

# Chapter 14

## The Economics of Marine Litter

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**Abstract** This chapter aims to provide an overview of research into quantifying the economic impacts of marine litter. From an environmental economics perspective it introduces the difficulties in measuring the economic costs of marine litter; reviews those sectors where these costs are notable; and considers policy instruments, which can reduce these costs. Marine litter is underpinned by dynamic and complex processes, the drivers and impacts of which are multi-scalar, trans-boundary, and play out in both marine and terrestrial environments. These impacts include economic costs to expenditure, welfare and lost revenue. In most cases, these are not borne by the producers or the polluters. In industries such as fisheries and tourism the costs of marine litter are beginning to be quantified and are considerable. In other areas such as impacts on human health, or more intangible costs related to reduced ecosystem services, more research is evidently needed. As the costs of marine litter are most often used to cover removing debris or recovering from the damage which they have caused, this expenditure represents treatment rather than cure, and although probably cheaper than inaction do not present a strategy for cost reduction. Economic instruments, such as taxes and charges addressing the drivers of waste, for instance those being developed for plastic bags, could be used to reduce the production of marine litter and minimise its impacts. In any case, there remain big gaps in our understanding of the harm caused by marine litter, which presents difficulties when attempting to both quantify its economic costs, and develop effective and efficient instruments to reduce them.

**Keywords** Marine litter · Environmental economics · Economic instruments · Plastic bags · Waste · Polluter pays

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## 14.1 Introduction

In addition to the environmental and health issues discussed in previous book sections (Galloway 2015; Kühn et al. 2015), marine litter can cause a range of economic impacts that both increase the costs associated with marine and coastal activities, and reduce the economic benefits derived from them. This chapter aims to provide an overview of the results of research performed to date, which has attempted to quantify the economic impact of marine litter. It provides a brief analysis of the marine litter problem from an environmental economics perspective, and discusses the use and design of economic-based policy instruments to tackle the problem.

## 14.2 Estimating the Economic Impacts of Marine Litter

Measuring the full economic cost of marine litter is complex due to the wide range of economic, social and environmental impacts, the range of sectors impacted by marine litter and the geographic spread of those affected. Some of the impacts are easier to evaluate in economic terms because they are more direct, such as increased marine litter cleaning costs. Others are more complex, for example, the less direct and/or more intangible values such as the impacts of ecosystem deterioration or reductions in quality of life. Furthermore, the spatial and temporal complexity of the impacts related to marine litter result in costs, which may not always be immediate or conspicuous but are nevertheless significant for sustainability (National Research Council 2008). As regards ecosystem degradation, it is useful to differentiate between impacts on biodiversity (species and habitats) and the impact on the ecosystem services flowing from the ecosystem (e.g. provisioning services such as food provision, regulating services such as water and waste purification; and cultural services such as tourism and recreation). As regards economic costs it is important to differentiate between actual economic costs linked to expenditure (e.g. costs of cleanup of beaches; costs associated with damage to or loss of fishing gear or obstruction of motors; eventual cost of hospitalisation from marine debris related health impacts), economic costs of loss of output or revenue (e.g. loss of revenue from fish or loss of income from tourism) and assessment of welfare costs in economic terms (e.g. health impacts from marine debris; assessing the economic value of loss of cultural values such as recreation or landscape aesthetics).

While marine litter has become an increasingly important issue in policy discussions, there is only a very sketchy (albeit growing) body of knowledge on the costs of the impacts. Because of a lack of recording even the direct economic costs of marine litter tend not to be measured (Mouat et al. 2010). Furthermore, even though there is a growing interest in ecosystem services (Costanza et al. 1997; MA 2005; TEEB 2010, 2011) little research has been done to date on the economic cost of marine litter on ecosystem service provision. Having said this, evaluations

of marine ecosystem services, which are estimated at €16.5 trillion in one study (Costanza et al. 1997), suggest that even fractional deterioration in provision would represent a significant cost (Beaumont et al. 2007; Galparsoro et al. 2014).

Thus far, studies undertaken to estimate the economic impacts of marine litter have generally focused on the direct losses borne by economic activities adversely affected by the presence of marine litter in the environment, within which they operate and rely upon (see Hall 2000; Mouat et al. 2010; MacFayden 2009; McIlgorm et al. 2011). Largely, such studies have not taken into account the often intangible costs of any social and ecological impacts. Some early studies allude to the need for research to explore these costs. For instance, Kirkley and McConnell (1997, p. 185) call for strategies, which account for the economics related to lost ecological functions driven by marine litter. The intricacy of developing such strategies can be illustrated with the example of alien invasive species. Marine litter provides additional opportunities for marine organisms to travel (including alien invasive species) up to threefold (Barnes 2002). Given that the introduction of alien invasive species can have a detrimental impact on marine ecosystems and biodiversity (Kiessling et al. 2015) and can result in serious economic losses to many marine industries, any estimates, which exclude such ecological impacts, will inevitably fall seriously short of the true cost of the marine litter problem. For example, the introduction of the carpet sea squirt (*Didemnum vexillum*) in Holyhead Harbour (Wales, U.K.) resulted in an eradication and monitoring program over a decade starting in 2009, which was expected to cost €670,000. This expenditure was economically justified as allowing the species to spread unpredated and smother organisms and marine habitats would have cost the local mussel fisheries up to €8.6 million alone over 10 years (Holt 2009). Goldstein et al. (2014) recorded the ciliate pathogen *Halofolliculina* (known to cause skeletal eroding band disease in corals) on floating plastic debris in the western Pacific and suggested that the spread of the disease to Caribbean and Hawaiian corals may be due to rafting on the enormous quantities of litter reported from the area. Increased coral mortality or the introduction of other pathogens via floating marine debris may lead to economic costs, for example through decreased revenues due to falling numbers of visiting tourists.

Despite their partial coverage, the studies that are available provide sufficient information to draw a number of important conclusions. The main economic sectors, which have been identified from the literature as being affected by marine litter are agriculture, aquaculture, fisheries, commercial shipping and recreational boating, coastal municipalities, coastal tourism sector and the emergency rescue services (Hall 2000; Mouat et al. 2010). The economic impacts affecting these sectors are described and quantified where possible (Hall 2000; Mouat et al. 2010; McIlgorm et al. 2011; Jang et al., 2014; Antonelis 2011). They also make attempts at aggregating economic impacts across sectors to provide regional cost estimates. Mouat et al. (2010) provide an estimate of marine litter costs for the Shetland (U.K.) economy, of €1–1.1 million on average per year, an estimate, which consists of actual expenditures and, in some cases, estimated lost income. This is only a single case study, and the sectors affected on Shetland would be affected to

varying degrees in other coastal areas. However, these findings clearly demonstrate that the economic impact of marine litter on coastal communities can be extremely high. McIlgorm et al. (2011) calculated the costs of marine litter for 21 economies in the Asia-Pacific region. Similarly to Hall (2000) and Mouat et al. (2010), they consisted of such losses as those from entangled ship propellers, lost fishing time, and tourism losses from deterring visitors, but not the cost of any harm to ecosystem services or other non-market values (McIlgorm et al. 2011). In total, McIlgorm et al. (2011) estimated the cost of marine litter in the Asia-Pacific region to be in the region of €1 billion per year to marine industries, equivalent to 0.3 % of the gross domestic product for the marine sector of the region.

