OECD Trade Policy Studies: Illegal Trade
in Environmentally Sensitive Goods


# *Chapter 5*

# The effect of domestic environmental policies on illegal trade

*This chapter assesses the role of domestic environmental policies on illegal trade in environmentally sensitive goods. The focus is on incentive-based mechanisms. The first section looks at the role of property rights regimes for resource management. It is followed by a review of selected taxes and charges related to pollutants and waste. In the third and final section, the case of economic incentives targeting trade flows directly is assessed, with a case study on the timber trade.*

## Introduction

In the previous chapter we focussed on measures which targeted trade flows through international policy mechanisms. However, as has been noted, policies introduced at the national level can have implications for illegal trade in environmentally sensitive goods.

The effects of national regulatory regimes on illegal trade depend largely upon the incentives for sustainable (or unsustainable) management of the resource or pollutant generated by the policy. A regulatory system which imposes costs on those exploiting the resource or emitting the pollutant will generate price differentials, which can provide incentives for non-compliance, with some of the output entering into international trade flows. This is, of course, a function of national enforcement capacity, supported in some cases by the international licensing schemes discussed above.

In recent years increased interest has been expressed in using economic incentives in the pursuit of environmental objectives, for example, to reduce pollution, protect biodiversity and habitats and promote the sustainable use of natural resources.[1] Such an approach is in contrast, or complementary, to more traditional command-and-control regulatory approaches. This chapter focuses on the effect of such measures on illegal trade.

While the evidence is scant there is some reason to suppose that the use of economic incentives at the national level may reduce illegal trade flows. On the one hand, some of the revenue generated by economic instruments (*i.e.* environmental taxes) can be used to reinforce enforcement capacity. On the other, the 'formalisation' of property rights implicit with the use of economic instruments can provide incentives for a longer-term view of resource management, and can even provide incentives for self-enforcement among those exploiting the resource.

Unfortunately, the possible impact of economic incentives on illegal trade has received little attention in the literature. This chapter attempts to explore this question in more depth in three sections. The first section looks at the role of property rights regimes for resource management. It is followed by a review of selected taxes and charges related to pollutants and waste. In the third and final section, the case of economic incentives targeting trade flows directly is assessed, with a case study on the timber trade.

## The establishment of property rights over environmental resources

One problem common to much environmental policy is the lack of clear property rights; although atmospheric pollution or the destruction of habitats, say, have clear negative impacts on human welfare, property rights – who "owns" the atmosphere or the habitats – are generally not well-defined. Establishing clear property rights and systems of governance can have benefits for environmental outcomes.

Indeed, the establishment of property rights is generally an essential precondition for any system of economic incentives to work effectively, as without it benefits and costs cannot be assigned to individual economic actors. Property rights are often particularly ill-defined or poorly protected in developing countries, and simply establishing and enforcing them may have similar effects to using economic incentives, creating a stable and predictable structure of costs and benefits.

The case studies below illustrate how property-rights-based approaches can be used to improve environmental outcomes and reduce illegal behaviour, with implications for trade flows. It is important to note, however, that in two of the cases discussed property rights are not vested in the individual, but rather in a broader community of resource users.[2]

### *Peruvian vicuña*[3]

Efforts to protect the vicuña – a small, doe-like Andean camelid which produces the finest quality wool in the world – from poaching provide a good example of how addressing the incentives behind environmental crime through better governance can yield dramatic results. They are all the more striking for being successfully driven by a relatively poor developing country, Peru, without much help from richer consumer states.

In the 1950s and 1960s, poaching for its wool drove the vicuña to the brink of extinction. It was listed as "endangered" in the 1970s International Union for the Conservation of Nature (IUCN) Red List, with a population of less than 10 000 animals remaining, most of which lived in Peru. Despite a ban on hunting, a large wool-processing industry remained, centred in Bolivia, and the wool continued to be widely available in luxury salons across the world. Use of an international trade ban, under an Appendix I listing in CITES, was essential to address this problem. However, the ban failed to stem the general decline of the vicuña because the animal was still under pressure due to competition for forage with alpaca herds owned by local *campesinos* (small, mainly subsistence, farmers) in the Andes.

A semi-autonomous unit created by the Peruvian government and tasked with coming up with solutions decided to rehabilitate the international market for the vicuña's wool. There were six key elements of their approach:

- The Peruvian government ceded the right to wool shorn from live vicuñas (and live animals only) to local *campesino* organisations, and revived the ancient Inca tradition of the *Chaccu*, whereby the local community encircls all the vicuñas in an area and shears them one by one. This gave local *campesinos* an interest in the animal's conservation since a live vicuña came to be worth at least five times the value of a dead poached one. Because only partial rights were transferred (to the wool from live animals), the incentive to cull vicuña herds was removed.
- The government maintained a monopoly right to buy this wool (for a guaranteed price) at the time of the *Chaccu*. The government thereby became the international market-maker for vicuña wool.
- Monopoly control over international trade was granted to a single international trading and processing cartel to maximise the exclusivity of the resulting products.
- All processed products were certified with a unique identification label.
- A clever use of the "stricter domestic measures" provisions under Article XIV of CITES imposed a double-check procedure on exports from Peru and assisted in policing the trade.
- As well as international trade co-ordination through CITES, in 1969, vicuña range states agreed the Convention for the Conservation of the Vicuña to co-ordinate all their conservation and market interventions. This also allowed Peru to provide focused technical assistance to other range states (especially Bolivia).

