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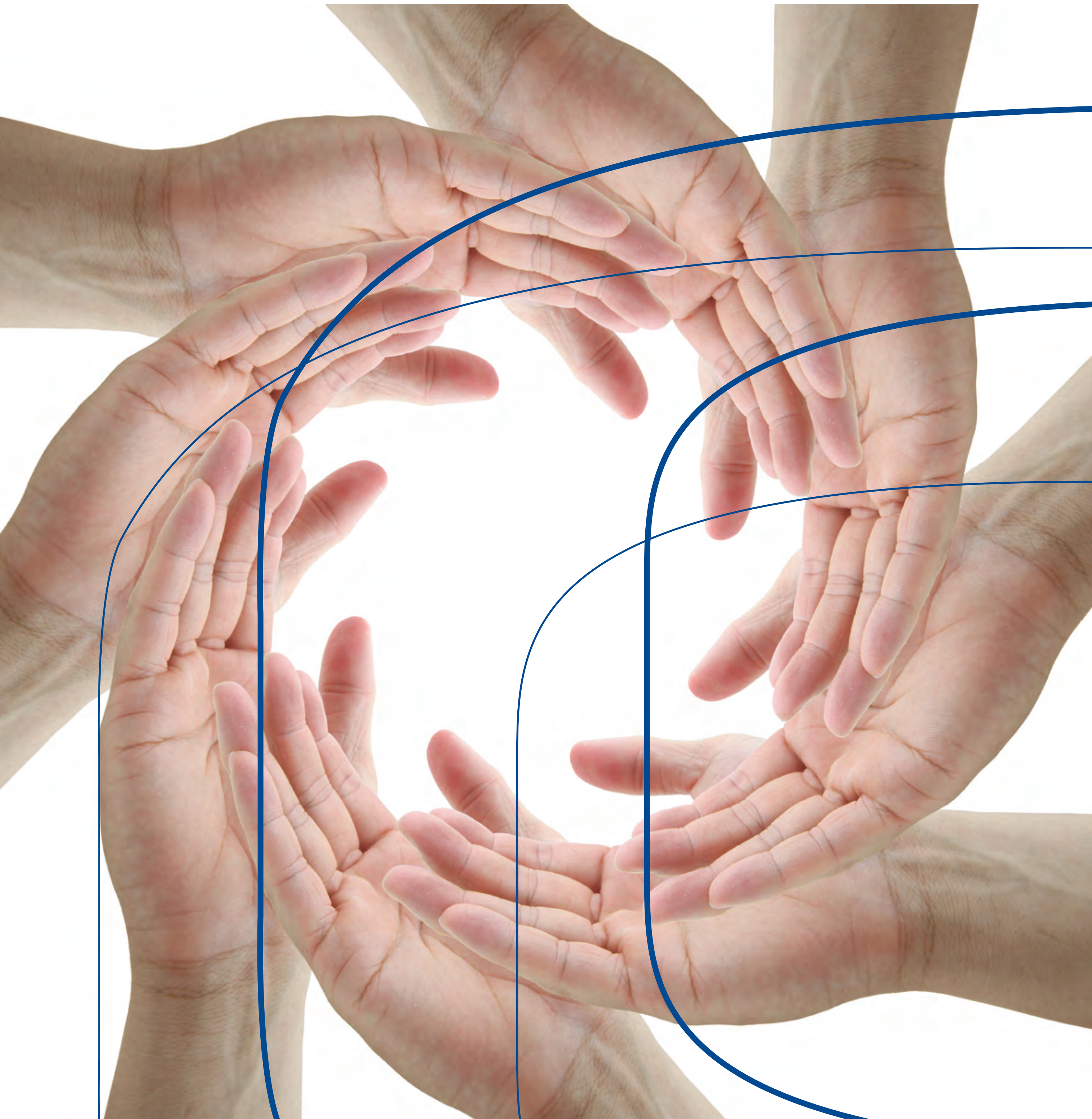
– A review with Nordic perspectives



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*Magnus Hennlock, Malin zu Castell-Rüdenhausen, Margareta Wahlström, Birgitte Kjær, Leonidas Milios, Eldbjørg Vea, David Watson, Ole Jørgen Hanssen, Anna Fråne, Åsa Stenmarck and Haben Tekie*

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# Content

# Foreword

Plastics are inexpensive and durable materials which easily can be shaped into a variety of products in a wide range of applications. As a result, production and consumption of plastics have increased significantly since the 1960s. Also waste plastic generation has grown significantly over the last decades creating the need for recycling in a sustainable circular economy. Recycling rates of plastics are currently relatively low in all Nordic countries. Plastics tend to be recycled into low grade products at the same time as at least Sweden, Denmark and Norway have relatively high incineration rates for waste plastics. Achieving a high quality of waste materials and recycling processes is a key challenge in closing resource loops for plastic. The Green Paper by the European Commission suggests looking at how economic instruments could be used to complement existing policy instruments in steering the waste flow through the waste hierarchy such that collection and material recycling of plastics can increase towards sustainable levels.

This report has been commissioned by the Nordic Working Group for Environment and Economics and the Nordic Waste Group. The report is structured in two parts. The first part provides a background on plastic waste flows in the Nordic countries. It presents an overview of existing policy instruments and the main challenges for designing policy instruments for improved recycling of plastic waste in the Nordic countries. The second part reviews the literature on policy instruments design and makes policy recommendations from a Nordic perspective.

The study has been written by the consultancy IVL Svenska Miljoinstitutet. Magnus Hennlock from IVL has acted as a project leader. The authors of the report are responsible for the content as well as for the recommendations which do not necessarily reflect the views and positions of the governments in the Nordic countries.

December 2014

*Magnus Cederlöf*
Chairman of the Nordic Working Group on
Environment and Economy

# Summary

This report summarizes the results from the project "Evaluation and design of economic instruments for increased recycling of plastic waste" initiated by the Working Group on Environment and Economy (MEG) and Nordic Waste Group (NAG). The purpose of the project is to evaluate and to identify overall design features of suitable economic policy instruments that may contribute to achieving socially efficient levels of recycling rates of plastic waste in Denmark, Norway, Sweden and Finland.

The first part (chapters 2–4) of the report provides a background on status and trends for plastic waste flows and treatment in Denmark, Finland, Norway and Sweden. It also gives an overview of existing policy instruments and an evaluation of the main challenges facing designs of policy instruments for achieving socially optimal recycling rates of plastic waste in these Nordic countries.

The second part (chapters 5–9) of the report uses the background results from the first part and reviews the economics research literature on policy recommendations for achieving optimal recycling rates. The policy recommendations naturally depend on the type of market failure that generates suboptimal allocations of recycling efforts across regions.

## *Existing National Targets and Policy Instruments in the four Nordic Countries*

Plastic waste is not specifically addressed by EU legislation and none of the four Nordic countries in this study has a specific plastic recycling target stated in their waste management plan. However, the Packaging Directive (94/62/EC amendments 2004/12/EC and 2005/20/EC) has a specific recycling target for plastic packaging. The minimum recycling target for plastics is 22.5% by weight, counting exclusively material that is recycled back into plastics. The national recycling targets for plastic packaging are higher than the EU requirements in both Norway and Sweden (30%). Denmark and Finland have the same target (22.5%) as set by the Packaging Directive. Based on the national reporting for 2011, the national targets are met in Norway and Finland, but not in Sweden. Denmark reported a slightly lower recycling rate in 2011 than required by the directive. The methods for calculating recycling rates differ be-

tween the Nordic countries though. Common methods in the Nordic countries for plastic waste streams could explore economies of scales in implementation. It would also open for using common instruments in the Nordic countries for exploring the larger scope of efficiency that exists at the Nordic scale rather than the scale of the single country.

The most commonly used economic policy instruments affecting waste plastic management in the EU-27 are producer responsibility schemes for specific waste streams such as packaging, deposit-refund systems for homogeneous products such as beverage bottles, charges and fees for waste disposal and treatment as well as landfill and incineration taxes and gate fees. When it comes to the use of major policy instruments affecting recycling of plastic waste in the four Nordic countries in the study, they can be summarized as follows:

- All four Nordic countries, except Sweden, use taxes on beverage packaging outside deposit-refund systems.
- All four Nordic countries have deposit-refund systems which include beverage packaging such as plastic bottles. Though the systems differ in the number of product types covered, the collection and recycling of packaging covered by the deposit system are in general high (85–95%).
- Plastic waste generation from packaging is part of an EPR scheme in Finland, Norway and Sweden. All producers and importers of plastic packaging (in Finland with a net turnover exceeding EUR 1 million) are legally responsible for organizing a collection and recycling system for the plastic packaging waste entering the markets. (In Finland the EPR system will cover household plastic packaging as of May 1st 2015.) In Denmark the municipality has the responsibility to establish a collection scheme for plastic packaging from households; the municipality is also responsible for the recycling of the collected waste back into plastic material. It is *likely* that this difference between the countries explains the somewhat lower material recycling rates seen in Denmark compared to the other Nordic countries in the study, which all have EPR schemes.
- All four countries have since several years back introduced landfill taxes. The taxes vary between countries from EUR 50–69/tonne. No studies have been identified which evaluate the effect of the tax on plastic recycling rates. It is difficult to evaluate the tax since most countries have had land fill bans at the same time.
- Sweden and Norway adopted incineration taxes for a number of years until they were abolished in 2010. They may have given

incentives for lower incineration rates. Denmark is now the only country that still has an incineration tax.

- Some municipalities use marginal-price instruments for garbage collection such as volume- or weight-based fees while other still use flat free-pricing. Evaluations show that municipalities that employ weight-based waste management fees generally experience higher collection rates than municipalities in which flat and/or volume-based fees are used.
- Denmark, Norway and Sweden have statutory bans or limitations on the landfilling of organic or combustible waste, while Finland will introduce a ban in 2016. It is *very likely* that this, in combination with high incineration capacities, explains the relatively high incineration rates seen in Denmark, Norway and Sweden. Similar patterns are seen in other countries with land fill bans. It is *likely* that an increase in incineration rate might be seen also in Finland after the implementation of the land fill ban 2016. It is recommended that the Nordic countries seek for policy solutions operative at the EU level to achieve optimal recycling rates across countries (see further section 9.3).

### *Design of Policy Instruments related to Producer and Consumer Choices*

The market failures related to producer and consumer choices illustrated in figure 6 in chapter 5 have been extensively analysed in the economics literature in the context of recycling rates. Input and output taxes or charges on production or consumption cannot in theory provide recycling incentives *per se* unless they affect behaviour towards substituting to other input materials that are more recyclable. In addition, plastics are used in a vast of applications, and the design of a tax or charge would need to take careful consideration such that it does not give incentives to unintended switches to other materials with even larger externalities.

In achieving optimal recycling rates, the economics literature rather points towards two-tiered instruments combining upstream and downstream measures as for instance deposit-refund systems, or in general upstream taxes/charges combined with downstream taxes/charges or subsidies in as illustrated figure 6 in chapter 5. The economic intuition behind these results is that the upstream-downstream design creates incentives for optimal allocation between reducing consumption and increasing recycling. The empirical evidences in the Nordic countries also shows that deposit and refund systems on beverage packaging have promoted recycling of plastic packaging in all four Nordic countries. The systems have ensured a high and uniform quality of collected plastic waste (85–95%).

Moreover, the Extended Producer Responsibility (EPR) systems with financial responsibility used in Finland, Norway and Sweden can to some extent be said to belong to this class of two-tiered instruments using up- and downstream measures. This is because the upstream charges, paid by producers, are used to finance the downstream collection and recycling of the waste generated by the products.

The major disadvantage of these two-tiered instruments is larger administrative costs connected to the monitoring and the verifying needed for charging and refunding. This is likely the reason why the deposit-refund systems in the Nordic countries have been used mainly for homogeneous standardised products such as beverage bottles and standardised reusable packaging such as pallets in the industry sector. The standardisation brings down the administrative costs for monitoring and verifying. It is also likely the reason why EPR systems tend to use simplified methods for calculating producer charges and fees, making the systems to deviate from marginal pricing that would be more optimal in theory (see chapter 8.2 for the case of EPRs in Finland and Sweden).

An extended use of deposit and refund systems, or EPR systems, based on marginal pricing in the Nordic countries is likely dependent on the possibilities of standardising and homogenizing products. The administrative costs would be high with deposit and refund systems for heterogeneous products, such as waste plastic in general. Using, for instance, weight as the pricing unit, the plastic content of the products would need to be weighted by production or purchase and again by recycling. One way to keep transactions costs lower in such systems is to reduce the number of actors in the system, suggesting that producers and recyclers, rather than households, could be actors in the systems. Moreover, simplified measures or proxies for measuring marginal quantities (weight, volume or units of products etc.) should be developed in order to keep monitoring and verifying costs low. Common methods in the Nordic countries for certain waste streams could explore economies of scales in the implementation.

When product design is essential for recyclability the upstream-downstream instruments could build on "recyclability indictors" that connect to product design. Literature suggests that several types of economic instruments based on up- and downstream measures can be designed as first best solutions to take into account aspects of product design externalities related to producer decisions in production. Instruments that connect to recyclability indicators could involve for instance, upstream taxes on production processes, charges, fees, disposal-content fees, subsidies for recycling in deposit refund systems as well as EPR schemes. Again the optimal design of such instruments would be a trade-off between the improved product design (in recyclability terms) and the increase in administrative costs for monitoring and verifying the recyclability indicators connected to product design.

There is an extensive body of economics research supporting that marginal cost pricing, or in general unit-based pricing, for collection services is more efficient than flat fees that do not give incentives to increased sorting. Unit-based pricing makes it possible in theory to set the price per unit of garbage collected equal to the social marginal cost of collection and recycling. Historically countries have used to charge flat fees or general municipal taxes to households for waste collection. However, recently variable fees have been implemented in several places. The majority of empirical studies, also in the Nordic countries, suggest that weight-based or volume-based pricing have larger effects than flat fees as predicted by economic theory.

The largest waste plastic flow, plastic packaging waste from households, also includes food containers where the plastic packaging many times has been in direct contact with food. Recycling these containers is combined with larger inconveniences (due to washing or the smell while storing it at home) for households. These inconvenience costs in combination with the low density of waste plastic (making the marginal incentive effect on waste plastic relatively smaller than for other more heavy substances in the mixed household waste) produce a caveat of relying on weight-based systems for promoting plastic waste recycling from households. Further analysis on the magnitude of these effects should be implemented before conclusions can made about the effects of weight-based pricing on waste plastic recycling from households.

### *Design of Policy Instruments related to Recyclers and Reprocessors Choices*

Common for the Nordic markets for waste plastics are demands for a relatively high quality of waste plastic compared to the quality levels supplied. Recycled plastic does not always meet the quality specifications that plastic manufacturers of technical high quality plastic products demand. There is also a tendency that demand is driven by the lower relative price of recycled plastic with respect to virgin plastic at the same time as demand for higher quality recycled plastic has increased during the last years.

The survey to managers in recycling and manufacturing industry in Sweden showed that one explanation for the insufficient supply of higher quality of waste plastic might be asymmetric information between recyclers and manufacturers about the quality of waste plastics on the market. An extension of the seminal Fullerton and Kinnaman (1995) model was developed in this project to analyse the effects and policy implications under asymmetric information. The result suggests that asymmetric information lead to adverse selection in terms of lower efforts in sorting by recyclers resulting in inefficient recycling rates due to lower quality of the waste plastic supplied. However, the presence of asymmetric information did not change the optimal instruments in the model (deposit-refund systems or in general combined upstream-downstream instruments).

In the survey in Sweden managers were asked to grade how effective they believed that different policy instruments would be to increase the supply of waste plastic. EU certification schemes for waste plastic quality were graded and ranked as the second most important instrument following weight-based pricing (managers in recycling industry) and a virgin tax (managers in manufacturing of final goods in plastics). It is notable that managers graded EU certification schemes as more important than subsidies to production. The call for EU certification schemes on waste plastic quality among sellers may be seen as a wish to better signal potential quality of their waste on the recycling market.

Before any policy recommendation is given, it is advisable to further analyse if, and if so to what extent, the lower quality of recycled plastic seen in the Nordic markets can be explained by inefficiency due to asymmetric information between recyclers and manufacturers in the Nordic countries.

### *Policy Instruments for Achieving Optimal Recycling Rates at International Markets*

Small countries, like the Nordic countries, in the EU face small volumes of waste plastic. In addition to this, a majority of the Nordic countries tend to be geographically large with relatively low populations resulting in higher transport and infrastructure costs in waste management relative to many other countries. Moreover, incineration of plastic as part of the residual waste often exhibit lower costs than local sorting and recycling (partly due to the relatively high labour costs for waste treatment).

From a global level it may then be more efficient to maintain lower material recycling rates in Nordic countries and/or export plastic waste to other countries that better can take advantage of the economies of scales in waste plastic recycling technologies. International designs of the economic policy instruments discussed above, for instance a single EPR or harmonized EPRs across countries, a single deposit-refund system or harmonized deposit-refund systems across countries, or alternatively international tradable quotas based on EPR, could in theory result in an efficient outcome at the international level. These systems could in principle be designed to operate at the Nordic level, the EU level, or at an even larger geographical area including also non-European countries. The policy recommendation is to further analyse such international policy instruments primarily at the EU level, eventually together with an EU-wide certification for quality of recycled plastic and/or the end-of-waste criteria for waste plastic as suggested by the European Commission (Villanueva and Eder, 2014).

However, for achieving socially efficient allocations of recycling rates across countries at the international level, the design of instruments used (including other instruments such as EU ETS) should take into account the social costs related to transport and export of plastic waste to other countries in the system. They should also take into account life cycle costs in the comparisons of domestic treatments (nevertheless incineration in the Nordic countries) and treatments in importing countries. For instance, LCA studies carried out for plastic packaging waste treatment systems shows that material recycling has advantages compared to incineration, both with regard to GHG emissions and energy resources (WRAP 2006, Raadal *et al*. 2009, Lyng & Modahl 2011, Rigamonti *et al*., 2014). However, these results are sensitive to the amount of virgin materials that is really substituted by recycled materials, as substantial amounts of plastic waste in some cases has to be sorted out due to low quality and ends up in incineration plants.

# 1. Introduction

## 1.1 Background

Recycling rates of plastics are currently relatively low in all Nordic countries. Plastics tend to be recycled into low grade products at the same time as at least Sweden, Denmark and Norway have relatively high incineration rates for waste plastics. The treatment of waste plastic could move up in the waste hierarchy to be in accordance with EU legislation and in line with the EUs Green Paper "On a European Strategy on Plastic Waste in the Environment". Achieving a high quality of waste materials and recycling processes is a key challenge in closing resource loops for plastic. This report reviews the findings in economics research literature on policy instruments that may have potential to lead to socially optimal recycling rates of waste plastics.

The Green Paper puts a caveat on landfill bans that may lead to a dominance of energy recovery over recycling which is in conflict with the waste hierarchy. At the same bans and restrictions for disposal of waste plastic on landfills have been implemented leading to an increase of using plastic waste in incineration as the last resort while material recycling has not increased by the same amount. This pattern is seen in Sweden, Denmark and Norway with relatively high incineration rates for waste plastics. The Green Paper suggests looking at how economic instruments could be used to complement existing policy instruments in steering the waste flow through the waste hierarchy such that collection and material recycling of plastics increases. This project responds to that call with specific focus on Nordic countries.

## 1.2 Purpose

The review has taken a broad view of the economics research literature on policy instruments when it comes to designing policy instruments for achieving socially optimal recycling rates of waste plastics from a Nordic perspective. The scope of the project is waste plastic from industry and plastic packaging from households in Denmark, Norway, Sweden and Finland, hereinafter sometimes denoted as "the Nordic countries". The project includes policy designs for achieving socially efficient output levels of high quality plastic materials from recycling systems in the four Nordic countries. As long as possible the review is based on existing research literature and results. In a few cases, existing literature did not provide answers. In these cases, a survey and some research were conducted within the constraints allowed by the limited time and resources in the project.

## 1.3 Implementation of the Project

The project was implemented in two steps, the first step collected background information on technologies and key actors in waste plastics recycling and reprocessing as well as current national and EU policy affecting plastic waste recycling in the four Nordic countries. The country-specific background information, which was collected by the help of IVL partners Copenhagen Resource Institute, Denmark, VTT Technical Research Center, Finland and Ostfold Research, Norway, was used to identify the major market failures and externalities in the recycling process of waste plastics from a Nordic perspective.

The second step performed the review of the economics research literature on the empirical effects of existing policies as well as the theoretical literature on the optimal design of efficient policy instruments.

The literature contains several studies of the effects of policies on household recycling behaviour while there are barely any studies of policy effects on the industries of plastics recycling, reprocessing and manufacturing. A survey was therefore conducted to managers in these industries exploring their experiences from the markets of plastic recycling and manufacturing in Sweden. The reason for choosing Sweden was that IVL could arrange contacts with managers in Sweden within the limited time and resources in the project. Since the waste plastics market is highly international we expect that the results would be representative also for Denmark, Norway and Finland. The survey was sent to 90 managers, of which 62 completed the survey corresponding to a response rate of 69%. Despite the high response rate it was not enough for running an experiment with randomised treatments.

The project was carried out by IVL Swedish Environmental Research Institute, Sweden (chapters 1–9) with partner organisations Copenhagen Resource Institute, Denmark, Ostfold Research, Norway, and VTT, Finland (providing country-specific information in chapters 2–4). Birgitte Jørgensen Kjær, Leonidas Milios, Eldbjørg Vea and David Watson at

Copenhagen Resource Institute, Denmark, Ole Jørgen Hanssen, Ostfold Research, Norway, Malin zu Castell-Rüdenhausen and Margareta Wahlström at VTT Technical Research Center, Finland and Anna Fråne and Åsa Stenmarck at IVL Sweden have contributed with invaluable contributions to the background material in chapters 2–4.

Haben Tekie at IVL, Sweden and Ida Muz at the Department of Economics, University of Gothenburg, Sweden assisted with contacting managers of recyclers and manufacturers for the survey. Magnus Hennlock at IVL has been the project leader of the project.

## 1.4 Structure of this Report

Chapter 2 provides a background on status and trends for plastic waste flows and plastic waste treatment in Denmark, Finland, Norway and Sweden. Chapter 3 provides an overview of existing policy instruments in these countries while chapter 4 makes a comparison and an evaluation of the main challenges facing designs of efficient policy instruments in these countries. Chapter 5 contains a theoretical introduction to the problem by identifying the major private decision-makers in a circular economy for plastics and potential major market failures. Chapter 6 presents a review of the research literature which has studied policy effects on household behaviour, while chapter 7 presents the results from the analyses of recyclers of waste plastic and manufacturers of plastic products. Chapter 8 summarize the review of policy instruments highlighted in the economics research literature as potential instruments for achieving optimal recycling rates. Chapter 9 provides a concluding summary with policy recommendations. Finally, chapter 10 contains references with a reference list of scientific (peer-reviewed) articles and a reference list of other sources such as books, reports and government documents.