### ***14.2.1 Beach Cleaning, Tourism and Recreation***

Coastal municipalities are impacted economically by marine litter primarily through the direct cost of keeping beaches clear of litter and its wider implications for tourism and recreation. Direct costs include the collection, transportation and disposal of litter, and administrative costs such as contract management. Ensuring that beaches are clean, attractive and safe for visitors is prioritised by municipalities when the economic case for protecting the local economy and tourism industry justifies the costs of removing the litter. In areas where coastlines make a significant contribution to the economy, the costs incurred through marine litter can be substantial.

In the U.K., the cost of removing beach litter to all coastal municipalities is estimated to be in the region of €18–19 million (Mouat et al. 2010). This equates to an average cost per municipality of €146,000 (Mouat et al. 2010). The majority of this cost was accounted for by labor costs. Mouat et al. (2010) also calculated the cost annually per km of coastline. Although the average cost of litter removal was between €7,000 and €7,300 per km per year, there was a lot of variation, with costs ranging from €171 to €82,000 per km per year (Mouat et al. 2010). Higher costs correlated with more intense cleaning operations on small areas of coastline, particularly in tourist areas. In Belgium and The Netherlands, the total cost of beach litter removal was estimated to be €10.4 million per year, at an average of €200,000 per municipality per year (Mouat et al. 2010). Per km, the cleaning costs came to €34,000 per year on average, again with great variation (e.g. from €600 to €97,300 in Den Haag) (Mouat et al. 2010). This average is much greater than that in the U.K. as municipalities in Belgium and The Netherlands removed litter from a much higher proportion of their coastline (because it is more densely populated). The great variation in amounts spent by municipalities on different beaches reflects the variation in importance of different stretches of coastline to the tourism industry. Of course many areas of coastline worldwide do not have anything spent on them to provide a litter cleanup service.

It is important to recognize, however, that beach cleaning is not necessarily performed by municipalities alone, and that voluntary organisations tend to play

a large role in removing litter (see Hidalgo-Ruz and Thiel 2015). This is an economic impact on society that comprises operational expenditure, financial assistance or some sort of ‘in kind’ assistance such as materials or insurance, and the value of volunteers’ time. There may also be an opportunity cost where volunteer time could be spent servicing the community in other ways. Mouat et al. (2010) estimated the value of volunteers’ time in two annual beach clean operations in the U.K. at which a substantial quantity of litter from the U.K. coastline was collected, to be around €131,000. As this estimate includes neither financial assistance nor operational management costs, it is likely to be a substantial underestimate.

In coastal municipalities, particularly those where beaches contribute significantly to the local economy, the indirect economic impacts of marine litter are more important. A few studies have attempted to calculate the costs incurred to coastal areas as a result of marine litter. Jang et al. (2014) considered the economic impact of a single marine litter event in South Korea, in which heavy rainfall resulted in an unusually high level of marine litter to be washed on the beaches of Goeje Island, a popular tourist destination. Based on government figures and a number of surveys they assessed multiple economic effects of marine litter such as lost expenditure on hotels and lodging, which could be influenced by marine litter. Lost expenditure was expressed as the product of decreased visitors and average visitor expenditure, in this case between €23 and €29 million of lost revenue in 2011 compared to 2010, as a result of over 500,000 fewer visitors to the island. In 2013, a study of 31 beaches in Orange County (California, USA) considered how marine debris influences their decision to go to the beach, and at what expense (Leggett et al. 2014). It applied the *travel cost model*, which estimates the value people derive from recreation at a particular site based on the utility they expect to experience in relation to alternative sites. They showed that marine debris had a significant impact on residents’ beach choices, and that a 75 % reduction in marine litter at six popular beaches generated over €40 million in additional benefits to Orange County residents over just 3 months. These two studies clearly demonstrate the value people place in the clean marine and coastal environments and the potential for costs to communities, which derive utility for the services which these environments provide. This is an interesting finding given that many of these visitors may also be responsible for some of the pollution, by littering during their beach visits, for example.

### ***14.2.2 Shipping and Yachting***

The shipping and yachting industries also experience economic impacts as a result of marine litter pollution, with harbors and marinas incurring the cost of removing marine litter from their facilities in order to keep them safe and attractive to users, and vessels experiencing interference with propellers, anchors, rudders and blocked intake pipes and valves (Mouat et al. 2010). On occasion, some of these

vessel encounters pose navigational hazards that require the rescue services to become involved, thereby increasing costs dramatically.

Mouat et al. (2010) estimated that removing marine litter costs U.K. ports and harbors on average €2.4 million per year. But this can range from €0 (as not all harbors surveyed in this study took action to remove marine litter, and thereby incurred no direct costs) to almost €73,000 per year for individual harbors (Mouat et al. 2010). Higher costs tended to correlate with larger and busier harbors. Disposal and manual removal of floating debris were observed to make up the bulk of these costs, as dredging to remove items off the seabed, although expensive, is not performed very commonly. There is no estimate for the cost of removing marine litter to the U.K. marina industry as a whole, but data from a small sample indicate that it could be costly, with one marina reporting an annual bill of €39,000 (Mouat et al. 2010). When factoring in the cost of undertaking rescue operations the cost of marine litter to shipping and yachting rises further. An estimate for the U.K. Royal National Lifeboat Institution in 2008 calculated that 286 rescue operations to vessels with tangled propellers cost between €830,000 and €2,189,000 (Mouat et al. 2010).

### 14.2.3 Fisheries

The fishing sector is more commonly viewed as a source of marine litter, but it is also subject to economic costs itself. Direct economic impacts faced by the sector arise from the need to repair or replace gear that has been damaged or lost due to encounters with marine litter; repairing vessels with tangled propellers (Fig. 14.1), anchors, rudders, blocked intake pipes, etc.; loss of earnings due to time diverted to deal with marine litter encounters; and loss of earnings from reduced or contaminated catches resulting from marine litter encounters including ghost fishing



**Fig. 14.1** Diver removes derelict rope wrapped around the propeller of a ship (*left* photo: NOAA Marine Debris program). Derelict rope crab pot “ghost fishing” at 1,091 m in Astoria Canyon, off Oregon (*right* © 2006 MBARI)

(Fig. 14.1). Wallace (1990) reported that in the Eastern US, over 45 % of the commercial fishers had their propellers caught, over 30 % had their gear fouled, and almost 40 % had their engine cooling system clogged by plastic debris at some point in time. The sector also experiences indirect losses of earnings due to the impact of loss and abandoned fishing gear on fish stocks (MacFayden et al. 2009; Sheavly and Register 2007).

There are potentially also costs associated with loss of value of fisheries resources (provisioning services under the ecosystem service nomenclature), whether through reductions in fish and shellfish numbers or reduced value due to impacts on quality of fish and shellfish (e.g. through ingested plastics or contamination with persistent organic pollutants, POPs). The body of literature describing the contamination of commercially exploited fish and shellfish by microplastic ingestion is growing rapidly, as is the literature analysing the consequences of this contamination on the health of individuals and populations (Galloway 2015; Lusher 2015; Rochman 2015). However, as yet there have been no economic assessments to estimate the costs of these impacts.

