives of the Convention. Notwithstanding the need to address all perverse incentives, in the first instance it is recommended to concentrate on identifying government subsidies with perverse effects on biological diversity.

In this vein, much attention has been focused lately on perverse incentives. The interest of policy-makers can be seen in two main sources. **Firstly**, when the incentives come in the form of subsidies, they draw on public finances, and are a potential source of sub-optimal use of a society’s resources — possibly causing more socially desirable goods and services (e.g. biodiversity amenities) to be provided in insufficient quantities. Moreover, expenditures on subsidies contribute to budgetary pressures, which, during the 1990’s, led most OECD countries to undertake a broad re-examination of public spending to ensure that taxpayers were getting good value. That process is still ongoing. Buttressing the need for that review of subsidies has been the work of policy analysts who have pointed out that their extent and impact on economic activity is considerable (Steenblik and Coroyannakis, 1995; Lee, 2002; Myers and Kent, 2001). Those studies highlight that subsidies are found in a wide range of economic activities in a number of different forms. They can keep entire industries in business long after they have ceased to be

1. See SCBD (2000), p. 11.

2. *Ibid*.

competitive with their rivals. For example, Anderson (1995) estimated that, prior to recent reforms, the subsidies given to the coal industry in West Germany had reached USD 90 200 per miner. **Secondly**, the reason why incentives that have perverse impacts on biodiversity are receiving increased attention is that they degrade a facet of the environment that is valued though not fully priced. Lack of explicit accounting for biodiversity's inputs into market transactions causes over-use; reducing welfare by leaving a lower quality of environmental amenities than would otherwise be desired — assuming the environmental impact was not foreseen.

The use of environmental/biodiversity inputs into economic processes has, of course, always been an essential part of growth and development. Some loss of biodiversity is inevitable if standards of living above subsistence are to be achieved. The loss of biodiversity, however, may occur as part of a *purposeful* decision taken by society to use its natural resources on the basis of tradeoffs it deems beneficial (e.g. *Hartwick's rule*, Hartwick, 1977). Or it may be *inadvertent*, such as when the consequences of policies on biodiversity either were not foreseen or were not properly assessed. This paper is primarily, though not exclusively, concerned with these latter, inadvertent, impacts where there is a clear need for public policy to correct an undesirable outcome. It also attempts to illustrate succinctly the negative impacts of many incentive measures on biodiversity in order to encourage their integration into decision-making so that the *right balance* between economic development and biodiversity sustainable-use can be achieved. From a biodiversity perspective, getting the balance right implies ensuring that market transactions fully reflect the impacts that production and consumption have on biodiversity.

While much of the relevant literature in perverse incentives focuses on subsidies, biodiversity is often impacted by other measures that can be just as important. Government policy that is intended to encourage regional development, for example, can be a source of impacts on biodiversity when the creation of employment emphasises resource use (e.g. fisheries, forestry, etc.). Such policies do not always have an obvious monetary component and may be "implicit" in conveying benefits to targeted areas. Clearly terminology is an important element of the discussion of subsidies and perverse incentives: how far should the definition of a subsidy be stretched to capture the notion of a transfer? Steenblik (2003) notes the heavy influence of existing (often conflicting) terminology related to subsidies and calls for building on the definition of a subsidy given in the WTO Agreement on Subsidies and Countervailing Measures. The objective definition that is advocated would account for a wide range of influences that could be termed perverse incentives. It would be a well-founded definition but still not be a complete measure of *inadvertent* impacts on biodiversity: the focus would be on measures of adjusted prices as indicators of support. It is the accounting of all sources of impacts (direct and indirect) that the study of perverse incentives is intended to address.

Perverse incentives are, of course, part of a broader discussion involving basic elements of social-choice. Economists argue that a necessary condition for maximising social welfare is for market prices to reflect all processes that were engaged in producing goods and services. In other words, prices should fully reflect individuals' preferences — both for the environment as well as for the things they buy. Perverse incentives, when they are unintended, lead to prices which do not reflect social preferences and thus lead to a loss of welfare through overuse of some goods and services.

The policy context in which to consider incentive measures, however, poses some analytical challenges. Since all economies have numerous policy instruments already in place, the discussion of perverse incentives must occur within a "second-best" policy setting. That is, other policies will create pre-existing distortions in the economy, making it difficult to know whether the full elimination or a simple reduction in incentive measures is called for. In other words, incentives need to be considered for the full range of their impacts in conjunction with existing policy measures and other sources of sub-optimal outcomes.

This paper outlines where (and, when possible, *how*) subsidies and incentive measures impact on biodiversity. It does so, however, while pointing out that policy changes often require some analytical underpinning in order to ensure that the changes will always cause a net gain. While there is little doubt that many subsidies and other policies did not consider biodiversity impacts when they were first introduced, a systematic appraisal of policies is nonetheless important to avoid undesirable consequences. Section (2) briefly explores the impacts of incentives and discusses the underlying conditions that affect their outcomes. It provides a predominantly descriptive treatment of **subsidies,** leaving biodiversity-perverse incentives as an important sub-text that is treated indirectly. More detail, and a somewhat more formal treatment, is given in Annex 1. Section 3 surveys available studies regarding the magnitude and impacts of incentive measures on biodiversity. Where possible, some observations are made regarding motivations for the measures. The section also highlights that the lack of available data, as well as detailed analysis, clearly call for more work in this area. That discussion is then followed by Section 4, which outlines some methodological issues in removing subsidies. Some concluding observations are given in the final Section (5).

2. Inter-linkages and incentive measures

The need for detailed attention to be given to various incentive measures, and their impacts, is underscored by the inter-connectedness of policies and economic activity. An incentive measure given in one economic sector will have implications beyond the sector in which it is applied. Since market economies are generally either at, or close to, what can be termed a “dynamic” equilibrium, impacts in one sector will *necessarily* spill over into others. This occurs because in a general equilibrium, all markets are in balance in the sense that prices ensure that demand and supply are roughly equal: if more money is spent by firms and consumers in one sector, less money *must* be being spent in other sectors.[3] Whether this impact is distributed across many sectors (and is, therefore, imperceptibly small for any one) or concentrated in a few (where it becomes substantial) is an empirical question: it depends on elasticities in those other sectors. Moreover, in an economy with a number of taxes and other government policies impacting on production and consumption, it may become difficult to determine the *net* impact of a particular measure. That is, general equilibrium implies that many sectors will be indirectly impacted by policies introduced elsewhere; therefore, determining the amount of harm done by a particular subsidy needs to begin by first accounting for the impact of other policies.

From a more traditional “macro” perspective, there is also consideration of the fiscal impact of a perverse incentive, particularly when it is given in the form of a subsidy. As was mentioned earlier, funding will not be available for other programmes that a government may want to undertake. The removal of subsidy measures, therefore, will either free up funding for other priorities or, by permitting tax reductions in other areas, provide a fiscal stimulus to the economy. In addition to the social benefits of internalising the impacts of perverse incentives on biodiversity, there are also economic benefits worth considering.

These observations are made particularly relevant by the fact that governments impose incentive measures for a wide range of reasons. To distinguish between measures that are, on the whole, welfare-enhancing from those that are not requires some analytical effort. Even more important is the fact that when the measures are deemed beneficial for the economy, mitigation of their impacts *on biodiversity* would be desirable. Whether this calls for replacement of the incentive (by an alternative instrument), or a fine-tuning of the incentive will be difficult to determine without careful consideration. Lack of good analyses should not be allowed to undermine cases where reform is clearly needed, but the difficulties associated with undertaking them should not be a basis for proceeding indiscriminately. Indeed, only

3. As is discussed in Annex I, the repercussions across various sectors will not exactly offset each other.

explicit accounting of all interactions, and sources of distortion in the economy, will ensure that changes in policy result in the realization of maximum social benefits.

Achieving an improvement in economic outcomes through a review of subsidies, however, is not sufficient to ensure that in the long term those gains will be maintained. If a review of subsidies occurs once and then is left without additional consideration, new subsidies may arise that could slowly erode the gains made during the review. A process of examining all new subsidies is called for to consider their *potential* impact on biodiversity before they are implemented. Because many of the impacts of existing subsidies could not have been predicted when they were first put in place, it will also be necessary to undertake periodic reviews. This would ensure that both the original purpose remains socially desirable, as well as a re-evaluation based on the most recent information that is available concerning ecosystems.