These controls represent a coherent attempt to govern both supply and demand pressures and to align the incentives of the various actors involved. *Campesinos* came to see the live animal as more of an asset than a competitor over resources. Meanwhile, the trade cartel had an interest in policing the international marketplace and preserving the exclusivity of their product. The government also built a series of double-checks and safeguards, enabling it to cross-check wool production and trade flows.

Although there have been some challenges along the way[4], the results speak for themselves: in 2008 the vicuña population reached almost 350 000 animals and the IUCN Red List reclassified its status as of "least concern".[5]

### *The CAMPFIRE initiative in Zimbabwe*

The Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) was initiated in Zimbabwe in the late 1980s. Its aim was to give rural people management rights over wildlife so that they would have an incentive to ensure its sustainable use, and so help to prevent poaching.

The establishment of national parks, game reserves, and safari areas in Zimbabwe (then Rhodesia) in the late 1920s may have helped avert biodiversity loss, but it also displaced rural communities from land that had been traditionally theirs (Fischer *et al.*, 2005). Cultivation and grazing land was expropriated, and subsistence hunting became illegal. Wildlife from the parks roamed freely in surrounding areas, destroying crops and threatening livestock and people. Central government owned the wildlife in trust for the country and reaped all the benefits, by selling licenses for hunting (forbidden in national parks, but allowed to a limited extent in safari areas) and charging fees for wildlife services such as tourism. Illegal poaching became a major problem, and since wildlife posed a nuisance, local communities would often turn a blind eye or even collaborate with the poachers.

CAMPFIRE focused on communal areas adjacent to the national parks, where wildlife intrusion was most severe. Communities were given co-ownership, with local councils, of the natural resources, and these provided the basis for a variety of income-generating activities, including trophy hunting concessions, natural-resource harvesting, tourism, live-animal sales, and raising animals for meat. District councils were responsible for management strategies and received any resulting income, although the intention was that they would devolve decision-making and benefits to the communities which they represented. Standard practice was that 50% of revenues were kept by the managing authority and 50% allocated to communities (Shyamsundar *et al.*, 2005). Communities decided themselves on how their share of the profits was to be used, whether for community projects or as direct payments to households.

The first CAMPFIRE project was established in 1988, and the initiative spread rapidly, with projects covering 75 wards within four years. Early projects focused on sport hunting of large mammals, but in the 1990s eco-tourism initiatives were also set up. As communities started to reap economic benefits from the legal use of wildlife they began to perceive game as a resource. Consequently, opposition to poaching increased, with public arrests of poachers by some communities and incidences fell drastically in some areas (Fischer *et al,* 2005).

CAMPFIRE was initially regarded as a success, and over USD 20 million was paid to participating communities between 1989 and 2001, 89% of which was generated by sport hunting (Frost and Bond, 2008). In some areas, the implementation of CAMPFIRE resulted in higher incomes for community members and improved resource management (Child, 1993). This was put down to effective decentralisation of decision-making powers to local communities, which meant that they instituted resource-management systems where previously there had been open access.

In other areas, CAMPFIRE had less of an impact. This was partly because of differences in the availability of natural resources, but also due to different approaches to decentralisation (Alexander and McGregor, 2000; McDermott Hughes, 2002). In some areas councils did not delegate responsibilities and decision-making powers to the communities, so that they felt alienated from the project. In the context of a long history of excluding people from the land, some communities considered CAMPFIRE to be just another initiative to undermine traditional tenure rights. Furthermore, some councils only shared a low proportion of the income generated with communities, and there were also conflicts between communities over the distribution of profits.

Another factor has been that some communities did not wish to become engaged in wildlife management – this being regarded as a backward way of life. Consequently, in parts of the country, CAMPFIRE projects were met with hostility or rejected, and poaching of wildlife and other natural resources continued, and in certain cases, increased. Presumably much of the resulting material entered into international trade, although no hard data on this are available.

Overall, however, the experience of CAMPFIRE in the 1990s was felt to be largely positive (Fischer *et al.*, 2005). Unfortunately, once the government's land-reform programme began in 2000, which included the seizing of game parks, CAMPFIRE fell into disarray. Coupled with the collapse of the tourism industry, and the general political instability, hunting and poaching became widespread.

### *Individual transferable fishing quotas*

In 2002 it was reported that over 15 countries had established market-based instruments for fisheries, and that these were being used to manage some 60 species (Newell *et al.*, 2002). Since then a large number of additional schemes have been introduced. Under such systems, a total allowable catch (TAC) is decided and this is divided up into individual fishing quotas. In many fisheries, these systems have brought benefits through improved incomes for fishers and more sustainable management of the resource.

However, if not enforced effectively these incentives may also encourage illegal behaviour. It has been estimated that the cost of IUU fishing may be as much as USD 10-23.5 billion a year (Agnew *et al.*, 2009). In addition to exceeding quotas, fisheries can face problems with poaching (harvesting by ineligible fishermen), unreported high-grading (discarding low-valued fish to make room for higher-valued fish) and the discard of by-catch (non-targeted species). These latter practices in particular are difficult to control as they take place at sea.