Derelict fishing gear (DFG) constitutes a considerable portion of marine litter and can result in economic losses for fisheries. DFG includes any equipment, which can catch (shell-)fish, which is lost by fisheries, including trawl nets, gill nets, traps, cages and pots (National Research Council 2008). As a result of their functional design, DFG can continue to trap marine life after they have been lost (a phenomenon known as ghost fishing). Increasingly durable materials used in fishing equipment means that it can continue to ghost fish for some time; in this way it presents particular challenges as marine waste. Fisheries incur costs, firstly in having to replace the fishing gear they have lost at sea, and secondly in a reduction in their potential harvestable catch, and indeed the sustainability of that catch (Butler et al. 2013; Arthur et al. 2014; Bilkovic 2014). One study in Puget Sound, Washington, estimated that over 175,000 dungeness crab (*Metacarcinus magister*) were killed each year by derelict fishing traps, equivalent to around €586,000 or 4.5 % of the average annual harvest (Antonelis 2011).

Mouat et al. (2010) focused on estimating the direct economic impact of marine litter on Scottish fishing vessels (i.e. costs of repairs and direct losses in earnings, not indirect losses due to ghost fishing) and estimated that on average marine litter costs each fishing vessel between €17,000 and €19,000 per year (Mouat et al. 2010). Two-thirds of this cost (€12,000) was incurred through time lost clearing litter from nets (calculated using the average value of 1 h's fishing time as estimated by vessels surveyed during this project). Aggregated, this costs the Scottish fishing industry as a whole between €11.7 and 13 million every year (Mouat et al. 2010). To put this in perspective, marine litter knocks 5 % off the fleets' total annual revenue. This is clearly a substantial cost to an industry that is already under high pressure and important in coastal communities.

Similar to voluntary beach cleanup operations, there are a number of 'Fishing-for-Litter' schemes in operation whereby fishermen voluntarily agree to collect the litter, which they catch in their nets during their normal fishing activity, and dispose of this safely on the quayside at designated waste disposal sites (Fig. 14.2).



**Fig. 14.2** OSPAR Fishing for Litter program. From left to right catch with a tyre from a *Nephrops* trawler in the Clyde (U.K.) (Photo: M. Bergmann, AWI); fisher from *FV Andrea* sorting litter into bags provided by Fishing for Litter scheme (Photo: G. Lengler, NABU, DSD); portside container for litter collected by fishers (Photo: K. Detloff, NABU)

These schemes are currently in operation in the U.K., Sweden, Denmark, The Netherlands, Belgium and Germany, and potentially other EU countries as there is EU support to fund such operations (KIMO International 2013; European Commission 2011).

The fishers benefit from being involved as they reduce the volume of litter accumulating in the oceans and on beaches, and they thereby reduce the amount of time they spend untangling litter from nets and reduce the risks of other marine litter related costs described above. Best-practice guidelines indicate that Fishing-for-Litter schemes also have the benefit of changing the culture within the industry to adopt good waste-management practices (OSPAR 2007). These schemes are not without costs to operate, however, as coordinators are needed to promote and run the programs, and there are of course costs associated with waste disposal (OSPAR 2007). These costs will vary depending on the country and the amounts of litter collected, but the schemes in Europe to date demonstrate that Fishing-for-Litter is a significant and cost-effective measure to reduce litter (OSPAR 2007).

#### 14.2.4 Aquaculture

Marine litter can result in costs to the aquaculture industry, through entangling propellers and blocking intake pipes, and time spent removing debris from and around fish farm operations. Mouat et al. (2010) surveyed finfish and shellfish aquaculture producers in Scotland and estimated that marine litter costs the sector on average €156,000 per year, which amounted to approximately €580 per year per producer. Ninety percent of this cost was due to time spent untangling fouled propellers on workboats and repairs. Removing marine litter from aquaculture sites was less of an issue overall, but this was highly variable, and in some areas it



was a regular problem. These figures demonstrate that in comparison to other sectors such as fisheries, and even agriculture, the direct cost imposed by marine litter on aquaculture is relatively low.

### ***14.2.5 Agriculture***

As a terrestrial economic activity, agriculture is not the most obvious sector to suffer economic losses because of marine litter. Indeed, similarly to the fishing sector it is more frequently seen as a source of marine litter (de Stephanis et al. 2013). However, in some locations, debris can blow, drift or get washed up on coastal farmland, causing damage to property, equipment and presenting a risk to livestock through ingestion and entanglement. These impacts may all lead to economic losses in addition to the cost of preventative litter removal. The Shetland Isles are one such location where marine debris litters agricultural lands, due to the prevalence of strong winds. Through interviewing farmers from Shetland with land by the coast, Hall (2000) estimated annual losses to be €500 per farmer, and a total of €770,000 for islands as a whole. This comprised time spent cleaning land, clearing ditches, freeing entangled animals, additional vet bills, and repairs to fences damaged by litter. Other losses to farmers not factored into this estimate were the loss of seaweed as a fertilizer due to plastic entanglement, and limits on the practice of grazing livestock on seaweed on beaches. Mouat et al. (2010) estimated that marine litter cost each smallholding an average of €841 per year and the agricultural industry of the Shetland Islands as a whole a total of approximately €252,000 (based on the assumption that 25 % of the 1,200 active crofters were operating in areas subject to marine litter damage). The cost to small-scale agricultural producers is of particular concern given that they have small profit margins. Although the scope of these estimates is restricted to one Scottish archipelago and the costs of marine litter to coastal agricultural businesses elsewhere is unknown, anecdotal evidence from the south of England and Sweden suggests it is a big problem in other coastal regions, too (Mouat et al. 2010). Clearly, more research would be required to determine the impacts and costs of marine litter in other coastal agricultural enterprises.

### ***14.2.6 Human Health***

Whilst the impact of marine litter on human health is a relatively new area of research this does not negate its potential for generating economic and welfare costs. At a local level ocean collisions with marine litter can seriously injure or kill mariners (Gold et al. 2013). This is particularly the case with impacts between smaller vessels and larger objects, such as semi-submerged lost shipping containers, which is a known danger to recreational sailors and fishers.

Plastic pollution poses a number of more nuanced risks, which could directly and indirectly impact on human health (Teuten et al. 2009; Thompson et al. 2009; Gold et al. 2013; UNEP 2014: p. 50; Galloway 2015). Firstly, the physical and chemical properties of polymers lends to their ability to facilitate the accumulation of contaminants already present in sea water. Industrial and agricultural chemicals, including polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT), and aqueous metals, have been linked to health impacts such as disease and reproductive abnormalities (Teuten et al. 2009). Marine litter acts as nuclei of accumulation for such toxins, which become several orders of magnitude more concentrated on the surface of plastics (EPA 2011). For instance, plastics can contain up to 1 million times the concentration of PCBs in contrast to sea water (Gold et al. 2013: p. 5; EPA 2013). Secondly, chemicals used in the production of polymers can increase local concentrations of harmful toxins which are also known to impact on health. Additives such as bisphenol A (BPA) and flame retardants, such as polybrominated diphenyl ethers (PBDEs), commonly found in plastic waste, can dissociate in the environment and are linked to endocrine disruption in both wildlife and humans (Gold et al. 2013). Both sources of chemicals increase the potential for bioaccumulation of toxins within food chains when marine litter is ingested by smaller organisms.