The inter-linkages that have been noted here, and in Annex 1, suggest that impacts of incentive measures are empirical questions, which call for the use of empirical tools. In other words, engaging a process of subsidy removal would call for considerable effort with tools such as data collection, traditional econometric analysis, and numerical modelling.

3. Perverse incentives and their impacts

A number of authors have attempted to quantify the amount of subsidy given to various economic activities, as well as some of their influences. For example, OECD (1998a) and Porter (2003) outlined some of the major areas where subsidies can be rigorously measured based on data that is already available. Beers and Moor (2001), as well as Myers and Kent (2001), on the other hand, attempted to account for areas where data are less readily available. While each of these studies found that the amounts given were very large, the latter studies claimed that total subsidies were not only a major factor in environmental outcomes, but in economic development as well.

Biodiversity, however, needs to be distinguished from the general environmental impacts that are outlined in much of that work. Concerns regarding human health make fresh air and clean water important but, since environment-related issues in air and water quality are often concentrated around population centres, they are not as pressing for biodiversity. Since government priorities are often focused on human-health related environmental issues, by not distinguishing longer term problems from more immediate ones, the former may never get the attention they need.

The remainder of this Section is organized around economic sectors where perverse incentives have important consequences. An attempt is made to quantify the incentives being given (where measurable), and discuss the manner in which they impact on biodiversity. One observation worth making at the outset is that the most important factor influencing biodiversity loss is how extensive an activity has become. For example, in general, policies that cause geographic expansion in economic sectors such as agriculture, forestry or fishing will be an important source of biodiversity loss. A significant potential exception to this, however, is greenhouse gases emissions which can induce climate change. Incentives that encourage such emissions have the potential to cause large losses in biodiversity by changing local ecosystems at a rate too rapid for adaptation to occur.

3.1 Agriculture

3.1.1 Crops and livestock rearing

Although direct measurement of the impacts of incentive measures on biodiversity is difficult to undertake, some partial and indirect studies have been undertaken. When combined with broad discussions of agricultural subsidies and the environment (OECD, 1998b; and 2000), a reasonable appreciation can be obtained of the magnitude of the policy issue.

OECD (2001a) provides definitions and estimates of support for agriculture in OECD countries. There are three broad categories that are used to group the data (using definitions given therein): producer support estimate, consumer support estimate, and general services support estimate.

Table 1. Estimates of support to agriculture (EUR Million, 2002)

Producer Support Estimate	***249.2***
Market Price Support	157.8
Payments based on output	9.4
Payments based on area/animal numbers	35.2
Payments based on historical entitlements	11.6
Payments based on input use	22.7
Payments based on input constraints	7.9
Payments based on overall farming income	4.4
Miscellaneous payments	0.3
Consumer Support Estimate	***-145.8***
Transfers to producers from consumers	-152.5
Other transfers from consumers	-23.9
Transfers to consumers from taxpayers	29.9
Excess Feed Cost	0.6
General Services Support Estimate	***58.6***
Research and development	5.8
Agricultural schools	2.1
Inspection services	2.0
Infrastructure	17.0
Marketing and promotion	24.9
Public Stockholding	2.1
Miscellaneous	4.8

Source: OECD (2003b).

For biodiversity, some of the more prominent subsidies are those given on the basis of output, per area or animal or input use — in the terminology of the WTO Agreement, *Amber Box* and *Blue Box* support measures. These payments tend to encourage farming practices that are either not sustainable in the long run, or adversely affect the environment off the farm.

Boardman *et al.* (2003) emphasize economic influences which they argue underlie most sources of damage to farm soils and other farm features. Some of the main biodiversity-related impacts are outlined in OECD (2001a) and Portugal (2002) as losses in:

- soil quality (erosion, nutrient supply, salinity);
- water quantity (lowering of water tables);
- diversity of plant and indigenous animals;
- habitats for plants and animals.

Stoate *et al.* (2003) and Donald *et al.* (2002) draw a more direct link between the payments made under the European Common Agricultural Policy (CAP) and damage to farm environments as well as to biodiversity. While this damage is being reduced as a result of reforms to the CAP, the subsidies and the consequent damage are still considerable. Some recent initiatives, however, attempt to correct for adverse impacts by purchasing biodiversity "services" from farmers. In Switzerland, for example, farmers are paid to meet certain agri-environmental targets that include biodiversity (OECD, 2002). In the US, an Environmental Benefits Index has been developed that scores the environmental consequences of changes in farmland use and allows farmers to be paid for those changes (through an auction bidding system). In Europe, national agro-environmental programmes have been implemented following the CAP reforms — part of which concern biodiversity conservation and the protection of elements of landscape such as hedges. More recently, the Natura 2000 programme has also begun to pay farmers for putting aside and improving farmland for its biodiversity benefits.[4]

Historically, the most significant biodiversity-related impacts of assistance to agriculture are caused by subsidies that encourage the extension of agricultural lands. By extending agriculture, those subsidies result in land being converted from forests, rainforests, and wetlands into agricultural production (Runge, 1994; Oregon State University, 2001). In the U.S., for example, some 50 per cent of wetlands have been lost mainly due to agricultural conversion (OECD, 1999)[5], while in Europe that number is closer to 60 per cent — in both regions the process continues. However, the relationship between agriculture and biodiversity may not always be straightforward. Some authors (Donald *et al*., 2001; among others) have argued that the long periods of traditional agriculture in parts of Europe have created a new balance for many species — they now depend on the altered open landscapes. For example, in the current European landscape, threatened species are sometimes found in farmed areas, notably grasslands, which have been argued to benefit from the continuation of traditional, extensive farming activities.[6] It has even been argued that the small farms that can be found in many OECD countries contribute to biodiversity by creating a wider variety of mini ecosystems (e.g. alpine meadows). This argument holds that extensive farming supports biodiversity when it creates a high number of small farms, or varied agricultural activity in an otherwise marginally diverse landscape.[7] The question, of course, is whether social welfare is improved by such an outcome because its benefits are greater than its cost.

Subsidies that cause more intensive forms of agriculture also have an impact on biodiversity. The high-input intensive modern farming technique, with its reliance on monoculture, mechanization, and the use of an extensive battery of agro-chemicals, has an impact on biodiversity through the simple large-scale conversion of farm-related ecosystems (Srivastava *et al.,* 1996). Subsidies also lead to greater use of fertilizers and pesticides and even to reduced crop rotation (Runge, 1994; Faeth, 1995). This leads to increased rates of loss of soil productivity (Liebhardt, 2002) which may not only hasten its withdrawal from agricultural production, but also impact the types of ecosystems that can return afterwards. It has also been found to be associated with adverse changes in landscapes, such as the removal of hedges, field edges and ponds — all of which provide habitats. Moreover, any subsidy that favours conventional agriculture to the disadvantage of alternatives (e.g. organic farming) damages biodiversity without necessarily providing additional agricultural output (Jones, 2003). Nonetheless, some have argued that intensive agriculture can

4. It should be pointed out that the net effect of these programmes may be to pay farmers twice to achieve the same outcome that no policy at all would have had on the environment.

5. This is an area comparable to the landmass of Germany.

6. Extensive farming habitats even represent an important part of the proposed "sites of community interest" in the context of the "Habitats" directive in the alpine, Mediterranean and Atlantic biogeographic regions.

7. It can be pointed out that the designation of a number of areas around the Mediterranean as biodiversity "hot spots" is, to a significant degree, the result of farm biodiversity — sustained by extensive agriculture.

benefit biodiversity by allowing the world's food supply to be produced in a smaller area than would otherwise be necessary.

An inescapable conclusion of these observations is that subsidies that encourage either intensive or extensive agriculture can be either damaging or beneficial. The outcome depends on the context in which they are applied. Nonetheless, on a global scale it seems clear that extensive agriculture that leads to replacing biologically diverse areas with monoculture (or grazing areas) has the largest negative impact on biodiversity. A good place to *start* subsidy reform for agriculture in many countries, therefore, is to focus on those subsidies that lead to agriculture's extension. However, given the amounts of money that are spent in support of all forms of agriculture (Table 1), and the amount of damage done to biodiversity, it is also clear that broad reform is needed — even in the short-term.