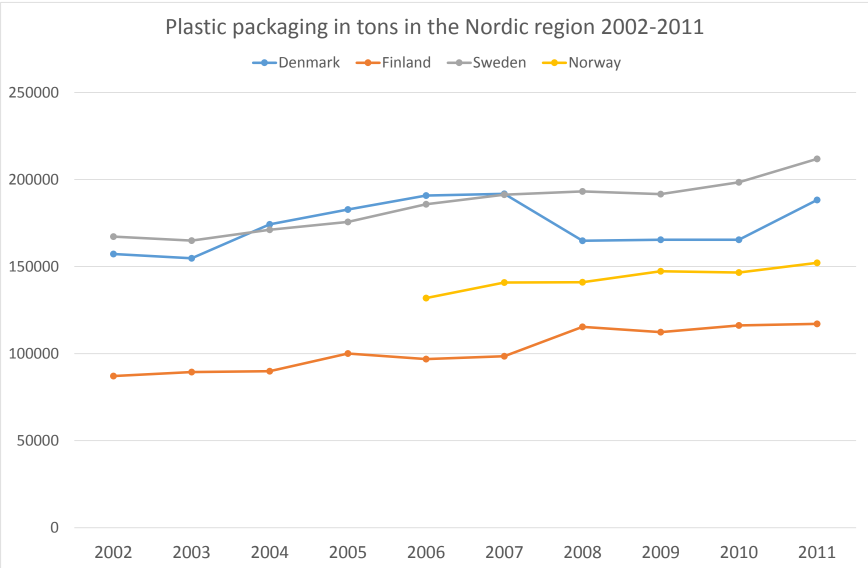
# 2. Status and Trends of Plastic Waste in Nordic Countries

## 2.1 Plastic Packaging Waste flows

Plastics are inexpensive, lightweight and durable materials, which easily can be shaped into a variety of products that find use in a wide range of applications. Not surprisingly, production and consumption of plastics have increased significantly over the last 60 years (Hopewell *et al.*, 2009). As a result, waste plastic generation has grown significantly over the last decades.

The total amounts of plastic packaging waste in the four Nordic countries in this study increased during the period 2002–2011, and varied between 120,000 and 220,000 tons in 2011. Data on development in amounts generated of plastic packaging waste in the four Nordic countries and EU27 during 2002–2011 is presented in Figure 1, based in data from Eurostat statistics.

***Figure 1: Plastic packaging waste from Denmark, Finland, Sweden, Norway and EU27***




Source: Eurostat.

Eurostat statistics are collected from each country according to standard categorization of waste flows and economic sectors, but comparisons between countries should in general be done with care as the methodologies for data gathering might differ from country to country (see Hanssen *et al*. 2013).

The total amounts have increased with 20–35% in the period 2002–2011. In Norway, it has been registered an increase in amount of plastic packaging from 0.5 tonnes/million NOK in turnover to 0.6 tonnes between 2005 and 2011, based in figures from a number of large packaging materials users. All countries showed increases in amounts of plastic packaging waste per capita in the period, ranging between 5–10 kg or between 15% and 30% as shown in table 1.

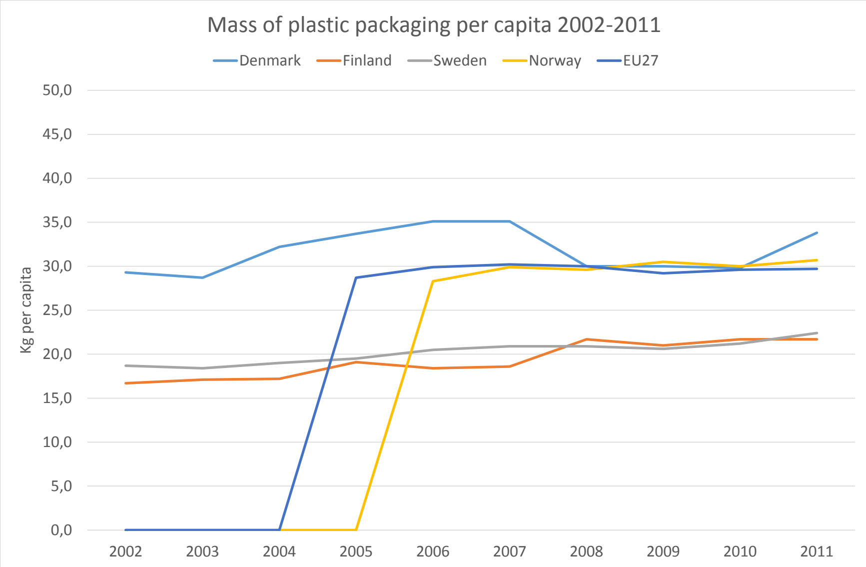
**Table 1: Percentage increase in plastic packaging waste 2002–2011**

| Country | Changes kg per capita (% change between 2002–2011) | Changes in total amount (% change between 2002–2011 |
|---|---|---|
| Denmark | 15.4 | 19.7 |
| Finland | 29.9 | 34.5 |
| Sweden | 19.8 | 26.7 |
| Norway* | 8.5 | 15.3 |

Source: Eurostat.

The packaging waste flow per capita during the same period is shown in figure 2. Data are lacking from Norway before 2006 and from EU27 before 2005. Finland and Sweden have quite similar, steadily increasing, but quite low amounts per capita, about 20 kg + 3 kg in the whole period.

*Figure 2: Plastic packaging waste per capita in Denmark, Finland, Sweden, Norway and EU27*



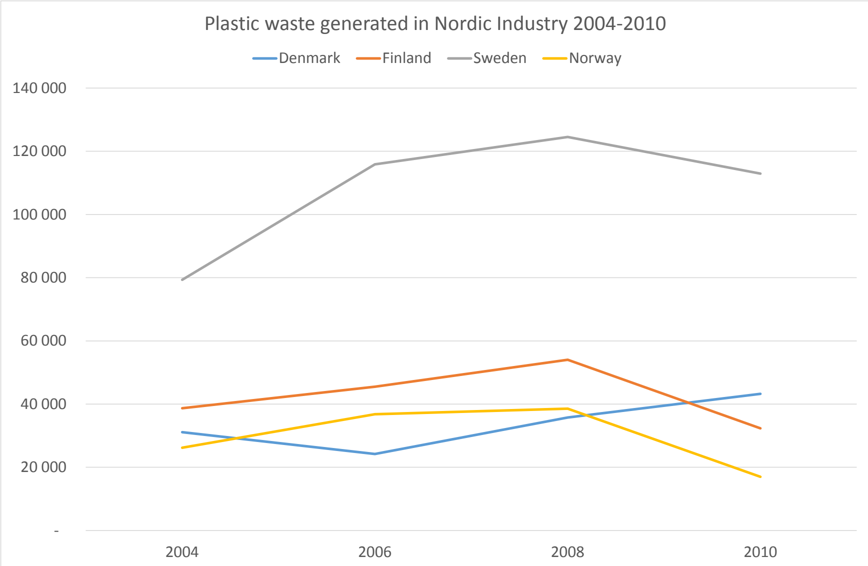

Source: Eurostat.

Norway and EU27 are also quite stable and similar, but on a significant higher level than Finland and Sweden, with about 30 kg per capita. Finally Denmark shows more fluctuating figures between 30 and 35 kg per capita, but with about 30 kg over the last two years from 2010–2011.

## 2.2 Industrial Plastic Waste Flows

Data on industrial plastic waste is generated from the statistics for "Manufacturing Sector" under Eurostat, and cover the years 2004–2010. Most countries showed an increase in total amount of plastic waste from 2004 to 2008, followed by a slight decrease up to 2010, with Denmark as an exception as shown in figure 3.

*Figure 3: Total mass of industrial plastic waste from Denmark, Finland, Sweden and Norway 2004–2010*



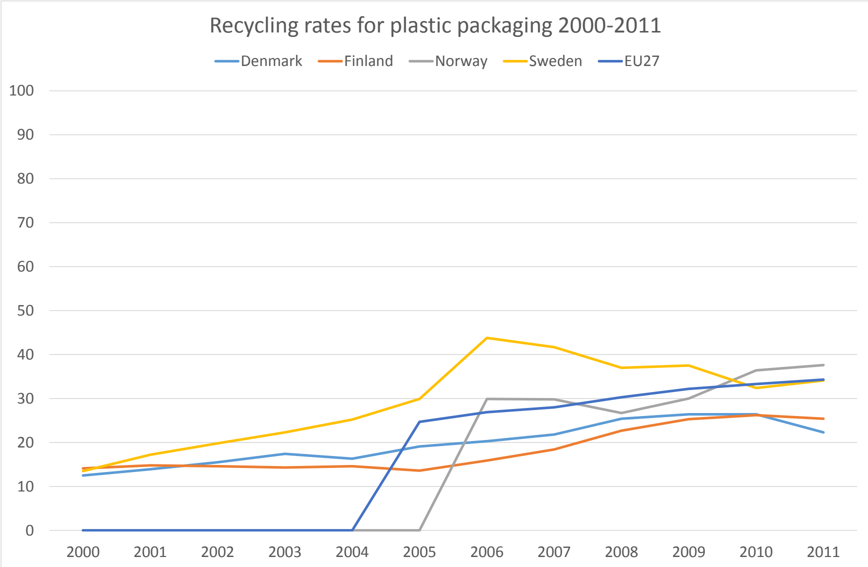

Source: Eurostat.

The statistics cover only producers of plastic and plastic products, including packaging producers. It is not possible from Eurostats statistics to get data on how plastic waste from the industry is treated in the different countries.

## 2.3 Plastic Packaging Waste Treatment

Sweden, Finland and Denmark had about the same recycling rate of plastic packaging waste from the starting point in 2000, but recycling of plastic packaging waste in Sweden increased steadily up to almost 45% in 2006 in figure 4. Since then the recycling rate was reduced to about 35% in 2010 and 2011. In the last part of the period, Norway had the highest recycling rate with about 38%, followed by Sweden with 30%, whereas Finland and Denmark had about 25% each. The recycling rate increased from between 12–15% to about 22–35% in 2011 for the group of countries. The joint EU27 rate was about 35%, increasing steadily from the first figures in 2005 as shown in figure 4.

*Figure 4: Rate of recycling of plastic packaging waste in the Nordic region and EU27*




Source: Eurostat.

Table 2 shows the total amount of plastic packages waste in Denmark, Finland Norway and Sweden that was collected for recycling and incineration.

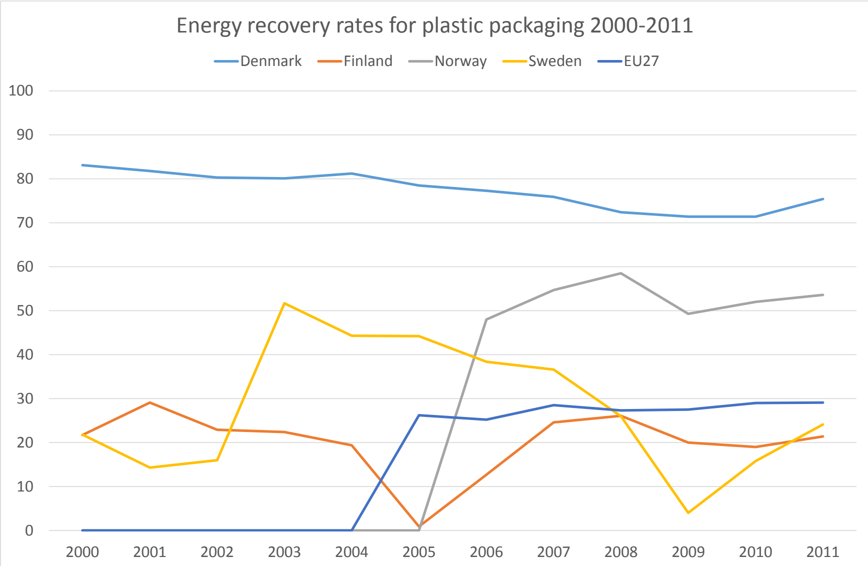
**Table 2: Collected amounts of plastic packaging waste in Denmark, Finland, Sweden and Norway to recycling and incineration**

| Country | Separately collected plastic packaging recycled (tonnes) | Separately collected plastic packaging incinerated (tonnes) | Total amount of plastic packaging waste collected (tonnes) |
|---|---|---|---|
| Denmark (2009) | 45,937 | 0 | 45,937 |
| Finland (2011) | 29,726 | 25,000 | 53,768 |
| Sweden (2010) | 45,560 | 77,030 | 123,500 |
| Norway (2012) | 54,424 | - | 54,424 |

Source: Miljøstyrelsen (2009) and Fråne *et al*. (2013).

With regard to energy recovery, Denmark has had the highest energy recovery rate among the countries the whole period, although the trend decreased steadily from 2000–2011. In the other countries there have been more fluctuations over time. Sweden showed a big increase in recovery rate between 2002 and 2003 from about 17% to about 52%, but this has later on been reduced back to about 20% in 2011. The recovery rate in Finland has varied between 20 and 30% (except for the year 2005 where data is lacking) and has been quite stable around 20% from 2009–2011 in figure 5. There was no data available for Norway before 2006. The energy recovery rate has varied between 50 and 60% over the years up to 2011.

***Figure 5: Energy recovery rate for plastic packaging waste in the Nordic region and EU27***




Source: Eurostat.

For EU27, the recovery rate has been quite stable slightly beyond 30% between 2005 and 2011. It should be noted that statistics is not fully comparable between the Nordic countries, as the energy recovery rate is based on source-sorted plastic packaging waste in Sweden, whereas the rate. In Norway includes plastic packaging in combustible residual waste fractions, leading to a relatively high energy recovery rate for plastic packaging waste in total.

## 2.4 Industrial Plastic Waste Treatment

In Eurostats databank for the manufacturing sector it is not possible to get access to data on treatment of pre-consumer plastic waste from the industry. Data on treatment has thus been collected from national statistics in each country, which is not always up to date. The data from the manufacturing sector include data from packaging industry and packaging users, which then will be double-counted in statistics from the industry as a whole. Data from Statistical Bureau in Norway shows that about

45% of the plastic waste was recycled to new materials in 2008, which is the last year with statistics available. The rest was incinerated, where 22% was energy recovered and 31% was incinerated without any energy recovery. This has probably changed since 2008, since the incineration capacity with energy recovery has increased since 2008. The figures from Sweden come from the latest official waste statistics from 2010 (Naturvårdsverket 2010). Pre-consumer industrial waste that is recycled within the factory itself (internal recycling) is not included. The amount of total recycling of plastic waste is actually lower or very similar to the total amount of plastic packaging waste reported as recycled the same year. The figures do not seem that reliable. The figure to energy recovery includes both plastic *and* rubber, both to energy recovery for heat and power production and use as fuel in the industry, e.g. mineral sector. There are no authoritative data available from Denmark, but it is estimated a total amount of 129,000 tonnes of plastic waste are generated in the business sector, of where 50% or about 65,000 tonnes is being recycled and about 51,000 tonnes incinerated per year. About 10% is assumed to be landfilled (shredder waste etc.). Also here is the statistics quite uncertain and must be treated with care, as the data are incomplete. In Finland, about 44% of a total of 41,000 tonnes was recycled, whereas about 51% was incinerated (table 3).

**Table 3: Treatment rates and amounts send to recycling and incineration of plastic waste from the industry in Nordic countries**

| Country | Plastic waste recycled (tonnes/percentages) | Plastic waste incinerated (tonnes/percentages) | Total amount of plastic waste (tonnes) |
|---|---|---|---|
| Denmark | 64,500 (50%) | 51,600 (40%) | 129,000 |
| Finland[12] | 18,000 (44%) | 21,000 (51%) | 41,000 |
| Sweden[3] (2010) | 45,000 | 110,000 | 155,000 |
| Norway (2008) | 16,000,(45%) | 19,000 (55%) | 35,000 |

Source: Statistical Bureau in Norway, Statistics Finland and Naturvårdsverket (2010).

[1] http://www.stat.fi/til/jate/2012/jate_2012_2014-05-15_tau_002_en.html
[2] http://www.stat.fi/til/jate/2012/jate_2012_2014-05-15_tau_001_en.html
[3] Statens Naturvårdsverk (2010).

# 3. Existing Policy Instruments in the four Nordic Countries

This chapter contains an overview of existing policy instruments in Denmark, Finland, Norway and Sweden which have relevance to recycling of plastic waste. The first section starts with a short overview of EU legislation relevant to recycling of waste plastics as this will be a binding framework also for implementing national polices in Nordic EU countries. Denmark, Finland and Sweden are EU Member States and must implement EU legislation. While not a Member State of the EU, Norway must, via the European Economic Area agreement, implement all environmentally related EU Directives. An overview of the green paper of plastic waste by the EU commission as well as an outlook to made EU initiatives affecting plastic waste is also included.

Chapter 4 makes a comparison between existing policies in Denmark, Finland, Norway and Sweden and can therefore be read as a summary of this chapter.

## 3.1 Framework for Plastic Waste in EU Legislation

The EU waste legislation is usually presented in three categories: the first category A) contains the Waste Framework Directive (2008/98/EC) on waste, and other directives, providing the general framework of waste management requirements, basic waste management definitions, classification system for wastes, and distinctions between hazardous and non-hazardous wastes. The second category B) contains legislation on waste management operations while category C) contains legislation on specific waste streams. The waste legislation and policy of the EU member countries shall apply as a priority order the following waste management hierarchy, 1) prevention, 2) preparation for reuse, 3) recycling, 4) recovery, and 5) disposal as the priority order in waste management.

The Directive also introduces the "polluter pays principle" and the "extended producer responsibility". It includes two new recycling and recovery targets to be achieved by 2020: 50% preparing for re-use and recycling of certain waste materials from households and other origins similar to households, and 70% preparing for re-use, recycling and other recovery of construction and demolition waste. The target has to include at least paper, metal, plastic and glass components of household waste and similar streams. A large part of household waste of these four materials is included in packaging. Therefore, recycling of packaging waste from households will have a strong influence on the overall recycling rate for household waste. The Directive further requires that Member States adopt waste management plans and waste prevention programmes.

There is still no specific EU legislation that strategically addresses plastic waste. The issue of plastic waste is spread over the legislation, for instance the separate plastic waste collection target 2015 in the Waste Framework Directive and the 50% household waste collection target by 2020. When it comes to packaging, which includes the largest plastic waste stream, the EU Packaging and Packaging Waste Directive (94/62) was the first directive that put forward quantified targets for certain substances in packaging, as well as prescribing by what means packaging waste should be recycled or recovered. The directive also includes qualitative objectives for the prevention of packaging waste and promoting of reuse.

According to article 174, the Treaty for the European Union underlines the principle of the polluter plays principle which is in line with achieving the targets and objectives in the directive by economic instruments. This is also put forward in article 15, and furthermore by allowing for the introduction of the Extended Producer Responsibility (EPR) on packaging in article 4 and article 6 of the Packaging and Packaging Waste Directive (EC 94/62). While EPR is not mandatory in the Packaging and Packaging Waste Directive it is so in the legislation for Waste Electrical and Electronic Equipment, End of Life Vehicles and Batteries. These directives cover waste steams which include plastic waste, although they have no specific targets for recycling of plastics. Finally, plastic waste can be included in the target for recycling and recovery of C&D waste in the Waste Framework Directive (Article 11(2)).

When it comes to packaging, the EU member countries have been given certain degrees of freedom to meet the requirements of the Packaging and Packaging Waste Directive. The most common economic policy instrument used among EU member countries are different designs of Extended Producer Responsibilites (EPR), other deposit-refund systems, and taxes on for instance packaging and land fill.

## 3.2 The Green Paper – On a European Strategy on Plastic Waste in the Environment

The EU Commission recently presented the Green Paper "On a European Strategy on Plastic Waste in the Environment" which is a reflection on possible responses to policy challenges regarding plastic waste which are at present not specifically addressed in EU waste legislation. The Green Paper is also an integral part of the review of the waste legislation that will be completed in 2014.

A general aim of the Green Paper consultation was to obtain answers to how policies on plastics can be brought in line with the Roadmap's objectives and how plastics can have a future in a circular economy. The consultation received a large number of responses from the industry, NGOs and public authorities in the member countries. The support of the waste hierarchy was confirmed; there was also consensus that landfilling and to some extent incineration of plastic waste should be reduced as much as possible, while recycling rates increase as much as possible. Although the views were diverse on to what extent voluntary versus mandatory measures are needed to reach these objectives, a majority of the replies signalled support for the following instruments:

- Plastic waste landfill ban.
- Improved doorstep collection and separation.
- More and higher targets for plastic recycling.
- Stricter export controls.
- Introduction of business systems (e.g. deposit and return schemes, leasing, pay-as-you-throw (PAYT).
- Better consumer information (e.g. on recyclability).
- Better use of eco-design instruments (better design, restriction of additives, abolish planned obsolescence).
- Increased use of market-based instruments.
- Define end-of-life criteria for plastic waste.

The Green Paper furthermore identifies that plastic landfilling rates remain high in those member countries which lack alternative treatment methods and lack effective economic instruments. Even member countries with high recovery rates and landfill bans (including several Nordic countries) achieve only modest plastic recycling rates (pp. 8–10, EC 2013). The Green Paper also puts a caveat on landfill bans that may lead to a dominance of energy recovery over recycling which is in conflict with the waste hierarchy. This can be clearly seen in Sweden, Denmark and Norway with relatively high incineration rates for plastics. For this purpose the Green Paper suggests looking at how economic instruments could be used to complement existing policy instruments in steering the waste flow through the waste hierarchy such that collection and material recycling of plastics increases.

The most commonly used economic policy instruments affecting plastic waste management in the EU-27 are charges and fees for garbage collection, producer responsibility schemes for specific waste streams (for instance packaging and beverage bottles) and charges and fees for waste disposal and treatment including landfill and incineration taxes and restrictions. Due to a lack of time series data on the change of the charges and tax levels the impact of policy instruments on municipal plastic waste cannot be estimated. Nevertheless the methods for measuring plastic waste flow is either lacking or differ across countries which would make statistical analysis across countries problematic. The national policy instruments are further discussed in section 3.5.

## 3.3 EU Legislation Initiatives

The Commission has reviewed key targets under the Waste Framework Directive, the Landfill Directive and the Packaging and Packaging Waste Directive. The basis for the review of the targets is twofold. Firstly, it is to respond to the review clauses set out in the Directives. The second objective is to bring these targets in line with the Commission's ambitions on promoting resource efficiency and reducing greenhouse gas emissions. In recent years the Commission has published a number of Communications which give a clear orientation for this exercise:

- The Resource Efficiency Roadmap including 2020 aspirational targets.
- The Raw Material Initiative highlighting the importance of recycling to secure access to materials in the future.
- The Report on progress against the Thematic Strategy on Waste Prevention and Recycling which identifies remaining challenges, and proposals for the future.