Thirdly, plastics could facilitate the transmission of, and act as a vector for, viral and bacterial diseases in areas where they would not naturally occur. In some locations plastic marine litter has developed its own habitat, the “plasticsphere”, supporting organisms which differ from those in the surrounding water (Gold et al. 2013). Lippsett (2013) found a plastic sample dominated by bacteria, which cause cholera and gastrointestinal disease. Consequently invasive species and foreign substrates linked to marine litter could pose significant health threats. The economics of such health risks are difficult to formulate but figures are urgently needed to assess these additional cost associated with marine litter.

### ***14.2.7 Summary***

So far we have presented the results of recent research estimating the direct costs of marine litter to the key coastal and maritime economic sectors affected. Estimates of economic impacts on a national or regional scale are hard to come by, however. It is clear that we require more monitoring of the costs associated with marine litter, both in terms of direct costs incurred on losses to outputs and income, and in terms of assessing health, ecosystem services, wellbeing, and welfare impacts.

What is also very clear from this review is that marine litter exerts substantial economic impacts on coastal sectors, and that the polluters or producers of plastics do not pay for these costs. Furthermore, those who do pay are often operating with tight budgets, such as municipalities, small-scale agricultural businesses and fisheries. In addition, the costs described and quantified in the literature consist primarily of the costs of cleaning up marine debris or recovering from marine litter

damage. As explained above, the cost of cleaning up is justified by the even higher costs of inaction. However, this expenditure does not address the underlying issue, and does not act to prevent litter from entering the marine environment in the first place. When prioritizing action and the allocation of funds, the costs of damage and clean up needs to be weighed against the cost of prevention.

### 14.3 Marine Litter and Economic Incentives

From an environmental economics perspective, marine litter arises, like other waste or pollution problems, through market failure. The marginal price of goods on the market, and that of disposable plastics in particular, does not reflect the full marginal cost to society of producing that good. In other words, there is an external cost to society not borne by the producer (or consumer) as demonstrated in the previous section. Furthermore, clean seas and beaches are public goods, which are vulnerable to free-riding, whereby those disposing of waste inappropriately benefit from the good without paying the full cost, thereby causing contamination and degradation of the marine environment.

Like other environmental problems, marine litter can be prevented and controlled using measures that limit and control this sort of behavior (command and control measures), by awareness raising and other information tools, and by using market-based measures that aim to encourage a change in behavior by altering the economic incentives in place and/or to raise revenue to bring the market price in line with the social cost.

There are a range of market-based instruments that can be used to address marine litter. Landfill taxes, if set at adequately high levels, can disincentivize the final disposal of waste and help to incentivize recycling and recovery, reducing the risk of waste reaching the marine environment (although care should be taken to set taxes at a level that does not give significant encouragement to illegal dumping of waste). Product taxes and charges can be used to discourage the consumption of certain products that frequently end up as marine litter, such as plastic bags, packaging and fishing tackle. Infrastructure charges, for example, for the use of port-waste facilities, help to ensure that waste management infrastructures and facilities are developed and maintained. Deposit-refund schemes, which are most often applied to packaging items such as bottles, can encourage return and reuse by consumers, and therefore reduce the number of such items ending up as litter. Hardesty et al. (2014) evaluated the effectiveness of South Australia's container deposit scheme in reducing waste lost to beaches and reported a threefold reduction in this dominant plastic item in the environment. Direct investment in infrastructure, such as rubbish bins and secure waste collections on beaches and in coastal areas, can help to keep coastal areas free of litter and reduce the risk of items reaching the seas. Such investment can be financed for example by tourist taxes or car parking fees. High fees and fines for littering, illegal waste disposal and fly-tipping help to dissuade behaviors that result in waste escaping from formal waste management

processes, reducing the risk of waste reaching the marine environment as litter (ten Brink et al. 2009).

The following sections of this chapter focus on a small selection of market-based instruments, which can potentially have an impact on the amount of litter that reaches the marine environment: landfill taxes, instruments addressing plastic bags (including charges/taxes and bans), producer responsibility schemes, and fees for the use of port waste reception facilities.

### ***14.3.1 Landfill Taxes/Levies***

A significant proportion of marine litter originates from land-based sources; a global figure of 80 % is frequently cited, although the origins of this are unclear (Arthur et al. 2014) and figures may vary considerably regionally. The National Marine Debris Monitoring Program, which analyzed marine litter on US beaches,<sup>1</sup> determined that 49 % was from land-based sources and 18 % from ocean-based sources, with a further 33 % for which the source could not be identified (Ocean Conservancy 2007). Up to 95 % of the litter found on Australian beaches comes from suburban streets through the stormwater system (Clean Up Australia 2009). For this reason, measures to promote improved waste management on land have an important role to play in preventing land-based waste from reaching the seas.

One of the most common economic instruments used in the waste sector is the application of a tax or levy on waste sent to landfill. Landfill taxes/levies can help to tackle marine litter by increasing the price of landfill to encourage the diversion of waste to other forms of treatment that are higher up in the waste hierarchy, including closed-loop waste-management processes such as recovery, recycling or reuse. If lightweight items in particular, such as many small packaging items, can be kept out of landfills, this eliminates the risk of them being blown by the wind from the surface of landfills, preventing them from reaching water courses and eventually entering the sea. It should be noted that landfill taxes can incentivise illegal landfilling and fly-tipping as a means of tax avoidance. Estimates of the amount of marine litter that comes from landfill and fly-tipping are limited; one estimate from the Scottish Government is that around 1.6 % of marine litter comes from fly-tipping incidents (Scottish Government 2013). To stop these unchecked methods of waste disposal from resulting in more waste being blown or washed into rivers and seas, landfill taxes should be accompanied by measures such as the closure of illegal landfills and enforcing fines on those who fly-tip or dump illegally. Producer-responsibility schemes can also help promote recycling (see below).

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<sup>1</sup>The NMDMP ran from September 2001–September 2006. The US was divided into nine coastal regions; within each region a random selection of between 12 and 23 beach sites was chosen for surveying (175 sites in total). Over 600 volunteers conducted surveys at 28-day intervals, covering a 500-m stretch of beach at each study site, and collected and recorded the various marine debris items found.

Landfill taxes are typically charged per tonne of waste landfilled, and there are often different rates for active (e.g. biodegradable) and inert (e.g. mineral/construction) waste, to reflect the varying environmental impacts of different wastes. A brief overview of the use of landfill taxes is provided here. It should be noted that this is not intended to be a fully comprehensive review of the global situation.