A secondary, though important, question concerns the issue of *when* the reforms have achieved social-welfare maximising outcomes. That is, when all things are considered, at what point are the externalities being addressed by subsidies overwhelmed by the distortions that are caused by the subsidies? The fact that subsidies to extensive agriculture can have both good and bad consequences for biodiversity — as can subsidies to intensive agriculture — implies that this is a difficult question that calls for considerable analysis. There is much scope in this area for classical cost/benefit analysis that consists of internalising a wide array of non-marketed aspects of agriculture (both good and bad).

Some of the more severe permanent effects of unsustainable agriculture can be seen clearly in the desertification that has occurred in some regions as a result of farming practices (Sombroed and Sene, 1993). In an extensive survey, Oldeman (1994) found that globally some 15 per cent of all soils used for economic purposes (i.e. approximately 1.2 billion hectares) were degraded beyond the ability of individual farmers to repair. However, Crosson (2003) argues that farmers who are making a living from the productivity of their soils have a vested interest in not degrading the soil too rapidly. The fact that soil degradation is occurring when it is counter to a farmer's long-term best interest suggests that there is a public-policy problem. To see why, consider the fact that, for a farmer, the soil is in fixed supply but represents a means to a livelihood. Some farming techniques largely preserve soil quality, causing only a slow rate of degradation, while others degrade it more rapidly. For a farmer, the soil (i.e. farming) must provide a rate of return that is comparable to other means of earning a living (after accounting for preferences in lifestyle). If a rapid rate of degradation is occurring, it must be because short-term objectives are strong enough to overcome long-term interests — inducing farmers into a choice that will lead to leaving the business. This is most clearly seen in subsidies or other incentives that are tied to farm output. In that case, the increased short-term return to farming accelerates the rate of soil degradation by raising the immediate return to the soil. Similarly, Wilson and Tisdell (2001) argue that pesticide use creates a cycle that farmers can not escape which inevitably leads to long-term environmental damage.

Causal links between production-based subsidies and the level of agricultural intensity have been observed (Heerink *et al.*, 1993; Nutzinger, 1994). For some OECD countries, these studies even find evidence for a relationship between estimates of the level of producer support and the use of fertilizers and pesticides.

The policy shift from market-price support to direct payments for farmers in OECD countries led to an increase from 5 per cent of total support in 1990 to 25 per cent in 1998. The change occurred primarily as a result of the Uruguay Round Agreement on Agriculture which encouraged governments to find alternative means of providing support to farmers. Its relevance for biodiversity is that the outcomes show that social objectives — farm income support — can be achieved without necessarily damaging biodiversity and the environment. It reinforces the notion that economic policy levers are effective in achieving environmental goals: the farmers responded, as predicted, to economic signals. For example, OECD (1998b) found that when price supports are removed, farming intensity decreases and a number of repercussions follow. These include lower groundwater pollution, less fertilizer use and in some cases there is less pesticide use.

3.1.2 Irrigation water

An important area where subsidies are given to farmers, either directly or indirectly, is for the irrigation of crops. The form in which these subsidies are given vary considerably both within and across countries. A few examples include: providing low-cost electricity that runs pumps; direct payments to offset the cost of surface-water extraction (equipment, operating cost); and conservation-incompatible water charges that are based on the type of crop and the area irrigated; or volumetric bulk-pricing that effectively gives discounts for larger quantities of water-use. Table 2 provides descriptions of the nature of subsidies and their impacts.

Table 2. Water-related impacts of agricultural subsidies

Description of subsidy	Channel for environmental harm	Impact on environment (biodiversity)
Agricultural price support policies	Incentives for farmers to grow water-inefficient crops.	Salinization, water-logging and/or decline in groundwater (GW) tables leading to changes in local ecosystems.
Surface water price	Overuse of water. Use of inappropriate technologies.	Pollution and depletion of water bodies leading to habitat destruction. Salinization and water-flow problems.
Electricity price	Substitution of surface water (SW) with GW. Overuse of GW due to excessive pumping.	GW levels are lowered and aquifers are depleted. Ecosystems altered by loss of water.
Pesticide prices	Overuse of pesticides and inefficient application leading to leaching.	Pesticides contaminate GW aquifers and impact ecosystems.
Fertilizer prices	Overuse of fertilizer and inefficient application leading to fertilizer leaching.	Fertilizers can increase soil salinity and contaminate GW aquifers, impacting ecosystems.

Source: Sur, *et al.* (2002), with modifications.

These impacts have been observed in developed countries such as the U.S. (SJVPD, 1991) and Australia where subsidies have been linked to groundwater depletion, over-tapped rivers, water logging, and salinization. They have also been observed in other parts of the world where the link has been made to the destruction of ecosystems (Postel, 1999). Indeed, empirical work on water subsidies finds that farmers (globally) rarely pay more than 20 per cent of the real cost of water (Postel, 1999).

From the perspective of effective policy instruments, Garrido (2001) argues that water demand for agriculture is elastic but only when its price is high enough for farmers to notice. He claims that previous studies caused observers to conclude that price changes had no effect on demand by irrigators because the initial and final prices were inconsequential for farmers.

Studies have observed that few countries in the world have “rational” water-pricing schemes where, on the margin, costs reflect benefits (Arlosoroff, 2002). In countries where users face a market-determined price for water (e.g. Israel), the use of water is significantly lower for the production of crops that are similar to those grown elsewhere. Since irrigation is a disproportionate use of water in OECD as well as non-OECD countries, the cost of water to irrigators is the dominant factor in overall water use. Moreover, when irrigation water is priced at levels similar to that for households and industry (i.e. the marginal price of water is equal between uses), water will be allocated to the application where it has maximum value.

The high, fixed-cost, of installing irrigation systems means that in most countries, governments must provide some funding in order to establish the infrastructure. The important policy question, therefore, is whether the government will attempt to recover that cost from water users. Subsidized provision of irrigation systems, where cost-recovery is not intended can, by itself, have a significant impact on biodiversity by leading to over-capitalized systems that draw water more heavily than otherwise. It can be the causal factor in the lowering of water tables - impacting wetlands and water-dependent near-surface processes.

Pricing schemes that do not reflect the marginal cost of provision of water (or, if the complementary inputs such as energy for irrigation pumps are subsidized), can also cause damage to biodiversity by concentrating water use in activity that is harmful (Rosegrant, 1997). For example, over-irrigation of farmland can result in the salinization of soils that would then have long term impacts on the soil's ability to return to pre-agriculture ecosystems.

Subsidized irrigation can also have an impact on biodiversity by its ability to expand areas profitable for agriculture. This promotes extensive agriculture that, in ecologically sensitive regions, will lead to damage to biodiversity. Another indirect impact on biodiversity is through reduced, more-polluted and warmer water flows in rivers from which there are significant withdrawals (and returns) of water connected with irrigation.

3.2 Energy

Assistance given to energy sectors has long been founded on the belief that low-cost energy is an important element of an industrialization strategy. This led to a wide range of government programs that provided the energy sector with a number of benefits, including: direct payments to consumers and producers; various tax benefits; favourable regulations; and assistance in the form of infrastructure and R&D. While the impacts of this assistance on biodiversity is not immediately clear, the discussion in this section will suggest some important links that connect subsidies to biodiversity loss. The quantitative presentation that is made of assistance to the energy sector is, therefore, intended to provide an appreciation of the magnitude of measures leading to the impacts. More detailed and quantitative discussion of the link between energy subsidies and the *environment* can be found in studies such as IEA (1999) and OECD (2001c).

Subsidies in the energy sector, therefore, come in many forms: ranging from those that lower the cost of producing energy, to those that affect the price faced by producers and consumers of energy. Table 3 provides some examples of monetized subsidies, additional work that indirectly measures subsidies is also available. Kosmo (1987), for example, reported on proxies for the energy industry that implied high levels of support. While these results are addressing indirect impacts, as is discussed below, support in the energy industry affects the manner and quantity of energy used — contributing to the biodiversity/environmental consequences of energy.