The Roadmap to a Resource Efficient Europe COM (2011) 571 set an aspirational target that by 2020 waste generation per capita in the EU is reducing; reuse and recycling are at their "maximum levels"; European waste policy has been fully implemented, energy recovery is limited to wastes which could not otherwise be recycled and the use of landfill has been "virtually eliminated".

It also suggested several ways to increase resource productivity and reach a sustainable development by decoupling economic growth from resource. When it comes to waste the objectives of the Roadmap raises reuse and recycling, to avoid recyclable materials for energy recovery and to landfill only residual waste. One of the major movements is a shift from taxation of labour towards environmental taxation in accordance with polluter pays principle. Environmental taxation has been implemented before in waste management in several member countries, for instance landfilling and incineration. Still there are only a few examples on taxation of goods (e.g. taxation on packaging) with the aim to reduce waste generation.

The European Commission adopted the Communication "Towards a circular economy: a zero waste programme for Europe" in July 2014 (COM/2014/0398 final). As part of the circular economy package, the Commission also put forward a legislative proposal to review recycling and other waste-related targets in the EU (COM/2014/0397 final). The legislative proposal only included specific targets for plastic waste in the field of packaging. The minimum target for preparing for re-use and recycling of plastic packaging waste is proposed as 45% by 2020 and 60% by 2025.

For municipal waste (household waste and similar streams) a recycling target of 70% is proposed for 2030. Plastic waste is part of municipal waste. Furthermore, a landfill ban of recyclable waste is proposed from 2025. This also includes a landfill ban on recyclable plastic waste which might be a driver for more recycling (COM/2014/0397 final).

The Commission has not yet proposed specific recycling targets for plastic waste from industry, agriculture or fisheries.

## 3.4 Implementation of EU Legislation in Nordic countries

As EU member states Denmark, Finland and Sweden must implement EU legislation, while Norway, via the European Economic Area agreement, must implement all environmentally related EU Directives. In 2004, the EU Packaging and Packaging Waste Directive (94/62/EC) was reviewed to increase the targets for recovery and recycling of packaging waste. The recycling target to fulfil by 2008 was: 55% for total packaging waste, 60% for paper and cardboard, 60% for glass, 50% for metal, 22.5% for plastic and 15% for wood. Denmark, Finland, Norway and Sweden have all implemented the Directive and fulfilled the recycling targets set for 2008.

The Waste Framework Directive (2008/98/EC) (WFD) does not set a specific target for plastic waste from households. However, the WFD sets a 50% recycling target for household waste and similar waste streams by 2020 (Article 11(2a)). Denmark, Finland, Norway and Sweden have all implemented the WFD. Countries are given four alternative methods by which they can calculate and report on achieved recycling percentages (EC 2011b). One of the calculation methods is based on recycling of municipal waste. Based on this calculation method the level of recycling of municipal waste in the four countries in 2012 was in the range 33–48% (Eurostat 2014). As such, all of the countries have yet to reach the 2020 recycling target.

## 3.5 National Policies in Nordic Countries

The Nordic countries have a similar institutional structure for the plastic waste management, with a legislative body implementing EU regulations and supervising authorities responsible for bringing waste politics and regulations into practice and implementing follow-up by control. Finland, Norway and Sweden have introduced an EPR for packaging; the EPR schemes are responsible for collecting and recycling of plastic packaging waste. In Denmark the packaging target is pursued through a voluntary agreement between industry and authorities.

### *3.5.1 Denmark*

**National targets and visions**

Denmark has the same national recycling targets for plastic packaging (22.5%) as set by the Packaging Directive. The new Danish waste management plan (Denmark uden affald) from 2013 includes no specific recycling targets for plastic. However, it is highlighted that recycling of household waste and the service sector has to increase significantly. Part of the increase needs to come from plastic packaging (Regeringen 2013).

### Economic policy instruments

Denmark has a packaging tax on all beverage packaging to encourage use of refillable beverage packaging. However, the tax has been reduced several times over the last 10 years and the effect seems to be small. Currently, the tax for plastic beverages packaging is DKK 0.13–1.60 for packaging outside the deposit system and DKK 0.05–0.64 for packaging included in the deposit system (Skat.dk, 2014).

The Danish Landfill tax and Incineration tax were introduced on 1 January 1987. The aim was to create an incentive to help reduce the amount of waste going to landfills and incineration plants and so promote recycling (ETC/SCP, 2012). The landfill tax is currently at DKK 475/tonne. The level of the tax has increased progressively (ETC/SCP 2012):

- DKK 40 (~EUR 5.3)/tonne in 1987.
- DKK 160 (~EUR 21.3)/tonne in 1993.
- DKK 335 (~EUR 44.7)/tonne in 1997.
- DKK 375 (~EUR 50.0)/tonne in 1999.
- DKK 475 (~EUR 63.3)/tonne in 2010.

Since 2010 the Danish incineration tax has been calculated based on energy content in the waste. The incineration tax was changed to an energy tax. The idea was to charge waste in a more similar way to fossil fuel used for heat. The energy tax is a combination of a tax on heat from waste, an additional tax and a CO2 tax. The tax is paid per GJ. The calculation of the tax is very complicated. For calculation of the tax in 2014 see Skat (2014).

There is no EPR system for household packaging in Denmark.

### Landfill ban

A total ban on the landfilling of combustible waste was decided in 1994 and coming into effect on 1st January 1997. The long lead-time between law-making and the implementation date gave time to municipalities to build sufficient recycling and incineration capacity (Fischer *et al*. 2012).

### Policy instruments for household plastic waste

There is no EPR system for household packaging in Denmark. Municipalities are responsible for the collection and recycling of plastic waste from households, except beverages packaging which is handled by the deposit system (Statutory order on waste 2012). The municipalities are responsible for establishing collection schemes for plastic packaging from households with specific focus on plastic bottles and jars (Article 31 Statutory order on waste 2012). Furthermore, municipalities are responsible for the recycling of the collected waste back into plastic material.

Special legislation applies to the deposit system for beverages packaging (Statutory order on the deposit system 2014) which include PET bottles for soft drinks, water etc. Dansk retursystem is in charge of the deposit return system in Denmark.

**Policy instruments for business plastic waste**

All firms are responsible for recycling their recyclable waste under the Danish regulation (Statutory order on waste 2012) as well as to ensure that a significant part of their separated waste are recycled (Article 65). The article specifies a number of important waste streams including recyclable PVC waste and recyclable plastic packaging waste (Statutory order on waste 2012).

Small firms can deliver waste similar to household waste at the recycling centres for household run by the municipalities. However, only cars below 3,500 kg are allowed to enter the recycling centre (Article 40 Statutory order on waste 2012). The municipality then has the responsibility for the treatment of the waste.

Plastic waste from WEEE and ELV are managed by the EPR systems. From 1985 to 2010 the municipalities had more duties and authorities regarding recyclable waste from private enterprises. The change in regulation by 1st January 2010 resulted in the municipalities no longer having responsibility for ensuring that sufficient recycling capacity is available either at publicly or privately owned plants for waste from private enterprises (Kjær 2013, Fisher *et al*. 2012).

There are no national instruments identified for plastic waste from agriculture. However, a number of municipalities have stated in their local regulation a requirement of sorting plastic from agriculture for recycling (4-S 2014). Esbjerg Municipality has included plastic from agriculture as a focus area in their waste management plan. They will provide information to farmers during their inspections and provide better facilities for delivering plastic waste at the recycling centres. The municipality has the target to significantly increase the recycling from 2013–2018 (Esbjerg Municipality 2013).

The achievement of the target for plastic recycling was also supported by different initiatives to improve recycling of plastic packaging among those were major information campaign for firms in 2006 and 2007 (Plastindustrien 2014).

**Voluntary agreements**

Based on a former voluntary agreement on PVC a collection and recycling system for hard PVC was set up in 1997. Five of the largest Danish plastics processors formed the organization WUPPI in 1997. A common feature of these five companies are building products made of hard PVC which represent a large part of their business, and since then a number of importers were associate members. Today, the entire industry stands behind the scheme. The WUPPI system includes today more than 1,000 users. The majority of users are private builders, contractors, etc., but also there is a vast majority of local authorities and waste management companies in the scheme (Wuppi 2014).

In 1994, the Ministry of Environment, Danish Industry, the Plastic Industry and the Packaging Industry formed the so-called Transport Packaging Covenant. The agreement has served as a positive example of voluntary agreements between industry and authorities. The agreement has helped to ensure that all recycling in the EU Packaging Directive were met already in 2008.

### *3.5.2 Finland*

**National targets and visions**

The national recycling target for plastic packaging in Finland is 22.5% (NCM 2014) which is in line with the minimum target of recycling of plastic packaging waste in the EU Packaging Directive. Non-packaging plastics are not covered by producer responsibility or packaging recovery targets.

The Finish waste management plan "Towards a recycling society – The National Waste Plan for 2016" from 2009 has no specific plastic recycling targets or any specific mention to plastic waste recycling in general; the target for MSW treatment is: material recycling 50%; Energy recovery 30%; and landfilling 20% (Ministry of the Environment of Finland, 2009). It furthermore includes an examination of more effective recycling of plastic packaging, and based on this examination, a target for plastic packaging recycling is going to be set. Among the many recommendations about increasing recycling in the Waste Plan of Finland, there is also a provision concerning the promotion of greater use of recycled plastics (Ministry of Environment 2009).

There are no specific recycling targets for industrial plastic waste.

**Economic policy instruments**

Drinks packaging taxes are currently paid on packaging outside the deposit systems for alcoholic beverages, beer, bottled water, soft drinks and certain other beverage packaging. This form of taxation aims to further encourage the re-use of beverage packaging, to reduce the quantities of such materials ending up in landfill, and to prevent litter. The taxation level currently amounts to EUR 0.51 per litre. This tax does not apply to packaging covered by approved returnable deposit systems that involve the collection of packaging for refilling or material recycling. Finland's returnable deposit system is defined in special legislation on the taxation of the manufacture of certain types of drinks packaging (1037/2004) (Ympäristö 2014b).

Recycling of plastic waste from households is mainly represented by PET bottles collected within a deposit return system for PET bottles. Suomen Palautuspakkaus Oy (PALPA) promotes and administrates the recycling of beverage bottles (deposit return system) (Plastic Zero 2012).

In the EPR scheme, the producer responsibility organization PYR collects registration and annual fees based on the company's turnover; for companies with a turnover of <EUR 1 million, EUR 1–1.7 million, EUR 1.7–17 million and > EUR 17 million; the registration fees are EUR 40, EUR 68, EUR 155 and EUR 223 + VAT and annual fees are EUR 0, EUR 228, EUR 457 and EUR 696 + VAT respectively. In addition to this recovery fees are collected based on the packaging quantities, in 20124 the recovery fee for plastic packaging is EUR 25/t +VAT. (PirELY, 2013; PYR, 2014).

There is a tax on landfilling of waste in place since 1996. All waste specified in the tax schedule appended to the Waste Tax Act (1126/2010) is subject to tax (Ympäristö 2014b). The level of the tax has increased progressively (ETC/SCP 2012):

- EUR 15.15/tonne in 1996.
- EUR 23/tonne in 2003.
- EUR 30/tonne in 2005.
- EUR 40/tonne in 2011.
- EUR 50/tonne in 2013.

In Finland there are no taxes for waste incineration in a waste power plant.

### Landfill ban

There is planned a ban on landfilling of biodegradable municipal waste which limits the organic content of landfilled waste to 10% as of the 1st of January 2016 (Finlex 2013). In practice, waste plastic must then either be recycled as material or utilised in energy recovery.

### Policy instruments for household plastic waste

The municipalities are responsible for the collection and treatment of household waste, excluding source-sorted plastic packaging waste discarded in the EPR system. Some municipalities have chosen to arrange for collection of plastic packaging waste themselves. Municipalities are not responsible for types of waste covered by producer responsibility. Producer responsibility obliges manufacturers and importers to organise and pay for the management of waste resulting from their products (Ympäristö 2014a).

The majority of plastic waste from households is collected within an energy waste fraction or in mixed household waste, of which part is recovered as energy. Some of the municipalities contracted waste management companies arrange for collection of plastic waste (packaging and non-packaging together) at recycling stations as a separate plastic waste fraction or as an energy waste fraction. The plastic waste is not subject to recycling in either of the two cases (NCM 2014).

The Waste Act [646/2011]: currently plastic packaging is covered by a partial producer responsibility covering only industrial waste, but the producer responsibility is to be complete and cover also household packaging as of 1st of May 2015. The waste act stipulates the producer to see to the recovery of 22.5 mass-% of the plastic packaging put on the market, as of 2016 this target is proposed to be 30% (Blauberg, 2013).

Plastic bags (LDPE) are collected for recycling in Finland. Plastic bag collection is common in recycling stations, as well as in supermarkets close to the deposit bottle return machines. The plastic bag collection is commonly organised by the supermarkets and included in their own waste management programme. The bags are mixed with other flexible plastic packaging waste from the supermarkets, and transported for recycling. Recovered LDPE is suitable for production of new plastic bags; in Finland plastic bags are made with approximately 60% recycled LDPE (NCM 2014).

### Policy instruments for business plastic waste

In accordance with the Waste Act (646/2011), waste holders or firms are primarily responsible for the management of waste. The producer responsibility obliges producers and importers of packaging to collect and recycle packaging waste put on the Finnish market (Finlex 1997).

Firms fulfil the obligation by joining the producer responsibility organisation or by taking care of the treatment themselves (and reporting to the supervising authority). The Centre for economic development, transport and the environment for Pirkanmaa monitor compliance with provisions on producer responsibility on a national level.

The producer responsibility organisation Pakkausalan ympäristörekisteri, PYR Oy (The Environmental Register of Packaging), organises the collection and treatment of plastic packaging waste. The producer responsibility organisation only arranges for treatment of industrial plastic packaging waste. The waste generator (e.g. industry) is obliged to collect and transport the plastic packaging waste to a treatment facility (NCM 2014). Plastic waste from WEEE and ELV are managed by the EPR systems (Ympäristö 2014a).

An overall waste producer responsibility covers the whole waste management chain of non-municipal waste and in practise concerns waste producers mainly in industry, commerce and agriculture (Plastic Zero 2012). The treatment or storing of plastic waste also needs an environmental permit, which is authorized by the ELY-centres.

Due to the new waste legislation, the producers (packers and importers of packed goods) will be obliged to take back plastic packaging waste from households as well (including collection, transport and treatment).

No specific instruments were identified for plastic waste from agriculture. However the Finnish youth organization 4H annually organizes collection of the agricultural plastics. The organisation collects specific type of plastic waste. Recent data indicated that 500–600 tonnes of agricultural plastic waste were recycled annually (Plastic Zero 2012). In Finland there are few waste management companies which collect the agricultural plastic waste for energy utilisation purposes. There is also a deposit-refund system for specific fodder containers (Plastic Zero 2012).

### *3.5.3 Norway*

**National targets and visions**

Norway has already achieved the national recycling target for plastic packaging waste of 30%. The Norwegian waste management plan from 2013 "From waste to resource," presents several measures to promote further plastic recycling. Besides the target for material recycling of plastic packing of 30% there is a target totally for recycling and energy recovery of 80%. For beverage packaging the target is a part of an agreement between the Norwegian EPA (Miljødirektoratet) and the producers, importers and users of plastic packaging organized through Grønt

Punkt and Plastretur. The EPA is assessing additional national instruments in addition to the EPR. For further increase of plastic packaging recycling, the EPA suggests to sharpen the requirements. More actors like households and farmers should be included and the municipalities should sort out specific plastic before incineration and landfilling, if it is economic and environmental feasible (Miljøverndepartementet, 2013).

**Economic policy instruments**

There is a fee for using plastic materials in packaging paid by all users of plastic packaging. The fess was in 2014 NOK 1.15 per kg.[4] The total fee paid in 2013 was NOK 155 million.

PET plastic bottles are regulated through a special tax on beverage containers and a required deposit refund system (Miljødirektoretet, 2013). There is also a tax on non-returnable plastic bottles of NOK 1.08 per unit and a special environmental tax that varies between 0 (>95% return) and NOK 3.16 (<25% return) depending on the percentage of bottles that are returned.

The landfill waste tax in Norway was introduced in 1999 in order to give incentives to reduce the amount of waste landfilled. Since July 2003, landfill tax rates have been differentiated according to the environmental standard of the landfill site to which the waste is delivered. The higher rate has been applied to sites not fulfilling the requirements with regard to site linings. Landfills that did not meet the new requirements were to close down by 16 July 2009. Since then all the landfills are classified as high standard sites (Kjær, 2013). The level of the tax has increased progressively:

- EUR 37/tonne in 2000.
- EUR 43/tonne in 2002.
- EUR 41–53/tonne in 2003.
- EUR 48–62/tonne in 2004.
- EUR 50–65/tonne in 2005.
- EUR 53–69/tonne in 2007.

The variation depends on the quality of the landfill (ETC/SCP. 2012).

[4] The fee increased from NOK 1.25 per kg in 2007 to NOK 1.70 per kg in 2010, and have then decreased to NOK 1.35 per kg in 2011/12.

In addition to the landfill tax, Norway also has had a tax on incineration of waste. The tax was introduced in 1999 and abolished on 1st October 2010. The reason for abolishment was mainly due to the fact that Sweden abolished its incineration tax, which created a competitive disadvantage to Norwegian incineration plants.

**Landfill ban**

A ban on landfilling of biodegradable waste was implemented from 1st July 2009.

**Policy instruments for household plastic waste**

The municipalities are responsible for the collection and treatment of household waste, excluding source-sorted plastic packaging waste discarded in the EPR system. Most municipalities have separate collection of different waste sources, where plastic packaging is among them. As there are no designated collection and recycling system for small items of plastic waste other than packaging, the fraction ends up in the residual waste fraction collected by the municipality or the municipalities contracted entrepreneurs, and is subject to incineration or landfilling.

**Policy instruments for business plastic waste**

Firms are responsible for the waste and can freely choose the treatment, as far as it is legal (Avfallsstrategi, 2013). Plastic waste generation from packaging is part of an EPR scheme. The EPR scheme is a negotiated agreement between the operators and the authorities which was first established in 1995 (Miljøverndepartementet, 2013). There is no specific regulation for these types of waste, other than in the general waste regulations in the Pollution Act. Grønt Punkt Norge AS ("Green Dot Norway plc") is a privately owned non-profit company responsible for financing the recovery and recycling of used packaging on behalf of the industrial sector (Grønt punkt 2014).

According to the Norwegian legislation, the municipalities are not obliged to accept industrial waste at the municipal recycling centres, but it is not prohibited. The firm is charged, as a minimum, the expenses the recycling centre has for receiving and treating the waste.

Small firms are obliged to deliver their waste to approved collection points. The firm is charged for this service (CRI, 2011).

Plastic waste from agriculture, like solar cells and silage film, is already recycled through the system "Green Dot Norway plc". There is a potential to recycle more plastic from this sector like foil and films for plastic tunnels (Grønt Punkt 2014).

### 3.5.4 Sweden

**National targets and visions**

The Swedish waste management plan "From waste management to resource efficiency – Sweden's Waste Plan 2012–2017" has no new specific requirements on plastic packaging recycling targets or any specific mention to plastic waste recycling in general (Naturvårdverket 2012a). The national objectives for recycling of plastic packaging are 70% recovery of which 30% recycling (Ordinance 2006:1273 on packaging).

**Economic policy instruments**

Taxes on beverage packaging outside deposit refund systems aim to encourage the use of refillable beverages packaging on behalf of disposable. While Denmark, Finland and Norway have such taxes Sweden does not have this.

Producers and importers of plastic packaging are since 1994 legally responsible for organizing a collection and recycling system for the plastic packaging waste entering the Swedish marketplace according to the producer responsibility on packaging (Ordinance 2006:1273 on packaging). The producer responsibility applies for all kinds of plastic packaging independently on end-consumer of the plastic packaging, i.e. if the plastic packaging is consumed by households or by businesses. The producers are responsible for the collection and treatment of the packaging waste discarded in their collection and recycling system. The fees for each type of packaging are the following (FTI, 2013):

- Consumer packaging (households): SEK 1.71/kg.
- Business packaging: SEK 0.03/kg.
- Service packaging: SEK 1.55/kg.

There is no registration fee, the annual fee is SEK 1,500 (EUR 166), recovery fees: consumer packaging (households) SEK 1.71/kg (EUR 0.190/t), business packaging SEK 0.03/kg (EUR 0.0033/t), service packaging SEK 1.55/kg (EUR 0.172/t). Companies with packaging fees (including FTI annual fee) less than SEK 8,500 (EUR 940) can choose a set fee instead of reporting actual packaging volumes. The set fees are divided into three levels: SEK 2000 (EUR 220), SEK 4000 (EUR 440) and SEK 8500 (EUR 940). Thus, all companies pay at least SEK 2000 (EUR 220) annually including the annual fee (FTI, 2013).

The system for collection and recycling of PET bottles is separated from other plastic packaging due to SFS 2005:22, Ordinance on deposit system for plastic bottles and metal cans. The ordinance is applied on PET bottles sold in Sweden with ready-to-drink beverages apart from bottles containing drinking dairy products, and drinks with a content of juice or vegetable parts exceeding 50%. The authority giving approval to deposit systems is The Swedish Agricultural Board (Jordbruksverket, 2013). The national objective for recycling of PET bottles stated in the ordinance is 90% recycling.

There is a tax on landfilling of waste in place since 2000 (ETC/SCP2012). The level of the tax has increased over time:

- SEK 250/tonne in 2000.
- SEK 288/tonne in 2002.
- SEK 370/tonne in 2003.
- SEK 435/tonne in 2006.

A tax on incineration on household waste was introduced in 2006 but was abandoned in 2010 (Milios 2013).

**Landfill ban**

There is a landfill ban in place since 2002 for separated combustible waste, which was stated in Ordinance (2001:512) on landfilling (Milios 2013). However, heterogeneous waste with a content of less than 10% (volume) combustible waste is exempted from the ban. (NFS 2004:04). There is also a ban on landfilling organic waste since 2005. Organic waste, according to Ordinance (2011:927), includes plastic waste as it contains organic carbon.

**Policy instruments for household plastic waste**

According to the environmental code (Ds 2000:61); Swedish municipalities are responsible for the collection and treatment of household waste, and waste that is similar to household waste from businesses. Waste generated by businesses (non-household waste) is the business responsibility. However, as mentioned below producers and importers of plastic packaging are legally responsible for organizing a collection and recycling system for the plastic packaging waste entering the Swedish marketplace according to the producer responsibility on packaging (Ordinance 2006:1273 on packaging).

As there are no designated collection and recycling system for small items of plastic waste other than packaging the fraction ends up in the residual waste fraction collected by the municipality or the municipalities contracted entrepreneurs, and is subject to energy recovery. Non-packaging plastics also ends up in the plastic packaging and plastic bulky waste fractions. In that case they follow the respective stream to recycling if the polymers correspond with the polymers sorted out of the plastic packaging and plastic bulky waste fractions (NCM 2014).