In **Europe**, many countries have introduced landfill taxes since the EU Landfill Directive (1999/31/EC) entered into force. The directive aims to encourage the prevention, recycling and recovery of waste by limiting its final disposal through landfills, including setting targets to reduce the amount of biodegradable waste sent to landfills and associated methane emissions. In addition, Annex I sets out requirements for the location (e.g. with regard to the proximity of water bodies and coastal waters) and technical specifications (e.g. design features to avoid pollution of soils and waters from landfills, including from wind-blown waste) of landfill sites. There are currently 20 countries<sup>2</sup> in **Europe** that tax waste sent to landfills. From 1995 to 2012, the number of EU countries implementing a landfill tax rose from 7 to 20 (Watkins et al. 2012). Over the same period, the amount of municipal waste sent to landfill decreased from around 63–33 % (Eurostat 2014a). In the majority of cases, the tax is collected by state tax authorities or regional institutions. However, only in some countries (Bulgaria, Finland, Belgium (Wallonia), France, Poland, Portugal, Spain and the United Kingdom) does part of the revenue go towards waste management and environmental initiatives (Fischer et al. 2012). The level of the tax varies considerably between countries: Watkins et al. (2012) identified a wide range of tax rates for municipal solid waste, from €3 t<sup>-1</sup> in Bulgaria to €107 t<sup>-1</sup> in The Netherlands. Higher landfill taxes tend to result in lower proportions of municipal waste being sent to landfills and higher rates of recycling and composting. The majority of countries with total landfill charges<sup>3</sup> of less than €40 t<sup>-1</sup> send over 60 % of their municipal waste to landfill. Countries are much more likely to meet a 50 % recycling target once landfill charges (or the cost of the cheapest disposal option) approach €100 t<sup>-1</sup> (Watkins et al. 2012). It should be noted that the best performing countries in terms of diverting waste from landfill usually also have other measures in place, such as bans on the landfilling of certain types of waste.

The **Australian** state of Victoria has a levy of €17 t<sup>-1</sup> for rural municipal waste, €29 t<sup>-1</sup> for rural industrial waste, €34 t<sup>-1</sup> for metropolitan municipal and industrial waste. All levies are paid into the Environment Protection Fund, with revenues from the levy contributing to improved waste management including upgrading of kerbside recycling systems, developing markets for recycled materials, and

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<sup>2</sup>Austria, Belgium (Flanders and Wallonia), Bulgaria, the Czech Republic, Denmark, Estonia, Finland, France, Ireland, Italy, Latvia, the Netherlands, Norway, Poland, Portugal, Slovenia, Spain (Andalusia, Catalonia, Madrid and Murcia), Sweden, Switzerland and the United Kingdom.

<sup>3</sup>Total landfill charge defined as tax plus ‘gate fee’ charged by landfill operator for receiving the waste.

studies into waste minimization, handling and disposal, organics recycling and litter control (EPA Victoria 2013). The levy in the Sydney metropolitan area has risen sharply since 2006/07 and is planned to reach €84 t<sup>-1</sup> by 2015/16 (New Zealand Ministry for the Environment 2014). The state of Western Australia applies a levy of €20 t<sup>-1</sup> (increasing to €39 t<sup>-1</sup> from 1 January 2015) for putrescible waste and €6 t<sup>-1</sup> (€28 t<sup>-1</sup> from 1 January 2015) for inert waste deposited in metropolitan landfills. The levy is paid by the owner of the landfill receiving the waste, but they may pass the cost on to customers. Not less than 25 % of revenues will be spent on initiatives to manage, reduce, re-use or recycle waste and to monitor/measure waste, and 7 % provided to the Office of the Environmental Protection Authority to assist in service delivery (Wastenet 2014). Evidence from Australia on the impact of landfill levies is somewhat mixed. The Western Australian Local Government Association claims there is limited evidence that a levy directly disincentives landfill, and that a lack of accompanying investment in waste management can actually be detrimental to waste diversion activities due to reduced expenditure on recycling infrastructure (WALGA 2012). The waste levy in Sydney helped to increase recycling by making waste recovery more financially attractive than landfill; the total quantity of waste to landfill was lower in 2010/11 than 2002/03, and waste recycled more than doubled over the same time period (New Zealand Ministry for the Environment 2014). Conversely, following the removal of a €25 t<sup>-1</sup> landfill levy in Queensland, there was a 20 % reduction in recycling (Ritchie 2014), which indicates that landfill taxes may indeed encourage recycling.

**New Zealand** applies a tax of €6 t<sup>-1</sup> to any waste deposited at a waste disposal facility. This is paid by disposal facility operators, but they may pass the cost on to the households/businesses that generate the waste. The levy's primary aim has been to raise revenue for waste minimization and recycling projects, but the levy was set at a relatively low level to avoid illegal dumping and to reduce the impact on businesses and households. Half of the revenues go to territorial authorities to assist with waste minimization. The rest (minus administration costs) is paid into a national waste minimization fund (New Zealand Ministry for the Environment 2013). A recent review of the effectiveness of the waste disposal levy found that the levy is estimated to be applied to only 30 % of total waste disposed of to land, and that the amount of waste landfilled has increased by around 6 % between 2010 and 2013. Revenues have supported a broad range of waste minimization initiatives, although the funding outcomes should be more effectively measured and monitored (New Zealand Ministry for the Environment 2014).

### ***14.3.2 Plastic Bag Initiatives***

Plastic is the most common litter type found in the marine environment. In European regional seas, for example, plastics comprise more than half of the marine litter. More than half of the plastic fraction of marine litter consists of plastic packaging waste, with bottles and bags being the most frequently found items (European

Commission 2013a). Plastics account for some 78 % by number of pieces, and 67 % by weight, of washed-up materials on the coasts of the Northwest Pacific Region (Kanehiro 2012). Lightweight plastic bags are particularly prone to becoming marine litter since they are seen by consumers as single-use, are often disposed of carelessly, are frequently not accepted in household recycling collections, and are easily blown by the wind into drains, water courses and the marine environment. During the Ocean Conservancy's 2013 International Coastal Cleanup,<sup>4</sup> 6 % of the total litter items found were plastic bags (grocery or other plastic bags) (Ocean Conservancy 2014). In China in 2012, plastic bags comprised 23 % of drifting marine litter found in the sea, and 59 % of that found on beaches (Meng and Chen 2013). In Shanghai, they accounted for between 15 and 29 % of all marine litter items recorded during coastal surveys between 2008 and 2012 (ICC 2013).

European Commission guidance for EU Member States on developing waste prevention programs, published in October 2012, suggests that **plastic bags** can be effectively targeted by waste prevention activities (European Commission 2012). Many countries have already taken specific action to tackle plastic bags: over 30 countries have introduced taxes/fees, and over 30 have introduced bans for single-use carrier bags, or bans on bags with certain characteristics, such as those made from plastic of less than a certain thickness (Earth Policy Institute 2013). In some countries, there is a mix of bans and charges, since this is an area of policy that is often dealt with at a local or city level. The following paragraphs summarise several of these initiatives, with a focus on those where information is available on their impacts.

In **Europe**, several countries have introduced taxes or charges on single-use (disposable) plastic carrier bags. Denmark has applied a charge for plastic and paper carrier bags since 1993 (the charge depends on the weight and material). The year after the tax was introduced saw an initial reduction in bag use of 60 % (Earth Policy Institute 2013).