Some of the known effects of energy subsidies include emissions of volatile organic compounds (VOC), NO_x, SO_x, and particulates (smoke, soot, dust, liquid droplets). For subsidies given to transport fuels, it also causes larger vehicles to be used (i.e. their marginal cost is lower so their benefits —

perceived or real — become more attractive; for example, sport utility vehicles) which in turn cause more $C0_2$ and related pollutants.

Some of the incentive measures have been quantified (Table 3).

Table 3. Sample of energy sector distortions

Source	Source of subsidy	Monetized distortion
		(USD million/year, various: 1988-95)
6 OECD countries DRI (1997)	Coal PSEs in Europe and Japan	5,800
Australia (Naughten *et al.*, 1997)	State procurement / planning Barriers to gas and electricity trade	133 1,400
Italy (Tosato, 1997)	Net budget subsidies to electricity supply ind. (ESI) VAT below market rate Subsidies to capital Excise tax exemption for fossil fuels use by ESI Total net and cross-subsidies	4,000 300 1,500 700 10,000
U.K. (Michaelis, 1997)	Grants/price support for coal and nuclear producers VAT on electricity below general rate	2,500 1,200
USA (Shelby *et al.*, 1997)	DFI (1993) analysis [a] DJA (1994) analysis [a]	8,500 15,400

Note: Subsidies are defined in non-comparable ways. All estimates in the table may have wide margins of error;
a - DFI and DJA analyse different sets of energy supports and use slightly different estimates for some of them.

Source: OECD (1997), see source for references given in Table.

Interest in energy subsidies for the purposes of this paper, however, is limited to the extent to which they lead to damages to biologically diverse areas. The most common form of energy subsidy supports the coal industry — sometimes to extraordinary levels, such as the USD 90 200 subsidy per coal miner that was being paid in West Germany in 1990 (Anderson, 1995). The link between coal mining and biodiversity degradation is, however, indirect. Acid rain and global warming are perhaps the most serious impacts on biodiversity but considerable challenges exist in estimating the cost their impacts. Strip mining in ecologically sensitive areas is, of course, a direct threat to biodiversity loss but it is not widespread.

Incentives for energy have their strongest impact on biodiversity when they encourage energy production in modes that require significant land conversion. For example, large-scale hydro-electric dams can result in the loss of substantial land surface to flooding. Often the construction of those dams is dependent on a limited accounting of purely private costs (i.e. negative environmental externalities are not accounted for, McCully, 1997). Even limiting consideration to private costs, the construction of hydro-electric facilities still requires government assistance to engage the private sector (Anderson, 1996). That is, publicly funded infrastructure such as roads, communication networks, etc., often have to be provided *gratis* in order for the final cost of the electricity produced to be competitive with alternatives. In some cases this is justified on the basis of values for public-good benefits, particularly recreation, fishing and hunting which some observers consider inflated (GAO, 1997).

Varangu (2002), however, observes that subsidies for renewable energy as well as subsidies to assist the poor can provide socially desirable benefits that would be difficult to achieve otherwise. In the case of developing economies, assistance for energy use by the poor can reduce pollution when it enables access to cleaner technologies without requiring the use of more expensive fuels. Many countries have enacted incentive measures to assist renewable energy development in the hope of alleviating problems in global environmental concerns such as climate change. Also important in that endeavour is the desire for a reduction in the reliance of many economies on fossil fuels from regions of the world that have historically unstable. These are, of course, important for considering options regarding incentives in the energy industry. As highlighted earlier, to ensure the maximum social benefit, many factors need to be considered in reforming perverse incentives.

3.3 Transportation

The primary role of transportation (along with communication) in economic development makes it a difficult issue for consideration in reforms of incentive measures — even when considering transportation's impact on biodiversity. Transportation draws producers and their various suppliers closer together, and those producers closer to their customers. From an economic perspective, this is essential element of efficient production because it facilitates specialization according to comparative advantage. When goods and services move freely, production will occur where it is most advantageous and not simply where it is closest to the consumer. Sachs (2000) points out that economic development has been most successful in regions where access to low-cost transportation allowed developing economies to build niches with their low-cost labour.

This justification for facilitating economic development and ensuring optimal resource use, however, needs to be distinguished from policies that make low-cost transportation an objective in-and-of-itself. When it is available for all purposes — regardless of whether or not it contributes to efficient production — it can become a source of reduced social welfare by intensifying negative externalities, including those related to biodiversity. In many economies, a close look at the predominant uses of transport suggests that assistance is primarily going to activities that have little apparent relevance to socially beneficial outcomes — they enhance the personal use of private transport. Many forms of *public* transport are under-funded relative to the social/environmental benefits they provide.

As with the earlier discussion regarding energy, this section begins with a consideration of some empirical results that measure subsidies in general. They are not linked directly to biodiversity loss. Instead these results suggest that, within these countries, there is a potentially significant level of assistance that is contributing to the impacts on biodiversity that are described below.

The assistance given to transport comes in two main forms:

- Direct
  - operating subsidies;
  - concessionary fares.
- Indirect
  - underpriced provision of infrastructure;
  - failure to internalise externalities;
  - foregone tax revenues.

Many of these incentives are difficult to measure. However, amounts given for some of these categories have been examined for European countries and are reported in Table 4.

Table 4. Assistance given to transport.

Country	Road		Rail		Public Transport
	Total Cost[a]	Reported revenue and taxes[b]	Total Cost[a]	Reported revenue and taxes[c]	Service subsidies concessionary fares
Austria	7287	4923	2946	1631	261
Belgium	5398	6239	2524	909	727
Denmark	2667	4558	815	586	241
Finland	2151	3626	600	592	361
France	47735	44016	12395	7367	4654
Germany	59273	41416	20284	13431	1622
Greece	7721	5520	261	135	193
Hungary	7609	1882	406	235	153
Ireland	1332	2393	169	127	0
Italy	30126	36185	6881	3441	0
Luxembourg	291	406	204	100	11
Netherlands	8311	10286	1372	1365	0
Portugal	3459	3819	198	188	33
Spain	15037	12870	3420	1495	512
Sweden	4107	5266	1825	1423	753
Switzerland	6310	4482	3812	2965	566
UK	28074	43983	7974	9125	979

Note: Figures are in EUR 1998. Amounts between columns are not strictly comparable;
a — infrastructure, air pollution, noise, global warming, and accidents;
b — infrastructure charges, vehicle taxes and fuel taxes;
c — ticket and freight revenues, track and station charges and fuel/energy taxes.

Source: Nash, *et al*. (2002).

Given its low *direct* impact on biodiversity, aviation transport has been omitted from the Table.[8] Since the bulk of subsidies given to that sector are in the form of exemptions to certain taxes on airline tickets and fuel, they impact on the marginal cost of air transport and therefore encourage overuse. In most countries, however, these subsidies are given to avoid putting domestic service providers at a comparative disadvantage relative to countries that have no, or lower, taxes on such products. Hopf *et al*. (2001) provide some examples of support given to airports in European countries. Aviation's main impact on biodiversity is through its contribution to high altitude pollutant emissions that contribute to global warming. If projected increases in air traffic occur, this source of emissions could begin to rival other modes of transport. The impact of aviation transport on biodiversity, therefore, is primarily as an indirect potential threat in the future.

Some of the major forms of transportation have significant environmental impacts through the emission of various pollutants (e.g. NO_x and Ozone). In many cases those transport modes receive substantial amounts of subsidization through preferential tax treatment or through government provision of infrastructure. OECD (2002) looks at available data on subsidy levels and concludes that, even in countries with substantial fuel taxes, subsidies to transportation are large. From an environmental perspective, however, certain subsidies are more harmful than are others. Those that go to rail transport, for example, have substantially less impact on the environment than those that go to diesel-fuel consuming trucks.

8. A distinction is made between impacts on biodiversity versus impacts on the environment. Noise and other forms of pollution are generally local in nature (except as noted) and the habitat destruction caused by airports is not on a sufficient scale to endanger species.

For biodiversity the direct impact of the subsidies to transport comes through two major sources, one of which is direct, while the other is indirect. Mader (1984) shows that roads can cause habitat fragmentation (isolated populations of flora and fauna) which MacArthur and Wilson (1967) argue become less resilient and more susceptible to extinction. Moreover, road density (i.e. length per unit area: km/km2) has been found to be a critical factor in the survivability of species within a given area (Forman and Herpsberger, 1996). Large predators such as wolves and mountain lions have difficulty maintaining viable populations in the United States when the density reaches 0.6 km/km2 (in the U.K. there are few areas that have densities below this level).