All plastic packaging (with a few exceptions) are covered by producer responsibility. There is one major EPR organization (FTI) and one smaller one (TMR) arranging for the collection and recycling of source sorted plastic packaging waste on behalf of their registered producers. The systems are separate and FTI and TMR are competitors. FTI represents "the base" with almost 6,000 recycling stations in Swedish municipalities. TMR works on a smaller scale and cooperate with some municipalities on curbside collection. FTI also works with curbside collection though.

**Policy instruments for business plastic waste**

Firms are responsible for ensuring that its plastic waste is managed in an acceptable manner from an environmental and health perspective. Alternatively, the operator may leave the waste to the municipal recycling centres, where it will be accepted upon payment of a charge. Small firms which generate a small amount of waste can deliver their plastic packaging waste free of charge to specific collection points managed by Förpacknings och Tidningsinsamlingen (FTI) (Naturvårdverket 2012a).

Plastic waste separated after sorting of WEEE and ELV are managed by the EPR systems.

The treatment or storing of plastic waste needs an environmental permit (depending on treated amounts etc.). The permits are authorized by the environmental courts or the county administrative boards.

However, a business generating plastic waste does not need an environmental permit. The collaboration between the business and the waste contractor is business-to-business oriented meaning that the business is free to contract a waste collector/contractor by choice. For plastic packaging waste the same general rule applies, but there are also around 100 collection points managed by FTI where businesses can leave up to 1 m3 of packaging waste free of charge (each time).

In agriculture there is a voluntary commitment for agricultural plastic (Naturvårdsverket 2012b). Agricultural Plastic consists of silage film, plastic bags, horticultural foil and the like. Agricultural plastic is not classified as packaging. But the sector has made a voluntary commitment that, by 2004, to collect and recycle at least 30% of agricultural plastic put on the market. The Environmental Protection Agency has promised to follow up on this commitment.

There is a voluntary agreement in the construction sector for the recovery and recycling of C&D waste, including the plastic waste of construction and demolition activities. These initiatives are on a very small scale and a minor amount of the generated plastic waste from the C&D waste sector is recycled. The existing initiatives are NRG Nordiska Plaströrsgruppen (www.npgnordic.com) where pipes of PVC, PE and PP to a certain extent are collected and recycled, and GBR Golvbranschen (www.golvbranschen.se) taking care of spillage of plastic flooring and wall coating.

There are also some initiatives for labelling of construction materials assessing its raw materials use and environmental impact, e.g. byggvarubedömningen (Byggvarubedömningen, 2014).

# 4. Evaluation and Comparison of Policies in Countries

The chapter compares the use of policy instruments in Denmark, Finland, Norway and Sweden. The last section gives a brief evaluation of experiences on the use of economic incentives as well as experiences of main challenges when designing policy instruments for recycling of plastic waste in the four Nordic countries of this study.

## 4.1 National Targets and Visions

Plastic waste is not specifically addressed by EU legislation and none of the Nordic countries has a specific plastic recycling targets stated in their waste management plan. All of the plans aim at promoting material recycling in general, but only Norway's plan mentions plastic waste with specific measures.

The Framework Directive on waste (2008/98/EC) establishes extended producer responsibility (EPR) as a key principle in waste management. The Packaging Directive (94/62/EC amendments 2004/12/EC and 2005/20/EC) has a specific recycling target for plastic packaging. The minimum recycling target for plastics is 22.5% by weight, counting exclusively material that is recycled back into plastics. The recycling targets of packaging waste do not specify specific targets for industrial and household waste. Finland, Norway and Sweden have introduced an EPR for all packaging; the EPR schemes are responsible for collecting and recycling of plastic packaging waste. In Denmark the packaging target is pursued through a voluntary agreement between industry and authorities. Country specific targets for the recovery of plastic packaging waste:

- Denmark: 22.5%.
- Finland: 22.5% (30% as of 2016).
- Norway: 30%.
- Sweden 30%.

Thus national recycling targets for plastic packaging are higher than the EU requirements in both Norway and Sweden (30%). Denmark and Finland have the same target (22.5%) as set by the Packaging Directive. No other targets for plastic recycling have been identified in the national policy in the four countries.

Norway is the only country which focuses specifically on plastic recycling in their waste management plan (WMP). The Norwegian WPN from 2013 "From waste to resource", present several measures to promote plastic recycling for packaging and plastic waste from agriculture. Furthermore, Norway plans to introduce an EPR scheme for fishing gear and discarded equipment from aquaculture (Avfallsstrategi, 2013).

The Swedish WMP "From waste management to resource efficiency – Sweden's Waste Plan 2012–2017" has no specific mention of plastic recycling (Naturvårdverket 2012a).

The Finnish WMP "Towards a recycling society – The National Waste Plan for 2016" from 2009, includes an examination of more effective recycling of plastic packaging, and based on this examination, a target for plastic packaging recycling is going to be set (Ministry of Environment 2009). A regulation on landfilling of biodegradable municipal waste is planned which will limit the organic content of landfilled waste to 10% as of the 1st of January 2016 (Finlex 2013) which might encourage more recycling (including energy recovery) of plastic.

The new Danish WMP (Danmark uden affald) from 2013 includes no specific recycling targets for plastic. However, it is highlighted that recycling of household waste and waste from the service sector has to increase significantly. A part of the increase will need to come from the recycling of plastic packaging (Regeringen 2013).

Plastic waste that is not recycled is either recovered as energy or landfilled. In order to prevent e.g. plastic waste from being landfilled Denmark, Norway and Sweden have a statutory ban (or limitation) on the landfilling of organic or combustible waste while Finland will introduce a ban 2016.

## 4.2 Economic Policy Instruments

Taxes on beverage packaging outside deposit refund systems aim to encourage the use of refillable beverages packaging on behalf of disposable. Denmark has a tax for plastic beverages packaging of DKK 0.13–1.60 for packaging outside the deposit system and DKK 0.05–0.64 for packaging included in the deposit system. Finland has a beverage packaging tax of EUR 0.51/l, which does not apply to packaging included in the deposit systems. Norway has a tax for plastic beverages packaging of NOK 1.08 for packaging outside the deposit system and NOK 0–3.16 for packaging included in the deposit system. In Sweden no tax is applied on beverage packaging.

Another type of economic instrument used to promote plastic recycling are packaging deposit systems. All four Nordic countries have deposit systems which include single-use beverage packaging such as plastic bottles. The systems differ in the number of product types covered. The collection and recycling of single-use packaging covered by the deposit system are high (85–95%).

Plastic waste generation from packaging is part of an EPR scheme in Finland, Norway and Sweden. All producers and importers of plastic packaging (in Finland with a net turnover exceeding EUR 1 M) are legally responsible for organizing a collection and recycling system for the plastic packaging waste entering the markets. (In Finland the EPR system will cover household plastic packaging as of 1st May 2015.)

Municipalities are responsible for the collection and treatment of MSW excluding source-sorted plastic packaging waste discarded in the EPR system (in Norway and Sweden) and beverages packaging which is handled by the deposit system. In Denmark the municipality has the responsibility to establish a collection scheme for plastic packaging from households; the municipality is also responsible for the recycling of the collected waste back into plastic material.

All firms are responsible for the management of their plastic waste (excluding plastic packaging waste which is covered by the EPR scheme in Finland, Norway and Sweden). The producer responsibility organizations collect registration and annual fees, as well as fees based on the packaging quantities:

- Finland: registration and annual fees based on the company's turnover, recovery fee EUR 25/t +VAT.
- Norway: recovery fee NOK 1.15 per kg (EUR 141/t).
- Sweden: no registration fee, annual fee SEK 1,500 (EUR 166), recovery fees: consumer packaging (households) 1.71 kr/kg (EUR190/t), business packaging 0.03 kr/kg (EUR 3/t), service packaging 1.55 kr/kg (EUR 172/t).

All four countries have introduced a landfill tax, which have been in place for many years. The tax varies between countries from EUR 50–69/tonne. The main objective with landfill and incineration taxes is to direct waste higher in the waste hierarchy towards recovery and recycling, through giving the other options a monetary benefit. No studies have been identified which evaluate the effect of the tax on plastic recycling rates. In general landfill taxes have demonstrated to be an efficient instrument to divert waste from landfill and to increase recycling rates (EC 2012). However, in countries with high incineration capacity (e.g. DK, SE and NO) the diverted waste might be incinerated instead. All four Nordic countries have implemented landfill taxes:

- Denmark: DKK 475/t = EUR 64/t.
- Finland: EUR 50/t.
- Norway: NOK 290/t = EUR 36/t.
- Sweden: SEK 435/t = EUR 48/t.

Sweden and Norway adopted incineration taxes for a number of years, which could give incentives for lower incineration rates, but which have now been abandoned. Denmark is the only country which still has an incineration tax. Unfortunately, no evaluation of the effect of the Danish tax on plastic waste recycling has been carried out.

## 4.3 Policy Instruments for Household Plastic Waste

The municipalities in the four countries are responsible for managing recyclable waste from households except waste covered by EPR. Finland, Norway and Sweden have established EPR schemes to collect and recycle plastic packaging waste from households (Finlex 1997, Avtale om plastemballasjeavfall 2003, Förordning (2006:1273)). In Denmark, there is no EPR for plastic packaging from households. Municipalities are instead responsible for establishing a collection scheme for plastic packaging from household with specific focus on plastic bottles and jars (Article 31 Statutory order on waste 2012).

## 4.4 Policy Instruments for Business Plastic Waste

The industrial waste management is different in the Nordic countries. Waste generators in Sweden are obliged to ensure that the waste is treated in an acceptable manner. In practice a firm contracts a legitimate waste collector/contractor for this task. In Denmark all waste producing enterprises are obliged to separate their waste at source and to ensure that a significant part of their separated waste is recycled. In Finland all industrial activities are issued an environmental permit specifying e.g. waste management requirements. In Norway, it is not mandatory to sort plastic from firms, but all plastic waste must be taken care of safely according to the Pollution Act and can be recycled or incinerated.

The treatment or storing of plastic waste needs an environmental permit in all Nordic countries; it is issued by the municipalities or the Danish EPA depending on the size and type of facility, by the ELY-centres in Finland, Miljødirektoratet or county administration in Norway, and by the environmental courts or the county administrative boards in Sweden.

Packaging waste is the only plastic waste stream from industry which is regulated in all four countries. In Finland, Norway and Sweden the waste is covered in the regulation by EPR schemes (Finlex 2011, Finlex 1997, Avtale om plastemballasjeavfall 2003, Förordning (2006:1273)). In Denmark, enterprises are obliged to separate their recyclable plastic packaging waste by law (Statutory order on waste 2012). In addition the regulation is supported by a voluntary agreement with the industry.

In Denmark, enterprises are obliged to separate recyclable (under current technology) PVC waste and ensure recycling of it by law (Statutory order on waste 2012). The industry has set up a take back system for recyclable PVC (WUPPI).

**Agriculture**

National initiatives have been identified for Norway and Sweden which cover agricultural plastic waste. Both countries have a voluntary commitment for recycling of plastic from agriculture (Avfallsstrategi, 2013, Naturvårdsverket 2012). No specific national instruments for recycling are identified for Denmark and Finland.

In Norway there is a voluntary commitment for agricultural plastic though the system "Green Dot Norway plc". Plastic waste from agriculture, like soil solarisation film and silage film, is already recycled through the system "Green Dot Norway plc".

In Sweden, the agricultural sector made a voluntary commitment that, by 2004 at least 30% of agricultural plastic put on the market should be collected separately and recycled. The Environmental Protection Agency promised to monitor the commitment. Data collection is managed by Swedish Silage Plastic Return AB (SvepRetur), and coordinated with the collection of (larger) plastic packaging from agriculture. By 2010, 79% of the agricultural plastic put on the market annually was material recycled by the system (Naturvårdsverket 2012).

In Denmark a number of municipalities have stated in their local regulation a requirement to sort plastic from agriculture for recycling and have initiatives in their local waste management plans (4-S 2014, Esbjerg Municipality 2013).

In Finland the youth organization 4H organizes an annual collection of agricultural plastic waste. The organisation collects specific type of plastics. There is also a deposit-refund system for specific fodder containers (Plastic Zero 2012).

## 4.5 Challenges when Designing Policy Instruments for Recycling of Plastic Waste in Nordic Countries

In the Nordic project on plastic waste recycling Phase I (Fråne *et al*. 2014), the main obstacles and challenges towards increased recycling of plastic waste were discussed, mainly related to household waste. Their points are only briefly presented here as input to the further analysis of policy instruments in chapters 5–9.

- Plastic waste is voluminous and has a relatively low density compared to other types of waste. This results in high transportation costs, and could also have an influence of people's tendency to source sort the plastic waste as it is difficult to compress the fraction at home or in the business sector.
- Misbelief about collection and recycling of plastics is also one of several factors that might contribute to low collection and recycling rates, and an obstacle for improved consumer behaviour.
- Better sorting solutions and increase capacity of sorting. The current sorting capacity is about to reach its limit as there is at present only one sorting facility in the Nordics accepting high volumes of plastic packaging waste Plastic packaging waste fractions need better sorting and processing due to mixed polymer types. The new solutions with central sorting of more or less pre-sorted residual waste (e.g. NIR technologies) might have potential to improve this bottleneck.
- An important bottleneck of increased plastics recycling is the relatively high costs compared to other waste management alternatives and virgin materials. Stakeholders are accustomed to profitable recycling of high-value materials (e.g. paper and metals) and respond negatively to the high cost of plastics recycling. It shows also often up in public purchasing of waste management services that

incineration of plastic as part of the residual waste has a lower cost than present sorting and recycling. The Nordic countries have a relatively scattered population and long distances to transport plastic waste increase costs although the waste is compressed to minimize volumes. Price of virgin materials is a fundamental factor. If the price for virgin material is relatively low, secondary raw materials will be in a disadvantageous position as the benefits will be perceived as marginal or non-existing.

- Quality of the recycled materials is also an important bottleneck for increasing recycling possibilities. Impurities and heterogeneousness of household plastic waste influences the characteristics of the material, thus, preventing recycling. The quality of the secondary plastic raw material is not as high as for virgin materials. There is a need of companies able to upgrade the plastic waste to qualities more comparable to virgin raw materials.
- Plastic waste is a very heterogeneous fraction and often contains other kinds of waste and non-recyclables why the fraction not always is perceived as attractive to recycle. Small items of plastic waste other than packaging are even more heterogeneous than plastic packaging waste and could contain a higher variety of additives that might not go well with recycling. Polymers in plastic packaging are dominated by three different polymers, but polymers in plastic products are more diverse. For plastic waste from the industry, the situation can be easier, as there often are more waste of homogenous types and qualities, making it easier to sort separately Decreasing the variety of polymers present in the plastic waste flows would facilitate for obtaining higher recycling rates, coming back to the importance of product design and collaboration between industry, consumers, collectors and recyclers. Plastic waste often have several polymers in one plastic product, as laminates with several layers of barrier materials is necessary to reduce oxygen transmission or transmission of other types of gases.
- Demand for plastic waste from recyclers and from product manufacturers is a criterion for increased recycling of plastic waste. European recyclers have over the last years maximised their capacity due to increasing amounts of source separated plastics in Europe. When possessing relatively small amounts, it may be a challenge to establish appropriate agreements with recyclers. The amount of European plastics sent to Asia for recycling has decreased and it is a trend that higher amounts of plastic waste resources are used in Europe. On the other hand there are indications that the market is saturated and that the demand is lacking.

- Existing recycling targets might be perceived as too low not creating enough incentives to increased collection and recycling of plastic packaging waste. 100% recycling may not be a viable target, but there are European countries reaching significantly higher recycling targets than the Nordics. Some regions in the Nordics have no legislation or formal incentive that requires recycling at all.
- Which of those factors that are the most important will probably vary from country to country and from time to time, but it is important to consider which incentives that are necessary to reduce the barriers to separate plastic materials for recycling.

# 5. Private Actors in the Circular Economy of Plastics

The chapter illustrates a simple overview of private actors and their positions in a circular economy of plastic. Chapters 6–8 then refer to potential market failures that may occur in this system.

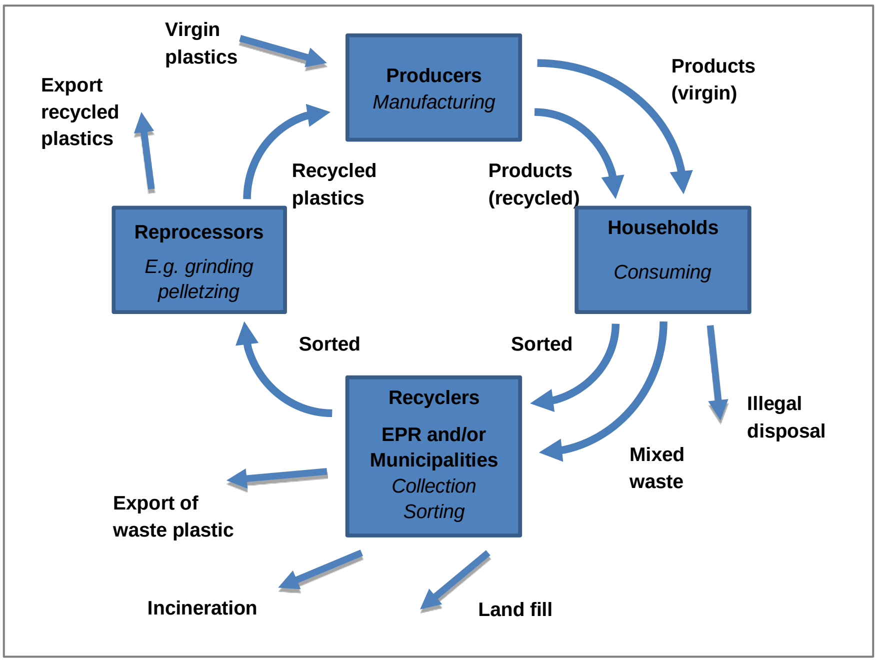
From an economic perspective, recycling is driven by the value of the recycled material. If a good becomes scarce, its price rises, creating incentives for reuse and recycling. If waste products lack value relative to current salaries recycling is not encouraged. In contrast, in a poor country such as India, a large fraction of what is thrown away is recycled (Sterner & Coria, 2012). In the case of economies with rising incomes, it is almost inevitably that the production of waste will increase over time. Bartone (1990) referred to these type of economies (specifically to the case of Japan's accelerated economic growth in the post war period), and pointed out that commitment to recycling is driven by the need to conserve resources and reduce imports of raw materials, save landfill space and reduce pollution from landfills, and make incinerators less costly and minimize their pollution. In such cases, a government may have to not only promote recycling but also establish and maintain a market for recycled goods (Sterner & Coria, 2012). In these markets, recyclers pay consumers a price for their used products that depend on the degree of recyclability (Calcott & Walls, 2000, 2005). Pittel *et al*. (2010) also highlight the role of the value of waste in these markets. In an economy where no value is allocated to waste, "final output producers as well as households do not take the value of waste generation for future recycling into account. Consequently, the opportunity costs of production and consumption are overestimated. On the other hand, resource and waste processing firms do not internalize in their supply plans the consequences of the reflux of material to the waste pile" (Pittel *et al*., 2010).

## 5.1 Private Actors in the Circular Economy of Plastic

The major actors in recycling of waste plastics are producers of plastic goods made of virgin and recycled plastic, consumers (or industry) consuming the goods, recyclers collecting and sorting the waste plastics generated by households (or industry) for further reprocessing into recycled plastics which can be used by producers in new goods. Each major actor type has its own decision node in the circular economy where his decision can break the circular flow of the recycling process in figure 6. The producers' decision node contains the choice between virgin and recycled plastics as input to production and then to sell goods to consumers (or industry). The consumers' decision node contains the choice of buying goods made of virgin or recycled plastics as well as the choice of sorting the waste for recycling, leaving it in the household mixed residual waste or even illegally dispose it, for instance, placing it next to an over-filled recycling container or burning it in the open fire.

The decision node for recyclers in figure 6, which contain actors in Extended Producer Responsibility (EPR) schemes and/or in the waste management systems of the municipalities as well as private recyclers, involves the major choices of recycling program, collection infrastructure, exporting the waste, treatment for selling it as input material in the reprocessing into recycled plastics or sending it to land fill or incineration. Finally, the reprocessors convert waste plastics to e.g. pellets that can be used as raw material in the manufacturing of new goods and sell it to domestic or foreign manufacturers of plastics.

*Figure 6: Major private actors and their major decision nodes in recycling of waste plastics*




The four major decision nodes of the circular economy in the figure are simplifications of the specialisation areas in the recycling process. In reality, some firms can be involved in collection, sorting and reprocessing while other firms do reprocessing and manufacturing. There are even firms involved in all steps from collection to manufacturing of final goods (for instance with homogeneous fractions of industry waste). However, for household waste which consists of more heterogeneous fractions, specialization in the processes above are more common with business-to-business contracts between different firms specialising in collection, reprocessing and manufacturing.

## 5.2 Potential Markets Failures in a Circular Economy

Without any market failures in figure 6 all decisions in the circular flow should in theory result in a price per unit of garbage disposed equal to the marginal social collection and disposal cost in the circular economy of plastic waste. There are however several reasons why this condition in reality may be violated in the recycling of waste plastics to manufacturing of goods with recycled plastics. The result would be suboptimal recycling rates. Potential market failures identified in the analysis and literature reviewed in chapters 6–8 are summarized hereinafter:

### *5.2.1 Market Failures related to Producers' Choices*

The literature and the analysis here mainly cover two types of market failures. The first type relates to producers' choice of input materials, production process and product design since these aspects may have environmental impacts as well later affect collection and disposal of the good. The second type refers to asymmetric information about quality and demanded specifications of the recycled plastics provided by recycling and reprocessing industry on input markets. Thus:

### *5.2.2 Market Failures related to Households' Choices*

Plastic packaging waste is for instance sorted at site by households in Denmark, Norway and Sweden and collected by the municipality in Denmark and by the EPR Scheme in Norway and Sweden. Households then sort their waste and leave plastic packaging waste through traditional bring systems or by different kinds of curbside collection. In theory, there are mainly two potential market failures that may arise at the households' decision node in the circular economy in figure 6. The first relates to negative externalities in household decisions over waste generation and disposal. When deciding on how much and what to consume in the shop, households might not fully take into account the type and amount of waste that will be generated. As a result more virgin material in goods, as well as more waste, is generated than is socially optimal. The second market failure relates to waste collection and recycling when the goods are consumed. In short, the household's choice consists of; sorting plastic waste, leaving plastic waste in the mixed household waste or illegal disposal (e.g. leaving plastic waste next to an over-filled container at an unmanned recycling station).