Ireland introduced a €0.15 levy per general purpose plastic bag in 2002, and increased the levy to €0.22 in 2007. The levy led to an immediate decrease in plastic bag use from an estimated 328 bags per capita per year to 21 bags per capita. Although per capita consumption increased again to 31 bags during 2006, an increase in the levy in 2007 led to a further reduction to 18 bags in 2010 (Department of Environment Community and Local Government 2013). Plastic bags constituted 0.3 % of litter pollution nationally in 2012 compared to an estimated 5 % in 2001 prior to the introduction of the levy (National Litter Pollution Monitoring System 2013). In 2001 (pre-levy) around 17 plastic bags were found per 500 m of coastline. This figure fell to around 10 bags in 2002 (the year the levy was introduced), 5 bags in 2003, and 2 in 2012 (Doyle and O'Hagan 2013).

In the U.K., a €0.06 levy on the use of single-use carrier bags (plastic and paper) was introduced in Wales in 2011, and in Northern Ireland in 2013; retailers in

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<sup>4</sup>Ocean Conservancy's (2013) International Coastal Cleanup involved nearly 650,000 volunteers at over 5,500 beach/coastal sites covering a total length of 12,914 miles in 92 countries and locations.

Scotland introduced the same charge in 2014, and supermarkets and larger stores in England will adopt the same charge in 2015. One year after the introduction of the **Welsh** charge, a 70–96 % decrease was observed in bag use at food retailers and a 68–75 % decrease at fashion retailers (Welsh Government 2012). While in 2010 (pre-charge), 0.35 million thin-gauge carrier bags were distributed in Wales; in 2011 (the year the charge was introduced), 0.27 billion bags were distributed and one year after the introduction of the charge the figure dropped significantly to 0.07 billion bags (WRAP 2013). This represents an 81 % reduction in only three years whereas no reduction was observed in any other nation of the U.K. over the same time period, which suggests that the charge has significantly contributed to the reduction in Wales. During International Coastal Cleanup days, 435 plastic bags were found in Wales in 2011, and 292 in 2012 (Ocean Conservancy 2012, 2013). These data only relate to a single day each year and therefore present a limited picture, but the reduction in plastic bag use due to the charge may have had at least some impact on the number of bags found.

In November 2013, the European Commission put forward a proposal for **European Union** legislation to reduce the use of lightweight plastic bags. The proposal would amend the existing Packaging and Packaging Waste Directive (1994/62/EC) (see Chen 2015), requiring all EU Member States to take action to reduce the consumption of lightweight plastic bags, but allowing them to choose the most appropriate measures to do this (European Commission 2013b). The proposal will be discussed in the European Parliament and Council in 2015.

Several nations in **Africa** have either totally banned the use of plastic bags (Eritrea in 2005; Somalia/Somaliland in 2005; Tanzania in 2006/Zanzibar in 2008; Democratic Republic of Congo in 2011) or banned the use of bags below a minimum thickness (Kenya and Uganda, both 2007) (see also Chen 2015). Botswana (2007) established a minimum thickness for bags and required retailers to apply a minimum levy to thicker bags (many retailers charged more than the minimum), to fund government environmental projects. A study of four retail chains 18 months after implementation of the charge showed that bag use had fallen by 50 % (Earth Policy Institute 2013). Morocco has implemented a minimum thickness standard. In 2004, South Africa implemented a tax of €0.002/bag for thicker bags, to run concurrently with a minimum thickness ban (Miller 2012). A portion of the tax funds environmental projects. Whilst bag use decreased by 90 % when the measures were first introduced, consumption has slowly risen again since (Earth Policy Institute 2013).

One-hundred and thirty-two cities and counties in the **US**, with a combined population of over 20 million people, now have plastic bag bans or fees. In California, the state with the largest number of anti-bag measures in place, plastic bag purchases by retailers fell from around 48,000 tonnes in 2008 to around 28,000 tonnes in 2012, a decrease of around 42 %. The Department of Public Works reported that the January 2012 ban on plastic bags in retail stores and the €0.08 charge for paper bags, implemented in unincorporated areas of Los Angeles County, had led to a sustained 90 % reduction in single-use bag use at large stores by December 2013. A €0.04 charge for plastic and paper carryout bags at all food/



alcohol retailers in Washington DC, introduced in January 2010, led to four out of five households using fewer bags, with two thirds of residents reporting seeing less plastic bag litter after the tax came into effect (Larsen and Venkova 2014). It is estimated that the bag tax in Washington DC reduced grocery bag sales by somewhere between 67 and 80 % after two years (Beacon Hill Institute 2012).

In **Asia**, several countries have taken action against plastic bags, including complete bans (Bangladesh: 2002; Papua New Guinea: 2009), minimum thickness bans (Taiwan: 2001) or taxes (Taiwan, tax of €0.02–0.08 since 2003; Hong Kong, tax of €0.05 since 2009). The Taiwanese charge increased the number of people who regularly took used plastic bags to reuse when they went shopping (rising from 18 % in 2001 to 72 % in 2006), and the Hong Kong charge successfully reduced plastic bag use by 75 % in affected stores (Earth Policy Institute 2013). In the first year following the 2008 ban on the provision of free plastic bags in all shops in **China**, the National Development and Reform Commission (NDRC) estimated that supermarkets had reduced plastic bag usage by 66 % (amounting to 40 billion fewer bags used) (Block, n.d.). Raw data, however, do not indicate that this reduction in usage translated into a reduction in bags found as marine litter. International Coastal Cleanup (ICC) surveys in the Nanhui District of Shanghai found 129 plastic bags in September 2007 (pre-ban), 286 in September 2008 (year of ban introduction), then (post-ban) an average of 400 per site per cleanup in 2009, an average of 245 per site per cleanup in 2010, and an average of 1,294 per site per cleanup in 2012 (NOWPAP DINRAC 2014). It is not clear whether these fluctuations (and significant increase in 2012) are due to variations in the number of volunteers participating in the cleanups, illegal selling of bags, ineffective implementation of the ban, or bags from non-Chinese sources being washed up on the coast. Indeed, bans in Bhutan (1999), India (attempted first in 1999 and several times subsequently) and Bangladesh (2002) have largely failed to bring about a decrease in use due to poor implementation and enforcement (Earth Policy Institute 2013).

In **Australia**, South Australia banned plastic bags in 2009, the Northern Territory and Australian Capital Territory followed suit in 2011, and Tasmania banned very thin plastic bags in 2013. Several cities and towns have also introduced voluntary bans. The South Australian ban purportedly encouraged customers to bring their own bags more often (Earth Policy Institute 2013). New Zealand has had a voluntary retailer levy of €0.03–0.06 since 2009 (Miller 2012).

There are also myriad voluntary initiatives worldwide undertaken by individual retailers to attempt to limit the number of disposable bags they hand out to customers, ranging from eliminating disposable bags altogether to charging for disposable bags or selling reusable bags. In Spain, a voluntary agreement between Catalonia's Waste Agency, regional and national business groups, plastic bag manufacturers, food distributors and supermarkets led to a 40 % decrease in the consumption of single-use plastic bags between 2007 and 2011, and 87 % reduction (equal to 1 billion individual bags) in annual supermarket plastic bag use (Earth Policy Institute 2013).