INRAS/IWW (2000) argues that transport-induced air pollution leads to impacts on forests, as well as changes to landscapes. The extent to which these are biodiversity-related (apart from those just highlighted) is difficult to determine.

An important issue for biodiversity loss is the activity that can follow from subsidized extensions of the transport system which lead to reduced costs for manufacturers and exporters (Sachs, 2000). When a transport system places a previously inaccessible ecosystem within range of low cost travel, the ecosystem will be subjected to *private* cost/benefit analysis regarding its use. Failure to account for *public* benefits results in a sub-optimal use of that geographic region — even on purely utilitarian grounds. The commercial, residential and agricultural use of that area affects habitats and thus is a source of biodiversity loss when the area is ecologically important.

Another source of biodiversity-related impacts of transport is one that is both indirect (often accidental) and not easily quantified. Perrings *et al*., (2000) explore the economics of invasive alien species and find that substantial costs to both economies and environment occur through the introduction of non-native species to environments where they have no natural predators. In some cases they cause economic damages which have been estimated to be very large on an annual and ongoing basis. For example, in the U.S., the European Zebra Mussel, *Dreissena polymorpha*, has infested over 40 per cent of internal waterways and may have required between USD 750 million and USD 1 billion in expenditure on control measures between 1989 and 2000. The damage to biodiversity is also very large because entire ecosystems are often impacted and changed by the intruder. Transportation of people and goods is the primary factor behind this phenomenon. Careful consideration would have to be given to determine the extent to which non-internalisation of the impacts of invasive species represent an indirect subsidy (and, therefore, a perverse incentive) to that industry.

3.5 Fisheries

As with many other sectors of the economy, the environmental impacts of the fisheries sector have become more acute as the technology for marine capture has become more efficient. These environmental impacts are now large enough that they are causing damage to biodiversity as well.

The forms of subsidies that are given to the industry include those described in Box 1.

Box 1 suggests that subsidies to fishing cover a broad range of transfers which are sometimes direct but other times subtle. They include tax benefits that result in over-capitalization of the fleet, as well as payments to fishers and free provision of infrastructure and services. Given the difficulties of measuring many of these channels, exhaustive studies are difficult to come by. Nevertheless, OECD, APEC and WTO provide data which suggests the subsidies are large. For example, the landed value of marine-capture fisheries was over USD 37 billion in 1996. Table 5 gives the proportions of that value which were given as transfers to fisheries. As is evident from the Table, substantial amounts of money are being given in assistance.

Box 1. Sample of transfers to fisheries in OECD countries

Direct Payments
Price support payments to fishers, grants for new vessels, grants for modernisation, vessel, income support, unemployment insurance, retirement grants for fisheries, compensation for closed or reduced seasons, compensation for damage from predators on fish stocks, disaster relief payments, grants to purchase second hand vessels, grants for temporary withdrawal of fishing vessels, grants to small fisheries, direct aid to participants in particular fisheries, price support payments, income guarantee compensation.

Cost-reducing transfers
Fuel tax exemptions, subsidised loans for vessel construction, subsidised loans for vessel modernisation payments to reduce accounting costs, provision of bait services, loan guarantees, underwriting of insurance costs, interest rebates, income tax deduction for fishers, transport subsidies.

General services
Research expenditure, management expenditure, enforcement expenditure, market intervention schemes, support to build port facilities for commercial fishers, expenditure on the protection of marine areas, expenditure on conservation and management.

Source: OECD (2000), more detail is given therein.

Table 5. Transfers to marine-capture fisheries (1996)

Country	Total transfers relative to total landed value (%)
Australia	7
Canada	46
Belgium	5
Denmark	16
Finland	92
France	19
Germany	40
Greece	13
Ireland	59
Italy	8
Netherlands	8
Portugal	21
Spain	8
Sweden	44
UK	12
Iceland	5
Japan	23
Korea	7
Mexico	1
New Zealand	3
Norway	13
Poland	4
Turkey	14
United States of America	24

Note: transfers include revenue-enhancing direct payments, cost-reducing transfers and general services.

Source: Cox (2002).

Given the inherent "problem of the commons" of a common-pool resource, one would expect over-fishing to occur in the absence of government intervention to protect the collective good. The impact, therefore, of the assistance to fisheries is difficult to distinguish from other sources of influence — they mostly operate through the same channels. The problem of the commons arises because each individual fisher fails to account for the impact of his/her fishing activity on others. The return to fishing is therefore overstated. With government assistance, either in the form of pricing-support policies or cost-reducing measures, the return to fishing is also artificially high.

On the other hand, high prices for fish and fish-products are likely to be reflective of a problem with the resource rather than a cause of it. That is, when scarcity increases, there will be an increase in price to reflect the conditions of demand and the ability of suppliers to meet it. Higher prices for specific species will reflect the fact that fishers are spending more time and effort in catching that species. Since (as mentioned above) the return that each individual fisher receives will always be greater than that which accounts for the impact on other fishers, the incentive to over-fish will remain until the price becomes exorbitantly high.[9] Moreover, if fish populations exhibit non-linearities in the impacts of continued population decline, permanent damage may occur which markets may be unable to avoid — government intervention would clearly be called for.

This is particularly the case in areas outside the 200-mile economic exclusion zones: intervention (though possible) is challenging because governments must look beyond short-term interests. The Convention on the Law of the Sea (1994) alleviated this problem, but it did not entirely eliminate it. Moreover, in areas within the exclusion zone, subsidies can aggravate problems and even lead to the severe depletion of major fisheries (such as the Altantic Cod, Mayo and O'Brien, 2000). FAO (1995) noted that, in the mid-1990's, 69 per cent of the world's marine fish resources were either fully- or over-exploited. This has a substantial impact on biodiversity through the reduction of species population — causing other species to also respond. Over-exploitation reduces the gene pool and alters ecological relationships with predators, symbionts, competitors and prey (Norse, 1993). The reduction or removal of one species shifts the balance of the ecosystem and impacts its resilience to shocks. Moreover, marine communities seem to exhibit multiple equilibria with different characteristics. In other words, when a system is disturbed strongly enough, the impact may be irreversible — some species currently over-exploited may have lost their ecological niche and, therefore, may never be able to recover.

One area where subsidies are known to have a large impact is in the way in which the industry catches fish. Some observers find that the capital used by fishers changes substantially in response to incentives thrown up by policy (e.g. Flaaten and Wallis, 2000). Perhaps the most serious impact of fishing on biodiversity is through by-catches that are discarded (lowering species population) and bottom trawling that damages seabed habitat (Waitling and Norse, 1998; Collie and Russo, 2000). These can be aggravated by subsidies when non- or marginally-profitable fishing is sustained by them.

Except for a few notable exceptions, most of the world's fisheries do not have well-defined property rights that cause fishers to internalize either the long-term consequences of over-fishing or the externalities caused by fishing techniques. In the cases where property rights have been established through the provision of fishing licenses by government, there can be an implicit subsidy given to fishers if no account is made of the damage to biodiversity or the broader environment from fishing (Milazzo, 1998). Since subsidies affect the way in which the industry catches fish, once a capacity is in place for one type of fishery, it may have long-term consequences. Munro and Sumaila (2001) suggest that there are significant

9. How high the price can go will depend on when the point may be reached where the number of remaining fishers is small enough to make co-operation feasible — co-operation would allow them to internalise all impacts on the species.

obstacles to moving fishing equipment to other fishing activity — implying that it is difficult to move to other fish stocks that are not under strain.

3.6 Forests

Though experiences vary considerably across OECD countries, the most frequently used and major sources of assistance given to the forestry sector include:

- Under-charging of resource rents.
- Roads and infrastructure provision.
- Export restrictions on raw logs.