Since 37% of the global fish harvest enters into trade, it is likely that a significant proportion of the catch from this IUU fishing activity is traded (FAO, 2010). Moreover, there may be greater incentives for those fishers engaging in IUU fishing to land their catch in foreign ports if this reduces the likelihood of being subject to enforcement. Indeed, a number of countries (Chile, United States, the European Union) have used trade measures in an effort to reduce IUU fishing in distant waters (FAO, 2010).

These problems are endemic to all fisheries-management systems and an effective system of enforcement is therefore crucial no matter what. However, incentives for IUU fishing differ according to the instrument used for fisheries management. By increasing the profitability of the sector, ITQs can increase incentives for non-compliance (Tietenberg, 2003). On the other hand the revenue generated by the ITQ system can (at least in part) cover this. In many fisheries, for example, those in Australia, Canada, Iceland, and New Zealand, the fees levied on quota owners pay for administration and enforcement.

Perhaps more importantly, in some instances, implementation of quotas has in fact resulted in improved co-operation between industry and enforcement agencies and better compliance. This is because the fishermen recognise that illegal fishing undermines the resource base and so damages the value of their quota rights (Tietenberg, 2003). If the rights are seen to be secure by the rights-holders they have incentives to ensure that IUU fishing is minimised.

### *Conclusions*

The case studies analysed above are examples of rights-based natural-resource management, in which there has been considerable interest in recent years. They illustrate a number of reasons as to why this approach can affect illegal activities and trade:

- Poachers decide to become legal harvesters: those engaged in illegal activities may decide to stop because the new governance regime means that they have more to gain from participating in the sustainable management of resources.
- Improved monitoring and control of resources: this is often an important factor in reducing illegal activities. Better controls may be instituted by a community because they are gaining economically from the resource and so they have the incentive to monitor it; or another factor may be that the new regime gives them the powers to monitor and control their resources, when previously they lacked them.

- The conversion of poachers and IUU fishers into legal operators: in some cases, rights-based natural-resource management can facilitate the conversion of poachers or illegal loggers into legitimate harvesters and resource managers.
- In some circumstances, however, rights-based natural resource management may encourage new types of illegality or create new opportunities for illegal activities – for example, where poachers' attention switches from large mammals (the focus of the new regime) to smaller game, which can be caught with snares rather than firearms and is therefore harder to detect (Shyamsundar *et al.*, 2005).

Studies of other examples of community-based natural resource management lead to some further conclusions. The institutional arrangements for revenue distribution are generally key to determining the success of the approach. In one example of participatory forest management in Kenya, young men who had been active in illegal harvesting of timber poles and charcoal saw few of the benefits of the new management regime, and consequently often did not change their behaviour (Schreckenberg and Luttrell, 2009). In another example, in the case of a community forest in Mexico, unfair distribution of the benefits from forest use resulted in widespread illegal harvesting of timber. The forest was managed by an elite from the central village, with most of the profits being invested and jobs generated there; outlying settlements saw little benefit and also perceived corruption amongst the village leaders and so felt justified in illegally felling trees (Klooster, 2000).

There are, however, examples of successful initiatives in this area, as the case study of Peruvian vicuña management shows in particular. A number of factors seem to contribute to the success of such initiatives:

- Adequate levels of governance and law enforcement are necessary, so that the resource owners can be sure that their rights will be upheld.
- Security of rights in particular is important, especially in community forest management. In some situations people may be willing to forgo short-term benefits, or to pay short-term costs, if they believe that there will be longer-term benefits, but they must be assured that they will be able to enjoy them.
- Decision-making powers and benefits need to be decentralised to those who are engaged in managing the resource.
- Co-ordination with broader national and international efforts can reinforce local management regimes, such as the control of trade in the resource in question. Such co-ordination can also help to avoid displacement of illegal activities from one region or country to another.

- Economic incentives are not the only factors in decision-making: cultural and social factors are also important in the decisions people make about resource use, and these need to be borne in mind when designing interventions.

## Taxes, charges and payments for environmental resources

This section reviews case studies of the effects of taxes and charges. Such measures are used to incorporate environmental costs into economic decision-making; increasing the price of the products by applying taxes or charges is one way of incorporating these environmental externalities into the price of a product, and thereby discouraging the production and consumption of those products which cause environmental damage. This section includes cases-studies where such measures have been applied explicitly to reduce environmentally damaging behaviour, and seeks to identify possible implications for illegal trade.

### *The ODS tax in the United States*

In 1989, the US Congress adopted a law applying an excise tax to those ozone-depleting substances (ODS) controlled by the Montreal Protocol, which had just entered into force: chlorofluorocarbons (CFCs) and halons (Hoerner, 1996). Carbon tetrachloride and methyl chloroform were added in 1990 after the Protocol was extended to those chemicals. The tax was initially set at USD 1.37 per pound (USD 3 per kilogramme) for 1990 and 1991, and escalated in value every year thereafter; later, legislation increased the rates further but retained its annual escalating nature. The tax also varied by a factor representing the ozone-depleting potential (ODP) of the chemical.[6]

The tax applied to manufacturers' and importers' sale or use of the chemicals and was levied when the chemicals were first sold or used in manufacturing. Imports of products manufactured with or containing the chemicals were also subject to the tax, but exports of the same products qualified for tax rebates, so as not to damage industry's international competitiveness. The tax immediately doubled the price of CFC-11 and CFC-12, the two most commonly used ODS, and by 1995 the taxed price was nearly triple the untaxed price.