### *5.2.3 Market Failures related to Recyclers' Choices*

At the recyclers' decision node in figure 6 recyclers take "upstream" decisions about the design of recycling program, collection infrastructure and pricing policies for collection services (which affect convenience costs and the incentives of households and industry for sorting and separation). Recyclers also take "downstream" decisions about treatments in the recycling process or incineration, landfill or even export of waste plastics. Market failures in these choices may occur if recyclers do not fully take into account the social cost of their choices besides their own operating costs in decisions about:

- design of recycling program and infrastructure that affect industry's and/or households' incentives and convenience costs to do sorting and separation
- pricing policies for collection services that affect industry's and/or households' incentives to do sorting and separation
- further treatment options incl. incineration, landfill and export of waste plastics for recycling in foreign countries and reprocessing.

Market failure may also result from the asymmetric information about quality and specifications of waste plastics and recycled plastics supplied to the reprocessing and manufacturing industry.

## 5.3 Deposit and Refund Systems for Beverage Packaging

Special legislation applies to deposit and refund system for beverages packaging (incl. PET bottles for soft drinks, water etc.) in all four Nordic countries. These systems form specific closed-loop systems with private actors in figure 6. The clear pellets are usually recycled as raw materials for new PET bottles while the colored pellets are recycled in e.g. the packaging and clothing industry. The recycling rates are usually 85–95%.

# 6. Household Behaviour and Recycling

Several theoretical and empirical studies in the economics literature seek to explain and understand how, and to what extent, household decisions are affected by different designs of policy instruments in the search for policy instruments that may achieve optimal recycling rates. Reviews of the theoretical literature on the economics of household waste management can be found in Choe and Fraser (1998) and Fullerton and Kinnaman (2002). As first best solution, a majority of the theoretical results identify deposit-refund schemes, a system with a tax or charge at production or consumer purchase and a refund to consumers that recycle and/or firms that collect or reprocess recycled materials. As an alternative first best solution (when illegal disposal such as dumping is not a problem), the results usually support the use of a virgin material tax or a tax on households' disposal choices. Even though deposit-refund systems generate the first best solution in resource allocations they have relatively high transaction costs and is in general more costly to administrate (monitoring and verifying for charging and refunding). This implies that large waste flows or other economies of scales in waste management systems (e.g. standardised product design such as a beverage bottles) are often needed for cost-efficiency. It is therefore not surprising that deposit and refund systems are usually implemented for goods with large waste flows such as packages and specifically beverage containers. There exist also several empirical studies in the economic literature on household responses to policy instruments and how household waste generation and recycling behaviour are influenced by attitudes and socio-demographic attributes in the context of present policy instruments. These studies often rely on community- or household-level data using probit or tobit models to estimate the frequency of recycling as a function of different policy instruments and household attributes (e.g. income level, value of time, education, and number of persons in the household, age, renting or ownership). Most studies that are presented here are from the 1990s and the 2000s.

## 6.1 Effects of Inconvenience Costs on Households Recycling

Judge and Becker (1993) conducted a controlled field experiment involving 1,000 households in Minnesota, USA (Rice County) to investigate the relationship between recycling convenience, solid waste diversion, and recycling program costs. The field experiment covered several voluntary recycling programs (treatments) which differ when it comes to convenience factors such as sorting requirements, frequency and location of recyclable collection as well as the amount of recycling information. The recycling volume of each household was monitored over a six-month period as a function of recycling convenience factors. A random sample of 20% of the households was drawn for a survey on further information about individual households such as household size, age, education level, and home ownership. The results indicated that convenience factors such as higher frequency of collection, lower sorting requirements (commingling in a single bin rather than several bins), and more convenient (closer) collection location contribute to recycling. No effect on recycling behaviour could be seen from information and educational efforts. Among the demographic variables household size and education level had a significant increasing impact on the quantity of recyclables.

Reschovsky and Stone (1994), conducted a survey with 1,422 households in the Finger Lakes region of upstate New York to study how different waste management policies (bring systems, kerbside recycling, mandatory recycling, and bag-based user fee) influence household decisions of recycling and sorting (newspaper, glass, plastic, cardboard, metal cans, and food/yard waste. The analysis also use socio-demographic attributes such as income, education, age, household size, marital status, gender, and storage space at home. Probit estimation showed that only married households and more highly-educated households tend to explain higher recycling (though being married had no effect on the recycling of plastics). Women tended to recycle more glass and plastic. Knowledge of a bring system within 5 miles of the home could explain higher recycling for all materials but newspaper. Moreover being well informed about the recycling programs is significant and positive for all the materials but food/yard waste. As for the policy instruments, mandatory recycling with kerbside collection with or without bag-based system were most effective however, only kerbside collection had a positive effect only on cardboard recycling.

Jenkins *et al*. (2003) estimate the intensity of recycling activities by material (glass bottles, plastic bottles, newspaper, aluminium, and yard waste) as a function of policy variables (disposal price, kerbside collection, bring systems, number of materials in kerbside collection, mandatory kerbside recycling and age of recycling programs) and socio-economic attributes (population density in the area, income, household size, age, detached home, home ownership, and education). The data is household survey data from 20 communities in the US and the estimation of recycling intensity is conducted with logit regression and by source. The results show that bring system recycling as well as kerbside recycling have significantly positive impacts on recycling efforts for each of the five different materials. The explanation is that bring system and kerbside recycling programs are convenience factors reducing the time and storage costs of recycling. Kerbside recycling though has a larger significant effect than bring systems due to lower transportation costs. They also find that in both programs the effects vary across the five types of sources. The probability of recycling materials increases for sources with larger transportation and storage costs, e.g. heavier materials such as glass and voluminous materials such as plastic bottles. Still they find that mandatory kerbside recycling program do not have a significant effect on recycling for any of the five sources. This is also supported by Kinnaman and Fullerton (2000) who found that mandatory recycling has a positive impact only on garbage but not on recycling. Another result is that the disposal price shows no effect on recycling which may be due to the fact that a majority of households lived in municipalities having unit-based pricing systems with subscription programs where households subscribe to a given number of bags resulting in a zero marginal cost for adding garbage.

## 6.2 Effects of User Fees and Marginal Pricing on Households Recycling

Dijkgraaf and Gradus (2004) compare the effect on recycling of four different unit pricing systems using data from Dutch municipalities. They found that a weight-based system or a bag-based system that prices both unsorted and compostable waste increased recycling by approximately 20%, a frequency-based system increased by 10% while a volume-based system had no significant effect. They also found a difference between a bag-based system that prices both unsorted and compostable and a bag-based system that price only to unsorted waste while compostable waste is free. Ando and Gosselin (2005) use survey data from 214 households in Illinois USA to analyse recycling efforts of several socio-demographic attributes (single-family housing, multi-family housing, age, education, single-gendered homes, interior storage space, distance to collection points, engagement in recycling activities when not at home). The results show that single-family housing, age, education, single-gendered homes and multi-family housing with interior storage age, households that engage in recycling when not at home (paper), households with two adults or only women had positive impacts on the recycling rate. On the other hand, distance to collection points, implying a larger cost in terms of time and efforts, had a negative impact on recycling rate.

Kinnaman and Fullerton (2000) estimate the demand for recycling collection as a function of socio-demographic attributes (income, size, age, education, home ownership, and density) and policy variables (particularity a user fee system, recycling program, mandatory recycling and a deposit/refund system). Using data from 959 communities in the US, the results show that age, household size, educations, house ownership, and user fee systems have a positive impact on recycling while income has not. They conclude that further research is needed to determine the cause for the positive impact of a user fee (e.g. whether it reduces consumption, shifts consumption towards less waste-intensive goods, increases composting, burning or dumping). The also control for municipal policy decisions about the user fee level, implementation of recycling programs and kerbside recycling and find that the two former do not yield statistically significant differences in results while the latter does.

Hage and Söderholm (2008) investigated the main determinants of collection rates of household plastic packaging waste in Swedish municipalities (cross-sectional data for 252 Swedish municipalities). The regression analysis suggested the collection rate appears to be positively affected by increases in the unemployment rate, the share of private houses, and the presence of immigrants (unless newly arrived) in the municipality. The analysis also showed that municipalities that employ weight-based waste management fees generally experience higher collection rates than those municipalities in which flat and/or volume-based fees are used.

One example of a municipality that early introduced a weight-based system is Varberg (southwest of Sweden). Bartelings and Sterner (1999) conducted an empirical case study in Varberg, which in 1994 introduced a weight-based billing system for household waste charging SEK 1/kg. Results from this study showed that, at the household level, the following were important explanatory variables: 1) if whether kitchen (and garden) wastes were composted, 2) living area (square meters of domicile), 3) age of the residents (recycling efforts increase with age), and 4) the resident's attitudes about the difficulty of recycling (Bartelings & Sterner, 1999; Sterner & Coria, 2012). In addition, households require a proper infrastructure that facilitates recycling.

Nestor and Podolsky (1998) compare estimated price effects of bag-based (pay per filled bag) and subscription systems (pay for a fixed number of emptying cans per period) in Marietta in Georgia in kerbside collections. They conclude the bag-based system has no significant effect while the can-based (or subscription) system increases on-site and off-site recycling. Among the demographic variables they found that income appears to be a significant and positive determinant of recycling, although the estimated coefficient is extremely low. Hong *et al*. (1993) use an ordered probit model to estimate how the frequency of recycling participation is influenced by the disposal fee of an additional can as well as the demographic attributes household size, education level, tenure status (renting or owning), income, and value (opportunity cost) of time. The results show that the disposal fee and the number of people in the household and the level of education contribute to more frequent recycling while the value of time and renting contribute to less frequent recycling.

Other studies that found more recycling activities in response to user fee systems for waste disposal are Miranda and Aldy (1998) and Hong (1999), the latter using survey data from over 3,000 Korean households. Hong and Adams (1999) found that the in a block payment system the disposal fee (the increase in collection fee between the contracted can size and the next largest can size) is the only variable with a significant and positive effect on recycling rate in a study involving 944 households in Oregon, USA. Van Houtven and Morris (1999) studied the effects of unit price used price scheme using data on the household level from the experimental implementation of a unit pricing program (bag-based or subscription) in Marietta, Georgia. They found that both bag-based and subscription-based unit pricing systems were significant in increasing the probability of recycling although there was no significant effect on the amount of recycling. Ownership and beliefs in the importance of waste reduction result in positive impacts on recycling while urban households and households size has negative effects.

## 6.3 Combined Effects of Policy Mixes on Households Behaviour

Some economic studies have also tested for combined effects of pricing system and recycling programs and infrastructure on behaviour together with socio-demographic attributes. Callan and Thomas (1997) conduct a regression analysis involving 324 Massachusetts communities in which recycling efforts was estimated as a function of municipal policies unit pricing, availability of kerbside recycling and collection services and demographic attributes (income and education) and municipal attributes (housing values, housing age, density and population). The results showed that unit pricing and kerbside recycling, especially when implemented together could explain higher recycling efforts. While unit pricing contributed to a 25% increase in recycling rate, unit pricing in combination with kerbside recycling contributed to a 45% increase. On the other hand, kerbside recycling contributed to a 15.6% increase alone and to a 36% increase if implemented in combination with unit pricing. On the other hand, the provision of kerbside trash disposal had no effect on recycling compared to bring system disposals. Among demographic variables, income, education, and housing value had a positive effect on recycling efforts while housing, age and population had negative effects. Finally, small municipalities tended to have higher levels of recycling than large municipalities

Also Ferrara and Missios (2005) conduct a probit regression analysis estimating the probability of recycling each of a range of sources (newspaper, glass, plastic, aluminium, tin cans, cardboard, and toxic chemicals) using policy variables (user fee, weekly and biweekly collection, number of free units in a unit pricing system, unit disposal limit, and mandatory recycling) and socio-demographic attributes (home ownership, income, education, household size, age) as explanatory variables. The data set contained household survey data set from 12 municipalities in Ontario, Canada.

They found that a user fee has a significant positive effect on the intensity of recycling for all the seven sources but toxic chemicals (which are excluded from the unit pricing system). Secondly, mandating kerbside collection results in increased recycling of all sources but glass. Thirdly, increasing the frequency of collection from every second week to every week (and reducing storage cost of sorted waste) results in increased recycling rate of glass, aluminium, and toxic chemicals. Fourthly, limiting on the number of units of garbage that can be placed at the kerb or bag had a negative and significant effect on the recycling of plastic and toxic chemicals. Finally, introducing free units below a threshold in a user fee system also had a negative significant impact on recycling.

## 6.4 Effects of Attitudes, Moral Norms and Social Contexts on Households Recycling

Finally, these economic studies have found that aspects like environmental awareness and moral norms may play a key role in recycling decisions and behaviour (Hage *et al*., 2009; Halvorsen, 2008, 2012; Viscusi *et al*., 2012). Indeed, Halvorsen (2012) explored the factors affecting household recycling activities across 10 OECD countries and showed that the most important motivations for household recycling are the belief that recycling is good for the environment and that recycling is a civic duty. Viscusi *et al*. (2011, 2012); Viscusi *et al*. (2013) made a differentiation between private values (individuals placing a higher value on the environment) and social norms such as the concern of how other people might judge individuals if they did not recycle. They concluded that these private values appear to be more influential than social norms. Besides the role of private values, Viscusi *et al*. (2012) also discusses about the role of education, income, and residential location as determinants of recycling behaviour across households. Regarding education, they argue that it may augment knowledge of the environmental benefits of recycling. Regarding income, there are competing effects on the recycling decision but overall, income has demonstrated to have a positive effect on recycling rates. The competing effect arises because in the one hand, it is expected that the time opportunity costs of recycling are higher when income increases, reducing recycling rates. But in the other hand, two income effects favour recycling rates. First, Viscusi *et al*. (2012) claim from their empirical study in the US that communities with high income levels have greater resources and may address such costs by making recycling more convenient through measures such as the availability of kerbside recycling. Second, there is positive income elasticity with respect to the valuation of the environment. This argument is based in the household's "warm-glow" feeling and benefit from recycling. Regarding low-income households, empirical evidence in the United States suggests that recycling and bottle deposits play an important role on the finance of these households. Indeed, there is evidence that bottle deposit redemptions are an important income supplement for scavengers, including the homeless.

Sterner and Bartelings (1999) conducted a study on attitudes and habits when it comes to recycling and waste disposal in three Swedish municipalities with different user fee schemes (flat, frequency-based and weight-based fee). The found that the households in the two municipalities with unit-based pricing systems tend to have higher percentages of recycling. Other explanatory variables that were found to have positive impacts on recycling were the household's habit or previous experience with recycling and the household's information about waste problems (paper). While household size had a positive effect on paper recycling it had a negative effect on the recycling of textiles (a possible explanation is that younger children in the family inherit clothing from older siblings). Reducing the effort needed to recycle had a positive impact on the recycling of some sources; glass, paper, and batteries. Finally, the attitude about the importance of waste and recycling seemed to have positive impact for the recycling of textiles.

Berglund (2006) estimated households' willingness to pay to for waste sorting activities as a function of income, gender, age, education, type of housing, distance to recycling centre, whether waste sorting is perceived to be a requirement imposed by the authorities, whether recycling is perceived to be a pleasant activity, and the green moral index (GMI) as a measure of moral motivation for recycling. The survey was carried out in a municipality in northern Sweden. Every variable were significant except income, education, and whether recycling was a pleasant activity. Male, younger individuals, people living in apartments or further away from recycling centres, people who perceive sorting at source to be a requirement imposed by the authorities, and individuals with weaker moral reasons for undertaking recycling activities showed a higher willingness to pay. Hence moral motives for recycling results in a lower willingness to pay to have someone else take over the recycling activities, can then help explain why the real cost associated with recycling efforts, as captured by willingness to pay, is lower than the time cost of recycling which is given by the opportunity cost of lost leisure.

Finally, in a meta-analysis Hornik *et al*. (1995) found that internal facilitators (knowledge of and commitment to recycling) were the strongest explanatory variables of recycling, followed by external incentives (monetary incentives and perceived social influence) and finally internal incentives (control, ecological concern, and personal satisfaction). Among external facilitators (proximity of containers, frequency of collections, and distribution of materials), only frequency of collections was significant while the other two facilitators, along with demographic variables had the least explanatory power. The study was based on 67 empirical studies that were classified into the four theoretical groups above.

# 7. Recyclers and Manufacturers

Recyclers take “upstream” decisions about the design of recycling program, collection infrastructure and pricing policies for collection services (which affect inconvenience costs and the incentives of households and industry for sorting and separation). Recyclers also take “downstream” decisions about treatments in the recycling process or incineration, landfill or even export of waste plastics. Market failures in these decisions may occur since recyclers often only take into account their own operating costs and not the social externalities (figure 6 in chapter 5). The market conditions or its actors, recyclers and plastic manufacturers, are not so well studied in the economics research literature as households. A literature review as well as a survey to 62 managers in recycling and plastic manufacturing industry in Sweden was implemented as part of the actor analysis in this project. Due to the limited size and time constraint of the project it was not possible to make a survey in all four Nordic countries. Sweden was chosen because IVL could use its existing contact lists to find managers in the plastic recycling and manufacturing industry. However, we imagine that these managers are representative also for the other Nordic countries since markets of recycling and manufacturing are international.

## 7.1 Recycling Technologies

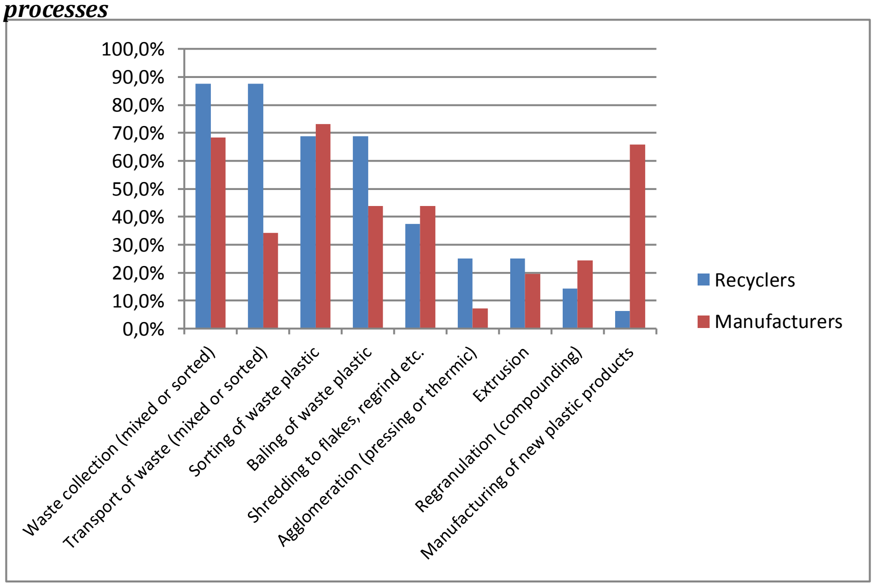
The transformation from waste plastic to recycled plastic occurs in several treatment processes – collection, sorting, baling, crushing, reprocessing, conversion and manufacturing of products made of recycled plastics. The transformation from waste to products is here divided in three steps depending on the type of market actors that usually specialize in performing them:

1. Collection, sorting, cleaning, size reduction and/or baling.
2. Reprocessing, for instance grinding, extrusion or pelletizing.
3. Manufacturing of new plastic products (extrusion, moulding or blowing).

There are no sharp distinctions between the steps. There are a few firms on the market that are involved in all three steps from collection of waste to manufacturing of products made of recycled plastics. However, the three steps still shows the common differentiation in specialization, especially when the source is household waste which consists of more heterogeneous fractions that require more effort in collection and sorting. This implies that firms specializing in step 1 and sometimes 2 sell the intermediate secondary raw material to recyclers or manufacturers specializing in plastic manufacturing in step 3.

Figure 7 shows the share of recyclers and manufacturers in the Swedish survey involved in the various waste plastic treatment processes. Recyclers tend to dominate early treatment processes and manufacturers later but can be involved in early treatment processes too when it comes to homogenous fractions of industry waste in take back in business-to-business contracts. Manufacturing of plastic products made of recycled plastics is dominated by manufacturers. Only 6% of the recyclers are involved also in plastic manufacturing. The steps in the recycling process from collection to manufacturing are connected by business-to-business contracts between recyclers and manufacturers.

***Figure 7: Share of recyclers and manufacturers involved in different treatment processes***



### 7.1.1 Collection and Sorting for Quality

During the recycling process, mixed plastic waste needs to be sorted by polymer type (the most common types are HD-PE, LD-PE, PP, PET, PS and PVC) to be kept separated during the whole reprocessing stage. Otherwise, when heated and melted together, they tend to separate like oil and water resulting in a weak output material. Waste plastics consisting of mixed polymer types are usually "down-cycled" to lower quality products than it originally was used for or even incinerated. Consequently, waste plastics sorted by polymer type have a higher market value than mixed waste plastics.

Waste plastic is also sorted by colour. Transparent waste plastic can be used to produce transparent or coloured recycled plastics while it is difficult to produce transparent recycled plastic from coloured or mixed coloured plastic waste. Moreover, black plastic waste cannot be detected by automatic optical sorting machines. In summary, transparent waste plastics have higher market values than coloured or mixed coloured waste plastics.

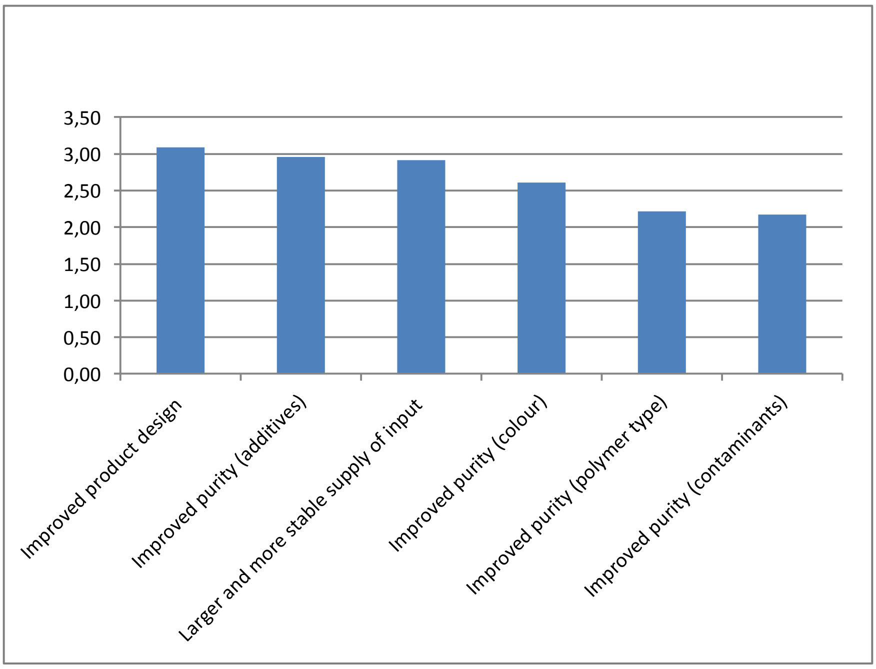
Waste plastics need also to be clean from contaminants (e.g. food, fat or oil previously contained in plastic bottles) otherwise the waste plastic will not keep the same quality as it had originally which might imply down-cycling or incineration. Contaminations with other non-wanted polymer types or contaminants from for instance paper labels on container can lower the market value of mixed waste plastic bale with up to 30–40%.