The standard Faustmann model, widely used in forestry economics, holds that the optimal rotation period — that is, the economically optimal age of harvesting an even-growth forest — is shorter when exploitation costs are lower and/or revenues are higher. Thus, *ceteris paribus*, direct or indirect subsidies to forestry tend to reduce the optimal time to cut a forest — which has a negative impact on biodiversity-rich older forests. Some, however, have argued that while the first category causes wood products to be under-priced, it does not constitute a major source of unsustainable management of forest resources (Ruzicka and Moura Costa, 1997). That is, even when resource rents are high, the use of the forest may be unsustainable — other elements of the management plan are also important. On the other hand, studies that look at the rent paid to resources in some auctions finds very low prices per hectare of forest (Hardner and Rice, 2002). The implication being that the choice between conservation of a forested area and harvesting is made in favour of the latter on the basis of very little gain. This latter study suggests that current logging practices in some areas put a very low value on biodiversity — at the prices quoted in the study, all the forests of Madagascar (a biodiversity "hot spot") could be preserved for less than EUR 50 million. Low resource rents (where they occur) may not be the only cause of unsustainable forestry use but reversing them would certainly be an important element of policies to achieve long-term forestry management objectives.

The provision of roads and infrastructure can be a crucial factor in forest use and the preservation of forest ecosystems. Rodgers (1997), for example, points out that, following a period of sustainable forestry management in Cote d'Ivoire, large-scale forest loss began to occur when roads were built that permitted easy access to densely forested areas. Given poorly defined property rights in those newly accessible areas, the outcome was predictable. As a result, 79 per cent of the forested areas have now been cleared; moreover, they have been left in a state that makes it difficult for the forest to regenerate itself. The provision of *subsidized* roads appears to have been the key factor that made the difference between a gradual harvest over a long period of time, and a rapid cutting down of the forest. The underlying implication is that when the forestry companies faced the full cost of getting the wood, its use was more sustainable. Succinctly, even if the impact on biodiversity is put aside, the value of wood products to society was lower than the full cost of producing those products. Unless a longer term goal was underpinning the activity, cutting down the forests was welfare reducing.

The final category of assistance to the forestry industry — restrictions on the export of logs — would not, in general, be expected to have an impact on biodiversity loss. The reason for this is that such restrictions serve primarily to allocate economic rents to the domestic wood-processing industry (Dean, 1995). Indeed, to the extent they result in higher domestic prices than would otherwise prevail, they could be a source of conservation of biodiversity.[10] Karsenty (2002), however, argues that this indirect support to domestic wood processing sectors, in addition to direct support measures, can lead to the building of excess capacity. This then creates increased pressure on the resource (including illegal logging which has been documented in Indonesia and Ghana)

The assistance given to the forestry industry that impacts on biodiversity is difficult to quantify given that the most important component involves measuring what the fees for logging *should have been,* as opposed to what was paid. In an interesting analysis of the difference between sustainable and non-sustainable forestry management Howard (1997) examines the opportunity cost of ecosystem management in the Pacific Northwest of the U.S. The study looks at the harvest value of forest resources and compares that to its economic value if 100 or 200 year harvest cycles were used. These longer cycles would reflect the need to maintain sufficient old-growth forest so that species dependent on it would not be threatened. The results found a substantial difference between management schemes, suggesting that if species-preservation is a socially desirable objective, then the implicit subsidy given to industry is large. A straightforward interpretation of this result suggests that stumpage fees are failing to reflect the social opportunity costs of the harvested wood.

Finally, it should be noted that important impacts on biodiversity loss in the forestry sector come not from the harvesting of trees but from the clearing of forests to extend agriculture. In many countries there are incentives in place to encourage such activity (Kaimowitz, 1995). Dealing with biodiversity loss from lost forests, therefore, requires adjusting the policies that have been put in place for sectors other than forestry.

4. Removing perverse incentives

These observations regarding incentives illustrate that the level of subsidies is large and the potential impact is significant. This is, in other words, an important policy issues for both the direct impact it has on welfare through the economy, as well as for its indirect impact on welfare through non-marketed impacts. Closer examination of incentives (i.e. subsidy measures) would seem warranted to ensure they are not causing perverse impacts on biodiversity.

Given the inter-connectedness of the economy that was observed earlier, however, some effort needs to be made to distinguish between incentive measures that are unambiguously harmful and those that provide some benefit. While many subsidies are harmful to both environment and economy, the positive effects cited in some studies (e.g. Fullerton and Mohr, 2003; Varangu, 2002) call for a careful consideration. Moreover, in many cases it may be possible to substitute a biodiversity-harmful subsidy for one that achieves the same social objective without the harm. It should be remembered that, from a purely theoretical perspective, since all policy instruments distort market outcomes (with the objective of improving overall market performance), when more than one is available, optimality would call for using each instrument until the marginal welfare loss is equal between them. Part of that consideration should be the extent to which subsidies are prone to over-use.

10. Prices *may* rise for two reasons. First, the demand for wood-products will remain unchanged. If domestic wood-processing is not as efficient as its foreign counterpart, then the supply curve will shift upwards and prices will rise. Second, when the restriction is effective in reducing demand, scale economies caused by fixed costs in moving logs (e.g. road building and capital equipment) to mills may create discontinuous supply curves that shift downwards at various increases in output. When the domestic market is considerably smaller than the export market, restrictions may thus lead to higher prices. Moreover, those scale economies may create a highly concentrated domestic market whose firm(s) are able to price on the demand curve.

The removal of subsidies has already occurred in a number of industries where the results have demonstrated considerable benefits for government finances as well as some tentative environmental gains. New Zealand, for example, removed many of the supports it was providing to the private sector across a wide range of industries. In sectors like forestry, responsibility for activities such as tree planting was given to the industry. Ten years after that change, more trees were being planted by the private sector than both public and private sectors had planted earlier (Rhodes and Novis, 2002). Other industries also successfully made the transition from receiving assistance to having to be self-sufficient.

Reforming biodiversity-harmful incentives, however, requires effort to ensure that outcomes maximize social benefits. Currently, quantifying incentives remains incomplete with a considerable amount of work yet to be done. In many cases that data are simply unavailable, while in others they are not comparable across sectors of the economy (Steenblik, 2003). In OECD economies, many incentive measures are explicit and, therefore, in principle, quantifiable. On the other hand, in less developed economies the measures are less obvious and dominated by cases where external effects on the environment are not accounted for in private decisions. Nonetheless, in cases in those economies where incentives are explicit, either through under-pricing of resources or through direct payments, they have considerable impacts on biodiversity.

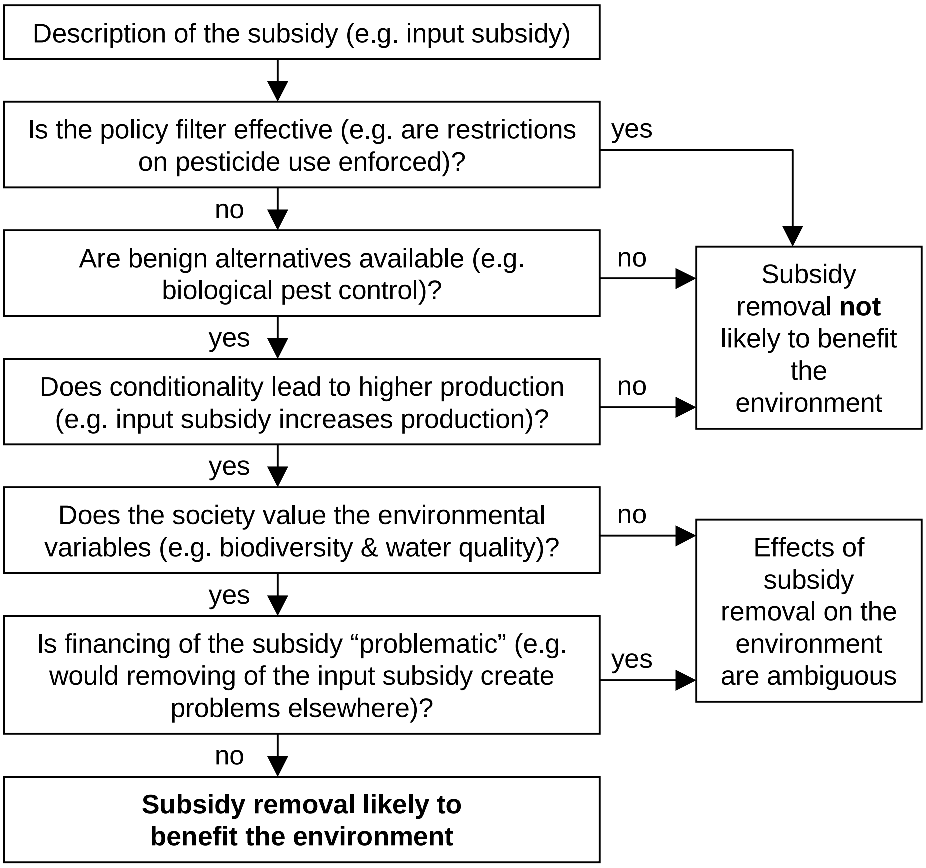
The complexity of interactions in the economy, however, calls for the use of numerical tools that are capable of illustrating the wide range of repercussions of policy initiatives. The tools that would be most useful would be those that could incorporate many of the existing sources of distortion in the economy so that analysis could explore policy in a “second-best” context. The OECD has in the past used models for a number of analyses that benefited from the use of quantitative tools (e.g. OECD, 2001a; Burniaux, 2000). Future work on the removal of subsidies would certainly proceed on a firmer footing if such tools could be used to help guide the selection of subsidies for reform or removal.