In line with its obligations under the Montreal Protocol, the United States also adopted overall caps on production and consumption, and achieved total phase-out of all the targeted ODS by the Protocol's target date of January 1996. It is difficult to identify the precise impact of the tax, given that allowable consumption limits were falling, and industry knew that a total phase-out was approaching. Some observers have argued that the tax contributed to a more rapid phase-out than would otherwise have been the case (Hoerner, 1996; Cook, 1998).

However, other developed countries succeeded in phasing out the use of the same chemicals just as quickly by using regulatory caps on production and consumption rather than taxes. In general, manufacturers found it easier and cheaper to phase out most uses of CFCs than had originally been foreseen, and the effectiveness of the excise tax compared with other measures in encouraging this cannot be determined definitively.

Illegal trade in CFCs was first detected in the United States in 1992, growing rapidly in the following years: by the mid-1990s it was estimated that CFCs were the second-most valuable commodity smuggled through Miami after cocaine (Brack, 1996). Estimates suggested that, in 1994, 20-40% of CFC-12 imported into the United States (9 000-18 000 tonnes) was illegal.

Although it is difficult to determine the drivers behind this illegal trade, it seems likely that the application of the excise tax was a factor. Black markets can be expected to develop where cost differentials between legal and illegal goods become significant and where enforcement is weak (the excise tax resulted in a near tripling of prices of CFCs 11 and 12). Indeed, one of the first indications of the extent of illegal trade in the United States was the failure of CFC prices quoted to retailers to rise in line with the tax increases. Weak enforcement also played a role. The network of small users of CFCs that characterised the US market – garages maintaining and repairing cars – represented a significant challenge for enforcement efforts.

However, these were not the only factors underlying the black market in the United States. In the early days of the ODS phase-out, it was not, in general, possible for CFC alternatives to be used simply as "drop-in" replacements in refrigeration or air-conditioning equipment; the systems themselves usually had to be replaced entirely. In 1995, the cost of replacing a vehicle's air-conditioning system was typically USD 200–300, but occasionally as much as USD 800 (Brack, 1996), whereas the cost of keeping the existing one topped up with CFCs was a few dollars a year. These costs alone created a powerful incentive for garages to source black-market CFCs in order to keep their customers' costs down.

The proximity of the United States to CFC markets in Latin America and the size of the CFC market within the United States were also factors that are likely to have encouraged the illegal trade. For example, Miami developed as a major entry point for illegal CFCs mainly because it is a common transit port for goods from Europe bound for Latin America, and it proved relatively easy to divert goods supposedly in transit into the domestic market. The market for CFCs in the United States was also far more extensive than that in other developed countries; in the early and mid-1990s, approximately 90% of US cars were fitted with air-conditioning systems which needed regular refilling with coolants, compared with about 10% in the EU.

Nevertheless, the fact that imports of illegal CFCs entailed tax evasion as well as other criminal behaviour gave the US authorities a powerful incentive to take action. Enforcement was slow to start, but it increased steadily; in the ten years to 2001, the US authorities seized 1 125 tonnes of CFCs, representing an estimated 11.5% of the total volume of these products entering illegally (this compares with an estimated seizure rate of 12-14% of narcotics, the area of illegal trade afforded the highest priority by enforcement agencies) (Montreal Protocol, 2002).

To conclude, it is difficult to disentangle the impacts of the US tax on illegal trade in CFCs. The price differential between legal and illegal products to which it contributed is likely to have encouraged the development of a black market in these products. However, as has been seen, there are several other possible reasons that could have contributed to the development of the black market here, including the challenges of enforcement. One lesson that can be learnt is that where a policy is likely to create incentives for a black market, enforcement measures should anticipate this and be strengthened from the outset.

### *Waste taxes and charges in the European Union*

Taxes and charges on waste disposal have been used in many countries to increase the incentives to reduce volumes of waste and to reuse and recycle products. Taxes are generally aimed at raising the costs of disposing waste through landfill. For instance, in the European Union, the aim of the 1999 Landfill Directive was to reduce landfilling through prioritising waste prevention, reuse, recycling and recovery. It set targets for progressively reducing the amount of biodegradable municipal waste landfilled up to 2016. Member states introduced a range of measures in response to this, including closure of landfills, increasing the costs of landfills, increasing incineration capacity and establishing separate collection of biodegradable waste (EEA, 2009a).