It is not possible to sort all waste plastics into pure qualities. For instance, more homogeneous waste plastics can be obtained by some sources (e.g. some take-back systems for industry waste) than others (household waste). Furthermore, plastics can be recycled a limited amount of times due to gradual breaking of polymer structures in each recycling phase. Waste plastics, once with high quality, will eventually reach lower quality levels as the number of reprocessing grows. Finally, plastics may often contain additives to enhance the properties, such as hardness, softness, UV resistance, flame formation resistance etc., of the plastic product. The content of additives can varies from less than 1% in PET bottles and up to around 50% in PVC. Additives can in some cases also affect recyclability.

## 7.2 Domestic Markets for Plastic Waste

The market supply of waste plastics potentially cover a range of different quality levels from high quality pure polymer plastics with the same quality as virgin plastic to low quality mixed waste plastics. From manufacturers' viewpoints the value of recycled plastics as input depends primarily on the quality and homogeneity of the polymer in the plastic waste. A homogeneous single polymer plastic free of contaminants has more options to be used in a range of new high quality products than a mix of different polymers and/or a mix of coloured plastics (especially darker colours) and/or is contaminated by materials (e.g. remains of food or motor oil) inside the plastic containers. The market price of a recycled polymer is determined by its quality (which in turn affects its substitutability with virgin plastic) and the price of virgin plastic. Figure 8 shows the most important quality factors of input waste plastics as stated by managers of manufacturing firms in Sweden. Product design, purity in colour, additives and a stable supply of input material over time got the highest grades.

***Figure 8: Manufacturers on the importance of quality factors for increased recycling rates***



From a recycler's viewpoint the incentives for engaging in collecting and sorting waste plastic is the increase in market value as it can be used as input in reprocessing and manufacturing of higher quality goods. However, for each potential quality level of the input waste material there is an upper threshold where sorting costs reaches a point where it is not profitable to raise the level of sorting further. When this threshold is passed the waste will usually be down-cycled or exported to foreign countries with lower treatment costs. This threshold will depend on a range of parameters, for instance, labour costs, the level of sorting by households, economic risks of investments due to volatility in domestic waste supply, international prices on recycled vs virgin plastics which in turn can depend on the presence of any market failures in terms of externalities in collection, production as well as asymmetric information between actors in the steps of the recycling processes.

There is a relationship between the value of the recycled material and the transaction costs of recycling. Market exchange of recyclables is assumed to be costly, because it requires recyclers to determine how valuable products are for recycling and pay a price based on that value (Calcott and Walls, 2005). Consumers may also incur costs in making items available to recyclers. Indeed, besides the benefits of the recycled material, the costs associated with recycling and processing the recycled material, and the costs associated with disposing of the material if it is not recycled, also determines the desirability of recycling from a policy perspective (Viscusi *et al*., 2012). Therefore, the value of the recycled material should justify these costs. From the consumer's perspective, they "may be deterred from selling some of their used products to recyclers, and hand them over for free instead, such as in a kerbside recycling bin. Other used products will be valuable enough, however, so that it is worthwhile to incur transaction costs" (Calcott & Walls, 2005). From the recycler's perspective, it should be noted that the prices of recycled raw materials are crucial. Moreover, these prices can be uncertain. A study from Lavee *et al*. (2009) based on 79 municipalities in Israel analysed the effect of price uncertainty and irreversible investment on the decision to switch from landfill waste disposal to recycling. To highlight the role of price uncertainty over decision-making, results suggested that uncertainty regarding the price of recycled materials might induce a risk neutral municipality to prefer landfill disposal, even when recycling is less expensive.

### 7.2.1 *Asymmetric Information in Recycling Markets*

In the survey to managers in Sweden, several recyclers and manufacturers highlighted problems with lack of information about the quality of recycled plastics on the market. The market supply of waste plastics could potentially cover a range of different quality levels from high quality pure polymer plastics with the same quality as virgin plastic to low quality mixed waste plastics. When recycled waste plastic is traded between sellers (typically recyclers) and buyers (typically reprocessors and manufacturers), the quality is usually specified in business-to-business contracts referring to specifications of national or European standards containing polymer type, colour and purity etc. The burden of testing conformity of requirements usually lies on the buyer, and it is also in the interest of the buyer who will pay the consequences of a lower-than-expected quality input in his production. Quality analyses of mixed plastic waste in bales add to the cost of reprocessing.

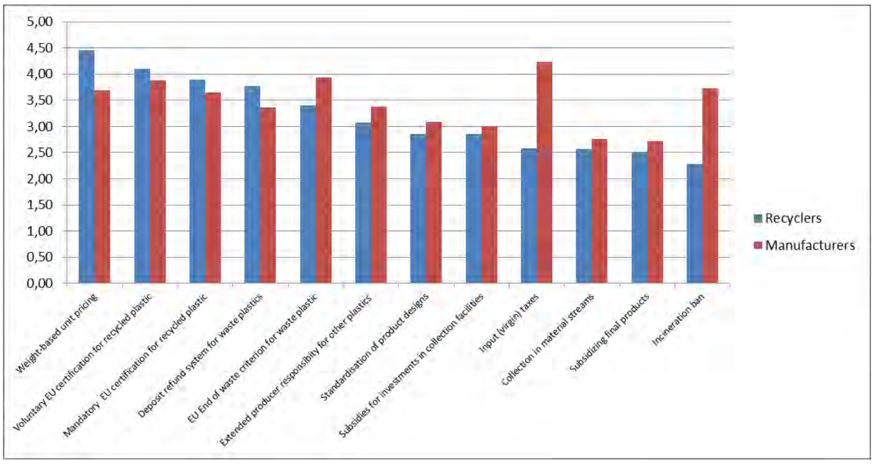
Asymmetric information was put forward both by managers of recycling firms (the sellers) and reprocessing/manufacturing firms (typically the buyers). Asymmetric information about the quality of recycled plastic waste may result in adverse selection and affects both sellers (that have more information) and buyers (that have less information) negatively. The adverse selection can be illustrated in the following simplified example. Assume that buyers of plastic waste (manufacturers) are facing contract offers from sellers of plastic waste (recyclers) of both low and high quality pre-treated waste plastics without knowing fully which quality a specific contract will bring. As a result, the expected market price of a contract offer would tend to be a value between the prices of a low and a high quality contract. Hence, recyclers that have invested in higher sorting levels and can offer a high-quality contract will not get a high-quality price. Since they would risk not covering their higher costs for sorting they would not invest in supplying high quality pre-treated waste plastics. As a result the market will contain lower levels of high quality pre-treated waste plastics and higher levels of high quality pre-treated waste plastics than in efficient markets with no market failure.

As the models on policy instruments for waste management have not yet analysed asymmetric information, the Fullerton and Kinnaman (1995) model was, in this project, extended to waste plastics with different quality levels. The result showed that asymmetric information between recyclers and manufacturers lead to adverse selection with inefficient effort levels in sorting (lower quality levels of the waste plastic supplied) and lower recycling rates than efficient level. However, the optimal designs of deposit-refund systems (or other upstream-downstream policies) remained under asymmetric information.

One way for recyclers (sellers) to try to overcome asymmetric information on the market is signalling and/or building reputation in long term business-to-business contracts. Another reason for long term contracts is that it increases the possibility to keep the same application for a plastic material as the one it once had (e.g. business-to-business contracts for take-back schemes in industry) makes it easier to meet the technical and legislative requirements of that application again. However, these benefits need to be balanced against the cost of maintaining homogeneous waste plastics streams for each application and the long term contracts. Other possibilities to reduce asymmetric information are voluntary certifications schemes and/or quality classifications of waste plastic.

Figure 9 shows which policy instruments managers believed to be most effective for increasing recycling rates. With the exception of a few differences, recyclers and manufacturer have similar beliefs. First, manufacturers graded virgin taxes higher while recyclers graded weight-based fees for garbage collection higher. The second highest graded policy among recyclers and manufacturers were EU certification schemes for plastic secondary raw material. Finally, manufacturers grade incineration bans significantly higher than recyclers as an effective instrument for increasing recycling.

***Figure 9: Effective policy instruments as stated by recyclers and manufacturers in Sweden***



Notably, subsidies for increased recycling/manufacturing got the second lowest grade among recyclers and the lowest grade among manufacturers. This may seem somewhat remarkable as subsidies are usually seen as popular instruments by industries. However, if there are tendencies towards adverse selection on the market, counteracting efficient recycling rates domestically, the effects of (and the gain to the industry from) output-based subsidies become uncertain.

## 7.3 International Markets for Plastic Waste

Common for all Nordic markets is a demand for a relatively high quality input and output plastic waste by plastic manufacturers. The Nordic markets for recycled plastic are characterised by low volumes and quality of input material as well as fluctuating supply of input material to recyclers. The supply of waste generation into the recycling system tend to be inelastic leading to high price volatility from demand shocks generating economic risk leading affecting investments in the recycling industry negatively.

FTI of Sweden and Grønt Punkt Norge have coordinately contracted four sorting facilities for their collected plastic packaging waste, one operator in Sweden and three in Germany. Swerec in Sweden accepts around 30,000 tonnes of plastic packaging from FTI and around 10,000 tonnes from Grønt Punkt Norge per year and the remaining is sent to Germany. At the Swerec facility the incoming plastic packaging is to 50% rigid plastics and to 50% flexible plastics (LDPE) and approximately 11% of the incoming material consists of non-plastic contaminants. Around 75% of the remaining 89% is sold for recycling. The polymers sorted out are LDPE, HDPE, PP and PET. Of the plastic packaging waste sent to the German facilities around 20% was sorted out as flexible plastics (LDPE), 32% as rigid plastics (HDPE, PP, PET and PS), and 29% as mixed plastics for recycling and 18% for energy recovery in 2012.

Swerec sells about half of the flexible plastics to the Nordic market. Almost all rigid plastics are sold to the European market, but all of it is not subject to recycling within the EU. Less than 5% of the rigid plastics are sold to the Nordic market. The secondary raw material from rigid plastic packaging waste is commonly recycled into plastic products and not back into plastic packaging. Flexible plastic packaging waste is often recycled back into packaging in the form of plastic bags.

Recyclers in Denmark are small enterprises which mainly recycle production waste from industry. The demand for recycled plastic is poor in Denmark and the number of Danish recyclers has decreased during the last 15 years. The main part (80–90%) of plastic packaging collected in Denmark is exported for recycling. The main part is transportation packaging and PET bottles (from the deposit system) of high recycling quality. The recyclable PVC waste collected by WUPPI is recycled in Germany and sold on the European market.

Recyclers in Finland mainly recycle production waste from industry. The industrial plastic waste is recycled at a recycling facility or receives treatment (sorting, pre-treatment) before export (mainly to Germany and China) for recycling, or is recycled at site at the production plant. It is common for the waste generating companies to buy back the recovered waste plastic as plastic recovered from industrial waste can be contaminant-free and have as good material properties as virgin raw materials.

Small countries in general in the EU face the problem of small volumes. Nordic countries tend to be geographically large countries with relatively low populations which result in higher transport and infrastructure costs in waste management systems relative to continental countries. Within the EU, countries with smaller populations tend to export to countries with large reprocessing capacities such as Germany, the Netherlands, Belgium and Italy which together import almost two thirds of the intra EU imports. In Finland and Sweden approximately 50–55% of all collected plastic waste is exported primarily to Germany, EU and the Asian markets especially China and Hong Kong. In Denmark approximately 80% of the waste collection is exported mainly to Sweden, Germany and the Asian markets. The large share of export of mixed plastic waste in Denmark is explained by the fact that sorting facilities are not developed to the same extent as in other Nordic countries. This is further explained by the fact the Denmark is the only Nordic country that does not have a producer responsibility scheme for packages (usually the largest plastic waste material flow) that give incentives for recycling. The survey to Swedish managers showed that 50% of the managers in recycling answered that they frequently sell to China.

Asian countries, and especially China, have since the 1990s become major importers of waste plastic from EU which has strongly influenced the waste plastic prices. In 2006 about 90% of the total EU waste plastic was exported to China and Hong Kong. Smaller countries in the EU with relatively low waste streams including the Nordic countries usually export the waste plastic either to other EU countries or Asian countries, especially China.

# 8. Economic Policy Instruments for Recycling of Plastic Waste

The chapter contains a review of the economics research literature on economic policy instruments for achieving socially optimal allocations of recycling rates. The review covers both theoretical and empirical results and covers the policy instruments that are most commonly highlighted in the literature on waste and recycling as potential instruments for efficient recycling.

## 8.1 Policy Mixes of Recycling Programs and Economic Instruments

The efficiency of economic instruments entails that recycling exists as an alternative option to disposal, e.g. by provision of kerbside recycling and the availability of nearby community recycling facilities. A tax or charge on garbage collection with no convenient recycling option available may instead result in increased illegal disposal of garbage (Kinnaman and Fullerton, 1995). Hence a recycling infrastructure that makes recycling possible and reduces the households' time and inconvenience costs associated with recycling may reduce any adverse effects of economic instruments due to high monitoring and enforcement costs. There are mainly two different types of recycling programs that are common in the four Nordic countries in the study:

- Bring system collection (manned and unmanned stations).
- Kerbside collection (commonly used by apartment buildings).

Bring systems are smaller unmanned or larger manned recycling stations placed at frequent visited areas and residential areas in the community where households can leave sorted waste for free. Manned cycling stations are larger and may use payment schemes. Kerbside collection implies that sorted waste can be left at stations within, or close to, dwelling buildings. For an overview of systems with corresponding statistics on recycling programs used in the Nordic countries from a waste plastic perspective (see Fråne *et al*., 2014).

Some empirical studies in the research literature have analysed the combined effects of recycling programs and economic instruments showing that, for instance, a policy mix of kerbside recycling and unit pricing becomes more effective than when only one of the instruments is implemented (Reschovsky and Stone, 1994; Callan and Thomas, 1997). A certain recycling program may decrease time (for instance by reducing the distance to the collection point) or effort, for instance by reducing the storage space needed at home for bulky waste, while a unit pricing system decreases the relative price of recycling. Thus depending on the characteristics, such as weight and bulkiness, the instruments may reinforce each other (see also chapter 6).

The households' knowledge about and experience within recycling programs tend to have positive effects on whether households recycle, as concluded in Reschovsky and Stone (1994), however experience with recycling programs increased the probability of recycling newspaper but not of recycling glass and plastic bottles and aluminium (Jenkins *et al*., 1993).

This suggests that efficient design of instruments to encourage recycling should take into account inconvenience costs and behavioural drivers (Halvorsen, 2008). This poses methodological challenges at the time of framing environmental policy. For instance, Pickin (2008) reviewed 37 cost–benefit analysis (CBA) as decision tools in the area of solid waste recycling and revealed how they differ in fundamental assumptions such as:

> "the types of environmental impact and their valuation; the relevance of upstream externalities; whether there is a scarcity externality; the economic significance of householder efforts; and the need to drive towards long-term sustainability through eco-restructuring"
>
> (Pickin, 2008)

## 8.2 Extended Producer Responsibility (EPR) Schemes

Producer responsibility schemes can be of two different types. The first type implies that the producer has a physical responsibility to take back his products through an obliged waste management treatment. This type is essentially a command and control instrument. The second type, which is most common in packaging, implies that the producer has a financial responsibility to finance the waste treatment of his products (usually a fee payment proportional to the product sales volume) basically making the EPR an economic instrument.

Producer responsibility schemes exists for several different types of waste streams in EU member countries, for instance batteries, WEEE, tyres, papers, oil, and medicines. Some of them are mandatory at the EU level and/or the national level while others are voluntary. Most relevant though for plastic waste are the producer responsibility schemes for packaging. EC (2012) found that 24 member countries used producer responsibility schemes for packaging with fees per tonne for plastic waste ranging from EUR 20 Romania to EUR 1,300 in Germany.

Household plastic packaging waste is collected within the extended producer responsibility (EPR) schemes (Finland, Norway and Sweden) or municipalities (Denmark). In Finland and Sweden the EPR systems are mandatory for producers by legislation and in Norway voluntary agreements between the producers and government (see also chapters 3 and 4). In the EPR system collection is managed by bring systems (single-family houses) which requires citizens to bring the recyclables and leave them separated at unmanned recycling stations or kerbside systems (apartments) which require citizens to leave and separate recyclables in closely located collection points.

Most EU member countries have introduced mandatory or voluntary EPR with financial responsibility schemes for packaging. The exemptions are Denmark and the Netherlands (which instead implemented packaging taxes and municipal responsibility for packaging waste). Like a packaging tax, the fees paid by producers in an EPR creates incentives for the producer to use less of, or different, material in production. However, since the revenue of the EPR is used to finance recycling, the incentive mechanism is also aimed at increasing recycling rates besides than just reducing waste generation as a packaging tax.

EC (2012) examined the relationship between the fees paid by producers in the schemes with the packaging recycling performance in the EU member countries but did not find a clear relationship. In theory, the cost-efficiency and internalisation level of a producer responsibility scheme depends on to what extent the fees cover the costs of recycling and the application of marginal pricing. Ideally the fees should be based on the amount of externality placed on the market by producers (either per unit, or per kg or tonne). EC (2012) furthermore found that the producer fees covered the full costs only in Austria, Belgium and Germany. In other countries it is not analysed whether there is full cost coverage by producers or marginal pricing is used. In Norway, the fees are differentiated to the type of polymer. In Sweden prices are differentiated between consumer, industry and service packaging as they correspond to different recycling costs. However, due to transaction costs simplifications in the methods for calculating annual fee payments are common (e.g. sampling and templates using firm turnover as proxy for packaging amount). Firms using small amount of packaging can pay a fixed annual cost. In Finland only firms with an annual turnover larger than 1 million euro pay to the producer responsibility scheme implying that the externalities of small producers are not internalised. Cost efficiency in this sense is a trade-off between increased efficiency of improved marginal pricing and the increased administrative costs of measuring and monitoring marginal units.

The UK has implemented a special type of producer responsibility scheme based on tradable quotas called "packaging recovery notes" (PRN) with the purpose to fulfil the EU targets for the recycling of packaging waste. The system which is basically an EPR but in terms of tradable quotas, was implemented in 1997 jointly with the Producer Responsibility Obligations (Packaging Waste) Regulations 1997. A PRN can be issued by accredited re-processors for each tonne of packaging recycled and recovered. A Packaging Export Recovery Note (PERN) can be issued by exporters of packaging waste for each tonne of material exported to foreign re-processors. Producers on the other hand are subject to material obligations (type and weight of materials) as well as recovery obligation. Both are determined by material specific and overall recovery targets set to meet the medium and long-term obligations under the EU Directive. The advantage of tradable quotas is that that the quantity constraints make the instrument more predictable to achieve certain targets for recycling rates than an EPR based on fees. The disadvantage is the higher monitoring and transactions costs following tradable quotas.

Similar to Germany's green dot program, the Swedish producer responsibility ordinance for packaging is an EPR that mandates producers to collect, remove and recover the packaging waste from consumers. As noted by Hage and Söderholm (2008), the producers are required to collect at least 70% of all plastic packaging waste (in terms of packaging weight). From these, at least 30% of the plastic packaging should be recycled, and hence used as input in new plastic products. The rest of the collected packaging, 40% of all plastic packaging, can be used for energy recovery purposes since in Sweden they are not allowed to end up in landfills. Regarding the scheme, producers need to provide suitable systems for collecting packaging waste and inform households about these systems. Municipalities are responsible for supervising the collection within their own borders, and households, who do not receive any economic compensation for their effort, have the responsibility to clean and sort the packaging waste (paper, plastic, glass, and metal) and transport it to drop-off recycling stations.

## 8.3 Product Design and Recyclability

Under the concept of "Design for Environment" (DfE), firms can, from the beginning, make products that are easier and less costly to recycle (Calcott & Walls, 2000). Disposal fees (e.g. weight-based or per unit) are downstream instruments that could, in theory, send signals upstream to producers. However, Calcott and Walls (2000) demonstrated that disposal fees alone cannot provide sufficient incentives for DfE without a fully functioning recycling market. Therefore, upstream instruments do appear to be necessary to spur product design activities on the part of producers. In the case of deposit and refund systems, they have traditionally worked as downstream systems where the consumers are generally those who receive the refund. However, deposit and refund systems can also be designed as upstream systems where the processors or collectors of recyclables – rather than consumers – receive the refund. According to Walls (2011), upstream systems may have lower transaction costs and better environmental outcomes than traditional downstream systems.

Fullerton and Wu (1998) present a general equilibrium model that encompasses the entire life-cycle of a product, from the design phase to production, consumption and disposal and that includes a negative externality from total waste generation. As it captures each price paid along the way, it shows conditions where the efficient solution can be obtained either by a "downstream" tax on waste disposal or by an equivalent "upstream" tax on production processes that give rise to subsequent waste. They analyse disposal-content fees, subsidies for recycling, unit-pricing of household disposal, deposit refund systems and manufacturer "take-back" requirements.

In their model, firms produce output using primary resources (labour or capital) and recycled materials. They also choose an amount of packaging and a level of "recyclability" intended to reflect the resources needed to implement a product design that would allow the subsequent recycler to take apart the item more easily. For example, separate the different types of plastic, and recycle a higher percentage of it. Households in the model supply primary resources (labour or capital), retain some resources for home production for leisure and generate amounts of garbage and recycling that depends upon the firm's choice of packaging and the firm's choice of recyclability. To achieve this social optimum in a decentralized economy, the government can implement taxes on packaging or on garbage disposal or use subsidies to garbage collection and recycling. In the case of taxes, revenues are returned to consumers in the form of lump sums. In the case of subsidies, they assume that government use lump sum taxes to raise any revenue to pay for subsidies.

## 8.4 User Fees for Collection Services

Historically many countries charged flat fees or general municipal taxes to households for waste collection. A flat fee-pricing scheme means that private households do not pay a variable price for waste collection. Instead garbage collection is financed by either a general municipal tax or a flat fee, or a combination of them; regardless of the amount of waste they actually generate (Bartelings *et al*., 2004). According to economic theory the absence of variable costs reflected in pricing for waste disposal leaves the households to perceive a zero marginal cost of the garbage collection services. Thus they have no incentives to reduce the amount of garbage by producing less garbage and/or recycling more.

On the other hand, unit pricing implies that a cost is imposed for each extra unit of garbage collected from the household. In the empirical research literature there exists strong evidence that unit pricing affects household behaviour towards lower waste generation and/or increased recycling as discussed in chapter 6, e.g. Callan and Thomas (1997), Dijkgraaf and Gradus (2004), Ferrara and Missios (2005), Fullerton and Kinnaman (1996), Hong *et al*. (1993), Hong (1999), Hong and Adams (1999), Jenkins (1993), Linderhof *et al*. (2001), Nestor and Podolsky (1998), Podolsky and Spiegel (1998), and Van Houtven and Morris (1999) and Skumatz (2008). There are only a few empirical articles that do not find a clear positive behavioural effect on recycling from unit (marginal) pricing (e.g. Jenkins *et al*., 2003); Kinnaman and Fullerton, 2000); Sterner and Bartelings, 1999).