A useful starting point, however, to examining incentives for potential removal would be the “Checklist” that was suggested by Pieters (2003) for exploring which subsidies do the most damage and are most easily removed. The Checklist is intended to identify significant instances of environmentally harmful subsidies. There are a series of questions for ranking the options for subsidy removal according to their possible environmental harm. It explores the link between subsidies and the regulatory and resource management frameworks already in place. It then tests whether the subsidy operates in a way that leads to an increase in production processes with negative environmental impacts. Finally, it leads to an assessment of whether the impacts are unavoidable, or if other measures could mitigate the harmful effects.

This Checklist is being actively developed and remains under review.

Governments create incentives that impact on the economy for many reasons. Sometimes incentives are justified on the ground that other distortions are impeding economic development, while at other times they simply reflect outcomes of consensus building for other policies. Whatever the reason, implementing the chosen incentive measures presumably reflected a considered choice of options (based on available information) *at the time they were implemented*. However, since governments have limited resources that must be allocated to changing priorities and changing circumstances, occasional review of incentive measures is warranted to ensure that the goals remain important and, as well, that they are being met.

Figure 1. "Checklist" to determine if subsidies are environmentally harmful



Source: Pieters (2003).

More importantly for the environment, and specifically for biodiversity, is the fact that often the full impacts of incentive measures are not known at the time of implementation. In that context, occasional review is imperative to ensure that the unforeseen costs to biodiversity become internalised into the public policy discussion. Efficient use of natural resources (a goal of the OECD *Environmental Strategy*) calls for tradeoffs between environmental amenities and other goods and services to be made explicit in public policy so that rational choices can be made. Only by fully accounting for the impacts of policies, and by including both direct and indirect costs (e.g. externalities) can those choices be made in the best interest of everyone.

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ANNEX I: THE INCIDENCE OF SUBSIDIES

Incentives in the form of subsidies can be given by many means. Their impact, both direct and indirect, will depend on a number of factors — including many that may not have been accounted for at the time the measure was implemented. Incentives will be particularly harmful to the environment in situations where they directly encourage overuse of environmental amenities. In other circumstance they may be harmful to the environment by causing the artificial expansion of industries that make heavier use of the environment as input. This Annex explores some issues related to the incidence of incentive measures (specifically subsidies) and the underlying conditions that contribute to their impact on biodiversity. To begin, consider the following Figures.

Figure 1

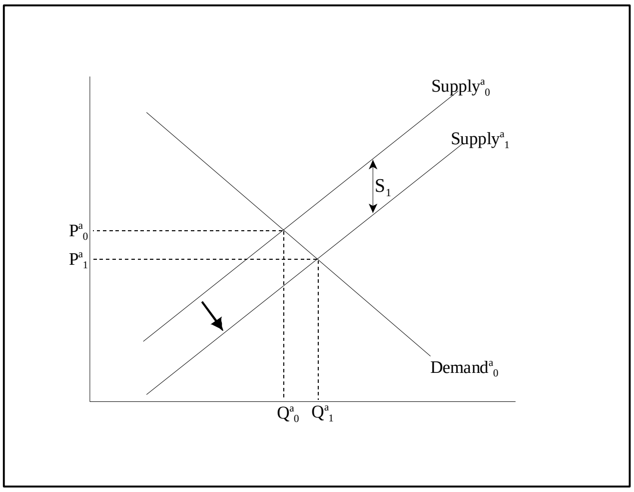


Figure 2

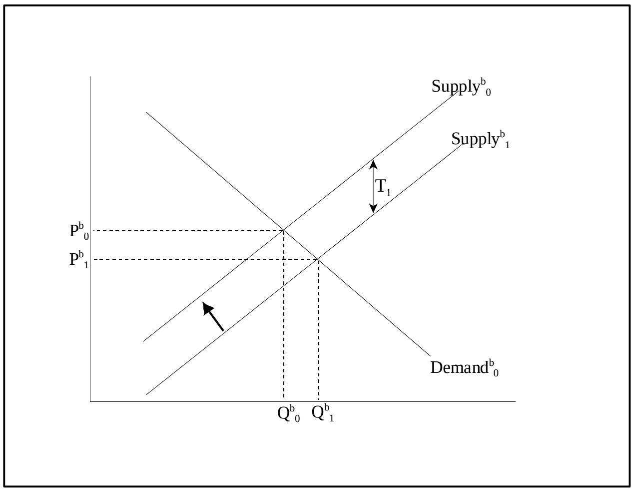


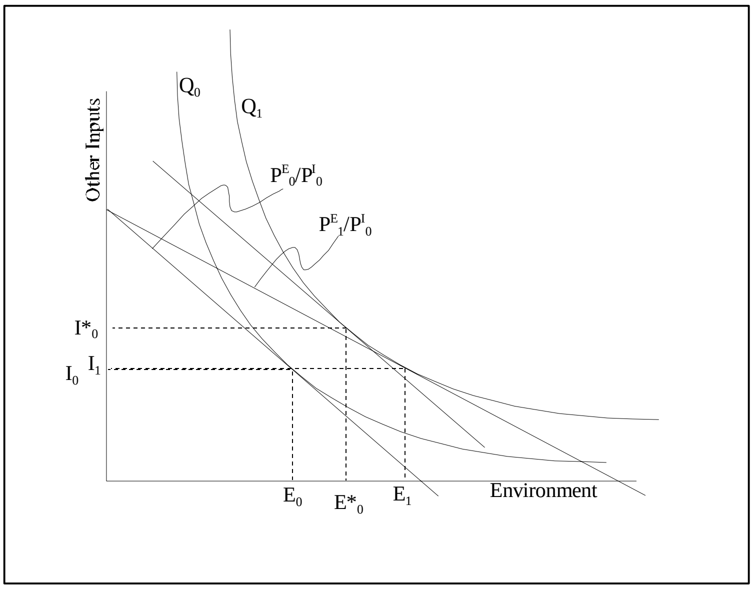
Figure 1 shows a situation where a subsidy in the amount of S_1 is given to firms for each unit of output that is produced. Initially the industry is producing at level $Q^a{}_0$ with the price at $P^a{}_0$. When the government provides the subsidy per unit of output, industry's output rises to $Q^a{}_1$, which is priced at the lower level of $P^a{}_1$. The increase of output requires a greater input of environmental amenities — the use of which would affect biodiversity; e.g. price protection (an implicit subsidy) for rice in the U.S. has led to excessive rates of water extraction in California, causing salinization downstream. Notice also that government policy that had not accounted for this expansion in output would have anticipated expenditures of roughly S_1 for each unit at $Q^a{}_0$ but instead it will have to pay for the larger volume at $Q^a{}_1$.