A study analysing the effectiveness of these measures concluded that landfill tax rates needed to be relatively high if they were to be effective, although public perceptions of the tax burden are also important (EEA, 2009a). Estonia, for example, has among the lowest landfill taxes in Europe. These stood at EUR 30-36 per tonne in 2004, compared with EUR 80-90 per tonne in Italy and Germany at the time. However, these had increased significantly since 1996 (by 700% in the decade to 2006, equivalent to an annual rate of increase of 23%). Therefore, the tax was perceived to be high and so it was effective, contributing to a drop in the amount of municipal waste being landfilled, from 90% in 2000 to 60% in 2006.

An increase in exports (legal and illegal) could also be expected to follow a rise in disposal costs and differences between countries. Exports of notifiable waste (mostly hazardous waste) from EU Member states increased fourfold between 1997 and 2004, mostly to other EU member states (EEA, 2009b).

A policy for diverting waste from landfills can only succeed, however, if the waste-management system is able to receive and manage the resulting waste flows. Thus, factors such as the existence of separate collection schemes and the system's recovery capacity also influence the effectiveness of the policy. If alternatives are not in place to manage the diverted waste flows, strict landfill policies can encourage illegal dumping and the export of untreated waste. For example, in Estonia, the closure of landfills resulted in an increase in illegal dumping because there were insufficient alternative means of waste collection.

In the United Kingdom, which has historically relied on landfill more than most other EU countries, a landfill tax was first introduced in 1996, applying to commercial, industrial and municipal waste. It was a weight-based tax, with different rates for inactive and active waste[7] (GBP 2 per tonne and GBP 7 per tonne respectively). A 2001 assessment found that the tax had reduced the amount of inactive waste going to landfill (largely due to increased reuse of construction and demolition waste), but the amount of active waste remained unchanged. There had also been an increase in illegal dumping (commonly known as "fly-tipping" in the UK) and in the misclassification of waste (as inactive rather than active) in order to reduce tax liabilities (Davies and Doble, 2004).

These findings led to recommendations for further increases to landfill taxes – which were then among the lowest in Europe – and for at least some of the revenue to be used to provide alternative waste-management options. Rates have subsequently been increased and now (FY 2010/11) stand at GBP 2.50 per tonne and GBP 48 per tonne for inactive and active waste, respectively; the rate for active waste is set to escalate by GBP 8 per year until at least FY 2014/15, when it will reach GBP 80 per tonne. The tax raised GBP 420 million in its first year of operation, and about GBP 1 billion in FY 2008/09. Some of the revenue raised has been allocated to various programmes to assist industry to reduce waste volumes. Since the introduction of the tax, the proportion of waste sent to landfills has fallen by around a third, accompanied by a similar increase in recycling (Seely, 2009).

It would be expected that the increase in the rates of landfill tax would lead to an increase in illegal disposal, as indeed was reported in 2001. Data were not collected systematically until 2004, and even these are not wholly reliable (for instance, there is no single definition of fly-tipping, so reported incidents vary widely in size; and data are only collected for fly-tipping on public land, not private). However, such information as is available does in fact suggest the reverse: in FY 2007/08, fly-tipping in England fell by 7.5% (Seely, 2009) and in FY 2008/09 by 9.3% (reaching 1.16 million incidents).[8]

There are a number of possible reasons for this. Over 60% of the reported fly-tipping incidents involved household waste, but householders do not pay landfill tax directly (their local authorities do) and so fly-tipping by householders will not have been influenced by the tax.[9] In addition, enforcement action has risen significantly; local authorities and the Environment Agency are devoting more attention to the issue than in previous years, and so this may account for the fall in the number of incidents.

Reported illegal shipments also grew: IMPEL (EU Network for the Implementation and Enforcement of Environmental Law) investigations and individual EU member state studies have suggested that as much as 85% of non-hazardous waste is shipped illegally or is non-compliant, whilst initial findings of the IMPEL Sea Port II project suggested a figure of around 40% (IMPEL, 2005). An investigation into the illegal shipment of waste among IMPEL member states (the EU plus Croatia, Macedonia, Norway and Turkey) found that the main drivers for the illegal trade were the high cost of treatment or disposal of waste, coupled with poor enforcement (IMPEL, 2005).

As with ODS, the application of waste-disposal taxes such as landfill taxes is likely to result in an increase in illegal disposal. Illegal disposal can be minimised if alternative means of disposal are provided, and industry and local authorities are assisted in learning to handle waste differently. Also important is ensuring that there is a sufficient level of enforcement. However, there are a number of other factors which also influence illegal behaviour, making it difficult to assess the precise impact of taxes. Thus, as countries increasingly limit the volume of waste going to landfills, the cost of this means of disposal would be expected to increase with or without taxes; and costs will also vary between countries, creating an incentive to export waste (legally and illegally).

### *Export taxes on timber*

There are various ways of intervening to affect a country's exports of timber. For instance at the international level, REDD+ can play a role in reducing illegal logging. A letter of intent signed between Norway and Indonesia committed the latter to develop its forest management enforcement capacity in return for $US 1 billion in support from Norway.[10]

More directly, export taxes or duties can be applied differentially to encourage particular categories and discourage others – for example, to discourage the export of logs and encourage the export of processed timber, thereby creating incentives for the domestic wood-processing industry. Alternatively, a government may attempt to achieve the same objective by a non-economic incentive such as banning the export of logs. The primary aim of such measures is typically not an environmental one, but they can have an impact on management practices and illegal trade. The following case study compares the experience with regulating exports through an economic incentive (export duties) and a non-economic incentive (trade ban).