From the examples outlined here, it is clear that bans and charges have had varying degrees of success, ranging from no discernible impact (failed bans in

Bhutan, India and Bangladesh) to reductions in bag use of over 90 % (charges/taxes in Ireland, Wales and parts of Los Angeles County). Sometimes impressive initial results are not sustained over time (Ireland, South Africa). This picture indicates that there is no one-size-fits all solution to the issue of plastic bags and that measures must be tailored to address different consumer/business behavior in different countries. It perhaps also suggests a need for measures that can be adapted to respond to failures following initial successes (e.g. increase in charges), and highlights the need for full implementation and proper enforcement of measures such as charges and bans to ensure their success.

### ***14.3.3 Packaging Producer Responsibility in the EU***

Extended producer responsibility (EPR) is an environmental policy approach in which a producer's responsibility for a product is extended to the post-consumer stage of a product's life cycle, meaning that they are responsible (financially and/or logistically) for dealing with the product when it becomes waste. This concept has been widely implemented in the EU over the past 20 years, with the introduction of a great variety of EPR schemes and the creation of producer-responsibility organisations (PROs), collective entities set up by producers or through legislation to meet the recovery and recycling obligations of individual producers.

Waste packaging forms a significant proportion of marine litter. Food wrappers, plastic and glass drinks bottles, bottle caps and drinks cans all regularly feature in the top ten most frequently found items during marine litter surveys; together these items comprised 31 % of all items found during the Ocean Conservancy's 2013 International Coastal Cleanup. When plastic and paper shopping bags, which may also be classed as packaging, are added, this figure increases to 37 % (Ocean Conservancy 2014). More than half of the plastic fraction of marine litter consists of plastic packaging waste (European Commission 2013a). This section therefore focuses on EPR schemes that deal with waste packaging.

All EU Member States have implemented EPR for packaging waste, since it is targeted by the Packaging and Packaging Waste Directive, 94/62/EC. In the 27 EU Member States in 2011 (prior to the accession of Croatia), 64 % of waste packaging was recycled (this figure includes composting for biodegradable packaging), and 77 % was recovered (this figure includes incineration with energy recovery). Within these figures there are of course variations between countries, ranging from only 41 % recycling in Poland to 80 % in Belgium, and only 45 % recovery in Malta to 97 % in Germany (Eurostat 2014b). Successful EPR schemes and the associated recycling infrastructures, including doorstep recycling collections, play a significant role in achieving high recycling and recovery rates and diverting packaging waste away from final disposal. Capturing packaging waste in closed-loop collection and recycling systems reduces the risk of items reaching the seas and becoming marine litter.

Typically in packaging EPR schemes, producers pay a fee to a PRO based on the amount of packaging they place on the market (for example, a fee per

tonne of paper/card, glass, aluminium packaging, with the fees for each material typically being different). These fees are then be used to cover, or contribute to, the cost of collection and treatment of waste packaging. Basing producers' contributions on the actual amount of packaging they place on the market ensures they pay their 'fair share' of the cost of waste management; this is the application of the producer-pays principle in practice, internalising the end-of-life costs into the cost of the product. This can incentivize producers to reduce the amount of packaging they place on the market, since this decreases the fees they pay.

A recent study (BIO by Deloitte et al. 2014) that looked at packaging EPR schemes in seven EU countries found that fees paid by producers ranged from just over €1 per capita per year in the U.K. to almost €20 per capita per year in Austria. The wide variation was primarily due to the different levels of cost coverage: fees from the purchase of Packaging Recovery Notes (PRN) in the U.K. were estimated to cover only 10 % of the total cost of the system, whereas in most of the other schemes reviewed, 100 % of the net costs of collection and treatment of separately collected waste were covered. Discussions with stakeholders during the study did not provide a consensus on whether producers should also finance the costs of dealing with packaging that is littered by consumers; measures that more directly target consumer behavior, such as deposit-refund schemes for packaging, have the potential to reduce littering. The more expensive schemes were not necessarily found to be the best in terms of recycling and recovery levels achieved. The highest recycling rates were achieved in Belgium: 85 % for household packaging and 82 % for commercial and industrial packaging, with costs per capita per year of around €8 and just over €1 respectively.

Many factors have an impact on the costs and performance of EPR schemes. Collection costs are typically higher in areas with lower population density. The historical development and quality of waste collection and treatment infrastructure is important, since economies of scale can be achieved through greater sorting and treatment capacities; EPR schemes can help to trigger infrastructure development and to finance improvements and maintenance. The value of secondary materials on national markets can be important, and can be influenced both by the demand for secondary raw materials and by the provision of high-quality materials once a recycling industry is in place. Citizens' awareness of separate collection schemes, and their willingness to participate, is also crucial, and investment in public communication can help EPR schemes to succeed. Other waste policy instruments, including those discussed in this chapter (disposal taxes) and others (pay-as-you-throw schemes, deposit-refund schemes, etc.) can complement EPR schemes and increase the efficiency of the general waste management system (BIO by Deloitte et al. 2014).

#### ***14.3.4 Charges for Port Reception Facilities***

With the abundance of ships using the oceans for shipping, transportation, tourism, military purposes and other maritime industries, there is a tremendous amount of

waste being generated at sea. It is therefore important for countries and their ports to provide adequate reception facilities for all of the types of ships that frequent those ports and all the types of waste they produce. It is also important to create proper incentives to encourage ships to use the correct facilities at ports, rather than dump waste into the sea. International regulations ratified by the signing members of MARPOL (73/78), the International Convention for the Prevention of Pollution from Ships, govern what kinds of waste can be discharged overboard and where they can be discharged. MARPOL prohibits the disposal of plastics at sea, as well as other garbage (though there are exceptions for food waste), and requires signatories to the convention to ensure that adequate reception facilities for ship-generated waste and cargo residues are established and that they are able to receive such waste from ships calling at the port without causing any undue delay. Although the convention provides a number of recommendations on how reception facilities can be established it does not specify how waste should be handled, and leaves all organisational issues to the responsible port authority. Compliance with the convention requires the efficient collection of waste from ships, and what happens thereafter is regulated by national legislation (for more details see Chen 2015).

One of the most important factors in incentivizing ship waste delivery is the waste fee system. Handling and disposing of waste is costly to ports (with the exception of oily waste, which, due to rising oil prices, has recently become economical to collect and recycle) (Øhlenschläger et al. 2013). Like the other services provided by ports, and in coherence with the polluter-pays principle, the cost of waste collection should rightly be recovered through the collection of port fees. However, high fees for waste collection can act as a disincentive to ships to discharge their waste at port, when they can throw their waste unseen overboard for free (if they can get away with it). It is therefore necessary to strike a balance between cost-recovery of waste handling and not discouraging disposal at port.