When choosing a unit pricing system, not only the user fee level needs to be decided but also which unit to use for the pricing system. The most commonly used are:

- Volume-based pricing – households pay according to the volume collected, for instance measured by each bag picked-up (sometimes called bag-based program).
- Frequency-based pricing – households pay a fixed fee per pick-up (regardless how much waste the bin contains) in a subscription program where the household subscribe to and pay for a specified number of pick-ups per period of a specific bin size (sometimes also called a block payment program).
- Weight-based pricing – households pay according to the weight of their garbage as the bin is identified by a microchip and weighted with contents upon each pick-up.

A study by EC (2012) showed that 17 member countries used some form of unit pricing. Specifically 16 used volume-based schemes, 15 frequency-based schemes, nine weight-based schemes, and six bag-based schemes. However, coverage differs largely between countries from zero up to 100% of the municipalities in Austria, Finland and Ireland.

It is clear that the various pricing systems create different behavioural incentives at the margin. For instance, in volume-based pricing a household that fills 1.5 bags per week can optimize by storing the half-filled bag until next week and dispose 2 bags every second week. Hence the household always face a positive marginal cost. In a frequency-based program the household can only choose to subscribe to either 1 or 2 pick-ups per week. When the household chooses 2 pick-ups per week the marginal cost is zero of increasing waste (or reducing recycling) by up to half a bag per week.

A frequency-based pricing program creates incentives for households to stay within their subscribed volume flow by using recycling as a way to adjust the garbage flow. Hence the incentive to recycle is largest for households whose usual waste flow is slightly below the subscribed flow (only a small increase in the waste flow would lead to the extra cost of extending subscription with another pick-up per period). Recycling incentives are smallest for household whose usual waste flow is slightly above the subscribed volume (thus they have almost a whole bin left to fill up at no extra cost in each period). This inefficiency is referred to as the “lumpiness” problem by Nestor and Podolsky (1998). Reducing the bin size would in theory reduce the allocation inefficiency, however, at the cost of increased transport costs.

There are currently around 20 municipalities that instead have introduced weight-based pricing in Sweden. Hage and Söderholm (2008) analysed the main determinants of collection rates of household plastic packaging waste in 252 Swedish municipalities. The regression analysis suggested the collection rate appears to be positively affected by weight-based pricing with larger effects than flat and/or volume-based pricing. Similar results were found by Hogg (2012) in Denmark. There are also follow-up studies in municipalities that have introduced weight-based and found a reduction in waste after the introduction of weight-based pricing (see e.g. Stare and Sundqvist (2013). However, these studies have in most cases looked at breaks in trends at introduction rather than applying statistical analyses with test of significance and control variables.

Oostzaan was the first Dutch municipality to introduce a weight-based pricing system. Linderhof *et al*. (2001) analysed this case distinguishing between compostable and non-recyclable waste collected at the kerbside. They found that the elasticity for compostable waste is four times as high as that for non-recyclable waste. This can be explained since the distribution of special composting containers has facilitated home composting. In addition, their study revealed that long-run elasticities are about 30% higher than short-run elasticities (Linderhof *et al*., 2001). Another empirical study for the Netherlands was conducted by Dijkgraaf and Gradus (2009). Their study was based on a pooled cross-section model for the Netherlands for the period 1998–2005. Results suggest that unit-based pricing systems have a significant effect on the quantity of collected waste. What is novel from their study is that they incorporated the effect of environmental activism and concluded that despite of this correction, which could have overestimated the effects on previous literature, the effect of the weight and bag unit-based pricing system on the quantity of total waste is still considerable and ranges between 28–35%.

Traditionally, households in the US used to pay fixed collection fees but they have demonstrated to be inefficient since the marginal price of waste disposal is zero, while the marginal collection and disposal cost is positive (Sterner & Coria, 2012). This problem is also highlighted by Bartelings *et al*. (2004) for the Netherlands since most households pay a fixed amount of money for waste collection through taxes. In the US, communities are increasingly charging for waste collection based on weight or volume of waste. According to Skumatz (2008), these programs are available to about 25% of the US population and about 26% of communities in the US – including 30% of the largest cities in the US. The first study to calculate empirically the elasticity of the price of waste upon waste quantities was conducted by Wertz (1976) for San Francisco. The result gave a negative elasticity of -0.15. Since then, many studies have been conducted. Bel and Gradus (2014) conducted a meta-regression analysis for unit-based pricing based on a sample of 65 price elasticities obtained from 21 studies. From these, 13 studies analysed unit-based pricing cases in the US. Elasticity estimations varied depending on the study but overall, these studies showed that US is better equipped to work with extrinsic motivation through price incentives. Yet, according to Bel and Gradus (2014) any general conclusions are hard to draw since there are large regional differences in the use of pay as you throw (PAYT) systems across the USA. Therefore, results on the effectiveness of these programs vary from study to study. Kinnaman (2006) demonstrated that in the US, efficiency gains from unit-based pricing over the last two decades have been quite small. Alternatively, the benefits of recycling accrue primarily as warm-glow utility gained by recycling households. Hong *et al*. (1993) conducted a study with a sample of 2,298 households in Portland, Oregon. Bel and Gradus (2014) estimated a price elasticity of -0.03 using the data from Hong *et al*. (1993). In the other hand, a more localized study from Sidique *et al*. (2010) conducted in Minnesota, USA, showed that variable pricing of waste disposal increased the rate of recycling. Streich (2006) investigated the effect of the "price-per-bag" garbage collection scheme that was implemented in Charlottesville, Virginia in 1992 and demonstrated that a unit-based garbage tax is regressive. Specifically, "the lowest income group spends an average of 0.64% of their income each year on waste management services, whereas the highest income group only spends 0.06%" (Streich, 2006).

In summary, the administrative costs differ between the unit-based pricing systems, weight-based pricings clearly being the more expensive (Dahlén, 2009). Thus the gains in allocative efficiency and effectiveness need to be evaluated against its larger administrative costs. This is nevertheless true for areas with apartment blocks usually applying kerbside collection where it would be costly to apply quantity-based pricing at the household level. Unfortunately the empirical research literature does not say much about administrative costs of different pricing systems.

A general drawback with unit-pricing is when using weight-based pricing as incentive for increasing recycling of waste plastics. Since waste plastic has relatively low density compared to other substances, it will get a relatively small marginal effect on the weight relative to other heavier substances.

Unit pricing makes it possible in theory to set the price per unit of garbage collected equal to the social marginal cost of collection and recycling. However, when municipalities in reality implement unit pricing they use average cost pricing and tend to include the costs of waste management treatment but not the external environmental damage costs of garbage such as the loss of biodiversity etc.

Unit pricing has been broadly discussed since Fullerton and Kinnaman (1996) highlighted that one of the concerns that policy makers had with unit pricing was the possibility of increases in illegal forms of garbage disposal. This concern was corroborated years later by Skumatz (2008). However, based on a survey of more than 1,000 communities, the author concluded that despite of this concern among elected officials and planners, illegal dumping is a bigger fear than reality since it is a problem in about 20% of communities and a significant issue in only 3% of the communities. The empirical literature, with the exemption of a study by Fullerton and Kinnaman (1996), where 28–43% of the garbage weight reduction is explained by dumping, show no, or small (up to 5%), waste transfers to for instance neighbour municipalities with flat fees (Reschovsky and Stone, 1994; Van Houtven and Morris (1999); Dijkgraaf and Gradus, 2004; Linderhof *et al*. 2001). Future research is needed to further examine to which extent unit pricing generate adverse effects in households when it comes to waste generation and collection. One way to reduce adverse effects could be implementing and informing about a convenient recycling infrastructure (lowering the costs of time and effort of recycling) before and at the introduction of unit pricing as well as improved enforcement of illegal disposal.

## 8.5 Deposit and Refund Systems

Deposit and refund systems are market-based instruments that combine a tax or disposal fee (deposit) when purchasing a product with a recycling subsidy (refund) when the product is collected and/or recycled (Walls, 2011). Deposit-refund systems are commonly used for beverage containers, batteries, motor oil, tires, various hazardous materials, electronics, and more (Walls, 2011). In the industry sector voluntary deposit and refund systems organized by firms themselves are also common for reusable packaging such as pallets and crates.

While the effects and allocative efficiency of deposit and refund systems have been analysed extensively in the theoretical literature they have not been analysed in the empirical literature to the same extent as e.g. unit pricing. In several theoretical papers deposit and refund schemes are found to be the best instruments to internalize the private and external cost of waste generation and disposal (Dobbs, 1991; Dinan, 1993; Atri and Schellberg, 1995; Fullerton and Kinnaman, 1995; Palmer *et al.*, 1997; Fullerton and Wu, 1998; and Ferrara, 2003; Kinnaman, 2010). The reason is that a deposit and refund system is an incentive mechanism covering both source reduction and recycling ensuring that resources (efforts) are allocated efficiently (Palmer *et al.*, 1997). In other words, a deposit and refund system creates incentives for the optimal allocation between reducing consumption and increasing recycling. Furthermore, Fullerton and Wu (1998) shows that a deposit and refund system can be designed such that it gives incentives for firms to increase recyclability e.g. by the choice of product design. Kinnaman (2010) showed that when recycling by centralized facilities is more cost-efficient than recycling by households, the refunds in a deposit and refund system should be paid to the centralized recyclers rather than households to preserve the efficient allocation of the deposit and refund system.

The empirical literature does not contain any studies that can verify the actual effects of deposit and refund systems on household behaviour. Still the results favouring deposit and refund systems are theoretically valid. A major difference between the incentive mechanisms of a unit pricing system and a deposit and refund system is that the latter can also ensure that the costs of illegal disposals are internalized (Kinnaman and Fullterton, 1995). This would be especially important when the households exhibit illegal disposal behaviour and monitoring and enforcement costs are high for detecting this (for instance in areas with low population density). Another advantage is that the refund can be "directed" towards specific sources and can then at least in principle be differentiated with respect to the different social costs of various sources. Unit pricing systems would impose a uniform unit marginal price for the mixed waste stream regardless the composition of sources.

A major disadvantage of deposit and refund systems is the larger administrative costs for administrating a charge as well as a refund. This is likely the reason why deposit and refund systems have been used mainly for homogeneous standardised products such as beverage bottles (where the standardised design of the bottles makes it possible to reduce transaction, monitoring and verification costs by for instance administrating the refund automatically in reverse vending machines located in food stores) or for standardised reusable packaging such as pallets in the industry sector. The administration and transactions costs would be even higher if deposit and refund systems would be used for a heterogeneous source, such as waste plastic in general. Using, for instance, weight as the pricing unit, the plastic content of the products would need to be weighted by purchase and again by recycling which would lead to high transaction costs. To keep transactions costs low in such a system, the number of actors in the system would need to be low which suggests that producers and recyclers rather than households would be the actors in the system. Moreover simplified measures or proxies for measuring quantities (weight, volume or units of products etc.) would be needed to be developed. Thus to keep transactions costs low, the design of a deposit refund system for a heterogeneous source, such as waste plastic, would likely take the shape towards more of an extended producer responsibility scheme than the original thought of a deposit refund system.

## 8.6 Taxes and Charges

Input and output taxes on production are sometimes mentioned in the policy literature on recycling. However, input and output taxes or charges on production do not provide recycling incentives per se unless they affect behaviour towards substituting to other input materials that are more recyclable.

**Input Taxes**

Input taxes on input materials in production such as taxes on virgin plastics, give incentives to reduce production and consumption of materials, and under certain circumstances encourage re-use and increase recycling of materials. However, policy instruments focusing only on input use, such as primary/virgin materials taxes cannot generate the optimal amount of disposal unless combined with a tax or subsidy on consumption (Palmer and Walls, 1997). Fullerton and Kinnaman (1995) also show that other sources must also be taxed to achieve the first best solution when it comes to recycling. Söderholm (2011) concludes that in cases where optimal policy instruments are not feasible, virgin taxes could play a role, particularly if they are complemented by downstream instruments to facilitate material recycling as well as the abolishing of harmful subsidies to virgin material extraction.

In practise, input taxes may also be an issue of political acceptance and other obstacles as it in general would influence trade of materials and products both within EU countries, and between the EU and the rest of the world. There may also be difficulties in achieving unanimity at the EU level for such taxes at the national level. In addition, since plastics is used in a vast of applications, the design need to take careful consideration such that it does not give incentives to unintended switches to other materials with even larger externalities. In practice, the effects of virgin taxes may therefore be more difficult to predict, besides that it may be more difficult to implement.

**Output taxes**

Output taxes or charges are examples of product taxes or charges on the final product sold to households e.g. a packaging tax such as a tax on plastic shopping bags. They are like input taxes upstream instruments and generate therefore similar incentives to reduce consumption and waste generation as well as switching to more environmental-friendly products provided they are applied to products, such as disposable items or plastic bags, for which there are substitutes which lead to lower levels of waste generation (Palmer and Walls, 1997). A tax may then not only discourage consumption of these products but also encourage the use of alternatives that contribute to a reduction in waste generation. However, again they do not per se affect incentives for recycling.

Some EU member countries have introduced packaging taxes. For instance, Denmark and Ireland have a tax on carrier bags while Belgium, Denmark and Norway have a tax on non-refillable packaging for beverages. Finland on the other hand has a tax for packaging that are non-recyclable and non-refillable (Finnish Ministry of Environment 2008). The packaging tax in Denmark introduced in 2001 with the purpose to discourage the use of packaging are not designed to internalise the environmental impact of the different packaging materials. The tax covers only about 13% of the retail packaging which corresponds to around 7% of the total packaging consumption (ETC/SCP, 2012). In total the different Danish packaging taxes are estimated to cover about 30% of total sales value of packaging and 15% of the total packaging by weight (Fischer 2010).

The Danish packaging tax consists of two parts. The first part is a volume-based packaging tax introduced already 1978 covering beverage packaging such as soft drinks, drinking water, wine, spirits etc. The tax rate is lower for beverage packaging subject to a deposit. The second part is a weight-based packaging tax that covers a broader range of packaging and types of commodity groups. This tax was introduced 1998. Since 2001 the weight-based tax is based on an environmental index, the weight of the packaging and the type of materials. A study by ETC/SCP (2012) could not find a reduction in the aggregate weight of packaging material since the tax was introduced. Due to lack of information on packaging amounts prior to the introduction of the tax and the missing information on refunds paid further analysis was not possible.

Another example is the packaging tax in the Netherlands that was introduced in 2008 and covers 95% of the packaging on the market. The current tax finances a waste fund from which municipalities are paid for the collection and registration as well as financing projects that prevents packaging waste. Such a policy mix could, like an EPR or deposit and refund system, have lager possibilities to affect recycling rate than isolated input or output taxes. Since the packaging tax was introduced recently there is still not yet enough data to carry out statistical analysis of the impacts of the tax.

### *8.6.1 VAT Differentiation*

One possibility to change the relative price of goods made of recycled plastic and goods made of virgin plastic would be by lowering VAT for goods made of recycled plastic. Within certain limits, the VAT Directive (2006/112/EC) opens up for member countries to introduce a differentiated VAT scheme. However, lowering VAT for final goods made of recycled plastic would be an imprecise instrument. Final goods are in general produced by a range of production processes and consist of a composition of different materials, all with different external social costs. Hence goods containing recycled plastic could have been produced in processes or contain other materials with other and sometimes larger environmental externalities than switching from virgin to recycled plastic. A reduction in VAT of such goods could in general increase inefficiency. For that reason, using VAT differentiation to internalise market failures due to environmental externalities cannot be recommended as a general rule. The aim should rather be to impose economic incentives, such as taxes or charges, closer to the sources of the externalities.

### *8.6.2 Taxes and Gate Fees for Landfill and Incineration*

Under efficiency, all producers of waste should face an incentive to reduce waste generation and to make use of cost-effective recycling services. Landfill taxes increases the range of recycling services that can be offered cost-effectively and the costs of providing some services may be fully supported through producer responsibility schemes. The same could be said for incineration taxes. Landfill taxes for disposal of municipal waste sent to legal landfills are currently used in 19 EU member countries. The tax levels range from about EUR 3 up to more than EUR 100 per tonne. Correlation analyses between EU countries using land fill taxes suggest a negative relationship between the tax level and the percentage of municipal waste sent to landfill (COM, 2012). Denmark, Germany and Sweden belong to the group of EU member countries that have the highest total tax levels and also the lowest percentage of municipal waste landfilled. However it should be noted that all these countries have introduced land fill bans or restrictions also being strong explanatory factors for the low percentage of waste sent to land fill. Besides land fill taxes there are also usually gate fees paid when entering the landfill facility which ranges from about EUR 20 to EUR 155 in the EU countries.

Incineration taxes are, or have recently been, in place in only six EU member countries as well as Norway. The tax levels vary, or have varied, from about EUR 3 to EUR 44 in Denmark. Sweden had an incineration tax in place from 2006 but abolished it 2010. Also Norway abolished its tax 2010. Just as with land fill gates there are gate fees set by the operator of the incinerator. In the Nordic countries, the gate fees as calculated per tonne are often in the same range as the tax level. All EU member countries that have, or have had, incineration taxes also have landfill taxes that are higher than the incineration tax (EC, 2102).

# 9. Concluding Summary

This chapter summarizes the results from the overview of existing policy instruments and main challenges in chapters 2–4 and the literature review on economic policy instruments for recycling and the analysis in chapters 6–8. It also provides policy recommendations for the Nordic countries connected to prevailing market failures which can be illustrated by simplified figure 6 in chapter 5.

## 9.1 Existing National Targets and Policy Instruments in the four Nordic Countries

Plastic waste is not specifically addressed by EU legislation and none of the four Nordic countries in this study has a specific plastic recycling target stated in their waste management plan. However, the Packaging Directive (94/62/EC amendments 2004/12/EC and 2005/20/EC) has a specific recycling target for plastic packaging. The minimum recycling target for plastics is 22.5% by weight, counting exclusively material that is recycled back into plastics. The national recycling targets for plastic packaging are higher than the EU requirements in both Norway and Sweden (30%). Denmark and Finland have the same target (22.5%) as set by the Packaging Directive. Based on the national reporting for 2011, the national targets are met in Norway and Finland, but not in Sweden. Denmark reported a slightly lower recycling rate in 2011 than required by the directive. The methods for calculating recycling rates differ between the Nordic countries though. Common methods in the Nordic countries for plastic waste streams could explore economies of scales in implementation. It would also open for using common instruments in the Nordic countries for exploring the larger scope of efficiency that exists at the Nordic scale rather than the scale of the single country.

The most commonly used economic policy instruments affecting waste plastic management in the EU-27 are producer responsibility schemes for specific waste streams such as packaging, deposit-refund systems for homogeneous products such as beverage bottles, charges and fees for waste disposal and treatment as well as landfill and incineration taxes and gate fees. When it comes to the use of major policy instruments affecting recycling of plastic waste in the four Nordic countries, they can be summarized as follows:

- All four Nordic countries, except Sweden, use taxes on beverage packaging outside deposit-refund systems.
- All four Nordic countries have deposit-refund systems which include beverage packaging such as plastic bottles. Though the systems differ in the number of product types covered, the collection and recycling of packaging covered by the deposit system are in general high (85–95%).
- Plastic waste generation from packaging is part of an EPR scheme in Finland, Norway and Sweden. All producers and importers of plastic packaging (in Finland with a net turnover exceeding EUR 1 million) are legally responsible for organizing a collection and recycling system for the plastic packaging waste entering the markets. (In Finland the EPR system will cover household plastic packaging as of May 1st 2015.) In Denmark the municipality has the responsibility to establish a collection scheme for plastic packaging from households; the municipality is also responsible for the recycling of the collected waste back into plastic material. It is *likely* that this difference between the countries explains the somewhat lower material recycling rates seen in Denmark compared to the other Nordic countries in the study, which all have EPR schemes.
- All four countries have since several years back introduced landfill taxes. The taxes vary between countries from EUR 50–69/tonne. No studies have been identified which evaluate the effect of the tax on plastic recycling rates. It is difficult to evaluate the tax since most countries have had land fill bans at the same time.
- Sweden and Norway adopted incineration taxes for a number of years until they were abolished in 2010. They may have given incentives for lower incineration rates. Denmark is now the only country that still has an incineration tax. Unfortunately, no evaluation of the effect of the Danish tax on plastic waste recycling has been carried out.
- Some municipalities use marginal-price instruments for garbage collection such as volume- or weight-based fees while other still use flat free-pricing. Evaluations show that municipalities that employ weight-based waste management fees generally experience higher collection rates than municipalities in which flat and/or volume-based fees are used.

- Denmark, Norway and Sweden have statutory bans or limitations on the landfilling of organic or combustible waste, while Finland will introduce a ban in 2016. It is *very likely* that this, in combination with high incineration capacities, explains the relatively high incineration rates seen in Denmark, Norway and Sweden. Similar patterns are seen in other countries with land fill bans. It is *likely* that an increase in incineration rate might be seen also in Finland after the implementation of the land fill ban 2016. It is recommended that the Nordic countries seek for policy solutions operative at the EU level to achieve optimal recycling rates across countries (see further section 9.3).

## 9.2 Design of Policy Instruments related to Producer and Consumer Choices

The market failures related to producer and consumer choices illustrated in figure 6 in chapter 5 have been extensively analysed in the economics literature in the context of recycling rates. Input and output taxes or charges on production or consumption cannot in theory provide recycling incentives *per se* unless they affect behaviour towards substituting to other input materials that are more recyclable. In addition, plastics are used in a vast of applications, and the design of a tax or charge would need to take careful consideration such that it does not give incentives to unintended switches to other materials with even larger externalities.

In achieving optimal recycling rates, the economics literature rather points towards two-tiered instruments combining upstream and downstream measures as for instance deposit-refund systems, or in general upstream taxes/charges combined with downstream taxes/charges or subsidies in as illustrated figure 6 in chapter 5. The economic intuition behind these results is that the upstream-downstream design creates incentives for optimal allocation between reducing consumption and increasing recycling. The empirical evidences in the Nordic countries also shows that deposit and refund systems on beverage packaging have promoted recycling of plastic packaging in all four Nordic countries. The systems have ensured a high and uniform quality of collected plastic waste (85–95%).

Moreover, the EPR systems with financial responsibility used in Finland, Norway and Sweden can to some extent be said to belong to this class of two-tiered instruments using up- and downstream measures. This is because the upstream charges, paid by producers, are used to finance the downstream collection and recycling of the waste generated by the products.