In 2005 the Russian Federation announced its intention to revise its forest policy. The government's stated aims were to develop the domestic timber-processing industry in order to increase employment and encourage economic growth, and also to improve the productivity of Russian forests and reduce illegal harvesting. From 2006 export duties on logs were increased. For example, for coniferous roundwood and birch logs exceeding 15 cm in diameter export duties were set at a minimum of EUR 10 per cubic metre (20% of the export value) from July 2007; and in April 2008, the duty was raised to a minimum of EUR 15 (25% of the export value). In January 2009 it was to be raised further to a minimum EUR 50 (80% of the export value) (Karjalainen *et al.*, 2010; Sokolov, 2010), but in fact this last increase was postponed twice.

Although these export duties have served to reduce exports of raw timber, the output of processed timber within Russia has not risen significantly. Exports of roundwood from Russia were estimated to have fallen from a peak of just over 50 million cubic metres in 2006 to just over 20 million cubic metres in 2009 (Sokolov, 2010), although the economic recession was also partly responsible for this. Exports of sawn timber increased slightly in 2007, but fell thereafter while exports of plywood fell slightly throughout the period and exports of newsprint rose slightly. This failure to increase exports of processed timber is the result of a shortage of capacity within the Russian timber-processing industry, while the planned investment largely failed to materialise (except for some Chinese investment, as outlined below).

Russian roundwood is exported predominantly to two countries, Finland (25% of exports in 2007) and China (55% in 2007). In Finland, the immediate impact of this policy was a decline in the competitiveness of Russian timber and a resulting fall in output of the Finnish processing industry, though the recession also played a role in this. Log exports to China fell similarly, but sawn timber exports began to rise. The response from China's industry was to invest heavily in timber processing facilities in Russia, and this resulted in a 13% increase in the volume of timber processed in Chinese-owned facilities in 2005-06 (Hongfan Li, 2007).

The actual impact on illegal activities is not known. Illegal logging in Russia was estimated at about 18% of the total timber harvest in 2004 (Seneca Creek, 2004), and is probably higher in the Russian Far East (a region which mainly exports to China) than in other Russian regions. A reduction in total trade from Russia should result in lower exports of illegal timber, but whether it is having any impact on illegal behaviour within the country is not yet known. If the trade simply switches from logs to sawn timber (as it may do in the Far East, given Chinese investment), then it may make no difference at all.

The Russian example can be compared with a regulatory instrument applied by the Indonesian government for similar goals. In late 2001, a complete ban was imposed on the export of logs, the government stating that its main goal was to aid law enforcement and reduce illegal logging and timber smuggling; at the time the country was suffering from a very high rate of illegal logging, probably of about 70-80%. Support for the processing industry, particularly through increasing the supply of legal logs to plymills, were also objectives[11] (Resosudarmo and Yusuf, 2006).

Trade data showed that illegal exports were significant before the ban was implemented; China reported importing up to 1 million cubic metres of logs from Indonesia in 2001 which were not reported as having been legally exported (Chatham House, 2010). Instead of halting the smuggling, however, the ban served only to change the methods used by the smugglers. This is demonstrated by the fact that while the discrepancy in reported log trade volumes between China and Indonesia rapidly declined after the ban, over the same period trade between China and Malaysia rapidly increased. Traders were either smuggling logs into Malaysia for export from this country or were simply declaring illegal Indonesian logs as Malaysian on arrival in China (EIA/Telapak, 2005). However, a major enforcement operation in Indonesia in 2005 led to a sharp fall in illegal log exports; trade data suggested the level fell to about 120 000 cubic metres a year (Chatham House, 2010).

In late 2004, three years after banning exports of logs, the Indonesian government also banned the export of most forms of sawn timber, again with the aim of reducing illegal logging and trade. As for log exports, discrepancies in trade data prior to the ban suggested that large volumes of illegally exported Indonesian sawn wood were being imported by various countries, including China. Trade data indicated that, after an initial lag, illegal exports fell steadily from 1.6 million cubic metres in 2004 to less than 0.4 million cubic metres in 2008 (Chatham House, 2010). However, improved enforcement since 2005 will have also played a role in this outcome.

As is common with studies of illegal behaviour, the shortage of data makes firm conclusions difficult to reach. The impact of the Russian export taxes on illegal logging and illegal trade is not known. Similarly, the

Indonesian log and sawn timber export bans do not appear to have had much impact by themselves on illegal exports. Rather it was improved domestic enforcement that seems to have been most effective.