Because of the increasing number of illegal discharges in the Baltic Sea, in the late 1990s the HELCOM Convention provided a number of recommendations regarding the introduction of a 'No Special Fee' or 'indirect fee' in Baltic ports, leading to the introduction of a 100 % indirect fee system for solid garbage waste ('household' waste, oily waste from machinery space and sewage) by several Baltic Sea ports (Gothenburg, Copenhagen, Klaipeda, Helsinki and Stockholm) (Øhlenschläger et al. 2013). An indirect fee means that the cost of delivering solid garbage waste to port reception facilities is included in the fee paid by all ships visiting the port, irrespective of the quantities discharged, and is not specified on the invoice (Ikonen 2013).

The no-special-fee system effectively prevents cost from becoming a disincentive for using port reception facilities. Given that all ships will pay the fee regardless of use, they therefore all contribute to the financing of waste collection facilities, and this approach also has the benefit of reducing the fee. The administrative burden on port operators also appears lower when a 100 % indirect-fee system is adopted, because operators simply collect waste without the ports' financial administrations needing to calculate fees based on the actual amounts of waste delivered (Øhlenschläger et al. 2013). One negative effect that has been observed,

however, is that ships are incentivized to deliver oily waste at each port, rather than accumulating the waste on board and delivering only when slop tanks are full (Ikonen 2012; Øhlenschläger et al. 2013). This results in a costly and inefficient situation whereby smaller amounts of oily waste are being delivered at each port. For solid garbage waste this is not really a problem as more frequent deliveries of smaller amounts of waste are almost as convenient and cheap to dispose of as less frequent larger quantities (Øhlenschläger et al. 2013). In addition, as the fee is not proportional to the amount of waste produced, it does not encourage waste reduction on board vessels (Ikonen 2012).

There is no evidence for the effectiveness of the no-special-fee system implemented in the Baltic on trends in waste delivered in ports, as quantities of waste delivered to ports is influenced by many factors, and it is almost impossible to detect illegal dumping of solid waste because essentially you would have to catch a perpetrator red-handed. However, there are data on detected incidences of illegal oil spills in the Baltic (which can be observed using aerial surveys), which indicate a decline in illegal spills following the introduction of the no-special-fee (HELCOM 2012; Ikonen 2013; Øhlenschläger et al. 2013). Given that solid waste is easier to deliver than oily waste and is often delivered at the same time, it is reasonable to assume that the number of illegal waste discharges at sea also dropped over this period (Øhlenschläger et al. 2013).

## 14.4 Choosing Economic Instruments

Economic instruments to tackle marine litter should be designed so as to deliver three objectives:

1. Minimize production of marine litter.
2. To minimize the harm caused by marine litter.
3. To avoid unintended consequences from the application of the instrument.

Achieving all of these objectives is a challenge. This chapter has provided examples of the use of economic instruments to reduce different types of waste that contribute to marine litter or to target specific sources of such waste. However, there is a difference between reducing waste arisings and managing disposal, which is the focus of economic instruments currently applied to waste, and addressing the harm caused by marine litter.

Reducing the quantity of marine litter may depend on targeting key sources. For example, where waste enters the environment affects its ability to contribute to marine litter. A plastic bag dropped from a ship is more likely to become marine litter than one dropped on coastal land which is, in turn, more likely to become marine litter than one dropped 100 km inland.

Targeting the economic instruments to address marine litter that causes the most harm is particularly problematic. Marine litter causes different types of impacts and the harm arising from these varies—ghost fishing, suffocation by

plastic bags, introduction of toxic substances—such impacts may be unique to some types of waste or focused around particular types of waste. In contrast, the impact of marine litter on tourism due to the presence of litter on beaches is largely a factor of its total quantity (although some types of waste are particularly unpleasant or unsanitary).

In addressing marine litter, economic instruments can be used to reduce the impacts of such litter in a variety of ways. Such instruments may:

- Incentivize industries to use less plastic (packaging) either through economic disincentives/subsidies (internalizing external cost);
- Target waste arisings generally—such as with a landfill tax;
- Target specific types of waste—such as plastic bags;
- Target sources of waste most problematic for marine litter—such as shipping;
- Target individual types of marine litter—such as to reduce ghost fishing;
- Pay for the collection of litter;
- Target the toxicity of litter;
- Discourage polluting behavior.

Economic instruments have been adopted for some of these types of waste/litter. However, the toxicity of waste/litter is usually addressed through regulation controlling the quality of products or materials. The use of a regulatory approach on this issue in Europe, for example, is strongly linked to the single EU market. However, differential taxes or charges for products with materials that would have different toxicities in water are theoretically possible.

All instruments can have unintended consequences, that is impacts other than those for which the instrument is designed. The most obvious are costs to businesses, administrations or individuals. Economic instruments may have such costs—charges are an obvious cost, but administrations may incur costs to administer an instrument. However, where charges or taxes are levied these can be used to pay for their administration or contribute in other ways (e.g. funding awareness raising to ensure compliance, monitoring of instrument efficiency).

The choice of economic instrument also needs to consider the acceptance of the instrument by those affected. An instrument that results in additional costs (a tax, charge, etc.) may be resisted by some stakeholders. For example, ‘pay-as-you-throw’ schemes are strongly opposed by some communities, but not others. However, those same communities may welcome a reward scheme to encourage ‘good’ behavior funded by local taxes (yet this still results in costs to people). Acceptance may change over time: for example, where plastic bag taxes were introduced early resistance has largely disappeared as communities have seen the benefits of the schemes.

Finally, it is worth noting that there is discussion on the use of economic instruments to manage litter on beaches, i.e. the financing of its removal. For example, Birdir et al. (2013) undertook a willingness-to-pay study of beach litter in Turkey. Their conclusions suggested local taxes and collection boxes as means to fund beach cleanups. However, while it is appropriate to consider how such services are funded, these are not economic instruments to tackle the problem at source.

## 14.5 Conclusions

Marine litter is a complex problem to address, which exerts significant economic costs, often borne not by the polluter but by coastal and marine industries such as fisheries, aquaculture, tourism, etc. (some of which also contribute significantly to marine litter). Economic instruments have a potentially important role to play in addressing marine litter, with initiatives in place in several countries proving that they can lead to significant reductions in waste entering the environment (ten Brink et al. 2009).

The development of effective and efficient instruments requires a strong link between the behavior change driven by the instrument and the harm caused by marine litter. However, there are several areas where there is a lack of sufficient information to make this link. At the heart of this is the problem of understanding the harm caused by marine litter. The harm caused by some forms of litter is known, however, there are large gaps in this understanding.

While the presence of litter in the marine environment and even its ingestion, etc., in species is documented, it is not clear what impact it is having on critical populations of marine organisms or indeed species higher up the food chain (including humans). Further, while some specific types of litter are identified as having some impacts (discarded nets, plastic bags, etc.), the impacts of other types of litter are currently poorly understood, which is most notable with the debate on micro- and nanoplastics. In relation to socio-economic impacts, impacts on tourism from beach litter are documented, but a quantitative link between the impact and levels of litter is poorly understood (Ballance et al. 2000).

These links between types, quantities and sources of marine litter and their varied impacts are important to understand if targeted economic instruments are to be developed. Otherwise an instrument may lead to a reduction in litter, but with a limited reduction in impact.

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