Figure 1 also helps to illustrate an issue mentioned earlier: taxes and subsidies can not easily offset each other, so subsidies generally lead to welfare loss unless they are correcting a source of market imperfection. To see why consider Figure 2, which replicates Figure 1 *for an alternative good b*, but assumes that a tax T_1 (which is equal to S_1) was imposed. For convenience, quantities and prices in the two markets are similar (i.e. $Q^a{}_i = Q^b{}_i$ and $P^a{}_i = P^b{}_i$). In the case of figure 2, the tax T_1 leads to output moving from $Q^b{}_1$ to $Q^b{}_0$. Notice that a government that imposes a tax T_1 in one market to pay for subsidy S_1 in another market will find itself short of revenue since $Q^a{}_1 > Q^b{}_0$. In the case of the two markets illustrated, the welfare *loss* caused by the tax is exactly offset by the welfare *gain* in the other (i.e. there is no net deadweight loss). However, since the government is short of revenue, either the tax will have to increase, or a third market will have to be taxed. In either case, the additional tax will create a welfare-reducing deadweight loss. In other words, a subsidy predisposes an economy to incur a welfare loss.[1]

Another interesting case that was discussed in the paper is where the government provides suppliers a fixed payment that is unrelated to production levels or price. Since the supply and demand curves relate price to quantity, there is no change in either curve in response to this policy. That is, there is no distorting impact on the market. In other words, social policy that is intended to support the livelihood of farmers would be less harmful to the environment and distorting to the economy if it simply transferred the money to farmers. This observation underlies some of the recent reforms of the European Common Agricultural Policy, which followed from negotiated outcomes for the Uruguay Round Agricultural Agreement.

In other cases where a subsidy is provided which is directly related to an input that has a close link to biodiversity, the impact may be more severe. Take, for example, the case illustrated in Figure 3 where the biodiversity-related environmental input is labeled E.

Figure 3



As in Figure 1, the industry is producing at output level Q_0 (illustrated as the lower isoquant[2]) where environmental amenities are being used at level E_0, and other inputs are being used at the level I_0. The subsidy that in Figure 1 caused output to move to Q_1 is also illustrated here. In this case it would cause the environmental input to be used at E^*_0 and other inputs at I^*_0. The relative price of those inputs remains unchanged so in that case there is a simple shifting of the price line P^E_0/ P^I_0.

1 Since the possibility exists of finding a market to tax where demand is highly inelastic, it may be possible that the tax induces a welfare loss that is smaller than the gain from the subsidy. For this reason, it is not possible to state that the subsidy unambiguously induces a welfare loss.

2. An isoquant shows all combinations of inputs (in this case *E* and *I*) that can produce a fixed quantity of output (in the Figure, either Q_0 or Q_1)

On the other hand, a subsidy given for the environmental input will cause relative prices to change and the line to tilt as firms see the cost of the environmental input decrease. For convenience, the subsidy in this case is illustrated so that it results in the same quantity of output. The new line will be $P^E{}_1/ P^I{}_0$ which will make contact with the isoquant Q_1 where the environmental input is E_1 and other inputs are at I_1. Figure 3 makes clear that the overuse of the environmental input in this case is more severe — impacting biodiversity more strongly.

The primary motivation for interest in subsidies from the perspective of economic policy is that they can result in reduced welfare. This is illustrated in Figure 4. The Figure shows the tradeoff in an economy of two goods where initially there is no subsidy (the economy is at X_0) followed by the introduction of a subsidy for good B (and a tax on good A to pay for it). The subsidy/tax combination cause production to move to X_1 which intersects a lower utility curve (not shown), implying a reduced level of welfare. Clearly for public policy this is not a desirable outcome and the subsidy should be removed.

Figure 4

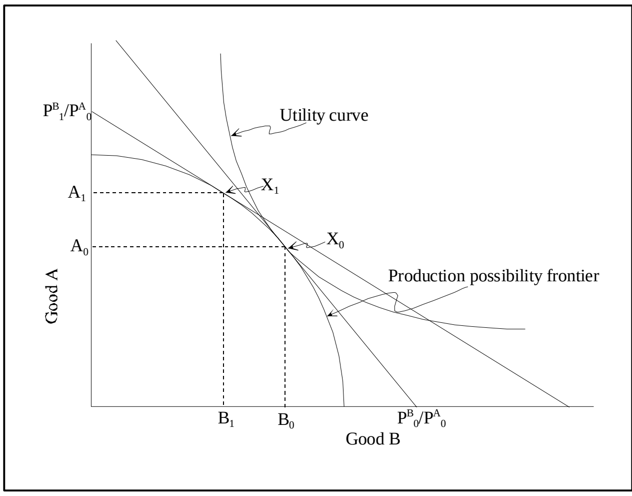
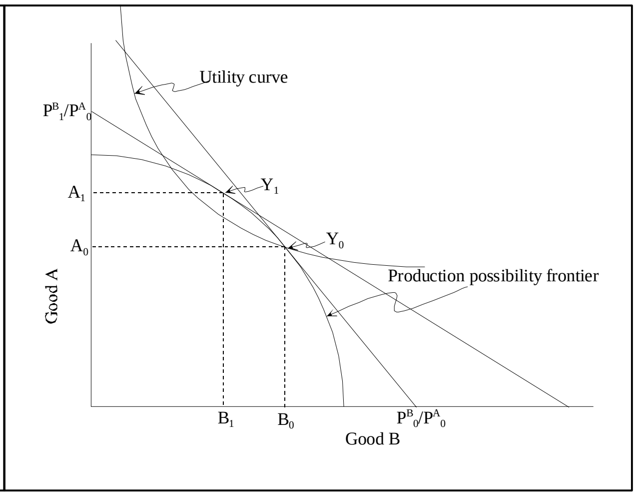


Figure 5



On the other hand, Figure 5 shows a similar situation where applying the subsidy/tax results in a net welfare gain. That is, at the relative prices which cause the economy to be at Y_0, there is a sub-optimal level of production occurring. This situation could occur when: either good B has a negative externality associated with it, or good A engenders a positive externality. In either case, the **market price** of the goods would not reflect their true (social) values. If the conditions for a Coasian solution to this situation do not exist (Coase, 1960), then policy intervention is warranted to improve economic outcomes.

In the general equilibrium of a national economy it is unlikely that there is a single good whose social value is different from its price in the market. Many goods are characterised by some level of either non-excludability or non-rivalry in use (i.e. they are, to some degree, **public goods**), making them potential sources of market imperfection. Moreover, the conditions for the market to correct the problem on its own are likely to vary between each situation so that some goods may already have partial corrections to their market-external characteristics while others may not. Policy that fails to account for these existing corrections would *itself* be a source of distortion from the social optimum.

It should also be noted that taxes imposed to sustain the subsidy can result in strong distortions which may be disproportionate in their impacts. The simple illustrations shown above give static examples of subsidies that cause, or correct, market imperfections. In dynamic models where human, or physical, capital are important for endogenous economic growth, taxes that are badly implemented can result in much more severe consequences by impacting growth (see OECD, 2003, for a survey).

Finally, it may be useful to illustrate the relevance of indirect consequences as well as their analytical challenge with a brief example. Consider the impact of a subsidy on a target sector. As was already outlined, the overall effects will depend both on demand and supply elasticities within that sector (OECD, 1998) as well as those in "related" sectors. This can be made concrete by considering an economy where private houses are predominantly built with wood. Take a simple case where at least 30 per cent of the material that goes into building a house is wood products. Suppose the government introduces a program that subsidises home ownership. For many people the impact will be to buy a larger home than they otherwise would since the potential return to housing capital will increase (Gervais, 1998). If houses become 10 per cent larger, then the impact could be to increase the use of wood for houses by 3 per cent, or to increase the price of wood (or both). The link to biodiversity is indirect but potentially significant since, over time, this change alone can cause the housing stock to raise the *annual* demand for wood products.

The upshot is that subsidising (by giving favourable tax treatment) a sector that has little apparent impact on biodiversity nonetheless leads to environmental harm through related industries.[3] A subsidy whose goals may have been laudable (housing for lower income groups) produces incentives that are unintentionally harmful for the environment when it is not very well targeted (it could have been made exclusive to the target group). Policies, therefore, have potentially substantial impacts on environmental inputs through multiple channels.

3. Furthermore, the larger house may require more electricity and other inputs on a day to day basis, again causing either the demand or price for those inputs to increase — with potential environmental (biodiversity) impacts depending on how those inputs are produced.