If the export bans had been reciprocally enforced by other countries – for example, if China and other export destinations had refused to import Indonesian timber after the bans – then the measure may have had more of an impact, though the disguising of Indonesian timber as Malaysian might have negated this. It is this kind of difficulty in excluding illegal timber from international trade that partly lies behind some of the recent actions taken by consumer countries, which have modified their legal and trade systems to make it easier to restrict trade. Examples includes the amendment of the Lacey Act in the United States (which makes importing illegally exported timber and timber products unlawful in the United States); the Voluntary Partnership Agreements being established between the EU and timber producing countries (which would provide a means of excluding illegal timber exports from the EU market; negotiations are currently under way with Indonesia, among other countries); and the EU "Timber Regulation" currently, which requires timber operators to establish due diligence systems to ensure they are not handling illegal timber.[12]

In theory, both export duties and export bans are likely to encourage illegal trade, the former making illegal timber cheaper (compared with their legal equivalents) and the latter potentially raising the value of illegal timber to importers. If the international policy framework continues to develop as it is, bans may prove more practical to implement and enforce than high export taxes, with destination countries assisting the exporting country in applying these. In either case, however, effective domestic enforcement is essential.

## Concluding remarks

There are relatively few studies and little data that look specifically at the relationship between market-based policy instruments and illegal trade. This situation is exacerbated by a lack of baseline information and before-and-after quantitative studies, making it difficult to link particular interventions to specific outcomes (Shyamsundar *et al.*, 2005). However, some trends can be observed and some tentative conclusions listed:

- Economic incentives can only work fully in a framework of good governance and law enforcement. Otherwise they risk exacerbating illegal activity, creating new opportunities for it, or shifting it to other regions or countries.
- As well as general good standards of governance, new governance structures can prove effective – *e.g.* community-based natural resource management, where local communities are given incentives to protect and manage the

resource. Security of tenure or other forms of resource ownership will be an important factor.

- Economic incentives will be more effective when they form part of a co-ordinated range of interventions – *e.g.*, where alternatives to illegal behaviour are provided (*e.g.* legal alternatives to poaching, or legal means to dispose of waste).
- Where international trade is a factor, co-ordination with other countries is an important means of ensuring the effectiveness of economic incentives, either to avoid displacement of illegal activities or to facilitate the creation of new incentives.
- Economics is not always the key driver of illegality, however. For example, poor governance or cultural values may also have a role. In such cases, economic incentives are likely to be less effective unless these other drivers are addressed.

In general, when designing economic incentives it is imperative that the potential for encouraging illegal behaviour (including trade) is considered, so that the consequences can be assessed and considered (*e.g.* whether ivory sales are likely to lead to increased poaching) and enforcement activities and other possible interventions can be better targeted.

### *Notes*

1. See, for example, Pearce *et al.* (1989). The OECD uses a definition of "economic instruments" in this context as: "fiscal and other economic incentives and disincentives to incorporate environmental costs and benefits into the budgets of households and enterprises. The objective is to encourage environmentally sound and efficient production and consumption through full-cost pricing. Economic instruments include effluent taxes or charges on pollutants and waste, deposit-refund systems and tradable pollution p+ermits." *http://stats.oecd.org/glossary/detail.asp?ID=723*.
2. See Johnstone and Bishop (2006) for a discussion.
3. This case study is extracted from Hayman, 2007, updated where appropriate.
4. These have included the collapse of management systems in Peru during the Shining Path insurgency in the early 1990s (Hayman, 2007); and ongoing poaching in some regions (Lichtenstein, 2010).
5. *www.iucnredlist.org/apps/redlist/details/22956/0*.
6. For instance, the ODP of CFC-11 = 1.0; that of halon-1301 = 10.0.
7. Active waste has the potential to undergo physical, chemical or biological changes when disposed of to landfill, *e.g.* timber, plastic, or paper; waste sites containing active waste need to be managed much more extensively than if they contain inactive waste.
8. UK government "Flycapture" website, at *www.defra.gov.uk/environment/quality/local/flytipping/flycapture-data.htm*.

9. *www.defra.gov.uk/environment/quality/local/flytipping/flycapture-qa.htm*.
10. For an assessment of the LOI see *www.wri.org/stories/2010/07/whats-next-indonesia-norway-cooperation-forests*.
11. *www.itto.int/en/news_releases/id=2610000*.
12. *http://ec.europa.eu/environment/forests/timber_regulation.htm*.

## References

Agnew D.J., *et al.* (2009), "Estimating the Worldwide Extent of Illegal Fishing", *PLoS ONE* 4(2): e4570. doi:10.1371/journal.pone.0004570, University of California San Diego, United States of America.

Alexander, J. and J. McGregor (2000), "Wildlife and Politics: CAMPFIRE in Zimbabwe", *Development and Change,* Vol. 31, Issue 3, pp. 605-627.

Brack, D. (1996), *International Trade and the Montreal Protocol*, Royal Institute of International Affairs, London.

Chatham House (2010), *Illegal Logging and Related Trade: Indicators of the Global Response*. Chatham House, London. *www.illegal-logging.info/item_single.php?it_id=938&it=document*.

Child, B. (1993), "Zimbabwe's CAMPFIRE Programme: Using the High Value of Wildlife Recreation to Revolutionize Natural Resource Management in Communal Areas", paper presented at 14th Commonwealth Forestry Conference, September 1993.

Cook, E. (1998), "Lessons from the CFC phase-out in the United States", in LePrestre, P., J. D. Reid and E. T. Morehouse (eds.), *Protecting the Ozone Layer: Lessons, Models and Prospects*, Kluwer.

Davies, B. & M. Doble (2004), "The development and implementation of a landfill tax in the UK", in *Addressing the Economics of Waste*, OECD, Paris.