The major disadvantage of these two-tiered instruments is larger administrative costs connected to the monitoring and the verifying needed for charging and refunding. This is likely the reason why the deposit-refund systems in the Nordic countries have been used mainly for homogeneous standardised products such as beverage bottles and standardised reusable packaging such as pallets in the industry sector. The standardisation brings down the administrative costs for monitoring and verifying. It is also likely the reason why EPR systems tend to use simplified methods for calculating producer charges and fees, making the systems to deviate from marginal pricing that would be more optimal in theory (see chapter 8.2 for the case of EPRs in Finland and Sweden).

An extended use of deposit and refund systems, or EPR systems, based on marginal pricing in the Nordic countries is likely dependent on the possibilities of standardising and homogenizing products. The administrative costs would be high with deposit and refund systems for heterogeneous products, such as waste plastic in general. Using, for instance, weight as the pricing unit, the plastic content of the products would need to be weighted by production or purchase and again by recycling. One way to keep transactions costs lower in such systems is to reduce the number of actors in the system, suggesting that producers and recyclers, rather than households, could be actors in the systems. Moreover, simplified measures or proxies for measuring marginal quantities (weight, volume or units of products etc.) should be developed in order to keep monitoring and verifying costs low. Common methods in the Nordic countries for certain waste streams could explore economies of scales in the implementation.

When product design is essential for recyclability the upstream-downstream instruments could build on "recyclability indictors" that connect to product design. Literature suggests that several types of economic instruments based on up- and downstream measures can be designed as first best solutions to take into account aspects of product design externalities related to producer decisions in production. Instruments that connect to recyclability indicators could involve for instance, upstream taxes on production processes, charges fees, disposal-content fees, subsidies for recycling in deposit refund systems as well as EPR schemes. Again the optimal design of such instruments would be a trade-off between the improved product design (in recyclability terms) and the increase in administrative costs for monitoring and verifying the recyclability indicators connected to product design.

There is an extensive body of economics research supporting that marginal cost pricing, or in general unit-based pricing, for collection services is more efficient than flat fees that do not give incentives to increased sorting. Unit-based pricing makes it possible in theory to set the price per unit of garbage collected equal to the social marginal cost of collection and recycling. Historically countries have used to charge flat fees or general municipal taxes to households for waste collection. However, recently variable fees have been implemented in several places. The majority of empirical studies, also in the Nordic countries, suggest that weight-based or volume-based pricing have larger effects than flat fees as predicted by economic theory.

The largest waste plastic flow, plastic packaging waste from households, also includes food containers where the plastic packaging many times has been in direct contact with food. Recycling these containers is combined with larger inconveniences (due to washing or the smell while storing it at home) for households. These inconvenience costs in combination with the low density of waste plastic (making the marginal incentive effect on waste plastic relatively smaller than for other more heavy substances in the mixed household waste) produce a caveat of relying on weight-based systems for promoting plastic waste recycling from households. Further analysis on the magnitude of these effects should be implemented before conclusions can made about the effects of weight-based pricing on waste plastic recycling from households.

## 9.3 Design of Policy Instruments related to Recyclers and Reprocessors Choices

Common for the Nordic markets for waste plastics are demands for a relatively high quality of waste plastic compared to the quality levels supplied. Recycled plastic does not always meet the quality specifications that plastic manufacturers of technical high quality plastic products demand. There is also a tendency that demand is driven by the lower relative price of recycled plastic with respect to virgin plastic at the same time as demand for higher quality recycled plastic has increased during the last years.

The survey to managers in recycling and manufacturing industry in Sweden showed that one explanation for the insufficient supply of higher quality of waste plastic might be asymmetric information between recyclers and manufacturers about the quality of waste plastics on the market. An extension of the seminal Fullerton and Kinnaman (1995) model was developed in this project to analyse the effects and policy implications under asymmetric information. The result suggests that asymmetric information lead to adverse selection in terms of lower efforts in sorting by recyclers resulting in inefficient recycling rates due to lower quality of the waste plastic supplied. However, the presence of asymmetric information did not change the optimal instruments in the model (deposit-refund systems or in general combined upstream-downstream instruments).

In the survey in Sweden managers were asked to grade how effective they believed that different policy instruments would be to increase the supply of waste plastic. EU certification schemes for waste plastic quality were graded and ranked as the second most important instrument following weight-based pricing (managers in recycling industry) and a virgin tax (managers in manufacturing of final goods in plastics). It is notable that managers graded EU certification schemes as more important than subsidies to production. The call for EU certification schemes on waste plastic quality among sellers may be seen as a wish to better signal potential quality of their waste on the recycling market.

Before any policy recommendation is given, it is advisable to further analyse if, and if so to what extent, the lower quality of recycled plastic seen in the Nordic markets can be explained by inefficiency due to asymmetric information between recyclers and manufacturers in the Nordic countries.

## 9.4 Policy Instruments for Achieving Optimal Recycling Rates at International Markets

Small countries, like the Nordic countries, in the EU face small volumes of waste plastic. In addition to this, a majority of the Nordic countries tend to be geographically large with relatively low populations resulting in higher transport and infrastructure costs in waste management relative to many other countries. Moreover, incineration of plastic as part of the residual waste often exhibit lower costs than local sorting and recycling (partly due to the relatively high labour costs for waste treatment).

From a global level it may then be more efficient to maintain lower material recycling rates in Nordic countries and/or export plastic waste to other countries that better can take advantage of the economies of scales in waste plastic recycling technologies. International designs of the economic policy instruments discussed above, for instance a single EPR or harmonized EPRs across countries, a single deposit-refund system or harmonized deposit-refund systems across countries, or alternatively international tradable quotas based on EPR, could in theory result in an efficient outcome at the international level. These systems could in principle be designed to operate at the Nordic level, the EU level, or at an even larger geographical area including also non-European countries. The policy recommendation is to further analyse such international policy instruments primarily at the EU level, eventually together with an EU-wide certification for quality of recycled plastic and/or the end-of-waste criteria for waste plastic as suggested by the European Commission (Villanueva and Eder, 2014)

However, for achieving socially efficient allocations of recycling rates across countries at the international level, the design of instruments used (including other instruments such as EU ETS) should take into account the social costs related to transport and export of plastic waste to other countries in the system They should also take into account life cycle costs in the comparisons of domestic treatments (nevertheless incineration in the Nordic countries) and treatments in importing countries. For instance, LCA studies carried out for plastic packaging waste treatment systems shows that material recycling has advantages compared to incineration, both with regard to GHG emissions and energy resources (WRAP 2006, Raadal *et al*. 2009, Lyng & Modahl 2011, Rigamonti *et al*., 2014). However, these results are sensitive to the amount of virgin materials that is really substituted by recycled materials, as substantial amounts of plastic waste in some cases has to be sorted out due to low quality and ends up in incineration plants.

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# Sammanfattning

Denna rapport sammanfattar resultaten från projektet ”Utvärdering och utformning av ekonomiska styrmedel för ökad återvinning av plastavfall” som initierats av miljö- och ekonomigruppen (MEG) och Nordiska avfallsgruppen (NAG). Syftet med projektet är att utvärdera och identifiera övergripande design hos lämpliga ekonomiska styrmedel som kan bidra till att uppnå samhällsekonomiskt effektiva nivåer av återvinning av plastavfall i Danmark, Finland, Norge och Sverige.

Den första delen (kapitel 2–4) i rapporten ger en bakgrund om status och trender för plastavfallsflöden och behandling i dessa länder. Den ger också en överblick över befintliga politiska styrmedel med en översikt av de viktigaste utmaningarna för att utforma styrmedel i dessa nordiska länder.

Den andra delen (kapitel 5–9) i rapporten använder resultaten från den första delen och granskar främst den nationalekonomiska forskningslitteraturen kring styrmedel för ökad återvinning av plastavfall utifrån ett Nordiskt perspektiv.

## *Befintliga nationella mål och styrmedel i de fyra nordiska länderna i studien*

Plastavfall tas inte specifikt upp i EU:s lagstiftning och ingen av de fyra nordiska länderna i denna studie har ett specifikt mål för återvinning av plast i sin nationella avfallsplan. Men Förpackningsdirektivet (94/62/EG med ändringarna 2004/12/EG och 2005/20/EG) har ett specifikt mål för återvinning av plastförpackningar. Minimikravet är 22,5 % materialåtervinning av vikten för plastavfallet. De nationella målen för återvinning av plastförpackningar är högre än EU:s krav i både Norge och Sverige (30 %). Danmark och Finland har samma mål (22,5 %) som fastställs i förpackningsdirektivet. Baserat på den nationella rapporteringen för 2011 är de nationella målen uppfyllda i Norge och Finland, men inte i Sverige. Danmark rapporterade en något lägre återvinningsgrad under 2011 än vad som krävs enligt direktivet. Metoderna för att beräkna återvinningsgrader skiljer sig dock mellan de nordiska länderna. Gemensamma metoder i de nordiska länderna för plastavfallsflöden skulle kunna utnyttja skalfördelar. Det skulle också öppna upp för att inom en

framtid kunna använda gemensamma styrmedel inom Norden och utnyttja den större effektivitetspotential som finns på den nordiska nivån jämfört nationella nivåer.

De vanligast förekommande ekonomiska styrmedlen inom EU-27 som påverkar plastavfallshantering är producentansvar, pantsystem för homogena produkter såsom dryckesflaskor, skatter och avgifter för omhändertagande och behandling av avfall samt deponi- och förbränningskatter. När det gäller användningen av styrmedel som påverkar återvinning av plastavfall i de fyra nordiska länderna i studien, kan de sammanfattas enligt följande:

- Samtliga nordiska länder i studien, utom Sverige, använder skatter på dryckesförpackningar utanför pantsystemet.
- Samtliga nordiska länder i studien har pantsystem som inkluderar plastflaskor. Även om systemen skiljer sig åt i antalet produkttyper som omfattas, så är återvinningsgraden inom pantsystemen i allmänhet höga (85–95%).
- Plastförpackningsavfall ingår i producentansvarssystemen i Finland, Norge och Sverige. Alla producenter och importörer av plastförpackningar (i Finland med en nettoomsättning som överstiger EUR 1 million) är juridiskt ansvariga för att organisera ett insamlings- och återvinningssystem för plastförpackningar som sätts på marknaden. (I Finland kommer producentansvarssystemet att omfatta hushållens plastförpackningar från och med den 1 maj 2015.) I Danmark har kommunen ansvar för att upprätta ett system för insamling av plastförpackningar från hushåll. Kommunen ansvarar också för återvinning av det insamlade avfallet. Det är sannolikt att denna skillnad mellan länderna kan förklara de något lägre materialåtervinningsnivåer som ses i Danmark jämfört med de andra nordiska länderna i studien, som har producentansvarssystem som inkluderar plastavfall. En tänkbar orsak kan vara att producentansvarssystemen skapar ett mer stabilt avfallsflöde vilket gynnar återvinningsindustrin.
- Samtliga nordiska länder i studien har sedan flera år tillbaka infört deponiskatter. Skatterna varierar mellan länderna från EUR 50–69/ton. Inga studier har identifierats som utvärderar effekten av skatten på plaståtervinningsgraden. Det är också svårt att utvärdera skatten eftersom de flesta länder under samma period också har infört deponiförbud.

- Sverige och Norge hade en förbränningsskatt under ett antal år tills dessa avskaffades 2010. Den kan ha gett incitament till lägre förbränningsnivåer för plastavfall i respektive land. Danmark är nu det enda land som fortfarande har en förbränningsskatt.
- Vissa kommuner använder styckprisinstrument för sophämtning såsom volym- eller viktbaserad avgift, medan andra fortfarande använder fast pris. Utvärderingar visar att kommuner som använder viktbaserade avfallsavgifter har högre insamlingsnivåer än kommuner där fast pris används.
- Danmark, Norge och Sverige har lagstadgade förbud eller begränsningar för deponering av organiskt eller brännbart avfall. Finland kommer att införa ett förbud 2016. Det är mycket sannolikt att detta, i kombination med hög förbränningskapacitet, förklarar de relativt höga förbränningsnivåerna för plastavfall i Danmark, Norge och Sverige. Liknande mönster ses i andra länder med deponiförbud. Det är sannolikt att en ökning av förbränningen av plastavfall kommer att ses även i Finland efter införandet av deponiförbudet 2016. Det rekommenderas att de nordiska länderna i förlängningen söker politiska lösningar på EU-nivå för en mer samhällsekonomiskt effektiv allokering och behandling av plastavfall inom EU-länderna (se vidare avsnitt 9.3).

### *Utformning av styrmedel relaterade till producentval och konsumentval*

De marknadsmisslyckanden som berör producent- och konsumentval, illustrerade i figur 6 i kapitel 5, har analyserats i den ekonomiska litteraturen. Skatter eller avgifter på produktion eller konsumtion kan i teorin inte ge incitament till återvinning i sig så länge den inte påverkar beteendet i riktning mot ett användande av andra produkter som är mer återvinningsbara i sig. Plast används i en stor mängd olika tillämpningar vilket gör det svårare att undvika att skatter eller avgifter även kan skapa incitament till att byta till oavsiktliga material med än större externa effekter.

För att uppnå optimala återvinningsgrader, pekar litteraturen snarare på styrmedel som kombinerar uppströms och nedströms insatser som till exempel pantsystem (eller generellt uppströms skatter/avgifter i kombination med nedströms skatter/avgifter eller subventioner) såsom illustrerats i figur 6 i kapitel 5. Intuitionen bakom dessa resultat är att instrument med en uppströms-nedströmsdesign kan skapa incitament för en optimal fördelning mellan minskad användning av råvaror och ökad återvinning. De empiriska resultaten i de nordiska länderna visar att pantssystem på dryckesförpackningar har främjat återvinning av plastförpackningar i alla fyra nordiska länder i studien. Systemen levererar en hög och jämn kvalitet på insamlat plastavfall (85–95%).

Producentansvarssystem med finansiellt ansvar som används i Finland, Norge och Sverige kan i princip räknas till denna typ av styrmedel med uppströms-nedströmsdesign. De avgifter som betalas ”uppströms” av producenterna används ”nedströms” för att finansiera insamling och återvinning av avfall som genereras av produkterna.

Den största nackdelen med dessa instrument är större administrativa kostnader i samband med den tillsyn och verifiering som krävs för hantera både avgifter och återbetalning eller finansiering. Detta är sannolikt orsaken till att pantsystem i de nordiska länderna har använts främst för homogena standardiserade produkter såsom dryckesflaskor och lastpallar inom industrin. Standardiseringen minskar de administrativa kostnaderna i systemet. Det är också sannolikt anledningen till att producentansvarssystem tenderar att använda förenklade metoder för beräkning av producentavgifter istället för marginalprissättning som skulle vara mer optimalt enligt teorin (se kapitel 8.2 för producentansvarssystemen i Finland och Sverige).

En utökad användning av pantsystem eller producentansvarssystem som i större grad bygger på marginalprissättning i de nordiska länderna är sannolikt beroende av möjligheterna att standardisera produkterna. De administrativa kostnaderna skulle annars bli höga med pantsystem för heterogena produkter, såsom plastavfall i allmänhet. Med t.ex. vikt som prissättningsenhet, skulle plastinnehållet i produkterna behöva vägas vid produktion och vid återvinning. System med färre aktörer i systemet, t.ex. bara tillverkare och återvinningsföretag, snarare än hushåll, kan ge lägre administrativa kostnader. Vidare bör förenklade åtgärder för att mäta lämpliga mängder (vikt, volym eller enheter av produkter etc.) utvecklas för att hålla kostnader för tillsyn och kontroll nere. Genom att utveckla gemensamma metoder i de nordiska länderna för vissa avfallsflöden skulle man även kunna utnyttja skalfördelar i systemen.

När produktdesign är en förutsättning för återvinningsbarhet kan styrmedel med uppströms-nedströmsdesign bygga på ”materialåtervinningsindikatorer” som kopplar till produktdesignen. Forskningslitteraturen har visat att flera olika typer av ekonomiska styrmedel som bygger på uppströms-nedströmsdesign kan utformas för att ta hänsyn till produktdesign. Det kan ske genom att man kopplar avgifter till materialåtervinningsindikatorer. Utformningen av sådana instrument skulle dock bli en kompromiss mellan förbättrad produktdesign (i termer av materialåtervinningsbarhet) och ökningen i administrativa kostnader för tillsyn och verifiering av materialåtervinningsindikatorer kopplade till produktdesign i systemet.

Det finns en omfattande ekonomisk forskning som stödjer användningen av marginalprissättning, eller enhetsbaserad prissättning, för avfallshantering framför fasta avgifter (vilka inte ger incitament på marginalen till ökad sortering). Enhetsbaserad prissättning gör det möjligt att i teorin sätta priset per enhet insamlat avfall lika med den samhällsekonomiska marginalkostnaden för insamling och återvinning. Historiskt sett har länder använt fasta avgifter eller kommunala avgifter för avfallsinsamling. Emellertid har nyligen variabla avgifter provats och införts i flera kommuner. Majoriteten av studierna tyder som väntat på att viktbaserad eller volymbaserad prissättning har större påverkan på sortering än fasta avgifter.

Det största plastavfallsflödet är plastförpackningar från hushåll. I denna ingår även plastförpackningar som många gånger har varit i kontakt med livsmedel. Återvinning av dessa förpackningar är förenat med större olägenheter för hushållen (på grund av diskning eller lukt vid lagring hemma). Dessa kostnader i kombination med den låga densiteten hos plastavfall försvagar den marginella effekten av ett viktbaserat system på plastavfall jämfört andra tyngre ämnen i blandat hushållsavfall. Denna kombination gör det tveksamt om man enbart kan förlita sig på viktbaserade system för ökad återvinning av plastavfall från hushåll. Ytterligare analys av dessa effekter bör genomföras innan man kan dra slutsatser om effekterna av viktbaserad prissättning på återvinning av plastavfall från hushåll.

### *Utformning av styrmedel relaterade till återvinningsindustri och tillverkare*

Gemensamt hos de nordiska marknaderna för plastavfall är det finns en efterfrågan på behandlat plastavfall med en relativt hög kvalitet jämfört med den kvalitet som finns på marknaden. Behandlat plastavfall uppfyller inte alltid den kvalitet som tillverkare av tekniskt högkvalitativ plast efterfrågar. Det finns också en tendens till att efterfrågan drivs av det lägre relativa priset på återvunnen plast i förhållande till jungfrulig plastråvara samtidigt som efterfrågan på återvunnen plast med högre kvalitet har ökat under de senaste åren.

Enkätundersökningen till chefer inom återvinnings- och plasttillverkningsindustrin i Sverige visade att en bidragande förklaring till bristen på behandlat plastavfall på marknaden kan vara asymmetrisk information mellan återvinningsindustri och tillverkare. En modifiering av Fullerton och Kinnaman (1995) modellen användes därför för att analysera effekterna av asymmetrisk information om kvaliteten på behandlat plastavfall på marknaden. Resultatet tyder på att asymmetrisk information kan leda till s.k. ”snedvridet urval” i form av färre åtgärder för sortering och behandling av plastavfall inom återvinningsindustrin vilket också innebär samhällekonomiskt ineffektiva återvinningsgrader.

I enkätundersökningen till chefer ombads de även att gradera hur effektiva de ansåg att olika styrmedel skulle vara för att öka utbudet av kvalitativt plastavfall. En kvalitetscertifiering på EU-nivå bedömdes som det näst viktigaste instrumentet efter viktbaserad prissättning (chefer inom återvinningsindustrin) och skatt på jungfrulig plastråvara (chefer i tillverkning av slutprodukter i plast). Det kan noteras att chefer graderade EU-certifiering som viktigare än subventioner till produktion vilket eventuellt kan ses som ett uttryck för att bättre kunna signalera potentiell kvalitet på återvinningsmarknaden. Innan man kan dra ytterligare slutsatser kring styrmedel bör man analysera om, och i så fall i vilken omfattning, bristen på högkvalitativt behandlat plastavfall på de nordiska marknaderna kan förklaras av asymmetrisk information mellan återvinnings- och plasttillverkningsindustrin i Norden.

### *Styrmedel för internationella marknader*

Små länder, som de nordiska länderna, i EU har små mängder plastavfall. Utöver detta har de stora nordiska länderna tendens att vara geografiskt stora med relativt små populationer som leder till relativt högre kostnader för transporter och infrastruktur inom avfallshantering. Förbränning av plastavfall som en del av restavfallet har ofta lägre kostnader än inhemsk sortering och återvinning, också delvis på grund av högre lönekostnader vid avfallshantering.

Utifrån internationell synvinkel kan det då vara effektivare att hålla lägre materialåtervinningsgrader i Nordiska länder och/eller exportera plastavfall till andra länder som bättre kan dra nytta av tekniska stordriftsfördelar inom plaståtervinning. Internationella utformningar av de ekonomiska styrmedel som diskuterats ovan, till exempel ett enda producentansvarssystem eller harmoniserade producentansvarssystem mellan länder, ett enda pantsystem eller harmoniserade pantsystem mellan länder, alternativt internationell handel med kvoter baserade på producentansvar, kan i teorin leda till en effektiv allokering även på internationell nivå. Dessa styrmedel skulle i princip kunna vara utformade på nordisk nivå, EU-nivå eller ett större geografiskt område som inkluderar även icke-europeiska länder. Rekommendationen är att ytterligare analysera sådana internationella styrmedel främst på EU-nivå, eventuellt tillsammans med EU-certifiering för kvalitet hos återvunnen plast och/eller de end-of-waste kriterier för plastavfall som föreslagits av Europeiska kommissionen (Villanueva och Eder, 2014)

För att uppnå samhällsekonomiskt effektiva allokeringar av behandling av plastavfall mellan länder på internationell nivå, förutsätts att utformningen av de styrmedel (inklusive andra påverkande styrmedel såsom EU ETS) tar hänsyn till samhällsekonomiska kostnader som uppkommer i samband med transport och export av plastavfall till andra länder inom systemet samt livscykelkostnader i jämförelserna mellan inhemska behandlingar (inkl. förbränning) och andra behandlingar i importerande länder. LCA-studier som genomförts för återvinning av plastförpackningar visar att materialåtervinning har fördelar jämfört med förbränning, både när det gäller utsläpp av växthusgaser och energiresurser (WRAP 2006, Raadal *et al*. 2009, Lyng & Modahl 2011, Rigamonti *et al*., 2014). Emellertid är dessa resultat känsliga för den andel av jungfrulig plastråvara som ersätter återvunnet plastmaterial med låg kvalitet och som därmed går till förbränning.

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# Economic Policy Instruments for Plastic Waste

Achieving a high quality of waste plastic materials and recycling processes is a key challenge in closing the resource loops for plastics. This report reviews the status and trends for plastic waste flows and treatment in Denmark, Finland, Norway and Sweden. Furthermore, it gives an overview of existing policy instruments and the main challenges for designing policy instruments for improved recycling of plastic waste in these Nordic countries. The report identifies potential market failures associated with closing the resource loops for plastics. It reviews the economics research literature on policy instrument design for achieving optimal recycling rates and makes policy recommendations from the Nordic perspective. Finally, it presents results from a survey on market conditions to managers in the recycling and plastic manufacturing industry in Sweden.

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