EEA (European Environment Agency) (2009a), "Diverting waste from landfill: Effectiveness of waste-management policies in the European Union", EEA Report No. 7.

EEA (2009b), "Waste Without Borders in the EU? Transboundary shipments of waste", EEA Report No. 1.

EIA and Telapak (2005), *The Last Frontier: Illegal Logging in Papua and China's Massive Timber Theft*, EIA, Telapak.

FAO (2010), *State of World Fisheries and Aquaculture,* FAO, Rome.

FAO (2010) *Global Forest Resources Assessment 2010: Key Findings, Food,* FAO, Rome.

Fischer, C., E. Muchapondwa and T. Sterner (2005), *Shall We Gather 'Round the Campfire'?: Zimbabwe's Approach to Conserving Indigenous Wildlife*, Resources for the Future.

Frost, P. & I. Bond (2008), "The CAMPFIRE programme in Zimbabwe: Payments for wildlife services", *Ecological Economics*, Vol. 65, Issue 4, pp. 776-787.

Hoerner, J. A. (1996), "Taxing Pollution", in Cook, E. (ed.), *Ozone Protection in the United States: Element of Success*, WRI.

IMPEL (European Union Network for the Implementation and Enforcement of Environmental Law) (2005), "IMPEL-TSF Threat Assessment Project: The illegal shipments of waste among IMPEL Member States".

IMPEL-TFS Seaport II (2006), "International Cooperation in Enforcement Hitting Illegal Waste Shipments", Project Report, Sept. 2004-May 2006. IMPEL. June 2006. Report No. 110643/CE6/0M9/000398. Available online at: *http://ec.europa.eu/environment/impel/pdf/seaport_2_report2.pdf.*

Johnstone, N. and J. Bishop (2006), "Private Rights and Private Markets for Natural Resource Management in Developing Countries", in *International Review of Environmental and Resource Economics,* No. 1, pp. 1-43.

Karjalainen, T. *et al.* (2010) "Economic Impacts of the Russian Customs Tariffs Programme for Roundwood Exports – a Finnish Perspective", (Presentation, Workshop on Emerging Trade Measures in Timber Markets. Geneva, 23 March 2010). Available at: *http://timber.unece.org/fileadmin/DAM/meetings/05Karjalainen_et_al.pdf.*

Klooster, D. (2000), "Community Forestry and Tree Theft in Mexico: Resistance or complicity in conservation", *Development and Change* Vol. 31, Issue 1, pp. 281-305.

Lichtenstein, G. (2010), "Vicuña conservation and poverty alleviation? Andean communities and international fibre markets", *International Journal of the Commons* 4(1). Available at: *www.thecommonsjournal.org/index.php/ijc/article/viewArticle/139/89.*

McDermott Hughes, D. (2002), "Cadastral Politics: The Making of Community-Based Resource Management in Zimbabwe and Mozambique", *Development and Change* Vol. 32, Issue 4, pp. 741-768.

Montreal Protocol (2002), "Study on monitoring of international trade and prevention of illegal trade in ozone-depleting substances, mixtures and products containing ozone-depleting substances (Decision XIII/12): Report of the Secretariat", UNEP/OzL.Pro/WG.1/22/4, April 2002.

Newell, R., J. Sanchirico & S. Kerr (2002), "Fishing Quota Markets", discussion Paper 02-20, Resources for the Future.

Pearce, D., A. Markandya and E. Barbier (1989), *Blueprint for a Green Economy*, Earthscan, London.

Resosudarmo, B. R, and A. A. Yusuf (2006), "Is the Log Export Ban Effective? Revisiting the Issue through the Case of Indonesia", Australian National University, Economics and Environment Network Working Paper EEN0602, June 2006.

Schreckenberg, K. & C. Luttrell (2009), "Participatory forest management: a route to poverty reduction? ", *International Forestry Review* Vol. 11, Issue 2, pp. 221-238.

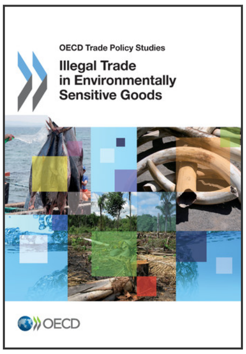
Seely, A. (2009), "Landfill tax : recent developments", House of Commons Library, SN/BT/1963, December 2009.

Seneca Creek Associates and Wood Resources International (2004), "Illegal logging and global wood markets: the competitive impacts on the US wood product industry", report prepared for American Forest and Paper Association.

Shyamsundar *et al.* (2005), "Devolution of Resource Rights, Poverty, and Natural Resource Management. A Review", Environmental Economics Series, Paper No. 104, World Bank Environment Department.

Sokolov, D. (2010) "Trade impact of Russian log export taxes, a Russian perspective", presentation, Workshop on Emerging Trade Measures in Timber Markets. Geneva, 23 March 2010. Available at: *http://timber.unece.org/fileadmin/DAM/meetings/06Sokolov.pdf.*

Tietenberg, T. (2003), "The Tradable-Permits Approach to Protecting the Commons: Lessons for Climate Change", *Oxford Review of Economic Policy*, Vol. 19, No. 3.

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